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UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF MONTANA

WILDEARTH GUARDIANS and WESTERN WATERSHEDS PROJECT, *Plaintiffs*, v. U.S. FISH AND WILDLIFE SERVICE, *Defendant*.	Case No. 9:24-cv-00066-DLC-KLD **DECLARATION OF JOHN G. CARTER**

I, JOHN G. CARTER, declare as follows:

1. I am personally aware of the matters set forth below, and if called as a witness, I would and could truthfully testify thereto.

2. I reside in Bondurant, Wyoming where I have lived since 2020.

3. My education includes a Bachelor of Mechanical Engineering degree from the Georgia Institute of Technology, an MBA in Finance from Georgia State University, and a PhD in Ecology from Utah State University.

4. I currently serve on the Board of Directors for Plaintiff Western Watersheds Project. I also serve on the boards of two other non-profit organizations, Yellowstone to Uintas Connection and Kiesha's Preserve. Yellowstone to Uintas Connection is a 501(c)(3) organization I established to promote and restore the regional wildlife corridor connecting the Greater Yellowstone Area to the Uinta Mountains in Utah, helping to bridge the Northern and Southern Rockies ecologically. Kiesha's Preserve, also a 501(c)(3), is a wildlife preserve and research area I established in southeastern Idaho beginning in 1993. It now encompasses over 1,000 acres.

5. For many years, I engaged in environmental consulting and research for clients in government and private industry (see my resume attached as Exhibit 1). I addressed ecosystem impacts from mining, manufacturing, and livestock grazing. Monitoring, data collection, and analysis were instrumental in obtaining permits, designing restoration strategies, evaluating environmental impacts, and meeting legal requirements. I was an expert witness for clients on these issues.

6. I have been exploring the western public lands and national forests since the 1960s. I was enthralled by their mountains and deserts, rivers and streams where I hiked,

climbed, fished, hunted, and watched wildlife. As my knowledge of these systems grew, I became aware of the massive scale of their degradation by livestock grazing. In the 1980s, I began to survey upland and riparian habitats across several states in the West and provide analysis and input to agency permitting processes on livestock grazing, addressing its effects on these ecosystems. I have collected large amounts of data on our public lands and have provided data and analysis to agencies and other organizations and individuals in the form of comments, data summaries, reports and papers.

7. In particular, this professional experience has included seeking out and collecting data on areas where livestock have been excluded in order to provide more accurate comparative analyses with landscapes impacted by cattle grazing.

8. Through my research, I have found livestock-grazed areas demonstrate a loss of keystone native species such as bluebunch wheatgrass, degraded riparian areas lacking instream and bank cover, high rates of sedimentation, and lack of fish habitat. Livestock can lead to the drying up of springs, seeps and streams. Uplands affected by livestock grazing lack ground cover, lose productivity, and suffer accelerated erosion. Plant communities are left in poor condition. And by contrast, in ungrazed areas I have researched, native species flourished, ground cover was near 100%, streams were shaded by vegetation with cover for fish, spawning areas were sediment-free and bank erosion rare.

9. I have witnessed this firsthand at Kiesha's Preserve through personal observations and through our monitoring program, which includes measuring stream flows, fish populations and fish habitat, aquatic invertebrates, water quality, sage grouse habitat structure and change over time, plant community production, and wildlife cameras to document location and timing of wildlife use. We removed cattle from Kiesha's Preserve 30 years ago. Since then, the riparian willows, other trees and shrubs have grown back and shade the streams. Riparian zones have expanded. Aspen stands that were dying out have regenerated and beavers have returned, accelerating expansion of the riparian areas and increasing water storage. Uplands have recovered with the return of native bluebunch wheatgrass and many other native flowers and grasses. Cheatgrass, which was a problem, has disappeared as ground cover of natives increased. The Preserve is grazed by hundreds of deer and elk in winter and during the transitional season in spring. Many remain on the Preserve to raise their young. Five sage grouse leks surround the Preserve. Sage grouse nest in the uplands and raise their broods in the meadows and riparian areas. Migrant birds are plentiful and are successfully nesting across the Preserve habitats. Small mammals are plentiful and function in all habitats. These support raptors and other predators. It's a beautiful and functioning ecosystem. It only needed time to recover from a century of livestock grazing.

10. I am familiar with the Red Rock Lakes National Wildlife Refuge and its livestock grazing program. When the Fish and Wildlife Service (FWS) released its

Environmental Assessment for Arctic Grayling Conservation in 2023, I provided analysis and comments on the draft EA on behalf of several organizations. Also, in early 2024, I again helped several organizations provide comments to FWS regarding the agency's analysis of plans to breach beaver dams at the Refuge to provide arctic grayling access to spawning habitats.

11. My research into the Refuge for such matters has included consideration of data from the State of Montana Water Rights Portal regarding irrigation diversions and areas irrigated on the Refuge, as well as data from the state Department of Environmental Quality on surface water quality. I concluded, for example, that a review of water rights, stream flows, and the needs of the ecosystem was needed due to concerns about stream dewatering and effects on grayling. I also learned that numerous streams within the Refuge and throughout the Centennial Valley are documented as impaired, with livestock grazing listed as a source of impairment.[1]

12. In preparing this declaration, I have reviewed numerous documents on this case docket, including the Complaint, FWS's Motion for Remand, the Krupp Declaration, Plaintiffs' Statement of Undisputed Facts, Plaintiffs' Memorandum in Support of Summary Judgement, and documents from the administrative record including

[1] *See, e.g.,* Exhibit 2 (attached) (Montana Department of Environmental Quality's Water Quality Standards Attainment Reports for two lakes and two streams that occur within the Red Rock Lakes National Wildlife Refuge).

FWS's Compatibility Determination for Prescribed Grazing and the Refuge's Comprehensive Conservation Plan.

13. I have prepared this declaration in order to provide additional perspective regarding points made in the Hogan declaration that FWS attached to its motion for voluntary remand. Based on my academic background and extensive professional experience and expertise in matters relating to livestock grazing on public lands, I think it is important that the generalized and unsupported statements in the Hogan Declaration are better contextualized and their limitations better understood. From my review of the materials described above, I understand that in seeking remand, FWS argues based on factual assertions in the Hogan declaration that the removal of cattle even for a couple years would have significant negative consequences for the Refuge. I provide the following comments and references in response to such claims.

14. In paragraph 12 of the declaration, Mr. Hogan explains that the agency's justification for grazing commercial livestock at the Refuge is the historical presence of bison. Mr. Hogan states that "[t]hese unique habitats evolved with some form of grazing, and grazing is necessary to maintain important ecological values, such as diversity of native vegetation species and habitats for native wildlife." However, I disagree with the conclusions that Hogan draws for several reasons.

15. First, the mere fact that bison had a presence in higher-altitude mountain valleys west of the Great Plains, like the Centennial Valley—where this presence was

likely more seasonal, transient, and less predominant on the landscape than on the plains—cannot reasonably justify the establishment of cattle grazing as a dominant land use purportedly representative of historical ecological conditions. The behavior of bison and their impacts on grassland and riparian ecology are significantly different from cattle. Cattle, for example, spend a disproportionate among of time in riparian habitats and on gentle slopes near water, while bison roam more widely and are less affected by slope or proximity to water. Such differences have been documented in research for decades, and I am familiar with recent studies that have noted how bison are "substantially more effective keystone species for diversifying the community than are domesticated cattle."[2] One recent study summarized some findings on the subject as follows:[3]

> [T]he largest behavioral differences between these species is their use of wetlands and associated woody vegetation There are several ecological and physiological differences between bison and cattle that support the hypothesis that their divergent grazing patterns and habitat preferences will differentially affect riparian systems. Cattle are known to degrade riparian areas due to damage or removal of riparian woody vegetation; effects that cascade into degradation of water quality and large fluctuations in stream temperatures and biogeochemistry. In contrast, bison are more drought and heat-tolerant, allowing them to graze farther from water, especially in hot conditions. Compared with cattle, bison select against areas with woody vegetation and standing water, spend less time

[2] Kyle E. Harms, *Bison outperform cattle at restoring their home on the range*, 119(40) PNAS e2213632119 (2022), https://doi.org/10.1073/pnas.2213632119 (attached as Exhibit 3).
[3] Boyce et al., *Bison Reintroduction to Mixed-Grass Prairie Is Associated With Increases in Bird Diversity and Cervid Occupancy in Riparian Areas*, 10 FRONT. ECOL. EVOL. 821822 (2022), https://doi.org/10.3389/fevo.2022.821822 (internal citations omitted) (attached as Exhibit 4).

> browsing, and specialize more on grasses, as opposed to forbs or woody vegetation.

16. Second, FWS's 2023 compatibility determination describes the grazing systems to be applied on the Refuge as follows: "Select grazing units may receive annual grazing treatments consisting of high intensity-short duration, extended rest, complete rest, and/or on a rotational grazing schedule." I find this approach to the grazing program particularly troubling given my familiarity with the issues from my own research. In 2014, I co-authored a research review of the issues surrounding short-duration grazing as well as other grazing systems such as rest-rotation.[4] That review covered, among other things, the findings and conclusions of two renowned range scientists, Dr. Jerry Holechek and Dr. David Briske. Dr. Holecheck authored numerous editions of the textbook "Range Management: Principles and Practices," which is used to teach range science in universities, and Dr. Briske is the former Editor in Chief of the Journal of Rangeland Ecology and Management. Both are Professors in Range Science, and as covered in the research review I published in 2014, these authors described issues with grazing systems like that deployed by FWS as follows:

> [Holecheck et al.] were also critical of government agencies for adopting [] unproven theories rather than basing management on "scientifically proven range management practices and principles."

[4] *See* Carter et al., *Holistic Management: Misinformation on the Science of Grazed Ecosystems*, INTERNATIONAL JOURNAL OF BIODIVERSITY (2014), https://doi.org/10.1155/2014/163431 (attached as Exhibit 5).

> Briske et al. stated, the rangeland profession has become mired in confusion, misinterpretation, and uncertainty with respect to the evaluation of grazing systems and the development of grazing recommendations and policy decisions. We contend that this has occurred because recommendations have traditionally been based on perception, personal experience, and anecdotal interpretations of management practices, rather than evidence-based assessments of ecosystem responses.

The approach described by FWS for Red Rock Lakes involves crowding cattle into individual pastures, which comes with numerous negative consequences as outlined in my 2014 study, and which does not mimic bison west of the Rockies.

17. Third, while Mr. Hogan's statement that the habitats evolved with some form of grazing is true, his intimation that this necessitates commercial livestock is unfounded. The Refuge and the West have herbivores including deer, elk, pronghorn, moose, grasshoppers, ants, rodents, rabbits, sage grouse, bears, and others. These species all consume herbaceous vegetation. During my professional career, I have analyzed herbivory in particular in parts of southern Utah, and my findings have included the revelation that insects and rodents are major consumers of herbaceous vegetation along with the larger mammals. I believe there would be no lack of herbivory in the Refuge as a natural system without livestock.

18. Mr. Hogan also claims that "Grazing has been shown to increase bird species diversity by enhancing habitat structural heterogeneity… ." Yet based on my expert understanding, Hogan's use of this general statement to make the unsourced

assumption that FWS's grazing program is a net ecological benefit at the Refuge is not sufficiently supported by relevant research. Well-known literature in the field of rangeland ecology has addressed the impacts of livestock grazing on plant communities, vertebrates, insects, foraging guilds, and ecosystem structure. Research has shown that the density and diversity of small mammals is reduced by livestock grazing. Similarly, songbird diversity and abundance increase after cessation of livestock grazing. Insects, which are important herbivores, have been shown to increase in number as livestock density decreases. Researchers as far back as the 1990s have found that the reduction in food, water quality and quantity, loss of perches, nesting sites and nesting cover and vegetational structure in the presence of livestock led to a reduction in biodiversity, replacement of riparian species by upland species and generalists, and loss of some neotropical migrants. Ground nesters tended to decrease in abundance with livestock grazing, while species that prefer open habitats such as cowbirds tended to increase.[5]

19. Mr. Hogan's generalized statement about an increase in biodiversity with grazing may reflect certain observed increases in habitat generalists that may move into grazing altered areas that are no longer usable by the native species adapted to the site in its natural state. But the whole picture is much more complicated; nest cover for ground-nesting native birds, for example, is reduced by grazing, which leaves nests exposed to increased predation on eggs and young.

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[5] Relevant science on these subjects is addressed in the 2014 study I co-authored as cited above.

20. Mr. Hogan claims that "vacatur of the Permits may result in the spread of invasive vegetation in the refuge and loss of biodiversity." But Mr. Hogan's assumption that cattle have net positive impacts on the spread of invasive and noxious species misrepresents the true state of the science on that subject. For example, I am familiar with research showing that cattle grazing fosters invasives because it reduces invasion resistance by decreasing bunchgrass abundance, increasing gaps between perennial plants while trampling reduces biological soil crusts.[6] These effects reflect increases in bare soil, lowered competition for space and water, and a greater area available for invasion by non-native plants. If a healthy soil cover and less soil disturbance is allowed to develop after removal of livestock, the native plant community can recover over time and resist invasion.

21. FWS's Comprehensive Conservation Plan for the Refuge claims that non-native grasses, smooth brome and Kentucky bluegrass are likely to expand if livestock are removed. But I have first-hand experience to rebut this notion. At Kiesha's Preserve, both these grasses remain but are not expanding as native plants and biological soil crusts increase in the absence of livestock. Moreover, sage grouse are using these areas for nesting and brood rearing because there is undisturbed cover. Moose, deer and elk bed in

[6] Reisner et al., *Conditions favouring Bromus tectorum dominance of endangered sagebrush steppe ecosystems*, 50 JOURNAL APPLIED ECOL. 1039 (2013), doi:10.1111/1365-2664.12097 (attached as Exhibit 6).

the taller grasses where they find secure hiding cover. While still present, these non-native species when un-grazed by livestock still provide habitat for native wildlife.

22. In paragraph 14 of Mr. Hogan's declaration, he claims as follows:

> Vacatur of the Permits would also likely result in the increased accumulation of wildfire fuel on the Refuge. Grazing has been shown to reduce wildfire risk in grasslands and other ecosystems, as livestock currently play a critical role in managing vegetation growth by consuming herbaceous plants and grasses. In their absence, vegetation could quickly become overgrown, leading to an excessive accumulation of dry, dead plant material. This build-up of vegetation would create a dense layer of potential fuel for wildfires.

Hogan's statement is written to imply that the Refuge would nearly immediately experience wildfire were it not for the grazing program. I disagree with Mr. Hogan's assessment for the following reasons.

23. First, according to FWS's Comprehensive Conservation Plan for the Refuge, wetland plant communities cover approximately half of the Refuge area. Moreover, maps and aerial imagery show clearly that there are roads along the Refuge boundary to the south (South Valley Road) and north side which can serve as fuel breaks. Also, as I have observed when I looked at the Montana water rights data during my preparation of previous comments on activities at the Refuge, there are many points of water diversion for domestic use, wildlife, fisheries, irrigation or stock watering on the Refuge. The combination of open water, wetlands, roads, fuel breaks, and irrigation reduces fire risk

in those areas. It is also important to recognize that natural fire cycles in native sagebrush, conifer and aspen communities which occur on and adjacent to the Refuge are on the order of decades to centuries.

24. Second, the claim of the "build-up of vegetation" as a fire hazard is a common but scientifically questionable tactic used to justify livestock grazing, fuels treatments, logging and other manipulations across public lands. I have watched Kiesha's Preserve wetlands and uplands for decades and do not see this "build-up." During winter snow cover, rodents and decomposition degrade and recycle the plant organic matter and nutrients back into the soil to support next years' growth. I have even measured this process quantitatively, comparing the standing crop of herbaceous vegetation at the end of summer with that after spring snowmelt: the 900 lbs/acre found at the end of summer declined to 130 lbs/acre by the next spring, and by the end of that summer, no residual production from the prior year was evident.

25. Even when grazing reduces fine fuels, a minimum fuel load remains that can carry fire if the weather is hot, dry, and windy.[7] Unless grazing is so intensive that it nearly removes all vegetative cover, which is an impractical and ecologically harmful level of use, enough fine fuel typically remains to carry fire across the landscape.

[7] *See* Strand et al., *Livestock Grazing Effects on Fuel Loads for Wildland Fire in Sagebrush Dominated Ecosystems*, 1 JOURNAL OF RANGELAND APPLICATIONS 35 (2014) (attached as Exhibit 7).

26. Thus, for the reasons detailed above and based on my experience and expertise, the relevant science and ecological considerations do not support Mr. Hogan's generalized assertions to argue that Red Rock Lakes would suffer from even the short-term removal of commercial livestock. I believe that the removal of livestock from the landscape at Red Rock Lakes would not result in any immediate or short-term ecological harm and would be more likely to benefit native species and habitats and to provide valuable new information about the behavior of Refuge ecology in the absence of effects from cattle.

27. Pursuant to 28 U.S.C. § 1746, I declare under penalty of perjury that the foregoing is true and correct.

Executed this 21st day of May, 2025,

John G. Carter

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John G. Carter

Exhibit 1

Resume
Dr. John Carter
PO Box 464
Bondurant, Wyoming 82922
435-881-5404
Jcoyote23@gmail.com

Education

PhD Biology/Ecology at Utah State University.
Master's Business Administration at Georgia State University.
Bachelor Mechanical Engineering at Georgia Institute of Technology.

Professional Experience

2017 – Present. Ecologist for Yellowstone to Uintas Connection and Kiesha's Preserve.

2012 - 2017. Manager of Yellowstone to Uintas Connection, a tax-exempt corporation dedicated to protection of the corridor connecting the Greater Yellowstone Ecosystem to the Uinta Mountains, Northern and Southern Rockies.

1993 – Present. Owner of Kiesha's Preserve, a 1,000-acre wildlife preserve and research area in Idaho and Wyoming.

1988-2017. Owner of Environmental and Engineering Solutions, LLC., a consulting practice dedicated to finding solutions to the environmental problems of industry.

2001-2010 and 2022-current. Utah Director and Member of the Board for Western Watersheds Project, Inc. a tax- exempt corporation dedicated to protection of watersheds, water quality and wildlife habitat.

1996-2001. President of Willow Creek Ecology, Inc., a tax-exempt corporation dedicated to wildlife habitat conservation and preservation.

1985-1988. President of ERI Logan, Inc. Principal endeavors involved the management of hazardous waste, directing Superfund remedial investigation/ feasibility studies and natural resource damage assessments.

1980-1988. Co-owner and Director of Ecosystem Research institute. Directing ecological research, monitoring, and permitting for watersheds, rivers and streams related to oil shale development, coal mines, hydroelectric and land management issues.

1979-1980. Principal Investigator and Coordinator of Aquatic Programs for Bio-Resources, Inc. A biological consulting firm in Logan, Utah.

Other Qualifications

Past Member of Idaho Academy of Science, Society for Range Management, Member Soil and Water Conservation Society.

Past Registered Mechanical Engineer in Utah and other states.

Member - National Committee for the updating and revision of the "Standard Methods for the Examination of Water and Waste Water." Published by the American Public Health association and the American Water Works Association.

Member - ASTM Committee on biological effects and environmental fate. American Society for Testing and Materials.

Publications and Presentations

Carter, J., Vasquez, E. and Jones, A. 2020. Spatial Analysis of Livestock Grazing and Forest Service Management in the High Uintas Wilderness, Utah. Journal of Geographic Information Systems, 12, 45-69. https://doi.org/10.4236/jgis.2020.122003

Carter, J, J. Catlin, N. Hurwitz, A. Jones, and J. Ratner. 2017. Upland Water and Deferred Rotation Effects on Cattle Use in Riparian and Upland Areas. Rangelands 39(3-4):112-118.

Jones, A., and J. Carter. 2016. Implications of Longer-Term Rest from Grazing in the Sagebrush Steppe: An Alternative Perspective. Journal of Rangeland Applications 3, 1 – 8.

Carter, J., Jones, A., O'brien, M., Ratner, J., and Wuerthner, G. 2014. Holistic Management: Misinformation on the Science of Grazed Ecosystems. International Journal of Biodiversity. 11 p. http://dx.doi.org/10.1155/2014/163431

Catlin, J.C., J.G. Carter, and A.L. Jones. 2011. Range Management in the Face of Climate Change. In: Monaco, T.A. et al comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18 – 20; Logan, Ut. Natural Resources and Environmental Issues, Volume XVII. Pp 207 – 249. S.J. and Jessie E. Quinny Natural Resources Research Library.

Carter, J., J. Chard, and B. Chard. 2011. Moderating Livestock Grazing Effects on Plant Productivity, Nitrogen and Carbon Storage. In: Monaco, T.A. et al comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18 – 20; Logan, Ut. Natural Resources and Environmental Issues, Volume XVII. Pp 191 - 205. S.J. and Jessie E. Quinny Natural Resources Research Library.

Watersheds and Native Trout Spawning Habitat Condition presented at the USU Non-Point Source Conference in 9/07.

Using GIS to Evaluate Capability Criteria, Watershed and Water Supply Outcomes Related to Livestock Grazing. Presented at 2006 International Conference of the Soil and Water Conservation Society in Keystone, Colorado and USU Runoff Conference in 2007.

Grazing Capacity and Rangeland Health, a Tool for Calculating Stocking Levels Using GIS. Presented at 57th annual meeting of the Society for Range Management. Authors: James Catlin, John Carter, and Allison Jones. 2004.

Insight, Development, and Discussion of BLM's "Determination Method": Determining Whether Grazing is the Cause of Impairment. Presented at 57th annual meeting of the Society for Range Management. Authors: James Catlin, John Carter, and Allison Jones. 2004.

Livestock, Degraded Ecosystems and Biodiversity. Presentation at 2002 Rangenet Conference in Boise, Idaho.

Carter, J. 2002. Stink Water: Declining Water Quality Due to Livestock Production. In "Welfare Ranching, Subsidized Destruction of the American West", Island Press. P189-193.

Lamarra, V., Liff, C., and Carter, J. 1986. The Hydrology of the Bear Lake Basin and its Impact on the Trophic State of the Bear Lake, Utah - Idaho. Great Basin Naturalist 46:4 pp. 690 - 705.

Carter, J., Lamarra, V., and Ryel, R. 1986. Drift of Larval Fishes in the upper Colorado River. Journal of Freshwater Ecology 3(4): 556-577.

Lamarra, V., Lamarra, M., and Carter, J. 1985. Ecological investigation of a suspected spawning site of endangered Colorado squawfish in the Yampa River. Great Basin Naturalist 45:127 - 140.

Carter, J., Valdez, R., and Ryel, R. 1985 Fisheries habitat dynamics in the upper Colorado River, Colorado. Journal of Freshwater Ecology 3:249 - 264.

Carter, J. and Lamarra, V. 1983. Environmental management: A data-based ecosystem approach from the oil shale industry. Journal of Environmental Management 17:17 - 34.

Medine, A., Lamarra, V., and Carter, J. 1983. Heavy Metal Distribution and Interactions in the White River Ecosystem. IN: National Conference on Environmental Engineering. (Eds.)

A. Medine and M. Anderson. American Society of Civil Engineers. 867-876 pp.

Carter, J., and Lamarra, V. 1983. An Ecosystems Approach to Environmental Management. In: Aquatic Resources Management of the Colorado River Ecosystem. Proceedings of the 1981 Symposium. P260-286.

Beedlow, P. and Carter, J. 1981. Monitoring Industrial Impacts: A Case History of an Ecosystem Approach from the Oil Shale Industry. IN: Proceedings of a national Workshop on In-Place Resource Inventories, Principles and Practices. Edited by Thomas Braun, Louis House, and H. Lund. Society of American Foresters.

PROJECT DESCRIPTIONS

Noise study to quantify impacts to the wildlife corridor in the Bear River Range, Idaho, delineate sources of impact and define impact zones related to travel corridors including roads, trails, and groomed snowmobile trails.

Monitoring of BLM and Forest Service livestock allotments in Idaho, Utah, and Wyoming. Focus is on determining impacts on stream and upland habitats, performance of grazing systems and water developments.

Survey of stream and riparian habitat conditions in cutthroat trout and bull trout streams in Idaho, Utah and Wyoming coupled to analysis of watershed function and impairment by roads, livestock grazing.

Data collection and research to develop a GIS-based model of watershed performance, soil erosion and runoff as related to ground cover and plant community composition. Based on a case study of Caribou National Forest capability criteria and livestock management in the Bear River Range, Idaho.

Survey of streams and watersheds in grazed and ungrazed watersheds in Utah's Uinta Wilderness to compare conditions, stream flow and determine potential for recovery of these damaged ecosystems.

Expert witness for Western Watersheds Project and Center for Biological Diversity challenge to BLM oil and gas leasing under the IM repealing essential provisions of the 2015 Sage Grouse Plan Amendments.

Expert witness for Western Watersheds Project in WWP v BLM Office of Hearings and Appeals hearing on the Duck Creek Allotment.

Expert witness for Advocates for the West in WWP and Idaho Bird Hunters v. BLM regarding livestock grazing decisions in the Nickel Creek Allotment, Owyhee Resource Area, Idaho.

Field Surveys, analysis, and testimony at hearing.

Expert witness for Earth Justice in Biodiversity Conservation Alliance and Center for Biological Diversity v. Fish and Wildlife Service regarding failure to list Bonneville Cutthroat Trout under the ESA.

Expert witness for Advocates for the West in WWP v K. Lynn Bennett, BLM et al regarding livestock grazing in allotments in the Jarbidge Resource Area, Idaho. Field surveys, analysis, development of settlement conditions and monitoring program design for 1.5 million acres.

Expert witness for Advocates for the West in WWP v. Carpenter regarding livestock grazing decisions for 70 allotments in northern Utah BLM lands. Design and implementation of surveys, analysis, monitoring program and ecological reference areas under settlement agreement for 3.2 million acres.

Expert witness for Advocates for the West in WWP v. BLM regarding range developments and grazing systems on the Duck Creek Allotment, Rich County, Utah.

Expert witness for Advocates for the West in WWP v State of Idaho regarding the Robinson Hole Allotment. Research and testimony on livestock grazing effects on the State Land Allotment in sage steppe, aspen and riparian areas.

Expert witness for Advocates for the West in WWP vs State of Idaho, Lacey Meadows Allotment. Research and preparation of testimony on livestock grazing, fire, and wildlife in northern Idaho forests.

Expert witness in Hells Canyon Preservation Council vs Forest Service. Site and document review to prepare testimony regarding livestock impacts to the Imnaha River, Oregon and its endangered fish.

Expert witness for Nucor Corporation in United States v. Nucor Corporation. The case involved literature review, technical analysis and testimony regarding building ventilation, meteorology, emission capture and control technologies and economics related to fugitive emissions regulatory requirements.

Principal investigator and expert witness for the state of Colorado Department of Law Natural Resource Damage Assessments. Lawsuits filed under Superfund against four major companies for hazardous waste damage (heavy metals and radiological) to natural resources. Issues addressed included data management and technical investigations for soil, surface water, vegetation, aquatic life and wildlife. Sites investigated included:

ASARCO Globe Smelter in Denver, Colorado

Idarado Mining and Milling facilities in Telluride and Ouray,
Colorado Cotter Uranium Mill in Canon City, Colorado
Umetco Uranium Mill in Uravan, Colorado

Detailed studies were conducted for the aquatic ecosystems including habitat, flow, chemical composition, algae, invertebrates, and fish. These were conducted in many tributaries of the Colorado River including: Dolores River, San Miguel River and Red Mountain Creek as well as the South Platte River and Arkansas River. Upland soil and plant communities were also characterized and damages assessed.

Expert for Firm of Hult & Marychild regarding environmental damages from trespass construction, spring and habitat damage on private property, Logan Canyon, Utah.

Expert for Thomas Fehr, Attorney representing clients with residences suffering exposure from below ground contamination from leaking gasoline tank.

Independent technical advisor for law firms of Barrett & Daines and Olson & Hoggan in Wellsville City vs. Dairy Product Services. Performed air and water sampling, process evaluations, analyses, and reporting to both parties as well as at City Council meetings to inform the public.

Expert for Firm of Don, Hiller & Gelleher regarding environmental contamination by metals, radiologicals and other uranium process-related compounds in soils, groundwater, surface water and biota associated with the UMETCO Uranium Mill, Uravan, Colorado.

Litigation and technical support to Envirocare of Utah in review of environmental and contamination characteristics of abandoned uranium mining, milling and disposal sites in Washington, Colorado, Tennessee, and Utah. Characterization of contaminant sources, transport and fate including analysis of geology, soils, vegetation, ground water, surface water, vegetation, and wildlife.

Survey of Kaibab Plateau, Arizona in relation to livestock grazing on the Grand Canyon Trust's newly acquired Kane Ranch Allotments. Determined current forage production versus potential. Used Kaibab National Forest capability criteria and current conditions in a GIS analysis to evaluate capability, stocking rate and design system for sustainable grazing management on the Central Winter and Central Summer Allotments.

Research into marsh habitat dynamics and development of a habitat management plan for a 42-acre marsh and surrounding upland ecosystem in Northern Utah. Management recommendations included developing constructed wetlands for storm water management. Client: Nucor Steel.

Research and enhancement of an existing wetland mitigation system to increase biological production and diversity in Mudflat/Playa habitats in Northern Utah. Client: Nucor Steel.

Monitoring and analysis of historical water quality data for the new Grand Staircase Escalante National Monument in Southern Utah. Willow Creek Ecology.

Research into the extent of water resources, baseline data collection, fisheries study in Indian Creek and the Colorado River for development of a long-term monitoring and management program for the Needles District, Canyonlands National Park. Client: National Park Service.

Ecosystem level study of heavy metal dynamics in the White River Ecosystem, Utah. Relationships were determined between ambient, dynamic concentrations of metals in the groundwater and river system and accumulation in sediment, algae, detritus, invertebrates, and fish. Chemical rate constants were developed for state changes in heavy metals related to physical factors. Client: Phillips Petroleum, Sohio, and Sun Oil.

An ecosystem level study of the White River, Utah examined physical, chemical and biological parameters over four years. The objective of the study was to determine important ecosystem relationships that could be used in long term environmental impact detection and monitoring. Client: Phillips Petroleum, Sohio, and Sun Oil.

In conjunction with a fisheries study being conducted by the U.S. Fish and Wildlife Service in the White River, Utah, and Colorado, studied habitat, water quality and fish food habits based on stomach contents and comparisons to benthic and drifting invertebrates. Client: Phillips Petroleum, Sohio, and Sun Oil.

An arid region watershed study to determine the factors influencing water quality and biology in coal strip mine lakes. The results of this study were used to develop the designs for reclamation of these lakes and their watersheds to promote cold water fisheries at closure. Client: Kemmerer Coal Company.

Habitat and water quality study of Spawn Creek, a tributary of the Logan River, Utah. Work performed for Bridgerland Audubon Society, Cache Angler's Chapter of Trout Unlimited and Citizens for Protection of Logan Canyon.

Development of a data management system and statistical ecological model for the White River Shale Oil Corporation Tracts Ua and Ub, Uintah Basin, Utah. The project objective was to integrate ten years of environmental data including air quality, meteorology, soils, vegetation, hydrology, terrestrial and aquatic biology into a model which could be used in a real-time impact detection and monitoring program. Client: Phillips Petroleum, Sohio and Sun Oil.

Aquatic biology baseline study for the Union Oil Shale Project on Parachute Creek, Colorado.

This study evaluated stream flow, habitat, algae, invertebrates, and Colorado cutthroat trout throughout Parachute Creek. Client: Union Oil of California.

Determination of herbicide impacts upon aquatic ecosystems, a laboratory and population modeling approach. Client: Ciba-Geigy Corporation of Basle, Switzerland.

Literature review and development of a document and data management system for Kennecott Copper smelter. This information was used to develop study designs to determine regional soil and surface water contamination by heavy metals that might be attributed to the smelter. Client: Kennecott Copper.

Fishery survey team member for an investigation of habitat and fish populations in the Escalante River near Harris Wash, Utah. Work performed for BioWest, Inc. Logan, Utah.

Reviewed and analyzed potential contamination pathways and ecological effects of metals and ammonia leaking from the Atlas Uranium tailings into the Colorado River and Moab Marsh. Special emphasis was on impacts to endangered Colorado squawfish. Client: Grand Canyon Trust.

Assessment of Egin Diversion Dam as a fish barrier. Study performed on the Snake River at the St. Anthony, Idaho hydroelectric project to relate discharge, velocity, and stage relationships to ability of cutthroat trout to negotiate barriers. Client: Utah Power and Light.

Study design for turbine mortality evaluation on cutthroat trout at the St. Anthony, Idaho hydroelectric project on the Snake River. Client: Utah Power and Light.

Instream Flow analysis for the Weber Hydroelectric Project to determine minimum stream flows for protection of Bonneville cutthroat trout fishery. Client: Utah Power and Light.

Stream rehabilitation design and construction for Teton Creek, Idaho to restore livestock-damaged stream habitat for spawning and maintenance of cutthroat trout populations. Client: Bonneville Pacific Power Company.

Investigation of fish migration at the Felt Hydroelectric Project on the Teton River, Idaho to determine if the diversion dam formed a barrier to cutthroat trout migration. Client: Bonneville Pacific Power Co.

An evaluation of mitigative alternatives for meeting terminus oxygen criteria for the Snake River at the proposed Island Park Hydroelectric Project in Island Park, Idaho. The study was designed to evaluate temperature, discharge and dissolved oxygen saturation to determine risks of gas bubble disease to cutthroat trout in Henry’s Fork. Client: Bonneville Pacific Power Co.

Environmental Assessment for the Island Park Hydroelectric Project. Assessment performed to assemble all historical ecological information for the Henry's Fork of the Snake River and the regions to address concerns of all interested private and public agencies. Client: Fall River Rural Electric cooperative and the U.S. Forest Service, Targhee National Forest.

Preparation of Exhibit E Environmental Reports describing the Environmental setting of three Utah projects for relicensing: Granite, Fountain Green, and Snake Creek. These reports described the environmental setting of the projects including vegetative cover, fish and wildlife resources, water quality and quantity, and scenic and aesthetic resources. Client: Utah Power and Light.

A study of larval fish drift dynamics in the Upper Colorado River, near Parachute Colorado was undertaken to determine the relationship between larval fish numbers with temperature, time and season, and position within the river (depth and horizontal distributions). This study was carried out as Consultation under Section 7 of the Endangered Species Act to determine the potential impacts on larval fish from a proposed water withdrawal. Client: Union Oil of California.

A study of habitat dynamics in the Upper Colorado River in relation to flow quantity. This study was carried out as Consultation under Section 7 of the Endangered Species Act to measure the changes in various habitat types with flow regime and determine the effects of water withdrawal to important habitats for threatened and endangered Colorado River fish. Client: Union Oil of California.

Assembled and evaluated the quality of the historical database for threatened and endangered fish species and their habitats in the Upper Colorado River. This project collected data from all known studies including government and private studies, evaluated data quality and assembled the final data base for analysis. Client: Denver Water Board.

Technical Adviser under an EPA TAG grant to South Weber Coalition from 1992 – 2016 for multiple Superfund sites on Hill Air Force Base, Ogden, Utah.

Investigation of potential soil and groundwater contamination and development of corrective action plan. Client: Logan Manufacturing, Co. Logan, Utah.

Development and preparation of a closure plan for potential soil and ground water contamination by organic solvents. Negotiations with the Utah State Department of Health, plan development and subsequent investigation and clean closure performed for Browning Arms, Morgan, Utah.

Investigation and bio-remediation of soil contamination from oil and diesel spills. Client: Nucor Steel Jewett, Texas.

Waste stream minimization through process modification and waste treatment alternatives including recycling to reclaim waste solvents. Engineering performed for Browning Manufacturing Co. Morgan, Utah.

Principal investigator and expert witness for an investigation of soil, vegetation, and ground water contamination at the old Wasatch Chemical Plant, a Superfund site in Salt Lake City, Utah. Client: Huntsman - Christiansen Co.

Development of environmental criteria for setting remedial action performance standards at Colorado hazardous waste sites. Projects coordinated efforts among a committee of technical consultants, Colorado Department of Health, and Colorado Department of Law. Client: Colorado Department of Law.

Investigation of potential liability for contamination of the city of Milwaukee water supply for an aluminum die casting operation in Portland, Oregon. Client: Browning Arms Subsidiary Bag Boy Golf, Milwaukee, Oregon.

Remediation of soil and ground water contamination for underground solvent leak at a large steel fabrication facility. Client: Vulcraft, Division of Nucor Steel, Brigham City, Utah.

Exhibit 2

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

ASSESSMENT UNIT INFORMATION

Reporting Cycle: 2020

Assessment Unit: MT41A005_030

Waterbody Name: Upper Red Rock Lake

Location Description: UPPER RED ROCK LAKE

Water Type:	Size (Miles/Acres)	Use Class:
FRESHWATER LAKE	2947 ACRES	B-1

Hydrologic Unit Code: 10020001

HUC Name: Red Rock

Watershed: Missouri Headwaters

Basin: Upper Missouri

TMDL Planning Area: Red Rock

Ecoregion: Middle Rockies

County: Beaverhead County

Lat/Long AU Start (U/S):

Lat/Long AU End (D/S):

MONITORING INFORMATION

Date Assessment Started: 12/12/1999

Assessed By: Tippie, Jessi

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

CITATIONS

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	quantitative physical data
Wells, Jerry D. ; Decker-Hess, Janet ; McMullin, Steve L. ; Oswald, Richard A. (1986), Southwest Montana Fisheries Study: Inventory and Survey of the Waters of the Big Hole, Ruby and Beaverhead Drainages: July 1, 1979 through July 1, 1986, F-9-R-28 through F-9-R-34 Job # I-b	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	quantitative physical data
Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	common ions, pH, conductivity, miscellaneous; quantitative physical data

Comments: Only formatted in c-s linkage

Montana DEQ - Water Quality Standards Attainment Record

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

DATA MATRIX
Biological Data

Comments:

Upper Red Rock Lake

Data Type	Comments	Ref Num	Citation
fish	arctic grayling, brook trout, sucker present; grayling numbers decreasing, brook and sucker numbers increasing; suckers are believed to be competing with grayling for food	526	Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106
fish	arctic grayling, brook, cutthroat, burbot and white sucker present; 1979 gill net statistics: 16 grayling (12.7 to 17.1 inches long), 3 cutthroat (6.0 to 7.0 inches long), 4 brook (8.7 to 17.5 inches long), 3 burbot (14.3 to 15.5 inches long), 184 white suckers (11.9 to 16.2 inches long)	1484	Wells, Jerry D. ; Decker-Hess, Janet ; McMullin, Steve L. ; Oswald, Richard A. (1986), Southwest Montana Fisheries Study: Inventory and Survey of the Waters of the Big Hole, Ruby and Beaverhead Drainages: July 1, 1979 through July 1, 1986, F-9-R-28 through F-9-R-34 Job # I-b
fish	grayling captured were 190 mm to 422 mm in length (age 1 year to 7 years);no population estimates available; grayling population has decreased dramatically in the past decades, and this may be an indication of imminent extinction	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

DATA MATRIX

Habitat Data

Comments:

Upper Red Rock Lake

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	approx. 2206 surface acres; max.depth = 6 ft (much of the lake is < 4 ft); historically, lake has been as deep as 25 feet, but increased sedimentation has led to decreased depth; lake bottom consists of mucky matter composed of decaying vegetation, other organic material and mineral soil; aquatic plants are so thick throughout lake that boats are unable to navigate; organic material builds up quickly in lake and adds to the decreased depth already caused by excessive siltation; lake habitat is considered marginal for grayling, brook and cutthroat trout; fish are only able to survive by taking advantage of the cool springs and inlets that feed the lake	526	Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106
riparian &/or instream surveys & physical features	approx. 4.3 km long by 3.2 km wide; encompasses 893 ha; max depth = less than 2.0 m; lake bottom consists of mud, peat and detritus and thick aquatic vegetation is present throughout the lake; bordered by marshes, meadowlands and smaller open water areas; water level of the lake was very low in the summer of 1994 and thermal conditions were believed to be more severe than in Red Rock Creek; drainage is naturally erosive and large amounts of sediment are deposited in the lake; poor conservation practices (ie cattle grazing and agriculture) have hastened the rate of sedimentation; lake is quickly becoming too shallow to support healthy grayling population due to accelerated eutrophication; thermal stratification rarely occurs during the summer since lake depth is decreasing and grayling are left without refuge from the heat; deterioration of lake habitat ,that could ultimately lead to grayling extinction, will likely continue without a change in land use practices and/or if the lake is not deepened through artificial means	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

DATA MATRIX

Chemistry Data

Comments:

Upper Red Rock Lake

Data Type	Comments	Ref Num	Citation
common ions, pH, conductivity, miscellaneous	DO often > 1.0 ppm in late winter	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)
quantitative physical data	max. T = 76 F (summer of 1952)	526	Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106

ASSESSMENT HISTORY

Cycle 2006

Not assessed this cycle

Cycle 2008

Not assessed this cycle

Cycle 2010

Not assessed this cycle

Cycle 2012

Not assessed this cycle

Cycle 2014

Not assessed this cycle

Cycle 2016

Not assessed this cycle

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

Cycle 2018

Removed Other Flow regime Alterations as a cause for Primary Contact Recreation (PCR). Changed PCR to Not Assessed as it no longer had any causes associated with it. Other Flow Regime Alterations cause was changed to Flow Regime Modification per EPA cause list

Cycle 2020

Not assessed this cycle

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

Overall Condition of Segment

Aquatic Life & Cold Water Fishery: BIOLOGY- fishery is severely impaired; arctic grayling numbers have significantly decreased; sucker numbers have increased; HABITAT- severe impairment; lake depth rarely exceeds 4 ft; lake habitat considered marginal for grayling, brook and cutthroat trout; lake is becoming too shallow to support healthy fishery; sedimentation due to overgrazing and agriculture is continuing to impair lake, thick aquatic vegetation also impairs the lake; CHEMISTRY- impairment level unknown due to lack of water chemistry data. Agriculture, Industrial & Drinking Water: need water chemistry data.

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

USE SUPPORT DECISION

Use Class B-1

Trophic Status: UNKNOWN **Trophic Trend:** Unknown

Uses	DQA	Method, Data, and Information Used	Assessment Type and Confidence	Use Support	Partial Flag	Use Support Certainty	Threatened
Aquatic Life				Not Fully Supporting	No		No
Agricultural				Not Assessed	No		No
Drinking Water				Not Assessed	No		No
Primary Contact Recreation				Not Assessed	No		No

Method Number and Description

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

IMPAIRMENT INFORMATION

Uses	Cause (Confidence): Source(Confirmed)	Observed Effects
Aquatic Life	371 (): 46 (N), 108 (N), 147 (N), 156 (N) 526 (): 46 (N), 108 (N), 147 (N), 156 (N)	
Agricultural		
Drinking Water		
Primary Contact Recreation		

Cause Number and Description	Source Number and Description	Observed Effect Number and Description
371-Sedimentation/Siltation 526-Flow Regime Modification	46-Grazing in Riparian or Shoreline Zones 108-Rangeland Grazing 147-Upstream Source 156-Agriculture	

DELISTING / STATUS CHANGES

Cause	Reason for Change	Date of Change

Reporting Cycle: 2020 **Assessment Record:** MT41A005_030.pdf **Status:** Unassigned

CATEGORY INFORMATION

Previous Cycle

Cycle	2018
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Current Cycle

Cycle	2020
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

ASSESSMENT UNIT INFORMATION

Reporting Cycle: 2020

Assessment Unit: MT41A005_020

Waterbody Name: Lower Red Rock Lake

Location Description: LOWER RED ROCK LAKE

Water Type:	Size (Miles/Acres)	Use Class:
FRESHWATER LAKE	2217.5 ACRES	B-1

Hydrologic Unit Code: 10020001

HUC Name: Red Rock

Watershed: Missouri Headwaters

Basin: Upper Missouri

TMDL Planning Area: Red Rock

Ecoregion: Middle Rockies

County: Beaverhead County

Lat/Long AU Start (U/S):

Lat/Long AU End (D/S):

MONITORING INFORMATION

Date Assessment Started: 12/09/1999

Assessed By: Tippie, Jessi

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

CITATIONS

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	quantitative physical data
Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	common ions, pH, conductivity, miscellaneous; quantitative physical data

Comments: Only formatted in c-s linkage

DATA MATRIX
Biological Data

Comments:

Lower Red Rock Lake

Data Type	Comments	Ref Num	Citation
fish	sucker, brook, arctic grayling present; grayling numbers decreasing, brook and sucker numbers increasing; suckers are believed to be competing with grayling for food	526	Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106
fish	grayling captured were 190 mm to 422 mm in length (ages 1 year to 7 years); grayling population in lake is small due to low water levels; most grayling spend their lives in the upper lake; grayling population definitely impaired in lower lake	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

DATA MATRIX

Habitat Data

Comments:

Lower Red Rock Lake

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	approx. 1126 surface acres; max. depth = 3 ft; airboat is required to travel on lake due to dense vegetation; historically, lake has been deeper (6 feet in places), but increased sedimentation has led to decreased depth; lake bottom consists of mucky matter composed of decaying vegetation, other organic material and mineral soil; organic material is found to accumulate quickly on lake bottom; lake has been known to go almost completely dry in the summer months, leaving grayling stranded on the shore; increased land use (grazing, agriculture, etc) in basin has made lake almost completely unable to support the grayling population	526	Vincent, Richard (1962), Biogeographical and Ecological Factors Contributing to the Decline of Arctic Grayling, Thymallus arcticus pallas, in Michigan and Montana, T-106
riparian &/or instream surveys & physical features	approx. 2.7 km long x 2.6 km wide; encompasses 456 ha; max.depth = 0.5 m; lake bottom consists of mud, peat and detritus and aquatic vegetation is present throughout the lake; bordered by marshes, meadowlands and smaller open water areas; most of the lake freezes solid in the winter, so grayling are not likely to over-winter in the lake; lake is too shallow to support healthy grayling population due to accelerated eutrophication; thermal stratification rarely occurs the summer since lake depth is decreasing and grayling are left without refuge from the heat; deterioration of lake habitat will likely continue without a change in land use practices and/or if the lakes are not deepened through artificial means	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

DATA MATRIX

Chemistry Data

Comments:

Lower Red Rock Lake			
Data Type	**Comments**	**Ref Num**	**Citation**
common ions, pH, conductivity, miscellaneous	DO often < 1.0 ppm in late winter	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)

ASSESSMENT HISTORY

Cycle 2006

Not assessed this cycle

Cycle 2008

Not assessed this cycle

Cycle 2010

Not assessed this cycle

Cycle 2012

Not assessed this cycle

Cycle 2014

Not assessed this cycle

Cycle 2016

Not assessed this cycle

Cycle 2018

Removed Other Flow regime Alterations as a cause for Primary Contact Recreation (PCR). Changed PCR to Not Assessed as it no longer had any causes associated with it. Other Flow Regime Alterations cause was changed to Flow Regime Modification per EPA cause list

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

Cycle 2020

Not assessed this cycle

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

Overall Condition of Segment

Aquatic Life/Cold Water Fishery: BIOLOGY - fishery is severely impaired; arctic grayling numbers have significantly decreased; numbers of suckers have increased; HABITAT - severe impairment; lake depth rarely exceeds 3 ft; lake is too shallow to continue to support grayling population; sedimentation due to overgrazing and agriculture is continuing to impair lake, thick aquatic vegetation also impact the lake; CHEMISTRY - impairment unknown due to lack of water chemistry data Agriculture/Industrial/Drinking Water: need water chemistry data.

Montana DEQ - Water Quality Standards Attainment Record

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

USE SUPPORT DECISION

Use Class B-1

Trophic Status: UNKNOWN **Trophic Trend:** Unknown

Uses	DQA	Method, Data, and Information Used	Assessment Type and Confidence	Use Support	Partial Flag	Use Support Certainty	Threatened
Aquatic Life				Not Fully Supporting	No		No
Agricultural				Not Assessed	No		No
Drinking Water				Not Assessed	No		No
Primary Contact Recreation				Not Assessed	No		No

Method Number and Description

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

IMPAIRMENT INFORMATION

Uses	Cause (Confidence): Source(Confirmed)	Observed Effects
Aquatic Life	371 (): 46 (N), 108 (N), 147 (N), 156 (N), 163 (N) 526 (): 46 (N), 108 (N), 147 (N), 156 (N), 163 (N)	
Agricultural		
Drinking Water		
Primary Contact Recreation		

Cause Number and Description	Source Number and Description	Observed Effect Number and Description
371-Sedimentation/Siltation 526-Flow Regime Modification	46-Grazing in Riparian or Shoreline Zones 108-Rangeland Grazing 147-Upstream Source 156-Agriculture 163-Low Water Crossing	

DELISTING / STATUS CHANGES

Cause	Reason for Change	Date of Change

Reporting Cycle: 2020 **Assessment Record:** MT41A005_020.pdf **Status:** Unassigned

CATEGORY INFORMATION

Previous Cycle

Cycle	2018
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Current Cycle

Cycle	2020
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

ASSESSMENT UNIT INFORMATION

Reporting Cycle: 2020

Assessment Unit: MT41A004_040

Waterbody Name: Corral Creek

Location Description: CORRAL CREEK, headwaters to mouth (Red Rock Creek)

Water Type:	Size (Miles/Acres)	Use Class:
RIVER	4.29 MILES	B-1

Hydrologic Unit Code: 10020001

HUC Name: Red Rock

Watershed: Missouri Headwaters

Basin: Upper Missouri

TMDL Planning Area: Red Rock

Ecoregion: Middle Rockies

County: Beaverhead County

Lat/Long AU Start (U/S): 44.576421 / -111.589099

Lat/Long AU End (D/S): 44.61376 / -111.609968

MONITORING INFORMATION

Date Assessment Started: 11/18/2019

Assessed By: McWilliams, Elizabeth

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

CITATIONS

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Rosgen, David L. (1996), Applied River Morphology	Box		riparian &/or instream surveys & physical features	Rosgen type; quantitative physical data
(2003), DEQ Field Assessment Form	Assessment Record	algae; chlorophyll; fish; macroinvertebrates; wildlife	Land use; photo points; riparian &/or instream surveys & physical features	Rosgen type; benthic sediment data; common ions, pH, conductivity, miscellaneous; major nutrients; metals; quantitative physical data
Bahls, Loren L. (2004), Biological Integrity of Streams in the Red Rock River TMDL Planning Area Based on the Structure and Composition of the Benthic Algae Community, Draft	WQPB Ebrary	algae		
Suplee, Michael W. ; Watson, Vicki ; Varghese, Arun ; Cleland, Joshua (2008), Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers	WQPB Ebrary	chlorophyll		major nutrients
(200n), Montana Interagency Stream Fishery Data for the Upper Missouri River Basin	DEQ PPA Data Archive	fish	Land use; photo points; riparian &/or instream surveys & physical features	quantitative physical data
Archer, Eric K. (2012), PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO-EMP) for Streams and Riparian Areas: 2012 Sampling Protocol for Stream Channel Attributes	WQPB Ebrary		riparian &/or instream surveys & physical features	quantitative physical data
Kusnierz, Paul ; Welch, Andy ; Kron, Darrin (2013), The Montana Department of Environmental Quality Western Montana Sediment Assessment Method: Considerations, Physical and Biological Parameters, and Decision Making-Draft, WQPBMASTR-02	WQPB Ebrary		riparian &/or instream surveys & physical features	quantitative physical data

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Montana Department of Environmental Quality (2014), Department Circular DEQ-12A: Montana Base Numeric Nutrient Standards, Circular DEQ-12A	WQPB Ebrary			major nutrients
Montana Department of Environmental Quality (2016), Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels, WQPBMASTR-01	WQPB Ebrary	General; algae; chlorophyll; macroinvertebrates		General; common ions, pH, conductivity, miscellaneous; major nutrients
(2019), Escherichia coli (E. coli) Assessment Method for State Surface Waters		e-coli		
U.S. Department of Agriculture, Forest Service (20nn), Forest Service PIBO Data Through 2001-2014 [Electronic Resource]	DEQ Metcalf Multimedia Case	macroinvertebrates	riparian &/or instream surveys & physical features	quantitative physical data
Govan, Sybil (nnnn), Natural Resources Information System	Montana State Library	fecal coliforms; fish	General; Land use; photo points; riparian &/or instream surveys & physical features	benthic sediment data; common ions, pH, conductivity, miscellaneous; major nutrients; metals; quantitative physical data
Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS	WEB	General; algae; chlorophyll; e-coli; macroinvertebrates	riparian &/or instream surveys & physical features	General; General; benthic sediment data; common ions, pH, conductivity, miscellaneous; major nutrients; metals; organics; quantitative physical data

Comments:

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

DATA MATRIX
Biological Data

Comments: ALGAE: In the years 2016 to 2018, 4 composite samples were collected for benthic chlorophyll-a and ash-free dry weight. The chl-a results are 2.3, 0.4, 0.4, 2.9 mg/m2. The AFDW results are 1.82, 0.09, 2.35, 3.76 g/m2.

MACRO: In the years 2016-2018, 3 samples were collected. The HBI scores are 4.37, 4.85, and 5.99.

PATHOGENS: In the year 2017, 6 samples were collected for E. coli. The results range 37.9 to 261.3 cfu/100mL. There is insufficient information to calculate a 30 day or seasonal geometric mean.

(Pre 2020 information) Fish: brook trout were present; rated "abundant". Resource value rating: "Outstanding". Historically (as early as 1907, with recent observations by MTFWP personnel), this stream served as a spawning location for grayling (likely adfluvial) from Red Rock Lake. Periphyton: Upper site: Both sites had one or more genera of filamentous green algae, and pollution-tolerant algae (Pollution Index: 2.65). Hydrurus foetidus, a cold stenotherm was collected at the upper Corral Cr. site, where it was abundant. Good biological integrity and minor siltation (Siltation Index: 20.86) is indicated here.Hannaea arcus was present, a pollution-sensitive species. Lower Site: The lower site had good biological integrity, but had a lower Pollution Index (2.50) and higher Siltation Index (27.35) than the upper site. Ground water inputs are indicated by the large number of non-motile, pollution-sensitive diatoms. The two sites shared 38% of their diatom assemblages.

Corral Cr, about 2 mi upstream for the mouth; on BLM land

Data Type	Comments	Ref Num	Citation
algae	Both sites had one or more genera of filamentous green algae, and pollution-tolerant algae (Pollution Index: 2.65). Hydrurus foetidus, a cold stenotherm was collected at the upper Corral Cr. site, where it was abundant. The largest percentage of abnormal diatom cells (0.96 %) was recorded in upper Corral Cr. Good biological integrity and minor siltation (Siltation Index: 20.86) is indicated here. This site had the highest Disturbance Index (31.53) of all sites in the study (likely natural). Hannaea arcus was present, a pollution-sensitive species.	10576	Bahls, Loren L. (2004), Biological Integrity of Streams in the Red Rock River TMDL Planning Area Based on the Structure and Composition of the Benthic Algae Community, Draft
fish	1980 Fish survey: brook trout were present; 7/24/80: 547 fish/1000, 8/01/80: 141 fish/1000; rated "abundant". Resource	4655	(200n), Montana Interagency Stream Fishery Data for the Upper Missouri River Basin

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	value rating: "Outstanding". Historically (as early as 1907, with recent observations by MTFWP personnel), this stream served as a spawning location for grayling (likely adfluvial) from Red Rock Lake.		

Corral Cr, 1/4 mi. upstream from the mouth

Data Type	Comments	Ref Num	Citation
algae	The lower site had good biological integrity, but had a lower Pollution Index (2.50) and higher Siltation Index (27.35) than the upper site. Ground water inputs are indicated by the large number of non-motile, pollution-sensitive diatoms. The two sites shared 38% of their diatom assemblages.	10576	Bahls, Loren L. (2004), Biological Integrity of Streams in the Red Rock River TMDL Planning Area Based on the Structure and Composition of the Benthic Algae Community, Draft

Entire Segment

Data Type	Comments	Ref Num	Citation
chlorophyll	4 Chlorophyll-a and 4 Ash-Free Dry Weight composite results from 2016-2018.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
e-coli	This document contains the assessment method with which E. coli data were analyzed for beneficial use determination.	15759	(2019), Escherichia coli (E. coli) Assessment Method for State Surface Waters
e-coli	2 E. coli samples (from one site) were collected in July 2017, and 4 E. coli samples (2 at one site, 1 at one site, 1 at one site) were collected from August 2017.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
macroinvertebrates	3 Macroinvertebrate samples were collected from 2016-2018.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

DATA MATRIX

Habitat Data

Comments:

SEDIMENT: Sediment assessment parameters were collected at one site in 2017. Site CRRL05-01a represents one reach type (Middle Rockies ecoregion, reach slope less than 2 percent, stream order 1, and unconfined). Fine sediment parameters (riffle fines less than 2mm, riffle fines less than 6mm, and pool tail fines less than 6mm) were evaluated against reference data; both riffle fines parameters are outside reference range (indicating excess fine sediment is being deposited in riffles) and the pool tail fines are within reference range. For coarse sediment and instream habitat parameters, width/depth ratio and pool count are within reference range but residual pool depth is outside reference range (indicating pools are not as deep as desirable and may be filling with excess sediment). Entrenchment ratio is within expected range given stream type. Additional sediment monitoring was attempted but much of the stream channel is influenced by beaver activity.

HABITAT: Habitat indicators for riparian condition and aquatic habitat were rated at three sites in 2016 and 2018. Most indicators for riparian degradation, loss of instream habitat, and alteration in geomorphology rated as optimal or sub-optimal. However, aerial images indicate areas that are experiencing riparian and channel disturbance due to high intensity livestock impacts which were not represented during site visits.

M01CORLC02

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01CORLC02 is located less than a mile above the mouth. The most apparent feature of this site is that it is in a state of recovery from past heavy grazing; soil disturbance is evident in the uplands and hummocking, pugging, and bank scars from previous access points are common. However, the riparian vegetation is recovering very well (nearly all channel margins, banks, and the wide riparian zone is densely vegetated with sedges and rushes) and hummocks are healing. Willows are common but lacking in areas where grazing influence was heaviest. The channel is influenced beaver, riffles are lacking due to beaver activity, and macrophyte beds and filamentous algae are common. The channel resembles a spring creek (crystal clear water, deep, slow-moving and the substrate is dominated by fine sand and silt). The channel appears stable and should continue to improve with continuation of appropriate land management.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

M01CORLC40

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01CORLC40 (sediment site CRRL05-01a) is located near South Valley Road. There are many willows, including saplings and seedlings, although some exhibit evidence of browse (umbrella-shaped and animal trails) and contain a lot of dead branches. Ground cover is primarily grass, clover and some rushes. Banks and channel features are impacted by livestock and hoof shear and pugging is prevalent. Pools are primarily shallow with fine sediment and the riffle substrate is small gravel and fine sediment.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

M01CORLC30

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01CORLC30 is between the mouth and South Valley Road. Surrounding vegetation consists of vigorous willows, sedge and rushes. The reach exhibits beaver influence (indistinct riffle-pool sequence, macrophytes, sand and silt substrate) but appears stable and at potential. The site appears to be recovering following historically heavy grazing which caused hummocking and clubbed willows. With the presence of beavers, the reach appears to be at potential.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Corral Cr, about 2 mi upstream for the mouth; on BLM land

Data Type	Comments	Ref Num	Citation
Land use	2003 MT DEQ sampling data and field assessment forms: Timber harvest and grazing indicated in the SRAF. Harvest occurred 10-15 years ago. Restoration project includes tree plantings, including willows, channel structures (logs), road closure. Good fish habitat in this reach. Roads are probably still a sediment source; trend is improving.	4650	(2003), DEQ Field Assessment Form

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
photo points	2003 MT DEQ sampling data and field assessment forms: Photos #1-4 show a montane, forested setting, a high-gradient Rosgen B channel with good flow, Spruce forest,stable stream banks, good riparian veg/shading. Photo 34 shows clean substrates, forest road nearby (about 50 feet distance). Photo # 3 shows an irrigation diversion, fabric dam.	4650	(2003), DEQ Field Assessment Form

Corral Cr, 1/4 mi. upstream from the mouth

Data Type	Comments	Ref Num	Citation
Land use	2003 MT DEQ sampling data and field assessment forms: Field photos indicate rangeland grazing.	4650	(2003), DEQ Field Assessment Form
photo points	2003 MT DEQ sampling data and field assessment forms: Photo # 21: shows hoof pugging and moderate turbidity, good flow volume. Photo # 22 shows grasses dominant in riparian area, fine sediment deposition at approx. 50 % of the channel width , starting at the channel margains. Sedges grazed heavily, appear reduced. # 23 & 24 , on State Land: Pugged, hummocked and slumped vegetated soil masses are shown. Willows present; no regeneration seen. # 25 , on State Land: Low stubble height, willows severely reduced, no regeneration seen. Left side shows evidence of incisement, flood plain is building below the eroded stream bank, but that vegetation is grazed hard, also. Bare or compacted soils are shown. Good flow volume. Cows are still on the allottment.	4650	(2003), DEQ Field Assessment Form

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

DATA MATRIX

Chemistry Data

Comments:

NUTRIENTS: In the years 2016 to 2018, 12 samples were collected for TN, TP, and NO2+3. TN samples ranged from 0.03 to 0.12 mg/L; TP samples ranged from 0.008 to 0.04 mg/L; NO2+3 samples ranged from <0.01 to 0.06 mg/L.

(Pre 2020 cycle) Metals: Sampling results: No exceedences of ("primary") numerical Human Health or Aquatic Life standards at either 2003 sampling sites. Other Field Parameters of concern: Dissolved Oxygen: meter readings of 13.97 & 13.77 mg/L at 2:15 PM and 4:30 PM might indicate excessive algae growth. Diurnal DO swings could be fairly high, especially is the lower-gradient downstream reaches.

Corral Cr, about 2 mi upstream for the mouth; on BLM land

Data Type	Comments	Ref Num	Citation
common ions, pH, conductivity, miscellaneous	2003 MT DEQ sampling data and field assessment forms: (Sampling date and time : 09/26/03, 1630) pH: 8.07, Specific Conductivity:184 umhos/cm, Water Temperature: 8.3 C, Chloride: <1.0 mg/L and Sulfate: 2.81 mg/L concentrations are well below the EPA guidance values (secondary Human Health) of 250mg/L. SAR value of 0.1 (ratio). All salinity indicator values are low. TDS: 110 mg/L, TSS: 3.50 mg/L. Turbidity: "clear". Discharge estimated as 3.0 cfs. Dissolved Oxygen: 13.97 mg/L	4650	(2003), DEQ Field Assessment Form
metals	2003 MT DEQ sampling data and field assessment forms: Sampling results: No exceedences of numerical Human Health or Aquatic Life standards	4650	(2003), DEQ Field Assessment Form
metals	Abandoned and Inactive Mines Database: No mines are located in this drainage	10090	Govan, Sybil (nnnn), Natural Resources Information System

Corral Cr, 1/4 mi. upstream from the mouth

Data Type	Comments	Ref Num	Citation
common ions, pH, conductivity, miscellaneous	2003 MT DEQ sampling data and field assessment forms: (Sampling date and time : 09/26/03, 1415) pH: 7.81, Specific Conductivity: 239 umhos/cm, Water Temperature: 10.4 C, Chloride: <1.0 mg/L and Sulfate: 2.01 mg/L concentrations are well below the EPA guidance values (secondary Human	4650	(2003), DEQ Field Assessment Form

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	Health) of 250mg/L. SAR value of 0.1 (ratio). All salinity indicator values are low. TDS: 154 mg/L, TSS: 67.9 mg/L. Turbidity notation: "slight". Discharge estimated as 4.0 cfs. Dissolved Oxygen: 13.77 mg/L		
metals	2003 MT DEQ sampling data and field assessment forms: Sampling results: No exceedences of numerical Human Health or Aquatic Life standards. Reported Total Recoverable Iron value of 0.79 mg/L is higher than the secondary Human Health Standard of 0.3 mg/L; may cause staining or affect taste.	4650	(2003), DEQ Field Assessment Form

Entire Segment

Data Type	Comments	Ref Num	Citation
major nutrients	This document contains information supporting the nitrate plus nitrite recommended criteria used to evaluate nitrate plus nitrite data while assessing wadeable streams in the Red Rock River watershed.	11934	Suplee, Michael W. ; Watson, Vicki ; Varghese, Arun ; Cleland, Joshua (2008), Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers
major nutrients	This document contains the numeric total nitrogen (TN) and total phosphorus (TP) standards for wadeable streams in the Middle Rockies level III ecoregion. These standards were used to analyze TN and TP data for beneficial use determination for wadeable streams in the Red Rock River watershed.	15265	Montana Department of Environmental Quality (2014), Department Circular DEQ-12A: Montana Base Numeric Nutrient Standards, Circular DEQ-12A
major nutrients	This document contains the assessment method with which nutrient data were analyzed for beneficial use determination.	15655	Montana Department of Environmental Quality (2016), Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels, WQPBMASTR-01
major nutrients	12 TN, 12 TP, 12 NO2+3 samples from 2016-2018.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division,

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
			Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
quantitative physical data	Entrenchment ratio literature values from the Rosgen stream classification system were used as delineative criteria to evaluate entrenchment ratio in streams in the Red Rock River watershed: < 1.6 for A, F and G streams, 1.2-2.4 for B streams, and > 2.0 for C and E streams.	10529	Rosgen, David L. (1996), Applied River Morphology
quantitative physical data	This document describes the sediment monitoring protocols used by the US Forest Service PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO-EMP).	15301	Archer, Eric K. (2012), PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO-EMP) for Streams and Riparian Areas: 2012 Sampling Protocol for Stream Channel Attributes
quantitative physical data	This document describes the DEQ assessment method for making sediment impairment listing decisions. The method involves a comparison of study site data against reference conditions using the one-sample Wilcoxon Signed Rank Test (alpha = 0.25) and qualitative observations, scientific professional judgment and other factors are considered when making impairment listing determinations; divergence of one parameter from the reference dataset does not necessarily equate to a determination of impairment.	14338	Kusnierz, Paul ; Welch, Andy ; Kron, Darrin (2013), The Montana Department of Environmental Quality Western Montana Sediment Assessment Method: Considerations, Physical and Biological Parameters, and Decision Making-Draft, WQPBMASTR-02
quantitative physical data	Data collected by the USFS PACFISH/INFISH Biological Opinion Effectiveness (PIBO) monitoring program in the Middle Rockies Level III Ecoregion from 2001 to 2018 was used to evaluate sediment conditions at sites in the Red Rock River watershed. Parameters for which PIBO data were applied are percent fine sediment < 6mm in pool tails via grid toss, width/depth ratio, residual pool depth, and pool frequency (pools/1000 feet). PIBO reference sites have catchment road densities less than 0.5 km/km2, riparian road densities less	10468	U.S. Department of Agriculture, Forest Service (20nn), Forest Service PIBO Data Through 2001-2014 [Electronic Resource]

Montana DEQ - Water Quality Standards Attainment Record

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	than 0.25 km/km2, no grazing within 30 years, and no known in-channel mining upstream of the site.		
quantitative physical data	CRRL05-01a (2017): Stream type: E4; Slope = 1; Bankfull width (ft) = 6.1; Riffle fines less than 2mm = 34; Riffle fines less than 6mm = 41; Pool tail fines less than 6mm = 3; Width/depth ratio = 6.4; Residual pool depth = 0.57; Pools/1000ft = 25.7; Entrenchment ratio = 14.77. Data collected by DEQ at reference stream sites from 2010 to 2013 were used to evaluate sediment conditions in the Red Rock River watershed. Parameters for which DEQ reference site data were applied are percent fine sediment < 6mm and < 2 mm in riffles via pebble count, percent fine sediment < 6mm in pool tails via grid toss, width/depth ratio, residual pool depth and pool frequency (pools/1000 feet).	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

DQA SUMMARY

Aquatic Life & Fishes

Nutrients	PASS
Metals	NOT ASSESSED
Sediment	PASS
Temperature	NOT ASSESSED
Other	NOT ASSESSED

Drinking Water

Metals	NOT ASSESSED
Other	NOT ASSESSED

Recreation

Nutrients	PASS
E.coli	NOT ASSESSED
Other	NOT ASSESSED

Agriculture

Common	NOT ASSESSED
Other	NOT ASSESSED

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

ASSESSMENT HISTORY

Cycle 2006

1996 Listing: Corral Creek was included on the 1996 303(d) List of impaired waterbodies. Aquatic Life and Cold Water Fishery uses were judged to be Partially-Supported for three stream miles. The Probable Causes were indicated as "Other Habitat Alteration" and "Siltation". Probable Sources of impairments were listed as Agriculture, Range Land (grazing), Streambank Modification, and Silviculture. 2006 303 (d) Listing: The 2006 Listing is a result of a MT DEQ reassessment effort using MT DEQ protocols, and all readily available data. The data assemblage used in the April, 2005 update provided the first use of sufficient credible data for determining beneficial use support/impairment for all uses prescribed for Corral Creek as a State waterbody under B-1 classification.
The 2006 303(d) Listing for this waterbody concludes Partial-Support for Aquatic Life and Cold Water Fisheries uses. Contact Recreation uses, Drinking Water, Agriculture, and Industrial uses are Fully-Supported. 2006 303(d) List Causes of Impairment to Aquatic Life and Fisheries uses are Sedimentation, Habitat Alteration, and Total Phosphorus. Total Phosphorus is an added pollutant, which did not appear on the previous (1996)impairment listing. It is probably associated with the high fine sediment loads to the channel.
Sources of Impairment are Grazing-Related Sources and Roads. Metals: No exceedences of Human Health (Drinking Water) or Aquatic Life Standards were reported. Industrial uses are fully-supported, though periods of moderate turbidity occur. Salinity levels are low. Livestock water use (that is, Agriculture use) is judged to be fully-supported for the reasons that the reported metals concentrations in the water column were below the recommended MCLs for livestock and poultry water, and salinity values were low. The water is excellent for stock water and for irrigation of all crops. Contact Recreation: Moderate flow alteration probably occurs but not to the extent that recreation uses are substantially diminished. (Long-lived macroinvertebrate taxa are present). Groundwater inputs augment in-channel flow. Although some filamentous algae are present at both sites, the resultant algae growth does not affect aesthetics of the stream for recreational uses.

Cycle 2008

Not assessed this cycle

Cycle 2010

Not assessed this cycle

Cycle 2012

Not assessed this cycle

Cycle 2014

Not assessed this cycle

Cycle 2016

Not assessed this cycle

Montana DEQ - Water Quality Standards Attainment Record

Cycle 2018

Not assessed this cycle

Cycle 2020

This assessment unit (AU) was assessed for nutrients, sediment, and habitat.

Total Phosphorus remains listed as a cause of impairment affecting Aquatic Life and Fish and Primary Contact Recreation.

Sedimentation/siltation and Alteration in stream-side or littoral vegetative covers remains as causes of impairment affecting Aquatic Life and Fish.

There is insufficient information to assess Escherichia coli.

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Overall Condition of Segment

Corral Creek is in the southeastern portion of the Red Rock watershed and flows northwest approximately four miles to its confluence with Red Rock Creek.

Assessment for the 2020 IR indicates that Corral Creek aquatic life use is limited by sediment. Sediment was monitored at only one site as much of the channel is influenced by beaver and therefore methods for evaluating sediment were not applicable everywhere. At the site monitored, two of three fine sediment parameters and one of four coarse sediment and morphology parameters differ significantly from reference condition. Some reaches where banks and channel features continue to be impacted by livestock (hoof shear and pugging). However, some reaches are in a state of recovery following historically heavy grazing; past soil disturbance is evident in the uplands and hummocking, pugging, and bank scars from previous access points are common. The channel appears stable and is reforming and revegetating and these improvements should continue with continuation of appropriate land management. The channel is influenced beaver and resembles a spring creek (crystal clear water, deep, slow-moving and the substrate is dominated by fine sand and silt).

Assessment indicates that Corral Creek is impaired for habitat, although riparian vegetation is recovering very well in places (nearly all channel margins, banks, and the wide riparian zone is densely vegetated with sedges and rushes) and hummocks are healing in areas of past impacts of heavy grazing. Willows are common but lacking in areas where grazing influence was heaviest.

Assessment indicates Corral Creek aquatic life and recreational uses are limited by nutrients, specifically total phosphorus. Water chemistry data were evaluated against numeric nutrient standards for TN and TP and to recommended NO2+3 nutrient criteria for the Middle Rockies ecoregion (TN = 0.30 mg/L; TP = 0.03 mg/L; NO2+3 = 0.10 mg/L). Of all the nutrient samples collected (n = 12 for TN, TP, and NO2+3), zero samples exceeded the TN standard, one sample exceeded the TP standard, and zero samples exceeded the NO2+3 recommended criteria. Benthic algae samples were evaluated against the recommended thresholds for benthic algal chlorophyll-a (120 mg/m2) and ash-free dry weight (35 g/m2). Of all the benthic algae samples analyzed (n = 4), zero results exceeded chlorophyll-a and ash-free dry weight thresholds.

A Montana-specific Hilsenhoff Biotic Index (HBI) score was calculated for each macroinvertebrate sample and evaluated against a recommended HBI threshold (HBI < 4) for the mountainous and transitional region of Montana. HBI scores above the threshold imply conditions that are limiting the natural macroinvertebrate community. Macroinvertebrate metric values are applied throughout the assessment process when evaluating whether or not a segment is supporting its aquatic life beneficial use support. Index scores may be affected by a variety of pollutant or pollution causes; therefore, failure to meet the biological threshold does not necessarily indicate that a particular cause of impairment is impeding aquatic life beneficial use support. Additionally, an exceedance of biological metrics may not always coincide with human caused biological limitations. Of all the macroinvertebrate samples collected (n = 3), 3 samples exceeded the HBI threshold.

Escherichia coli bacteria (E. coli) data were collected to update the 2020 IR. All E. coli bacteria samples were collected when the primary contact water quality standard for B-1 standards apply (April 1 - October 31). There is insufficient information to calculate a geometric mean with the 30 day or seasonal method; 1 out 6 samples exceeded the 252 cfu/100ml result (17%). E. coli will not be listed as a cause of impairment affecting the primary contact recreation beneficial use.

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

Probable sources of pollution identified in this watershed include: Grazing in Riparian or Shoreline Zones, Impacts from Hydrostructure Flow Regulation/modification, Natural Sources, and Unspecified Unpaved Road or Trail.

Pre 2020 cycle
Upper reaches: This is a montane, spruce forest setting. The stream is a high-gradient Rosgen B channel with good, cool flow. Stable stream banks, good riparian veg/shading, clean substrates characterize this reach. A forest road is near the channel (about 50 feet distance). Old logging harvest areas (about 15 years ago) are reestablished with vegetation . An irrigation diversion with fabric dam is located here. Restoration efforts here have included tree plantings (willows), channel structures (logs), road closure. Good fish habitat in this reach. Roads are probably still a sediment source; trend is improving.

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

USE SUPPORT DECISION

Use Class B-1

Trophic Status: **Trophic Trend:**

Uses	DQA	Method, Data, and Information Used	Assessment Type and Confidence	Use Support	Partial Flag	Use Support Certainty	Threatened
Aquatic Life	Pass			Not Fully Supporting	No	Medium	No
Agricultural				Not Assessed	No	High	No
Drinking Water				Not Assessed	No	Medium	No
Primary Contact Recreation	Pass			Not Fully Supporting	No	Medium	No

Method Number and Description

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

IMPAIRMENT INFORMATION

Uses	Cause (Confidence): Source(Confirmed)	Observed Effects
Aquatic Life	84 (Medium): 46 (Y) 371 (Medium): 46 (Y), 58 (N), 155 (N), 170 (Y) 462 (Medium): 46 (Y), 155 (N), 170 (Y)	
Agricultural		
Drinking Water		
Primary Contact Recreation	462 (): 46 (Y), 155 (N), 170 (Y)	

Cause Number and Description	Source Number and Description	Observed Effect Number and Description
84-Alteration in stream-side or littoral vegetative covers 371-Sedimentation/Siltation 462-Phosphorus, Total	46-Grazing in Riparian or Shoreline Zones 58-Impacts from Hydrostructure Flow Regulation/modification 155-Natural Sources 170-Unspecified Unpaved Road or Trail	

DELISTING / STATUS CHANGES

Cause	Reason for Change	Date of Change

Reporting Cycle: 2020 **Assessment Record:** MT41A004_040.pdf **Status:** Completed

CATEGORY INFORMATION

Previous Cycle	
Cycle	2018
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Current Cycle	
Cycle	2020
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

ASSESSMENT UNIT INFORMATION

Reporting Cycle: 2020

Assessment Unit: MT41A004_080

Waterbody Name: O'Dell Creek

Location Description: O'DELL CREEK, headwaters to mouth (Lower Red Rock Lake)

Water Type:	Size (Miles/Acres)	Use Class:
RIVER	16.09 MILES	B-1

Hydrologic Unit Code: 10020001

HUC Name: Red Rock

Watershed: Missouri Headwaters

Basin: Upper Missouri

TMDL Planning Area: Red Rock

Ecoregion: Middle Rockies

County: Beaverhead County

Lat/Long AU Start (U/S): 44.547484 / -111.852221

Lat/Long AU End (D/S): 44.639282 / -111.815172

MONITORING INFORMATION

Date Assessment Started: 11/18/2019

Assessed By: Makarowski, Kathryn

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

CITATIONS

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Decker-Hess, Janet ; Nelson, Frederick Allen ; Peterman, Larry G. (1981), Instream Flow Evaluation for Selected Waterways of the Beaverhead, Big Hole and Red Rock River Drainages of Southwest Montana	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	quantitative physical data
Montana Department of Fish, Wildlife, and Parks (1989), Application for Reservations of Water in the Missouri River Basin above Fort Peck Dam. Volume 2: Reservation Requests for Waters Above Canyon Ferry Dam	WQPB Ebrary	fish; macroinvertebrates; wildlife	Land use; riparian &/or instream surveys & physical features	benthic sediment data; quantitative physical data
Kaya, Calvin M. (1992), Restoration of Fluvial Arctic Grayling to Montana Streams: Assessment of Reintroduction Potential of Streams in the Native Range, the Upper Missouri River Drainage above Great Falls (Masters Thesis)	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	quantitative physical data
Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)	WQPB Ebrary	fish	riparian &/or instream surveys & physical features	common ions, pH, conductivity, miscellaneous; quantitative physical data
Rosgen, David L. (1996), Applied River Morphology	Box		riparian &/or instream surveys & physical features	Rosgen type; quantitative physical data
Shields, Ronald R. ; White, Melvin K. ; Ladd, Patricia B. ; Chambers, Clarence L. ; Dodge, Kent A. (1998), Water Resources Data: Montana Water Year 1997, USGS Water-Data Report MT-97-1	WQPB Ebrary	fish		benthic sediment data; common ions, pH, conductivity, miscellaneous; major nutrients; metals; quantitative physical data

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Montana Department of Fish, Wildlife, and Parks (2006), Montana Rivers Information System (MRIS): Montana Fisheries Information System (MFISH) - http://maps2.nris.mt.gov/scripts/esrimap.dll?name=MFISH&Cmd=INST	Assessment Record	fish; wildlife	Land use; riparian &/or instream surveys & physical features	benthic sediment data; common ions, pH, conductivity, miscellaneous; quantitative physical data
Montana State Library Natural Resouce Information System ; Montana State University (2006), Montana View at http://montanaview.org/	DEQ PPA Data Archive	chlorophyll; fecal coliforms; macroinvertebrates; other bacteriological data	photo points; riparian &/or instream surveys & physical features	benthic sediment data; bioaccumulation; common ions, pH, conductivity, miscellaneous; imagery data; major nutrients; metals; organics; quantitative physical data
Suplee, Michael W. ; Watson, Vicki ; Varghese, Arun ; Cleland, Joshua (2008), Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers	WQPB Ebrary	chlorophyll		major nutrients
U.S. Department of Commerce, National Oceanic and Atmospheric Admin. (2008), Screening Quick Reference Tables (SQuiRTs), OR&R Report 08-1	WQPB Ebrary			benthic sediment data; organics
(200n), Montana Interagency Stream Fishery Data for the Upper Missouri River Basin	DEQ PPA Data Archive	fish	Land use; photo points; riparian &/or instream surveys & physical features	quantitative physical data
Archer, Eric K. (2012), PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO-EMP) for Streams and Riparian Areas: 2012 Sampling Protocol for Stream Channel Attributes	WQPB Ebrary		riparian &/or instream surveys & physical features	quantitative physical data

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Citation	Location	Biological Data	Habitat Data	Chemistry Data
Drygas, Jonathan (2012), The Montana Department of Environmental Quality Metals Assessment Method-Final, WQPBMASTR-03	WQPB Ebrary			benthic sediment data; metals
Montana Department of Environmental Quality (2012), Montana Numeric Water Quality Standards: Circular DEQ-7, Circular DEQ-7	WQPB Ebrary			common ions, pH, conductivity, miscellaneous; major nutrients; metals; organics
Kusnierz, Paul ; Welch, Andy ; Kron, Darrin (2013), The Montana Department of Environmental Quality Western Montana Sediment Assessment Method: Considerations, Physical and Biological Parameters, and Decision Making-Draft, WQPBMASTR-02	WQPB Ebrary		riparian &/or instream surveys & physical features	quantitative physical data
Montana Department of Environmental Quality (2014), Department Circular DEQ-12A: Montana Base Numeric Nutrient Standards, Circular DEQ-12A	WQPB Ebrary			major nutrients
Montana Department of Environmental Quality (2016), Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels, WQPBMASTR-01	WQPB Ebrary	General; algae; chlorophyll; macroinvertebrates		General; common ions, pH, conductivity, miscellaneous; major nutrients
(2019), Escherichia coli (E. coli) Assessment Method for State Surface Waters		e-coli		
U.S. Department of Agriculture, Forest Service (20nn), Forest Service PIBO Data Through 2001-2014 [Electronic Resource]	DEQ Metcalf Multimedia Case	macroinvertebrates	riparian &/or instream surveys & physical features	quantitative physical data
Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS	WEB	General; algae; chlorophyll; e-coli; macroinvertebrates	riparian &/or instream surveys & physical features	General; General; benthic sediment data; common ions, pH, conductivity, miscellaneous; major

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Citation	Location	Biological Data	Habitat Data	Chemistry Data
				nutrients; metals; organics; quantitative physical data

Comments:

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

DATA MATRIX
Biological Data

Comments: ALGAE: In 2018, 2 composite samples were collected for benthic chlorophyll-a and ash-free dry weight. The chl-a results are 2.5, 10.5 mg/m2. The AFDW results are 7.62, 4.01 g/m2.

MACRO: In 2017-2018, 2 samples were collected. The HBI scores are 2.10 and 3.22. A Montana-specific Hilsenhoff Biotic Index (HBI) score was calculated for each macroinvertebrate sample and evaluated against a recommended HBI threshold (HBI < 4) for the mountainous and transitional region of Montana. HBI scores above the threshold imply conditions that are limiting the natural macroinvertebrate community. Macroinvertebrate metric values are applied throughout the assessment process when evaluating whether or not a segment is supporting its aquatic life beneficial use support. Index scores may be affected by a variety of pollutant or pollution causes; therefore, failure to meet the biological threshold does not necessarily indicate that a particular pollutant or pollution cause of impairment is impeding aquatic life beneficial use support. Additionally, an exceedance of biological metrics may not always coincide with human caused biological limitations. Of all the macroinvertebrate samples collected (n = 2), 0 samples exceeded the HBI threshold.

PATHOGENS: In 2017, 8 samples were collected for E. coli. The results ranged from 6.3 to 104.6 cfu/100mL. The 30 day (7/19/17 to 8/17/17) geometric mean is 26.96 cfu/100ml.

From Headwaters to Mouth (Lower Red Rock Lake)

Data Type	Comments	Ref Num	Citation
algae	2 Chlorophyll-a and 2 Ash-Free Dry Weight composite results from 2018 from EQuIS.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
e-coli	This document contains the assessment method with which E. coli data were analyzed for beneficial use determination.	15759	(2019), Escherichia coli (E. coli) Assessment Method for State Surface Waters
e-coli	2 E. coli samples (at one site) were collected from July 2017, and 6 E. coli samples (4 at one site, 2 at one site) were collected from August 2017.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
fish	brook, arctic grayling, cutthroat, rainbow x cutthroat hybrids present; 3971 ft electrofished in 1975- 40 brook, 2 arctic grayling and 2 cutthroat captured; 10,560 ft electrofished in 1976- 67 arctic grayling, 39 brook and 2 rainbow x cutthroat hybrids captured	1453	Decker-Hess, Janet ; Nelson, Frederick Allen ; Peterman, Larry G. (1981), Instream Flow Evaluation for Selected Waterways of the Beaverhead, Big Hole and Red Rock River Drainages of Southwest Montana

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
fish	brook, cutthroat present	3444	Kaya, Calvin M. (1992), Restoration of Fluvial Arctic Grayling to Montana Streams: Assessment of Reintroduction Potential of Streams in the Native Range, the Upper Missouri River Drainage above Great Falls (Masters Thesis)
fish	a box trap was installed in 1994; 11 total arctic grayling were captured (4 males and 7 females), as well as 14 cutthroat and 10 brook trout; all fish appeared in excellent condition; grayling numbers appear to be decreasing during the past twenty years (67 grayling in 1976 versus 11 grayling in 1994), but more data is needed; cutthroat population has increased (2 cutthroat in 1975 versus 14 cutthroat in 1994)	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)
fish	arctic grayling, brook, longnose sucker, westslope cutthroat, white sucker, yellowstone x westslope cutthroat present; 1975-1976 electrofishing results see above; electrofished in 1994- 40 westslope cutthroat per 1000 ft; genetics data collected from 10 fish in 1994 found 95% westslope cutthroat and 5% yellowstone cutthroat; trout numbers remaining stable from 1975 to 1994; no grayling numbers available for 1994	11355	Montana Department of Fish, Wildlife, and Parks (2006), Montana Rivers Information System (MRIS): Montana Fisheries Information System (MFISH) - http://maps2.nris.mt.gov/scripts/esrimap.dll?name=MFISH&Cmd=INST
fish	Montana Interagency Stream Fishery Data: arctic grayling, cutthroat, longnose sucker, mountain whitefish, brook, white sucker, rainbow x cutthroat present	4655	(200n), Montana Interagency Stream Fishery Data for the Upper Missouri River Basin
macroinvertebrates	2 Macroinvertebrate samples were collected from 2017-2018.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

DATA MATRIX

Habitat Data

Comments:

SEDIMENT: Sediment assessment parameters were collected at one site in 2017. The site represents a single reach type (Middle Rockies ecoregion, reach slope less than 2 percent, stream order 3, and unconfined). Fine sediment parameters (riffle fines less than 2mm, riffle fines less than 6mm, and pool tail fines less than 6mm) were evaluated against reference data and all three parameters were within reference range. Coarse sediment and instream habitat parameters were also evaluated against reference data and residual pool depth and pool count were within reference range, but width/depth ratio is outside reference range, suggesting the channel is becoming wider and shallower. Entrenchment ratio is within expected range given stream type. A site downstream was investigated but substantial beaver activity precluded it from monitoring and inclusion during assessment. An additional site near the mouth was also visited but the pool/glide system with indistinct riffle features made it undesirable for monitoring and inclusion during assessment.

HABITAT: Habitat indicators for riparian condition and aquatic habitat were rated at two sites in 2016 and 2018. Most indicators for riparian degradation, loss of instream habitat, and alteration in geomorphology rated as optimal or sub-optimal; several marginal and poor. Aerial images indicate areas of high intensity grazing impacts on riparian and channel which were not represented during site visits.

BEHI: ODLL09-02b: BEHI Index = 27; BEHI rating = moderate; near bank stress rating = high; sources of streambank instability = riparian grazing (100%); bank composition = 60% sand/clay, 20% fine gravel, 20% coarse.

GREENLINE: ODLL09-02b: 53.2% vegetated ground cover (18.2 wetland, 33.5 grass/forbs, 1.4 base shrub/tree); 4.3% nonvegetated ground cover (4.3 rock); 35% deciduous understory; 0% overstory; 7.5% invasive plant observations.

M01ODELC20

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01ODELC20 is located between the mouth and the South Valley Road crossing. Riparian vegetation is robust and beaver activity is prevalent.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

M01ODELC01

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01ODELC01 (sediment site ODLL09-02a) is located near the South Valley Road Crossing. The riparian area is vigorous and dominated by willows, sedge, horsetail and	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	mixed forbs. Canadian thistle is present. The channel is stable with a defined riffle-pool sequence and no evidence of downcutting or excess bank erosion. The channel splits near the road into two large channels but returns to a single channel. Silt accumulates in some places but mostly along channel margins. In places, there is apparent aggradation, occasional blow-outs, bank erosion, and mid-channel bars, suggesting the channel is not always effectively transporting and sorting sediment. Also, historic incisement appears to have left some very tall cut banks that are devoid of vegetation. Although human disturbances are not apparent at this site, it is suspected that natural factors (beaver activity) and upstream logging activities (active during site visit) and water diversions may be impacting the flow regime and sediment transport capacity. No evidence of heavy grazing in this vicinity.		Water Quality EQuIS

From Headwaters to Mouth (Lower Red Rock Lake)

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	originates in Centennial Mountains; 27.7 square mi drainage; 78% owned by BLM, 9% by USFWS, 9% by private individuals and 4% by the State of Montana; upper portion travels through floodplain vegetated with aspen, conifers, grasses and forbs; lower portion is vegetated with willow, grasses and forbs, and beaver dams are abundant; ave.gradient = 16ft/1000ft; ave.width = 23 ft; land use includes agricultural research, wildlife habitat, irrigated hay production, cattle grazing and recreation; grazing has resulted in trampled banks, loss of streamside vegetation and increased rate of erosion; sedimentation is severe in portions of the lower creek; majority of soils within the drainage are classified as unstable and sedimentation will continue to be a threat to aquatic life;	1453	Decker-Hess, Janet ; Nelson, Frederick Allen ; Peterman, Larry G. (1981), Instream Flow Evaluation for Selected Waterways of the Beaverhead, Big Hole and Red Rock River Drainages of Southwest Montana

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	irrigation system above Red Rock Refuge completely dewaters a stretch of the stream during low water years, but this data has not been published; minimum instream flow of 16 cfs requested		
riparian &/or instream surveys & physical features	ave. gradient is 1.6%; except for beaver dams, stream is unimpeded to the lake; lower portion has low gradient (<1%) and has numerous deep pools; silt makes up much of the substrate in this area; mean annual flow of 18 cfs	3444	Kaya, Calvin M. (1992), Restoration of Fluvial Arctic Grayling to Montana Streams: Assessment of Reintroduction Potential of Streams in the Native Range, the Upper Missouri River Drainage above Great Falls (Masters Thesis)
riparian &/or instream surveys & physical features	ave.gradient = 6.9% in upper portion, ave.gradient = 1.0% in lower portion; total streambank reach score was 94.9 (streambank condition is fair)-1.8% undercut banks, 9% overhanging vegetation, 64% bank erosion, 44.1% bottom covered with fines, stream width (mean) was 5.58 m and depth (mean) was .25 m; bottom materials-1.3% small boulders, 6.9% large rubble, 13.1% small rubble, 8.1% coarse gravel, 8.8% fine gravel and 61.8% sand, silt, clay and muck; stream drainage was found to be highly erosive and contributes large amounts of sediment to stream; cattle grazing, sheep grazing, timber harvest, phosphate mining and an undersized road crossing severely impact the stream; 1988 habitat surveys indicated fair to poor aquatic and riparian habitat due to bank erosion, filling of pools and covering of spawning gravels; condition of stream is improving through fencing riparian areas away from cattle and removing an undersized road culvert, allowing increased flows to flush out the majority of fine sediment	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)
riparian &/or instream surveys & physical features	land use-irrigated cropland, managed pasture, native pasture, general recreational use; water use-irrigation, stock watering; ave. channel width of 23 ft; riparain vegetation consists of	11355	Montana Department of Fish, Wildlife, and Parks (2006), Montana Rivers Information System (MRIS): Montana Fisheries Information System

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Data Type	Comments	Ref Num	Citation
	grass/herbaceous forms and bank vegetation consists of deciduous shrubs; vegetation species include silver sagebrush and an undesignated willow species; data collected at low flow period in 1985 found flow to be 7.5 cfs		(MFISH) - http://maps2.nris.mt.gov/scripts/esrimap.dll?name=MFISH&Cmd=INST
riparian &/or instream surveys & physical features	Montana Interagency Stream Fishery Data: habitat rating for arctic grayling = best; habitat trend is static; factors limiting fishery include lack of spawning areas, high erosive damage, turbidity, excess siltation, agriculture and overuse by stock	4655	(200n), Montana Interagency Stream Fishery Data for the Upper Missouri River Basin

M01ODELC05

Data Type	Comments	Ref Num	Citation
riparian &/or instream surveys & physical features	Site M01ODELC05 is near the mouth. The site is heavily grazed with riparian vegetation impacts including limited regeneration of willows, grass encroachment on sedges and rushes, heavy browse and clubbed willows. Bank stability varies depending on where cattle are concentrated. Pools and glides are dominant features. Substrate is comprised of silt to small gravel. Beaver activity upstream may be affecting flow velocities during runoff, limiting sediment transport. Aerial images indicate a stark fenceline contrast near the mouth where cattle have congregated and severely widened and trampled the channel.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

DATA MATRIX

Chemistry Data

Comments:

NUTRIENTS: In the years 2016 to 2018, 13 samples were collected for TN, TP, and NO2+3. TN samples ranged from 0.03 to 0.07 mg/L; TP samples ranged from 0.017 to 0.036 mg/L; NO2+3 samples ranged from <0.01 to 0.02 mg/L.

METALS: In the years 2015-2016, 10 dissolved Al samples, 10 Total Recoverable As, Cd, Cr, Cu, Fe, Pb, Se, Ag, and Zn, samples were collected using DEQ Sampling protocols. For all metals parameters; 4 samples were collected during high flows for, remaining samples were collected during baseflow conditions. Water column data for each metal was evaluated against numeric water quality standards (acute, chronic, and human health) according to DEQ assessment method for metals. For metals parameters Al, As, Cd, Cr, Cu, Fe, Pb, Se, Ag, Zn: zero samples exceed the aquatic life; acute or chronic standard, and human health standard.

SEDIMENT METALS: Two samples were collected and analyzed for As, Cd, Cr, Cu, Fe, Pb, Hg and Zn. Benthic sediment metals data was considered separately from water column metals data and is not explicitly included in the assessment decision-making process. Data was evaluated using NOAAs Screening Quick Reference Tables for Inorganics in Soil (Probable Effect Level). All concentrations are below the screening criteria used (17ug/g As, 3.53 ug/g Cd, 90 ug/g Cr, 197 ug/g Cu, 91.3 ug/g Pb, 0.486 ug/g Hg and 315 ug/g Zn).

From Headwaters to Mouth (Lower Red Rock Lake)

Data Type	Comments	Ref Num	Citation
benthic sediment data	This document contains NOAA's Screening Quick Reference Tables for Inorganics in Soil with which benthic sediment metals concentrations were evaluated in the Red Rock River watershed.	14365	U.S. Department of Commerce, National Oceanic and Atmospheric Admin. (2008), Screening Quick Reference Tables (SQuiRTs), OR&R Report 08-1
benthic sediment data	2 benthic sediment As, Cd, Cr, Cu, Fe, Pb, Hg, Zn samples in 2017.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
major nutrients	This document contains information supporting the nitrate plus nitrite recommended criteria used to evaluate nitrate plus nitrite data while assessing wadeable streams in the Red Rock River watershed.	11934	Suplee, Michael W. ; Watson, Vicki ; Varghese, Arun ; Cleland, Joshua (2008), Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers
major nutrients	This document contains the numeric total nitrogen (TN) and total phosphorus (TP) standards for wadeable streams in the	15265	Montana Department of Environmental Quality (2014), Department Circular DEQ-12A:

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Data Type	Comments	Ref Num	Citation
	Middle Rockies level III ecoregion. These standards were used to analyze TN and TP data for beneficial use determination for wadeable streams in the Red Rock River watershed.		Montana Base Numeric Nutrient Standards, Circular DEQ-12A
major nutrients	This document contains the assessment method with which nutrient data were analyzed for beneficial use determination.	15655	Montana Department of Environmental Quality (2016), Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels, WQPBMASTR-01
major nutrients	13 TN, 13 TP, 13 NO2+3 samples from 2016-2018 from EQuIS.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
metals	This document contains the assessment method with which metals data was analyzed for beneficial use determination.	14350	Drygas, Jonathan (2012), The Montana Department of Environmental Quality Metals Assessment Method-Final, WQPBMASTR-03
metals	This document contains the numeric water quality standards used to evaluate metals concentrations for beneficial use support in waters in the Red Rock watershed.	13619	Montana Department of Environmental Quality (2012), Montana Numeric Water Quality Standards: Circular DEQ-7, Circular DEQ-7
metals	10 Dissolved Al, As, Cd, Cu, Fe, Pb, Se, Ag, Zn samples from 2015-2016	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS
quantitative physical data	Q = 34 cfs (estimate)	1453	Decker-Hess, Janet ; Nelson, Frederick Allen ; Peterman, Larry G. (1981), Instream Flow Evaluation for Selected Waterways of the Beaverhead, Big Hole and Red Rock River Drainages of Southwest Montana

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Data Type	Comments	Ref Num	Citation
quantitative physical data	mean annual Q = 18 cfs	3444	Kaya, Calvin M. (1992), Restoration of Fluvial Arctic Grayling to Montana Streams: Assessment of Reintroduction Potential of Streams in the Native Range, the Upper Missouri River Drainage above Great Falls (Masters Thesis)
quantitative physical data	T = 3.3 to 15.5 C	2336	Mogen, James Tory (1996), Status and Biology of the Spawning Population of Red Rock Lakes Arctic Grayling (Master's Thesis)
quantitative physical data	Entrenchment ratio literature values from the Rosgen stream classification system were used as delineative criteria to evaluate entrenchment ratio in streams in the Red Rock River watershed: < 1.6 for A, F and G streams, 1.2-2.4 for B streams, and > 2.0 for C and E streams.	10529	Rosgen, David L. (1996), Applied River Morphology
quantitative physical data	mean daily Q for 1997 is 8.0 to 342 cfs; mean daily Q for 1993-1997 seasons is 5.5 to 350 cfs	2761	Shields, Ronald R. ; White, Melvin K. ; Ladd, Patricia B. ; Chambers, Clarence L. ; Dodge, Kent A. (1998), Water Resources Data: Montana Water Year 1997, USGS Water-Data Report MT-97-1
quantitative physical data	Q = 7.5 cfs during low flow stage in 1985	11355	Montana Department of Fish, Wildlife, and Parks (2006), Montana Rivers Information System (MRIS): Montana Fisheries Information System (MFISH) - http://maps2.nris.mt.gov/scripts/esrimap.dll?name=MFISH&Cmd=INST
quantitative physical data	Q = 7.02 to 102.7 cfs; T = 0 to 12 C	2471	Montana State Library Natural Resouce Information System ; Montana State University (2006), Montana View at http://montanaview.org/
quantitative physical	This document describes the sediment monitoring protocols	15301	Archer, Eric K. (2012), PACFISH/INFISH

Montana DEQ - Water Quality Standards Attainment Record

Data Type	Comments	Ref Num	Citation
data	used by the US Forest Service PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (PIBO-EMP).		Biological Opinion Effectiveness Monitoring Program (PIBO-EMP) for Streams and Riparian Areas: 2012 Sampling Protocol for Stream Channel Attributes
quantitative physical data	This document describes the DEQ assessment method for making sediment impairment listing decisions. The method involves a comparison of study site data against reference conditions using the one-sample Wilcoxon Signed Rank Test (alpha = 0.25) and qualitative observations, scientific professional judgment and other factors are considered when making impairment listing determinations; divergence of one parameter from the reference dataset does not necessarily equate to a determination of impairment.	14338	Kusnierz, Paul ; Welch, Andy ; Kron, Darrin (2013), The Montana Department of Environmental Quality Western Montana Sediment Assessment Method: Considerations, Physical and Biological Parameters, and Decision Making-Draft, WQPBMASTR-02
quantitative physical data	Data collected by the USFS PACFISH/INFISH Biological Opinion Effectiveness (PIBO) monitoring program in the Middle Rockies Level III Ecoregion from 2001 to 2018 was used to evaluate sediment conditions at sites in the Red Rock River watershed. Parameters for which PIBO data were applied are percent fine sediment < 6mm in pool tails via grid toss, width/depth ratio, residual pool depth, and pool frequency (pools/1000 feet). PIBO reference sites have catchment road densities less than 0.5 km/km2, riparian road densities less than 0.25 km/km2, no grazing within 30 years, and no known in-channel mining upstream of the site.	10468	U.S. Department of Agriculture, Forest Service (20nn), Forest Service PIBO Data Through 2001-2014 [Electronic Resource]
quantitative physical data	ODLL09-02a (2018): Stream type: C4; Slope = 1.05; Bankfull width (ft) = 22.3; Riffle fines less than 2mm = 9; Riffle fines less than 6mm = 12; Pool tail fines less than 6mm = 12; Width/depth ratio = 18.5; Residual pool depth = 2.19; Pools/1000ft = 15.4; Entrenchment ratio = 3.89.	15288	Montana Department of Environmental Quality, Planning, Prevention and Assistance Division, Water Quality Planning Bureau (nnnn), Montana Water Quality EQuIS

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Data Type	Comments	Ref Num	Citation
	Data collected by DEQ at reference stream sites from 2010 to 2013 were used to evaluate sediment conditions in the Red Rock River watershed. Parameters for which DEQ reference site data were applied are percent fine sediment < 6mm and < 2 mm in riffles via pebble count, percent fine sediment < 6mm in pool tails via grid toss, width/depth ratio, residual pool depth and pool frequency (pools/1000 feet).		

Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

DQA SUMMARY

Aquatic Life & Fishes

Nutrients	PASS
Metals	PASS
Sediment	PASS
Temperature	NOT ASSESSED
Other	NOT ASSESSED

Drinking Water

Metals	PASS
Other	NOT ASSESSED

Recreation

Nutrients	PASS
E.coli	PASS
Other	NOT ASSESSED

Agriculture

Common	NOT ASSESSED
Other	NOT ASSESSED

Montana DEQ - Water Quality Standards Attainment Record

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ASSESSMENT HISTORY

Cycle 2006

Not assessed this cycle

Cycle 2008

Not assessed this cycle

Cycle 2010

Not assessed this cycle

Cycle 2012

Not assessed this cycle

Cycle 2014

Not assessed this cycle

Cycle 2016

Not assessed this cycle

Cycle 2018

Removed Alteration in stream-side or littoral vegetative covers as a cause from Primary Contact Recreation. Removed Turbidity as a cause from Primary Contact Recreation. Changed the use Primary Contact Recreation from Not Supporting to Not Assessed as it no longer had any causes.

Cycle 2020

This assessment unit (AU) was assessed for nutrients, metals, pathogens, sediment, and habitat.

There are no nutrient, pathogen, or metals listings for this cycle.

Sedimentation/siltation is added as a cause of impaired affecting Aquatic Life and Fish. Turbidity is delisted since the sediment-related impairment is more accurately captured via the sedimentation/siltation cause.

Alteration in stream-side or littoral vegetative covers remains as a cause of impairment affecting Aquatic Life and Fish.

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Reporting Cycle: 2020 **Assessment Record:** MT41A004_080.pdf **Status:** Completed

Overall Condition of Segment

O'Dell Creek is in the southeastern portion of the Red Rock watershed and flows north approximately 16 miles to its confluence with Lower Red Rock Lake.

Assessment for the 2020 IR indicates O'Dell Creek remains impaired for sediment. Only one site was able to be monitored fully for sediment parameters and, at this site, all three fine sediment and three of four coarse sediment and morphology parameters did not differ significantly from reference conditions. The channel is stable with a defined riffle-pool sequence and no evidence of downcutting or excess bank erosion. Below this site, the channel is heavily influenced by beaver activity but is also relatively intact. However, below the beaver activity, the lowermost reaches were not suitable for measurement of sediment parameters but field observations and aerial images indicate substantial impact from livestock grazing which has led to riparian disturbance, bank erosion and failure, and channel widening. Recent visual observations noted clear water with turbidity noted only during spring runoff; TSS values low during baseflow site visits, ranging from non-detected to 12 mg/L. The upper reaches generally flow through a steep, confined, forested canyon and some logging activity was observed in the vicinity.

Assessment indicates O'Dell Creek is impaired for habitat. Field observations and aerial images indicate that there are areas where livestock grazing pressure is especially high in the lowermost reaches of the stream; in these areas, willow regeneration is limited and mature willows are heavily browsed, pasture grass is encroaching on riparian species, and the channel is trampled and widening. Upstream, riparian vegetation is robust and dominated by willows, sedge, horsetail and mixed forbs and beavery activity is prevalent. Mid-segment, the channel is stable with a defined riffle-pool sequence and no evidence of downcutting or excess bank erosion. Silt accumulates mostly along channel margins. The channel appears to have been historically incised and is now aggrading and it is suspected that natural factors (beaver activity), upstream logging activities (active during site visit), and water diversions may be impacting the flow regime and sediment transport capacity.

Nutrient assessment indicates O'Dell Creek is not impaired for nutrients. Water chemistry data were evaluated against numeric nutrient standards for TN and TP and to recommended NO2+3 nutrient criteria for the Middle Rockies ecoregion (TN = 0.30 mg/L; TP = 0.03 mg/L; NO2+3 = 0.10 mg/L). Of all the nutrient samples collected (n = 13 for TN, TP, and NO2+3), zero samples exceeded the TN standard, four samples exceeded the TP standard, and zero samples exceeded the NO2+3 recommended criteria. Benthic algae samples were evaluated against the recommended thresholds for chlorophyll-a (120 mg/m2) and ash-free dry weight (35 g/m2). Of all the benthic algae samples analyzed (n = 2) zero results exceeded the respective thresholds. Of all the macroinvertebrate samples collected (n = 2), 0 samples exceeded the HBI threshold.

Escherichia coli bacteria (E. coli) data were collected to update the 2020 IR. All E. coli bacteria samples were collected when the primary contact water quality standard for B-1 standards apply (April 1 - October 31). There is sufficient data to assess the data with the 30 Day approach. The dataset from 7/19/17 to 8/17/17 has a geometric mean of 26.96 cfu/100mL; 0 out of 7 samples exceeded the 252 cfu/100ml result (0%) E. coli will not be listed as a cause of impairment affecting the primary contact recreation beneficial use.

Metals assessment indicates O'Dell Creek is not impaired for metals. Aluminum (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), lead (Pb), selenium (Se), silver (Ag), and zinc (Zn)) data were collected to update the 2020 IR. All metals samples were collected during which the water quality standards apply. There is sufficient data to assess the data. Data had high flow samples at acceptable frequencies. For metals parameters

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dissolved Al, As, Cd, Cr, Cu, Fe, Pb, Se, Ag and Zn, no sample exceeded aquatic life chronic or acute or human health standards. Although not used for metals assessment, two benthic sediment samples were collected and analyzed for As, Cd, Cr, Cu, Pb, Hg, and Zn. All sediment concentrations are below NOAA screening criteria.

Probable sources of pollution identified in this watershed include: Grazing in Riparian or Shoreline Zones, Natural Sources, and Silviculture Activities.

Pre-2020 comments that were not addressed or changed during 2020 cycle:
Site Reach Name: From Headwaters to Mouth (Lower Red Rock Lake) Aquatic Life and Cold Water Fishery: BIOLOGY - moderate impairment in fishery; grayling population decreasing over past two decades, trout population increasing; grayling spawning area threatened due to severe erosion and sedimentation; HABITAT: dewatering has been observed in a stretch of the stream, but data is not documented; Agriculture: no high salinity or toxicant levels documented Industrial: no high salinity or turbidity levels documented.

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USE SUPPORT DECISION

Use Class B-1

Trophic Status: **Trophic Trend:**

Uses	DQA	Method, Data, and Information Used	Assessment Type and Confidence	Use Support	Partial Flag	Use Support Certainty	Threatened
Aquatic Life	Pass			Not Fully Supporting	No		No
Agricultural				Not Assessed	No		No
Drinking Water	Pass			Fully Supporting	No		No
Primary Contact Recreation	Pass			Fully Supporting	No		No

Method Number and Description

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IMPAIRMENT INFORMATION

Uses	Cause (Confidence): Source(Confirmed)	Observed Effects
Aquatic Life	84 (): 46 (Y) 371 (): 46 (Y), 155 (Y), 166 (N)	
Agricultural		
Drinking Water		
Primary Contact Recreation		

Cause Number and Description	Source Number and Description	Observed Effect Number and Description
84-Alteration in stream-side or littoral vegetative covers 371-Sedimentation/Siltation	46-Grazing in Riparian or Shoreline Zones 155-Natural Sources 166-Silviculture Activities	

DELISTING / STATUS CHANGES

Cause	Reason for Change	Date of Change
Turbidity	Data and/or information lacking to determine WQ status; original basis for listing was incorrect	11/19/2019

Montana DEQ - Water Quality Standards Attainment Record

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CATEGORY INFORMATION

Previous Cycle

Cycle	2018
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Current Cycle

Cycle	2020
Category	5 - Waters where one or more applicable beneficial uses have been assessed as being impaired or threatened, and a TMDL is required to address the factors causing the impairment or threat.
User Defined Category	N/A

Exhibit 3

PNAS COMMENTARY

Bison outperform cattle at restoring their home on the range

Kyle E. Harms[a,1]

High-diversity grasslands are increasingly recognized as "old-growth" communities, having assembled over at least millennia, and often much, much longer (1, 2). Human-caused disturbance frequently disassembles such communities, reducing them to lower-diversity states (3). A substantial challenge to conservationists interested in habitat restoration is to reassemble high-diversity communities that have been degraded by anthropogenic disturbance. Although we know that certain North American grasslands have the potential for very high diversity of plants and other organisms (1, 4), we know relatively little about the historic influences of extirpated large native herbivores, which can be keystone species in the communities they inhabit (5). Among the reasons for our ignorance are the geographic extent and nature of degradation of grassland ecosystems: Many have been extensively plowed, the natural fire regime has been altered, and large native grazers have long been absent from most (1–3, 5). In the case of native grazer removal, one of the most important unknowns is how well the effects of these missing animals can be matched by similar nonnative stand-ins. These considerations bear on broader questions in conservation biology about "rewilding" and the restoration potential of influential species of all sorts. In PNAS, Ratajczak et al. (6) share evidence that reintroduced American bison (*Bison bison*) more effectively diversified a Great Plains plant community—from which bison had long been extirpated—than did their nonnative counterparts, cattle.

The US westward expansion of the 19th century caused bison numbers to crash and cattle numbers to soar. However, without unmodified reference sites it has been difficult to fully understand the extent to which these anthropogenic disturbances disassembled previously intact tallgrass prairie communities (5). Ratajczak et al. (6) studied the influence of native bison and domesticated cattle at the Konza Prairie Biological Station (KPBS) in Kansas, part of the Flint Hills ecoregion, which contains the largest remaining tract of unplowed tallgrass prairie (Fig. 1). For 29 y they compared vegetation among sites with bison, cattle, or neither large herbivore. During the long-term experiment, clear and consistently different trajectories characterized the three treatments. Ungrazed sites changed relatively little through time. In contrast, bison caused native plant species richness to increase compared to ungrazed sites during the nearly three-decade period, culminating in 103% higher species richness at the 10-m^2 plot scale and 86% higher richness at the larger catchment scale (each catchment was >18 ha and sampled with 20 noncontiguous plots). At the two respective scales, cattle caused modest 41% and 30% increases in native plant species richness compared to ungrazed sites. Relatively few nonnative plants were present in any of their sites. Bison concomitantly reduced combined cover of the four dominant grass species and increased forb cover, whereas dominant grass cover in sites grazed by cattle remained at intermediate levels between the relatively low dominant grass cover with bison and the relatively high grass cover in ungrazed sites. Fire is a key process in the tallgrass prairie ecosystem (5), but prescribed fire frequency (1- to 4-y return intervals) did not qualitatively change the observed outcomes with respect to grazer treatments.

To foster thoughtful conservation and restoration, Aldo Leopold (7) developed the Land Ethic—his philosophy concerning humans' responsibility toward nature. One of his metaphorical dictums was "to keep every cog and wheel is the first precaution of intelligent tinkering" (8). Even so, a general principle of community ecology is that some species play outsized roles in the ongoing process of community assembly. Keystone species wield large and disproportionately large influences relative to their abundances (9, 10). Foundation species form structurally dominant populations around which the rest of their community's species assemble (11, 12). Ecosystem engineers create, maintain, or modify habitat by physically altering environmental materials (13). With respect to community assembly, these various species of unusual effect (SUEs) are especially important "cogs and wheels." To remove any of these species from an intact community generally has substantial consequences for species composition, diversity, and often ecosystem function; to add one back to a community from which it was previously removed could have a similarly large effect on reversing those consequences.

By increasing plant species richness relative to ungrazed sites in the Konza Prairie experiment (6), it could be argued that both bison and cattle can play keystone roles in this tallgrass ecosystem. A previous smaller-scale, shorter-term experiment at KPBS found that both bison and cattle had a diversifying influence on the prairie vegetation (14). However, within the longer-term experiment, the bison effect on native plant species richness was more than double the cattle effect at the plot scale (6). In addition, sites with bison were more resilient to extreme drought (6). During the 20th and 21st years of the study, the Konza Prairie experienced one of its most severe droughts since the Dust Bowl of the 1930s. Although the drought reversed the steady increase in species richness that had occurred before the drought in

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Fig. 1. KPBS (site of the Konza Prairie Long-Term Ecological Research program): reintroduced native bison (*Top*), the prairie landscape with bison and prescribed fire (*Middle*), and narrow-leaved purple coneflower (*Echinacea angustifolia*), a native forb species (*Bottom*). All images courtesy of Eva Horne (Division of Biology, Kansas State University).

the bison-grazed catchments, species richness relatively rapidly recovered to meet a linear extrapolation of its predrought, steadily increasing trend. In contrast, postdrought species richness with cattle oscillated but resulted in no net change at the end of the study. In other words, reintroduced native bison are substantially more effective keystone species for diversifying the community than are domesticated cattle.

The relatively sustained increase in plant species diversity when bison were added back to the community supports the parsimonious conclusion that bison also had a diversifying influence before they were extirpated. We may never know whether the bison effect in the current study fully aligns catchments with the community compositions and biodiversity properties that existed prior to the 1800s, but higher diversity clearly benefits tallgrass prairie conservation. With so little relatively untouched, old-growth grassland remaining, any tract of high native species diversity is valuable (3, 5). Besides, bison also need habitat in which to roam!

How does a single species of large herbivore change the processes of community assembly so drastically, sustaining nearly three decades of diversifying influence and doubling small-scale plant diversity? Why are bison so much better at it than cattle? Ratajczak et al. (6) acknowledge that their large-herbivore treatments represent two different, albeit typical, land-use types that differ for a variety of reasons beyond grazer identity. In their experiment, grazing seasonality and stocking density differed between the two species. Bison and cattle also behave differently; e.g., bison generally selectively consume a higher percentage of dominant native grasses, create wallows, and exhibit their own distinctive individual and collective grazing-patch dynamics, resulting in particular patterns of microdisturbances (e.g., trampling) and nutrient redistribution (e.g., deposition of dung and urine) (5, 6). Whether the hyperdiversifying effect of bison results directly from increased fine-scale environmental heterogeneity owing to their idiosyncratic behaviors, indirectly through competitive release of high-diversity forbs, or some other cause, understanding the potential mechanisms for the observed pattern of vegetation differences among treatments would help resolve one of the most elusive problems in ecology: the maintenance of diversity in high-diversity communities (e.g., ref. 15). The mechanistic explanation could also help identify management options for cattle (e.g., stocking certain breeds or altering their husbandry) to render their effects more like those of bison and to diversify working grassland landscapes at an even broader scale to include areas where bison are unlikely to be added. In addition, further comparisons between native SUEs and analog species will help clarify the circumstances under which "rewilding" with functionally similar substitutes for native taxa differentially influences native biodiversity.

Restoration ecologists generally agree that it is usually favorable to bring back lost elements (native species, natural processes, etc.) when restoring a community (16, 17). This may be especially true for SUEs. For example, what would a tallgrass prairie be without its foundational guild of dominant C_4 grasses (5)? Similarly, Ripple et al. (18) recommend rewilding apex predatory wolves and ecosystem engineering beavers in US western states. They marshaled evidence that bringing back those focal species across selected landscapes could substantially help achieve the goal of the Conserving and Restoring America the Beautiful initiative codified in President Biden's Executive Order 14008—to conserve at least 30% of our national lands and waters by 2030. The study by

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Ratajczak et al. (6) further supports the ecosystem management rule of thumb that SUEs in natural communities can be unusually effective at diversifying anthropogenically disturbed communities, with great potential to meet the challenging policy objectives that face us as we continue to strive to create a sustainably diverse planet.

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Exhibit 4

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Bison Reintroduction to Mixed-Grass Prairie Is Associated With Increases in Bird Diversity and Cervid Occupancy in Riparian Areas

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In grassland ecosystems, grazing by large herbivores is a highly influential process that affects biodiversity by modifying the vegetative environment through selective consumption. Here, we test whether restoration of bison is associated with increased bird diversity and cervid occupancy in networks of riparian habitat within a temperate grassland ecosystem, mixed-grass prairie in northcentral Montana, United States. We used a long time-series of remote sensing imagery to examine changes in riparian vegetation structure in stream networks within bison and cattle pastures. We then assessed how vegetation structure influenced diversity of bird communities and detection rates of mammals in these same riparian networks. We found that percent cover of woody vegetation, and native grasses and forbs increased more rapidly over time in bison pastures, and that these changes in vegetation structure were associated with increased bird diversity and cervid occupancy. In conclusion, bison reintroduction appears to function as a passive riparian restoration strategy with positive diversity outcomes for birds and mammals.

Keywords: grassland, songbirds, buffalo, cattle, conservation, ungulates, rewilding, American Prairie

INTRODUCTION

Restoration of evolutionary grazing processes, those that replicate or mimic effects of native herbivores, is a common goal of restoration efforts in temperate grassland systems (Freese et al., 2014; Fuhlendorf et al., 2018). Grazing is a ubiquitous natural process that creates and maintains habitat for myriad grassland species (Milchunas et al., 1988; Gao and Carmel, 2020). Whereas native grazers are the preferred option for restoration, in nearly all temperate grassland systems, native grazers have been replaced with cattle (*Bos taurus*) (van Zanten et al., 2016), which are raised for milk, meat or other animal products. When managed sustainably, cattle grazing can provide the disturbance regimes and vegetation heterogeneity necessary for diverse grassland systems (Milchunas et al., 1998; Porensky et al., 2020; Boyce et al., 2021). Furthermore, sustainable cattle grazing maintains healthy soils and resilient plant communities, resulting in more intact ecosystems than row-crop agriculture, which is the primary alternative land use in many grasslands

(Krausman et al., 2009; da Silva et al., 2015). Restoration efforts in temperate grassland systems in North America have often focused on the re-introduction of the native megaherbivore; plains bison (*Bison bison bison*) (Freese et al., 2014). There is some evidence for biodiversity benefits of bison reintroduction, including increased diversity in plants (McMillan et al., 2018) and increased abundance of some grassland obligate songbirds (Boyce et al., 2021) but opportunities to evaluate its biodiversity impacts are rare (but see Allred et al., 2013; Nickell et al., 2018). Furthermore, the bison reintroduction process is both expensive (Carbyn and Watson, 2001) and controversial (Ranglack et al., 2015), so it is critical to evaluate whether these efforts are resulting in increasingly diverse and resilient ecosystems.

Most comparisons between bison and cattle have focused on differences in biodiversity or vegetation structure in upland grasslands (Greibel et al., 1998; Lueders et al., 2006; Moran, 2014; McMillan et al., 2018; Nickell et al., 2018). However, the largest behavioral differences between these species is their use of wetlands and associated woody vegetation (Kohl et al., 2013). There are several ecological and physiological differences between bison and cattle that support the hypothesis that their divergent grazing patterns and habitat preferences will differentially affect riparian systems. Cattle are known to degrade riparian areas (Kauffman and Krueger, 1984; Fleischner, 1994) due to damage or removal of riparian woody vegetation; effects that cascade into degradation of water quality and large fluctuations in stream temperatures and biogeochemistry (Larson et al., 2019). In contrast, bison are more drought and heat-tolerant, allowing them to graze farther from water, especially in hot conditions (Allred et al., 2013; Kohl et al., 2013). Compared with cattle, bison select against areas with woody vegetation and standing water, spend less time browsing, and specialize more on grasses, as opposed to forbs or woody vegetation (Peden et al., 1974; Knapp et al., 1999; Steuter and Hidinger, 1999; Allred et al., 2011; Kohl et al., 2013; Ranglack and du Toit, 2015).

The above differences predict replacing cattle with bison will have net positive impacts on quantity and complexity of riparian vegetation, but this hypothesis is largely untested. Furthermore, we predict that increases in riparian vegetation will be associated with increased diversity of bird communities (MacArthur and MacArthur, 1961; Macarthur, 1964; Cooper et al., 2020). We also predict that deer use will increase with higher shrub and tree cover after accounting for distance to the Missouri River drainage, a forested landscape that serves as a source population for cervids. Here we combine contemporary data on vegetation structure, bird community diversity and ungulate occupancy, with a long-term remotely sensed vegetation dataset to test whether bison reintroduction in a mixed-grass prairie ecosystem has resulted in higher quality riparian habitat in comparison with areas seasonally grazed by cattle.

MATERIALS AND METHODS

Study Area

We studied vegetation and animal communities associated with ephemeral streams on the northwest glaciated plains subregion of the Northern Great Plains ecosystem (Forrest et al., 2004). Our study area included parts of Blaine, Phillips and Valley counties bounded by the Milk River in the north, the Missouri River in the south and the western boundary of the Fort Belknap Indian Reservation on the west (**Figure 1**). Land ownership is a mix of private lands concentrated in the vicinity of permanent water and alluvial soils, with large blocks of public land composed of mixed-grass prairie or sage steppe. The Charles M. Russell National Wildlife Refuge is an exception, as it contains nearly 1 million acres of rugged "breaks" canyons with conifer (*Pinus ponderosa* and *Juniperus scopulorum*) savannah and includes the extensive riparian bottomlands of the Missouri River. In the uplands, conversion from native grassland to row-crop agriculture is ongoing, and conversion of riparian vegetation to hay fields for cattle forage is widespread (Gage et al., 2016).

The study area contains many waterways, from small ephemeral streams which innervate the expansive uplands, to the Missouri River, one of the largest in North America. The small seasonal or ephemeral streams are isolated strands of riparian vegetation amidst large expanses of grassland or sage steppe. Typical riparian vegetation in the region includes woody shrubs like common snowberry (*Symphoricarpos albus*), sandbar willow (*Salix exigua*), and wild rose (*Rosa* spp.). Common tree species are eastern cottonwood (*Populus deltoides*), box elder (*Acer negundo*), and peach-leaf willow (*Salix amygdaloides*). In heavily grazed areas, woody vegetation can be entirely absent, with low-growing grasses and forbs found up to the channel edge.

American Prairie is a private non-governmental organization with the mission to create the largest nature reserve in the lower 48 states[1]. American Prairie owns 423 km^2 of private land and holds the grazing leases for an additional 1,275 km^2 acres of public land as of 2021. The goal of American Prairie is to manage its lands as a fully-functioning grassland ecosystem complete with keystone grazers (bison and black-tailed prairie-dogs *Cynomys ludovicianus*) fulfilling their ecological role (Knapp et al., 1999; Freese et al., 2018). From 2005 through 2021, American Prairie has reintroduced bison to three large parcels of either private land or a mixture of private land and leased grazing allotments managed by the Bureau of Land Management (BLM; Freese et al., 2018; Boyce et al., 2021). Bison herds vary in size; approximately 150, 200, and 400 animals within 2,349, 2,963, and 10,909 ha pastures as of 2021. These herds have corresponding stocking rates of 0.77, 0.81, and 0.44 AUMs per hectare, which correspond to normal-year precipitation estimates calculated by BLM staff for public grazing allotments and by a private contractor (EMPSi Inc., Boulder, CO, United States), and conform to NRCS methodology for private parcels. Bison populations are regulated by public hunting opportunities, donations of animals to other conservation herds, and temporary chemical contraception (Freese et al., 2018). The Aaniiih and Nakoda peoples, who live at Fort Belknap Community, also maintain a large conservation herd of bison (∼900 individuals) on its 8,903 ha Snake Butte pasture. This pasture has a substantially higher stocking rate (1.21 AUMs per hectare) as those managed by American Prairie and bison were

[1] www.americanprairie.org

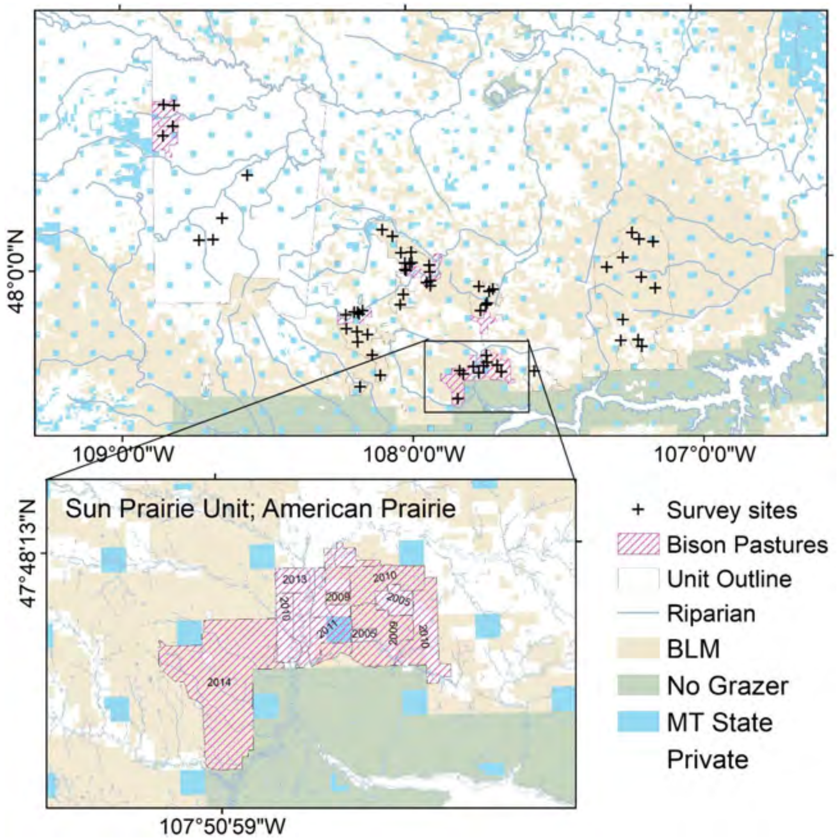

FIGURE 1 | Survey sites. Unit outlines include Fort Belknap Indian Reservation and American Prairie lands.

introduced nearly 30 years ago (Shamon et al., 2022). All bison herds are managed via continuous grazing, in which bison move freely within each grazing allotment throughout the year.

Cattle pastures within our study area were managed via deferred rotation grazing (Rhodes, 2020, BLM, personal communication). Deferred rotation refers to a management regime where allotments are divided into 3–4 pastures using barbed-wire fence, and cattle are moved sequentially through all pastures over the course of each grazing season (March–November). During winter, cattle are removed from public lands grazing allotments and sold or fed overwinter on private lands. Because private lands experience winter grazing, we separated BLM and private (Non-BLM) cattle pastures for our analysis. Cattle pastures in the region have been managed consistently using this protocol for 10+ years (Rhodes, 2020, BLM, personal communication). Cattle stocking rates on BLM lands are dictated by the same NRCS methodology used to determine bison stocking rates on both private and BLM lands. Specific stocking rates are variable over time in response to precipitation and across space in response to small differences in soil productivity. Because bison and cattle stocking rates are determined by the same methodology, we consider the effective stocking rates as similar, with the key difference that bison numbers per unit area are lower because their AUMs are distributed across a 12-month period as opposed to a shorter growing season grazing period in cattle. There are two exceptions to this principle within our study area. First, bison stocking rate on the Snake Butte pasture on the Fort Belknap Indian Reservation is substantially higher than on American Prairie bison pastures and BLM cattle pastures (Boyce et al., 2021; Shamon et al., 2022). Second, cattle stocking rates on private lands are variable depending on economic decisions by individual ranchers and details are not publicly available.

Vegetation

Current Differences in Vegetation Cover Amongst Treatments

We compared current vegetation cover proportions (2019) between treatments using a logistic regression with logit-link function; $f\left(veg\ cover\right) \sim Treatment$ (Warton and Hui, 2011). *Intercept* was set to BLM cattle, the most common form of management in the study region. We ran 8 model combinations (*2 distance categories* × *4 vegetation types*). Current vegetation cover estimates were derived from Range Analysis Platform (RAP) Vegetation Cover Dataset 2019 at 30 m resolution (Allred et al., 2021).

We assessed riparian vegetation structure across four general treatments: BLM cattle, private lands cattle, bison, and no bovine grazers. Bison pastures were then subcategorized according to the year at which bison were reintroduced to the pasture (2005, 2009, 2010, 2011, 2012, 2013, 2014), creating 10 total grazing treatments. First, we identified riparian segments using the National Hydrography Dataset (Moore et al., 2019). Next, we created two distance category buffers around streams: (1) 0 m from stream (e.g., within streambed); (2) <30 m from stream edge. Distance categories were chosen at 30 m scale which matches the resolution of the vegetation cover data used for the analysis. Distance categories highlight the gradient effects of grazing treatment on vegetation. Next, we extracted vegetation cover percentages for each pixel centroid that fell within the two distance categories of a streams using Google Earth Engine (Gorelick et al., 2017).

Long-Term Vegetation Trends

We assessed the direction and rate of change of vegetation cover using RAP Vegetation Cover Dataset 2000–2019 time-series (Allred et al., 2021). We computed a pixel-wise Mann–Kendall rank correlation and estimated Theil-Sen's slope for four vegetation types. Calculations were done using the "rkt" package (Marchetto, 2017) available from CRAN (R Core, 2018). Mann–Kendall rank correlation is a non-parametric test that is considered resilient to outliers and combined with Theil-Sen's slope has proven to be a successful predictor for time-series analysis such as vegetation and climate trends (Gocic and Trajkovic, 2013; Li et al., 2013). Mann–Kendall test provides an assessment of slope estimate uncertainty, and the slope indicates the rate of change and the direction of the trend where positive slope means increase in vegetation cover and negative slope means reduction. Overall, we calculated vegetation tends for 80 combinations of *grazing treatment (10)* × *distance category (2)* × *vegetation type (4).* Vegetation types included: (1) annual forbs and grasses; (2) perennial forbs and grasses; (3) shrubs; and (4) tree cover.

We modeled current vegetation cover and time-series slope estimates to determine differences between BLM cattle and the other nine treatments. The analysis was conducted at two distance categories, and four vegetation types separately. We only used estimates with significance level of <0.05 in this analysis. Slope estimates were used as a response variable against treatment categories using a generalized linear model framework (GLM); "stats" R base package (R Core, 2018). Intercept was set to BLM cattle grazed (most common form of management in the study region) and we ran 8 model combinations (*2 distance categories* × *4 vegetation types*).

Vertebrate Surveys

Occupancy of Grazers and Browsers

We detected ungulates with camera traps (model Reconyx HyperFire 2) during two growing seasons (July–October 2018, May–September 2019). We deployed camera traps at 78 riparian sites. We placed three cameras in each survey site spaced 250–400 m apart. Cameras were set at 50 cm above ground, facing north to minimize false triggers induced by direct sun exposure. Images were sorted, identified to species, and stored in the eMammal repository (Shamon, 2021). We collected habitat data at each camera location to assess detection bias. These data included the percentage of ground cover vegetation (bare ground, grass, forb, and shrub), percentage of canopy cover, mean shrub height within 5 m in front of each camera, distance at which the camera sensor was triggered in response to an approaching human, and whether or not the camera was set on an obvious animal trail. Percent cover class and mean shrub height were estimated visually. Riparian sites were used to model cervid occupancy in relation to vegetation cover and structure, and these data were used to compare cattle and bison activity with the riparian area. We only used camera deployments which functioned for ≥7 days, which resulted in 213 deployments in the riparian area for deer species. For the bison and cattle models, we eliminated deployments from areas where no cattle were present in the pasture due to rotation to other sub-pastures or removal from the pasture entirely, resulting in 198 deployments (107 bison and 91 cattle).

To compare between bison and cattle activity at streams we used detections counts for each species. Detection counts were modeled using N-mixture models, a family of models that can estimate counts while accounting for imperfect detection (Royle, 2004; Joseph et al., 2009; Zhou and Carin, 2015). Cattle and bison densities in the tested area remained steady during the study period, therefore we do not expect bias due to population fluctuations. Model calculations were done in a two-step process where first we identified the variables that affect detection probability (binominal distribution; see detection covariates above). Detection models were ranked by Akaike information criterion (AIC) and the best fitting model was used in each N-mixture model combination (count model using Poisson distribution). We modeled both species together using "*species*" as a categorical covariate to learn about the difference in activity by streams, and bison was set as an intercept. We assumed detection probability would be the same for bison and cattle given the similarity of body size and the consistency of camera setups, therefore detection was modeled for both species together.

We estimated occupancy for three deer species (mule deer; *Odocoileus hemionus*, white-tailed deer; *Odocoileus virginianus*, and elk; *Cervus canadensis*) in riparian habitats (MacKenzie et al., 2002). We used presence-absence data instead of detection counts to avoid bias of population fluctuations between years. Model calculations were done in a two-step process where first we identified the variables that affect detection probability (binominal distribution; see detection covariates above). Detection models were ranked by AIC and the best fitting model was used in each occupancy model. Second, we modeled deer presence-absence data against proportion of vegetation cover within 100 m buffer around the camera location derived from both the RAP vegetation cover dataset and the distance from major rivers. Different combinations were tested and competing models were ranked by AIC score for each species and we considered models with <2 delta AIC as equivalent.

Bird Diversity

To characterize riparian bird communities we conducted 10-min 200 m fixed-radius point counts laid out along ephemeral or seasonal streams such that the center points of each point count were at least 500 m from the nearest neighbor point count location to avoid double-counting (Hutto et al., 1986). Each cell was visited once per field season between May 30 and July 3, in 2018 and in 2019. All birds seen or heard were recorded and estimated distance and bearing to each individual was recorded to help prevent double-counting. Point counts began at 30 min before sunrise and all counts were completed prior to 8 am to minimize variation in detectability related to time of day. Following Hutto et al. (1986), point counts were not conducted during strong wind or precipitation. We aimed to describe communities of birds associated with riparian habitats, so we removed grassland obligate species under the assumption they were detected in adjacent grassland or sage steppe. We also removed species only detected as flyovers which we could not safely assume were using habitat in the point count area (primarily raptors and waterfowl).

We used three diversity metrics to describe the bird community at each survey location: species richness (SR), Faith's phylogenetic diversity (Faith, 1992; hereafter PD), and functional richness (Mason et al., 2005; Villéger et al., 2008, hereafter FRic). Species richness is simply the number of species detected at a given site. PD describes the total phylogenetic breadth of a community, calculated as the sum of all branch lengths in a phylogenetic tree including all species in a community. Under the general assumption that ecological roles are phylogenetically conserved, PD should be a proxy for overall niche space encompassed by a given community. FRic is analogous to PD, but distance among species is not defined by their phylogenetic relatedness, but by how similar or different they are based on a variety of functional traits related to a species' ecology (Villéger et al., 2008).

To facilitate calculation of PD, we downloaded a subset of 1,000 trees from www.birdtree.org (Jetz et al., 2012) containing all riparian species detected on our counts. We produced a consensus tree and estimated all branch lengths using the "consensus.edges" function in the phytools package (Revell, 2012). To estimate FD, we compiled seven functional traits for each species from the literature. Six of seven traits were morphological measurements linked to locomotion and diet: wing chord, tail length, tarsus length, bill length, hand-wing index, and body mass (Miles and Ricklefs, 1984, 1994; Pigot et al., 2020). We also compiled information on the primary diet of each species, classified into one of six categories: invertebrates, vertebrates, seeds, fruits, plants, and omnivores. Hand-wing index, body mass, and diet data were sampled from a comprehensive dataset compiled by Sheard et al. (2020). All other morphological traits were compiled from a large dataset of passerine morphology (Ricklefs, 2017), Cornell's Birds of the World (Billerman et al., 2020) or references therein. Body mass values were log-transformed before analyses and all analyses were conducted in R v. 3.5.1 "Feather Spray" (R Core Team, 2015).

Phylogenetic diversity and species richness were calculated using the "pd" function in the package "picante" (Kembel et al., 2010). PD can only be estimated for communities of at least two species, so we assigned a PD value of 0 to communities of only one species. FRic was calculated using the "dbFD" function in the package "FD" (Laliberte and Legendre, 2010). FRic can only be estimated for communities containing at least three species, so we assigned a FRic value of zero to all communities with fewer than three species (1 site).

To describe associations between riparian vegetation and bird communities we combined data from the remotely sensed RAP platform (Allred et al., 2021) with our riparian point-count data. We fitted generalized linear models with each diversity metric (SR, PD, and FRic) as the response variable, and three of the four vegetation metrics (tree cover, shrub cover, perennial grass/forb cover) as explanatory variables. Since percentages must add to 100, the fourth category, annual grass/shrub, was excluded from the model as redundant and the parameter estimated for each included category are in comparison to annual grass/forb. Models explaining variation in raw species richness values were fit using a Poisson distribution while all other models used a Gaussian distribution.

RESULTS

Contemporary Differences in Vegetation

We found perennial grass/forb cover was significantly higher in private cattle grazed pastures and bison pastures in areas at, or closer to, the stream (category 0 m and <30 m; excluding 2014 bison pastures; **Figure 2**). In contrast, annual grass/forb cover was significantly higher at BLM cattle streams; consistent for categories 0 m and <30 m (**Figure 2**). Differences in shrub cover were variable between BLM cattle and bison pastures but were significantly lower at private cattle pastures and higher in the no-grazer area (**Figure 2**). Differences in tree cover were variable between BLM cattle and bison pastures with a tendency to be higher at bison pastures and were significantly higher at private cattle pastures and the no-grazer area (**Figure 2**).

Long-Term Vegetation Trends

We found perennial grass/forb cover trends were significantly higher in bison pastures at both distance categories and no-grazer areas for distance categories <30 m (**Figure 3**). At stream (0 m) results were variable with a tendency to be higher than BLM cattle (**Figure 4**); the opposite trend was observed for private cattle pastures. Annual grass/forb cover trends were variable with a tendency to be higher for 0 m at bison pasture and lower for <30 m at bison and no-grazer areas (**Figure 3**). Annual grass/forb cover trends were significantly higher at private cattle pastures. Differences in shrub cover trends were variable between BLM cattle and bison pastures but were significantly lower at private cattle pastures and higher in no-grazer areas (**Figure 3**). Differences in tree cover trends were variable between BLM cattle and bison pastures with tendency to be higher at bison pastures and were significantly higher at private cattle pastures and no-grazer areas (**Figure 3**).

Ungulate Occupancy

Comparison of Cattle and Bison Detection Counts

The mean and standard deviation of cattle and bison detections per 100 camera nights at riparian cameras were 106.8 ± 376.7 and 40.0 ± 107.9, respectively (number of camera deployments: n_{cattle} = 91 and n_{bison} = 107). Because cattle graze seasonally

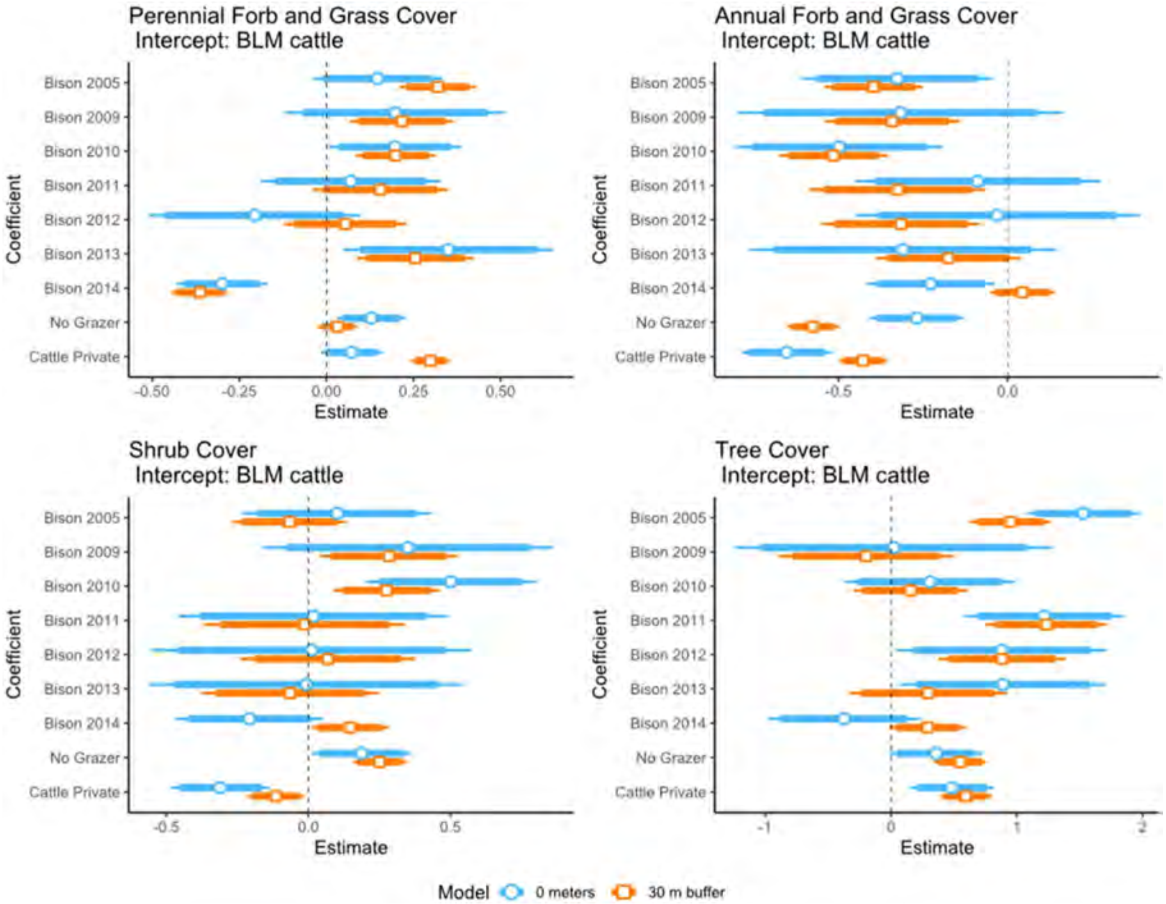

FIGURE 2 | 2019 percentage of vegetation cover in relation to grazing treatment. Intercept: BLM cattle grazed streams. Vegetation: perennial forbs and grasses, annual forbs and grasses, shrub cover, and tree cover. Treatment: BLM cattle, Bison (2005, 2009, 2010, 2011, 2012, 2013, 2014), no grazers, private lands cattle. Models were run for two distance categories: (1) 0 m from stream; (2) <30 m from stream edge.

and are rotated among multiple pastures, they were not always present during camera surveys. Accordingly, the data were subset to include only times when cattle were present, resulting in mean of 255.7 ± 553.2 cattle detections per 100 camera nights (number of camera deployments: $n = 38$). Modeling the full dataset that included times when cattle were not present at the pasture show there are no significant differences between bison and cattle overall activity in riparian areas (**Table 1**). However, modeling the dataset that only contained deployments when cattle were present in the pasture containing the camera deployment revealed that cattle are significantly more active in riparian areas than bison (coeÿcient = 1.4 ± 0.1 where bison category was set as the intercept; **Table 1A**).

Deer Occupancy in Relation to Riparian Vegetation Cover

Elk occupancy was primarily determined by distance to major rivers and the interaction between woody cover and distance to major rivers (**Table 1B**). Mule deer occupancy was unrelated to distance to rivers and did not show any specific relationship with vegetation cover (**Table 1B**). White-tailed deer occupancy increased with higher tree cover and higher perennial grass/forb cover (**Table 1B**).

Bird Diversity

We conducted 147 point-counts along prairie streams (**Figure 1**) and detected 59 species (mean species richness per count = 6.9; range 1–18; **Supplementary Material**). All three metrics of riparian bird community diversity (SR, PD, and FRic) increased with increasing tree cover, shrub cover, and perennial grass/forb cover in comparison with annual grass cover (**Table 2** and **Figure 4**). Across all three metrics tree cover had the largest positive effect on diversity, followed by woody shrub cover and finally perennial grass/forb (**Table 2** and **Figure 4**).

DISCUSSION

Reintroduction of bison and replacement of cattle with bison significantly impacted riparian systems in our study area. We found that streams in pastures grazed year-round by bison had a faster rate of increase of perennial grass/forb cover, as well as

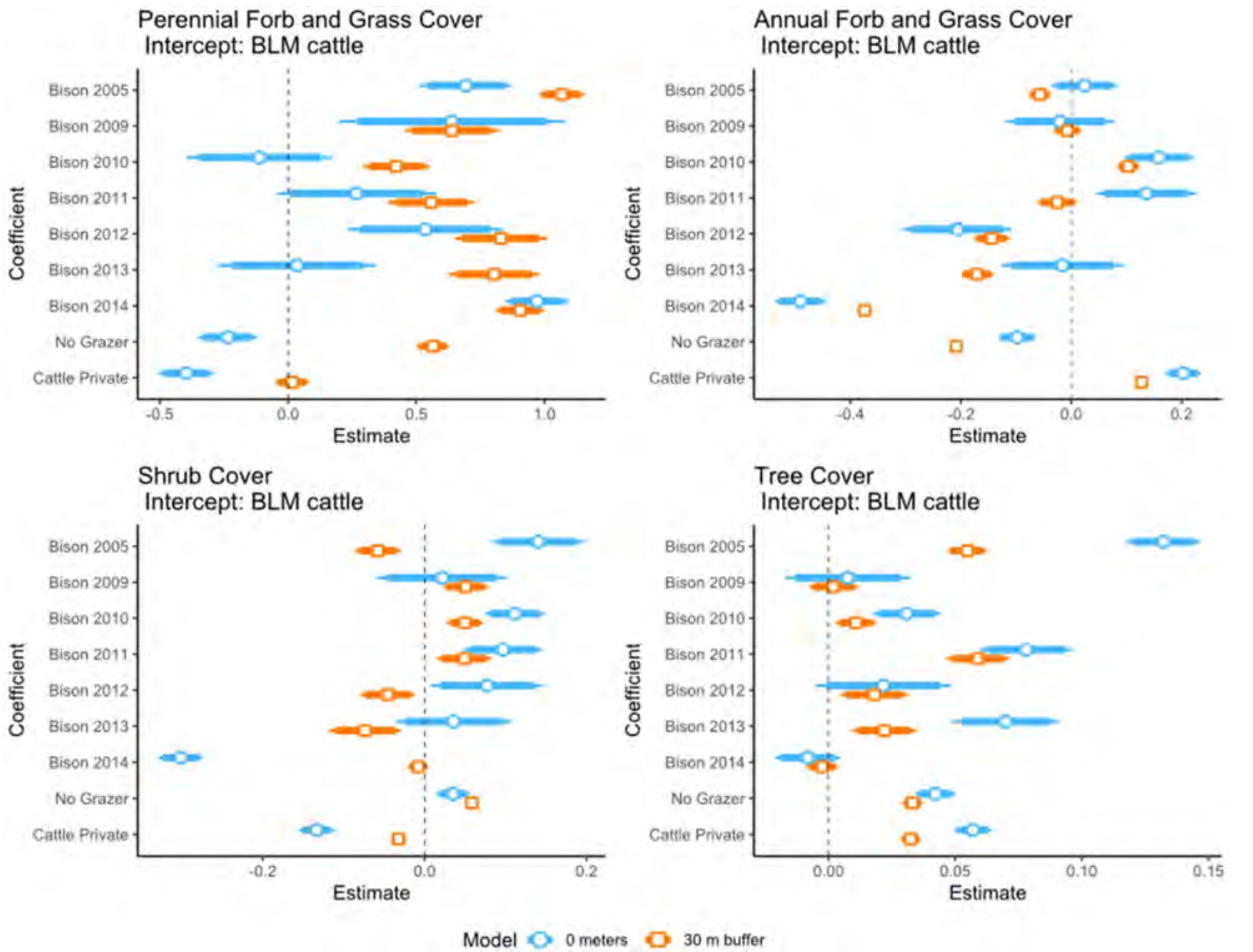

FIGURE 3 | Long-term vegetation trend estimates (slope) in relation to grazing treatment. Intercept: BLM cattle grazed streams. Vegetation: perennial forbs and grasses, annual forbs and grasses, shrub cover, and tree cover. Treatment: BLM cattle, Bison (2005, 2009, 2010, 2011, 2012, 2013, 2014), no grazers, private lands cattle. Models were run for two distance categories: (1) 0 m from stream; (2) <30 m from stream edge.

shrub and woody cover, when compared to streams in seasonally grazed cattle pastures. These trends are supported by camera trap records in streams that show cattle are detected significantly more within streams than bison and that activity is concentrated in time due to the rotation schedule. These results are consistent with previous studies showing that bison select against low-elevations and woody vegetation compared with cattle, and that bison forage farther from water (Knapp et al., 1999; Allred et al., 2011; Kohl et al., 2013; Ranglack and du Toit, 2015). We used remote sensing to document riparian area changes, but a ground-based study in the same area comparing vegetation plots within bison and cattle pastures found similar results of increased plant diversity and increased shrub cover within riparian areas of bison pastures (Yu, 2021). Furthermore, we show that streams with more trees and woody shrubs, and higher percent cover of perennial grasses and forbs relative to non-native annuals are associated with more diverse breeding bird communities and have higher rates of occupancy for two of three species of native ungulates (the exception was mule deer). The idea that greater amounts and complexity of vegetation drives increased bird diversity is hardly a new result (see Macarthur, 1964), but taken together, our results suggest that bison reintroduction, and a minimal intervention grazing management strategy, is associated with positive biodiversity outcomes in riparian habitats found within the Northern Great Plains.

Negative impacts of cattle grazing on riparian systems are well documented, particularly intense in arid regions (Fleischner, 1994; Belsky et al., 1999), negatively impact imperiled species (Ohmart, 1994; Wilcove et al., 1998; USFWS, 2002, 2020; Dettenmaier et al., 2017), and may increase in severity with climate change (Allred et al., 2013). Cattle grazing has been shown to have negative effects on riparian breeding bird communities in Montana (Fletcher and Hutto, 2008) and across the west (Tewksbury et al., 2002). In forested systems, removal of domestic grazers may present a suitable remedy for this issue since forest succession does not rely on intense grazing by bovines as a primary source of disturbance (Hessburg et al., 2019). Because of their ecological and physiological

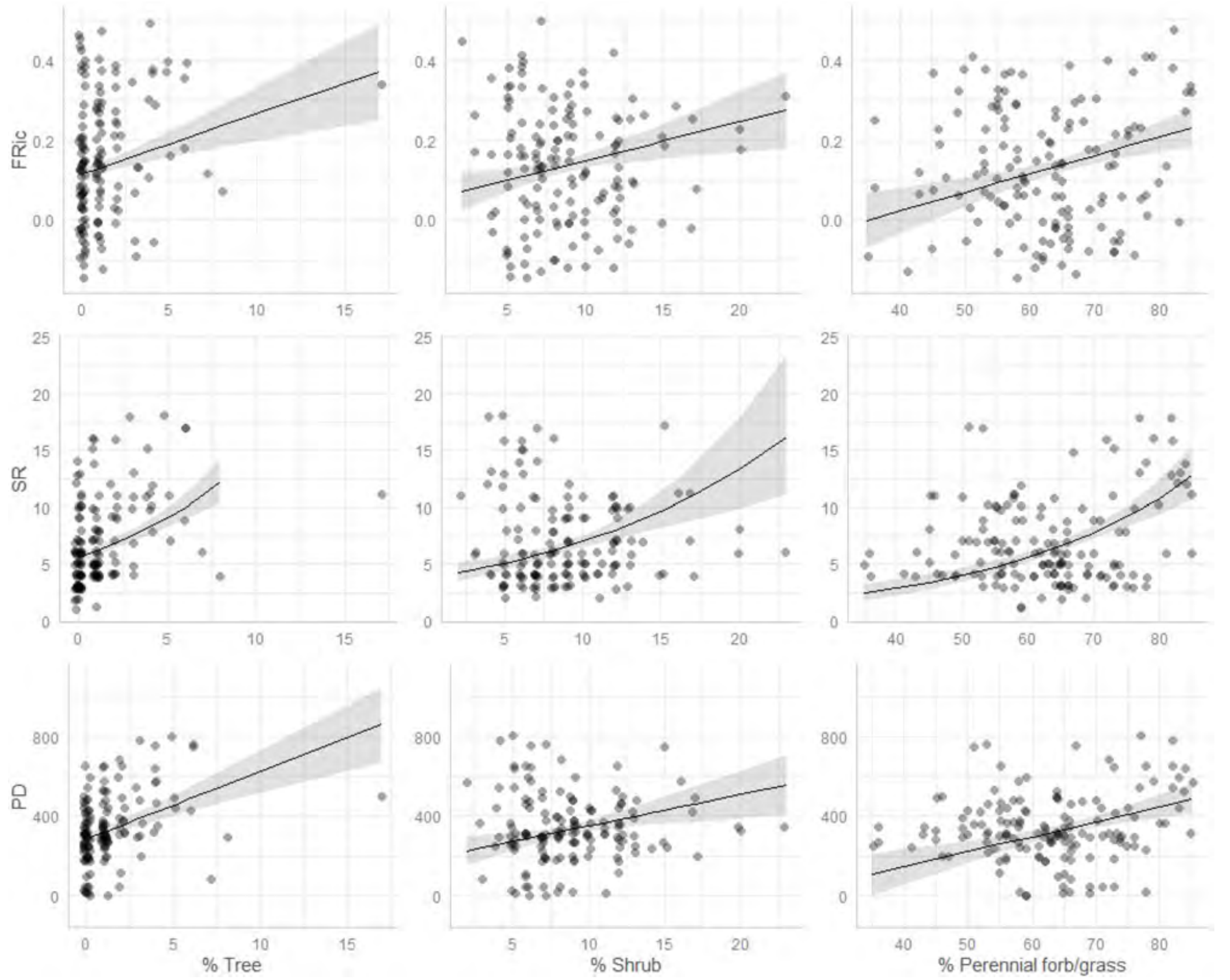

FIGURE 4 | Marginal relationships between three components of remotely sensed vegetative cover and three metrics riparian bird community diversity; functional richness (FRic), species richness (SR), and Faith's phylogenetic diversity (PD). Dots represent individual samples, lines and shading represent predicted relationships and associated 95% confidence intervals. All three vegetative components shown here are increasing in percent cover most rapidly in areas grazed by bison and areas with no grazers. Vegetation cover data are estimated from the RAP platform.

coevolution with arid grassland systems, bison are ideally suited to grassland ecosystems because they provide disturbance and thus heterogeneity in upland habitats (Gibson, 1989; McMillan et al., 2018), but have reduced impacts on riparian habitats, at least at agency standard stocking rates. Whether bison are reintroduced and managed as wild animals for ecosystem benefits, cultural benefits, or meat production, they are likely to provide ecosystem benefits if managed in a way that facilitates natural grazing patterns (Shamon et al., 2022).

Restoration in the Northern Great Plains will not only involve the introduction of large herbivores but also large carnivores. Due to their great mobility large carnivores can repopulate areas once given adequate movement corridors. Riparian systems are important movement corridors and refugia for dispersing, migrating, or resident wildlife (Machtans et al., 1996; Skagen et al., 2005). Linear features such as streams are used by both predator and prey species (Dickie et al., 2020) and forested riparian areas are key dispersal corridors for large mammals including species recolonizing grassland ecosystems in North America (Morrison et al., 2015; Gigliotti et al., 2019). Specifically, riparian corridors are used by black bears (*Ursus americanus*) for movement across grasslands in south-central United States (Gantchoff and Belant, 2017). Grizzly bears (*Ursus arctos*) use riparian areas to move through mixed use areas in British Columbia, Canada (McLellan and Hovey, 2001). Finally, mountain lions (*Felis concolor*) use riparian corridors to disperse between forest fragments (LaRue and Nielsen, 2008) and use riparian forest for dispersal through an agricultural/grassland matrix. The passive restoration of riparian corridors via bison reintroduction has the potential to increase landscape connectivity for large predators including grizzly bears and mountain lions which are actively recolonizing the

TABLE 1 | (A) Bison and cattle count model including the full dataset (all) and a subset that only includes deployments when cattle were active. **(B)** Occupancy estimation for deer species in relation to vegetation cover within a 100 m buffer around a camera and distance to major rivers.

(A)	**Detection**	**Counts model**					
	Intercept	**Intercept**	**Cattle**	**negLogLike**	**nPars**	***n***	**AIC**
Bison and cattle (all)	−3.23 ± 0.07	2.135 ± 0.07	0.127 ± 0.07	6565.58	3	179	13141.16
Bison and cattle (subset)	−3.62 ± 0.07	2.30 ± 0.08	1.43 ± 0.06	6098.20	3	128	12202.41

(B)	**Elk**	**White-tailed deer**	**Mule deer**			
Accumulative AIC models <Δ2.1	**Model 1**	**Model 1**	**Model 1**	**Model 2**	**Model 3**	**Model 4**
Intercept	−3.12 ± 0.32	−1.39 ± 0.08	−1.82 ± 0.07	−1.82 ± 0.07	−1.82 ± 0.07	−1.82 ± 0.07
Grass	0.01 ± 0.005					
Shrub		0.006 ± 0.004	−0.006 ± 0.004	−0.006 ± 0.004	−0.006 ± 0.004	−0.006 ± 0.004
Intercept	−1.55 ± 0.31	−0.54 ± 0.15	0.54 ± 0.16	0.54 ± 0.16	0.54 ± 0.15	0.54 ± 0.15
Distance to major river	−1.54 ± 0.44					
Perennial forbs and grasses		0.44 ± 0.16			−0.05 ± 0.15	
Shrub		0.19 ± 0.16		−0.01 ± 0.16		0.04 ± 0.16
Tree	0.29 ± 0.29	0.55 ± 0.19	0.23 ± 0.17	0.24 ± 0.17		
Tree × Distance to major river	0.93 ± 0.45					
negLogLike	395.43	1220.83	1479.00	1479.00	1480.00	1480.03
K	6	6	4	5	4	4
AIC	802.86	2453.66	2965.99	2967.99	2967.99	2968.06
delta	0	0	0	2.00	2.00	2.07
AICwt	0.50	0.76	0.42	0.15	0.15	0.15
cumltvWt	0.50	0.76	0.42	0.57	0.73	0.88

Northern Great Plains, and a conservation network of riparian systems would benefit the movement of multiple mammal species (Fremier et al., 2015).

In addition to movement corridors, riparian areas can serve as seasonal habitat for many species, as evidenced by our breeding bird surveys. Intact riparian and mesic areas within a grassland or sage steppe matrix are of critical importance for sage grouse brood-rearing (Aldridge and Boyce, 2007; Donnelly et al., 2016), but are subject to damage via cattle grazing (Beck and Mitchell, 2000). Sage grouse conservation is of critical importance to multiple stakeholders in the western United States (Duvall et al., 2017), but habitat manipulations for their benefit do not always improve overall biodiversity measures (Carlisle et al., 2018). To this end, grazing with bison has the potential for positive conservation outcomes for sage grouse via reduced damage to wet habitats within a grassland or sage-steppe matrix.

Intact riparian systems in grasslands also buffer temperature extremes and may serve as thermal refugia during increasingly frequent and severe heat waves (Turunen et al., 2021). Increased native plant cover along stream banks increased stream stability in North Dakota grasslands (Hecker et al., 2019). Grassland streams with forested riparian buffers have increased abundance of aquatic insects (Wahl et al., 2013). A high diversity stream in a tall grass prairie system in Illinois saw some improvements (and no degradation) in stream quality following bison restoration (Vandermyde, 2017). In this way, year-round bison grazing in large pastures and with appropriate stocking rates, may facilitate increased climate resilience in grassland ecosystems.

Despite the potential ecological benefits of year-round bison grazing in comparison to seasonal cattle grazing, bison are not a singular solution to grassland conservation or restoration of the system. Bison constrained to small pastures or maintained at high stocking rates can certainly have negative effects on grassland biodiversity (Beschta et al., 2020) or individual species

TABLE 2 | Models describing the relationship between raw and fitted values of avian community diversity and remotely sensed vegetation metrics from the Rangeland Analysis Platform (Allred et al., 2021).

Biodiversity metric		**Estimate**	**Std. error**	***z* value**	***p***
Species richness (SR)	(Intercept)	−0.87	0.40	−2.18	0.03
	Tree	0.10	0.01	8.37	**<2.00 e-16**
	Shrub	0.06	0.01	4.76	**1.93 e-06**
	Perennial forb/grass	0.03	0.01	7.22	**5.12 e-13**
Phylogenetic diversity (PD)	(Intercept)	−321.36	151.38	−2.12	0.036
	Tree	33.81	6.13	5.52	**1.58 e-07**
	Shrub	15.82	5.30	2.98	**<0.01**
	Perennial forb/grass	7.63	1.77	4.30	**5.59 e-05**
Functional richness (FRic)	(Intercept)	−0.26	0.10	−2.65	0.01
	Tree	0.02	0.004	3.88	**1.62 e-04**
	Shrub	0.01	0.003	2.86	**<0.01**
	Perennial forb/grass	0.01	0.001	4.11	**6.69 e-05**

All effects in model summaries are in comparison to the 4th category of land cover, annual grass/forb. Predictors significant at the P < 0.05 level are bolded.

(Powell, 2006). Furthermore, upland bird communities are similar between seasonally grazed cattle and year-round bison (Boyce et al., 2021), meaning that exclusion of cattle from riparian zones within a grassland matrix may result in similar overall ecosystem health compared with bison pastures. True restoration efforts of prairie riparian areas must include restoration of beaver (*Castor canadensis*) as major engineers of riparian systems (Pollock et al., 1995; Hood and Bayley, 2008).

Domestic livestock grazing has negative biodiversity effects across trophic levels (Filazzola et al., 2020), but paradoxically, grassland ecosystems require disturbance from grazers to produce the diverse vegetative niches required for maximal biodiversity (Becerra et al., 2017). Bison appear to resolve this paradox due to reduced preference for riparian habitats and vegetation, along with increased heat and drought tolerance that facilitates grazing far from water resources (Allred et al., 2013; Kohl et al., 2013). The use of bison as a restoration tool may therefore be particularly effective as northern grasslands become increasingly hot and dry, such that their role as a keystone grazer not only supports biodiversity, but also increases resilience to climate change in one the world's most imperiled ecosystems.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

AB, HS, and WM formulated the idea. AB and HS conducted the fieldwork, performed the analyses, and wrote the first draft of the manuscript. All authors contributed to revisions of the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2022.821822/full#supplementary-material

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Exhibit 5

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Review Article

Holistic Management: Misinformation on the Science of Grazed Ecosystems

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Over 3 billion hectares of lands worldwide are grazed by livestock, with a majority suffering degradation in ecological condition. Losses in plant productivity, biodiversity of plant and animal communities, and carbon storage are occurring as a result of livestock grazing. Holistic management (HM) has been proposed as a means of restoring degraded deserts and grasslands and reversing climate change. Th fundamental approach of this system is based on frequently rotating livestock herds to mimic native ungulates reacting to predators in order to break up biological soil crusts and trample plants and soils to promote restoration. This review could fin no peer-reviewed studies that show that this management approach is superior to conventional grazing systems in outcomes. Any claims of success due to HM are likely due to the management aspects of goal setting, monitoring, and adapting to meet goals, not the ecological principles embodied in HM. Ecologically, the application of HM principles of trampling and intensive foraging are as detrimental to plants, soils, water storage, and plant productivity as are conventional grazing systems. Contrary to claims made that HM will reverse climate change, the scientifi evidence is that global greenhouse gas emissions are vastly larger than the capacity of worldwide grasslands and deserts to store the carbon emitted each year.

1. Introduction

Lands grazed by livestock include 3.4 billion ha worldwide with 73% estimated to be suff ring soil degradation [1]. The solution presented during Allan Savory's February 2013 TED Talk was to use holistic management (HM) to reverse desertific tion and climate change [2]. He reported that we are creating "too much bare ground" (1:30 in video) in the arid areas of the world and, as a consequence, rainfall runs off or evaporates, soils are damaged, and carbon is released back to the atmosphere. Grasslands, even in high rainfall areas, may contain large areas of bare ground with a crust of algae, leading to increased runoff and evaporation. Desertifi ation is caused by livestock, "overgrazing the plants, leaving the soil bare, and giving off methane" (4:20 in video).

HM is also called holistic resource management, time controlled grazing, Savory grazing method, or short-duration grazing. It is designed to mimic the behavior of grazing animals that are regulated by their predators to gather in large groups. As Savory puts it [2], "What we had failed to understand was that these seasonal humidity environments of the world, the soil and the vegetation developed with very large numbers of grazing animals, and that these grazing animals developed with ferocious pack-hunting predators. Now, the main defense against pack-hunting predators is to get into herds, and the larger the herd, the safer the individuals. Now, large herds dung and urinate all over their own food, and they have to keep moving, and it was that movement that prevented the overgrazing of plants, while the periodic trampling ensured good cover of the soil, as we see where a herd has passed." (9:28 in video). In view of the large amount of attention received from the TED talk it is important to examine the validity of Savory's claims.

Savory's writings lack specific that could be used for implementation of HM or for scientific testing. Details regarding setting of stocking rates, allowable use by livestock, amount of rest needed for recovery, or ecological criteria to be met for biodiversity, sustainability, wildlife, and watershed protection are absent [3–7]. The e publications by Savory and his colleagues show that HM is based on the following assumptions: (1) plant communities and soils of the arid, semiarid, and grassland systems of the world evolved in the presence of large herds of animals regulated by their predators; (2) grasses in these areas will become decadent and die out if not grazed by these large herds or their modern day equivalent, livestock; (3) rest from grazing by these large herds of livestock will result in grassland deterioration; (4) large herds are needed to break up decadent plant material and soil crusts and trample dung, urine, seeds, and plant material into the soil, promoting plant growth; and (5) high intensity grazing of these lands by livestock will reverse desertific tion and climate change by increasing production and cover of the soil, thereby storing more carbon.

We address these five assumptions of HM with a focus on western North American arid and semiarid ecosystems, principally in the desert, steppe, grassland, and open conifer woodland biomes as described by [8]. We use the broad term, grassland, to be inclusive of these types.

2. Are Western North American Ecosystems Adapted to Herds of Large Hooved Animals?

Not all of today's grasslands, arid, and semiarid systems evolved with herds of large, hooved animals. The Great Plains of North America and subtropical grasslands in Africa that receive moisture during the long, warm, and moist growing season historically supported millions of herbivores [9–11]. Lands west of the Continental Divide of the USA, including the Great Basin, Sonoran, Mojave, and Colorado Plateau deserts, along with the Palouse Prairie grasslands of eastern Washington, western Montana, and northern Idaho, did not evolve with signific nt grazing pressure from bison (*Bison bison*) [9, 12, 13]. Th ugh bison were abundant east of the Rockies on the Great Plains, they only occurred in limited numbers across western Wyoming, northeastern Utah, and southeastern Idaho [12, 14]. The e low numbers and patchy occurrence would not have played the same ecological role as in the plains. Historically, pronghorn antelope (*Antilocapra americana*) were more widespread than bison west of the Rockies, but these animals are smaller and lighter than bison and are not ecologically comparable [9]. Evidence for this general lack of large herds of grazing animals west of the Rockies also includes the lack of native dung beetles in the region. Whereas 34 native species of dung beetle (g. *Onthophagus*) are found east of the Rockies on the plains where bison were numerous, none are found west of the Rockies [15].

Th supposition that current western North American plant communities are adapted to livestock grazing because the region supported a diverse herbivore fauna during the Pleistocene epoch ignores that the plant communities have changed in the intervening time [16]. The e was rapid evolutionary change following the Pleistocene glaciations in North America with "the establishment of open xeric grasslands in the west central part of the USA . . . less than 10,000 years ago" [17]. Many of the grasses of the Pleistocene have disappeared from the plains and western USA and the fauna of the Pleistocene was altered by the arrival of bison from Eurasia. The e were destructive to long-leaved bunchgrasses found west of the Rockies, while the rhizomatous grasses found east of the Rockies in the prairies were more resistant to their grazing pressure. These rhizomatous grasses are the types found in the prairies of the central and western USA in conjunction with fossil remains of bison. In summary, the western USA of the Pleistocene is not the western USA of today. Th climate was much wetter and cooler and the vegetation more mesic in the Pleistocene than today [8, 18]. Th drier periods following the Pleistocene as temperatures warmed have altered soil conditions and fire cycles and contributed to the changing fl ra [19, 20].

Grasslands cover large areas that could support forests but are maintained by grasses outcompeting woody species for belowground resources and providing fuel for fi es that limit encroachment of woody species [21]. Climate change and increasing CO_2 concentrations have been implicated in the post industrial expansion of woody species and invasive species into grasslands; however, studies from paired locations in grasslands have shown that under similar climatic and CO_2 environments, herbivory by domestic livestock has caused the shift in woody species and increased invasive species [20, 22, 23]. Elevated CO_2 and climate warming appear to contribute to, but do not explain, the shrub encroachment in these semiarid areas which is due to intensive grazing by domestic livestock [24].

Conclusion. Western US ecosystems outside the prairies in which bison occurred are not adapted to the impact of large herds of livestock. Recent changes to these grassland ecosystems result from herbivory by domestic livestock which has altered fire cycles and promoted invasive species at the expense of native vegetation.

3. Do Grasses Senesce and Die If Not Grazed by Livestock?

A major premise of HM is that grass species depend on large grazing ungulates in some way, and thus grasses become moribund and die if not grazed, leading to deterioration or eventual loss of the entire grass community [6]. Th dead or dormant residual leaves and stalks that remain attached to ungrazed grasses at the end of each growing season can be deceptive. The plants are still alive and healthy, with living buds at the plant base. Bunchgrass canopies collect snow and funnel rainwater to the plant base and soil and increase infi tration [25]. The plants and dead leaves in contact with the soil reduce overland flow and erosion [26]. Plant canopies moderate temperature and protect the growing points from temperature extremes [27]. Th standing dead litter provides cover and food for wildlife species including large and small mammals, ground-living birds, and insects. Th loss of these leaves and stems through heavy grazing by livestock which occurs under HM destroys these natural attributes.

Grasses with attached dead leaves are more productive than grasses from which the dead leaves have been removed. Loss of these dead tissues to grazers increases thermal damage to the growing shoots and reduces the vigor of the entire plant [28]. Dead leaves and flowering stalks on ungrazed grasses inhibit livestock grazing, allowing those grasses to grow larger than their neighbors [29]. Grazing and trampling by domestic livestock damage plants in natural plant communities [30–32], reduce forage production as stocking rates increase [33], and can lead to simplification of plant communities, establishment of woody vegetation in grasslands, and regression to earlier successional stages [20] or conversion to invasive dominated communities [23] and altered fire cycles [20]. In contrast to the assertion that grasses will die if not grazed by livestock, bunchgrasses in arid environments are more likely to die if they are heavily grazed by domestic animals [34, 35].

Conclusion. Grasses, particularly bunchgrasses, have structure that protects growing points from damage, harvests water, and protects the soil at the plant base. Removal of the standing plant material exposes the growing points, leading to loss or replacement by grazing tolerant species, including invasives.

4. Does Rest Cause Grassland Deterioration?

Another principle of HM is that grasslands and their soils deteriorate from overrest, a term that implies insuffici t grazing by livestock. However, grasslands that have never been grazed by livestock have been found to support high cover of grasses and forbs. Relict sites throughout the western USA, such as on mesa tops, steep gorges, cliff sides, and even highway rights of way, which are inaccessible to livestock or most ungulates, can retain thriving bunchgrass communities [36–38]. For example, herbaceous growth was vigorous on never-grazed Jordan Valley kipukas in southeast Oregon [37] and on a once-grazed butte called The Island in south central Oregon [36]. Published comparisons of grazed and ungrazed lands in the western USA have found that rested sites have larger and more dense grasses, fewer weedy forbs and shrubs, higher biodiversity, higher productivity, less bare ground, and better water infiltration than nearby grazed sites. The e reports include 139 sites in south Dakota [39], as well as sites that had been rested for 18 years in Montana [40], 30 years in Nevada [41], 20–40 years in British Columbia [42], 45 years in Idaho [43], and 50 years in the Sonoran Desert of Arizona [44]. None of the above studies demonstrated that long periods of rest damaged native grasslands. A list and description of such sites can be found in [45].

Th HM misinterpretation of the natural history of grazed and ungrazed grasslands is apparent in Savory's description of the Appleton-Whittell Research Ranch in southeastern Arizona [6]. This ranch has been protected from livestock grazing since 1968 with the grasses on the ranch described by Savory as becoming "moribund" (page 211), with "bare spots opening up" (page 211). In contrast to those claims, plant species richness on the ranch increased from 22 species in 1969 to 49 species in 1984, while plant cover increased from 29% in 1968 to 85% in 1984 [46]. Total grass cover on the ranch was signific ntly higher on ungrazed sites when compared to grazed sites ($P < 0.01$) [47]. The e well designed studies produced quantitative data showing that the HM view of the ranch is not the case.

Conclusion. Contrary to the assumption that grasses will senesce and die if not grazed by livestock, studies of numerous relict sites, long-term rested sites, and paired grazed and ungrazed sites have demonstrated that native plant communities, particularly bunchgrasses, are sustained by rest from livestock grazing.

5. Is Hoof Action Necessary for Grassland Health?

A key premise of HM is that livestock can be made to emulate native ungulate responses to predators by moving them frequently in large numbers and tight groups. This promotes very close cropping which is said to benefit grasses and other forage, as well as hoof action that breaks up soil crusts, increases infi tration, plants seeds, and incorporates plant material, manure, and urine into the soil [6]. Other than bison in the plains states, the evidence indicates a low frequency of large hooved mammals in the western USA during pre-Columbian times [48], so the opportunity for hoof action to sustain grasslands and deserts appears limited at best. In contrast to HM claims, elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), and other ungulates may avoid areas where predators have an advantage in capturing them [49]. Avoidance is not the same as a panicked flight or tight groupings of animals promoting hoof action. Rather, the major response is greater vigilance and sometimes avoidance of risky areas. While the presence of wolves (*Canis lupus*) affects elk behavior by reducing browsing on willows and aspen [50], snow depth and other ecological needs appear to outweigh the effect of wolves leading to grazing and browsing in areas of higher risk [51, 52]. We found no documentation of native animal responses to predators generating hoof action or herd effect in tight groupings in the western USA.

Soils in arid and semiarid grasslands oft n have signific nt areas covered by biological crusts [53–55]. The e are made up of bacteria, cyanobacteria, algae, mosses, and lichens and are essential to the health of these grasslands. Biological crusts stabilize soils, increase soil organic matter and nutrient content, absorb dew during dry periods, and fi nitrogen [53, 56–60]. Crusts enhance soil stability and reduce water runoff by producing more microcatchments on soil surfaces. They increase water absorbing organic matter, improve nutrient flow, germination and establishment for some plants, while dark crusts may stimulate plant growth by producing warmer soil temperatures and water uptake in cold deserts [61]. Some crusts are hydrophobic, shedding water [60]. Biological soil crusts are fragile, highly susceptible to trampling [61–63], and are slow to recover from trampling impacts [64]. Loss of these crusts results in increased erosion and reduced soil fertility. The loss of crusts in the bunchgrass communities of the western USA may be largely responsible for the widespread establishment of cheatgrass and other exotic annuals [23, 58, 65]. The rapid spread of introduced weeds throughout the arid western USA is estimated at over 2000 hectares per day [66], largely due to livestock disturbance.

The HM assumption that increasing hoof action will increase infi tration has been disproven. Livestock grazing can compact soil, reduce infiltration, and increase runoff, erosion, and sediment yield [67–71]. Major increases in erosion and runoff occur under normal stocking when comparing grazed to ungrazed sites [68, 71–74]. Extensive literature reviews report the negative impacts of livestock grazing on soil stability and erosion [75–77]. For example, a study of wet and dry meadows in Oregon found the infiltration rate in ungrazed dry meadows was 13 times greater and 2.3 times greater in ungrazed wet meadows, compared to similar grazed meadows [78].

Hoof action is not needed to increase soil fertility and decomposition of litter. It is well-established that soil protozoa, arthropods, earthworms, microscopic bacteria, and fungi decompose plant and animal residues in all environments [79, 80]. Even the driest environments contain 100 million to one billion decomposing bacteria and tens to hundreds of meters of fungal hyphae per gram of soil [81]. Brady and Weil [80] discuss the importance of mammals in the decay process, mentioning burrowing mammals, but not large grazers such as cattle and bison. Removal of plant biomass and lowered production resulting from livestock grazing can reduce fertility and organic content of the soil [70, 82–84].

Conclusion. We found no evidence that hoof action as described by Savory occurs in the arid and semiarid grasslands of the western USA which lacked large herds of ungulates such as bison that occurred in the prairies of the USA or the savannahs of Africa. No benefits of hoof action were found. To the contrary, hoof action by livestock has been documented to destroy biological crusts, a key component in soil protection and nutrient cycling, thereby increasing erosion rates and reducing fertility, while, increasing soil compaction and reducing water infi tration.

6. Can Grazing Livestock Increase Carbon Storage and Reverse Climate Change?

Among the most recent HM claims is that livestock grazing will lead to sequestration of large amounts of carbon, thus potentially reversing climate change [2]. However, any increased carbon storage through livestock grazing must be weighed against the contribution of livestock metabolism to greenhouse gas emissions due to rumen bacteria methane emissions, manure, and fossil fuel use across the production chain [85, 86]. Nitrous oxide, 300 times more potent than methane in trapping greenhouse gases [87], is also produced and released with livestock production. The livestock industry's contribution to greenhouse gases also includes CO_2 released by conversion of forests to grasslands for the purpose of grazing [86].

Worldwide, livestock production accounts for about 37 percent of global anthropogenic methane emissions and 65 percent of anthropogenic nitrous oxide emissions with as much as 18% of current global greenhouse gas emissions (CO_2 equivalent) generated from the livestock industry [85]. It is estimated that livestock production, byproducts, and other externalities account for 29.5 billion metric tons of CO_2 per year or 51 percent of annual worldwide greenhouse gas emissions from agriculture [88]. Lower amounts of greenhouse gas emissions due to livestock may be estimated by using narrower defin tions of livestock-related emissions that include feed based emissions only and exclude externalities [89].

Some suggest that grass-fed beef is a superior alternative to beef produced in confine animal feeding operations [90]. However, grass provides less caloric energy per pound of feed than grain and, as a consequence, a grass-fed cow's rumen bacteria must work longer breaking down and digesting grass in order to extract the same energy content found in grain, while the bacteria in its rumen are emitting methane [89]. Comparisons of pasture-finished and feedlot-finished beef in the USA found that pasture-finishe beef produced 30% more greenhouse gas emissions on a live weight basis [91].

It is estimated that three times as much carbon resides in soil organic matter as in the atmosphere [92], while grasslands and shrublands have been estimated to store 30 percent of the world's soil carbon with additional amounts stored in the associated vegetation [93]. Long term intensive agriculture can significantly deplete soil organic carbon [94] and past livestock grazing in the United States has led to such losses [95, 96]. Livestock grazing was also found to signific ntly reduce carbon storage on Australian grazed lands while destocking currently grazed shrublands resulted in net carbon storage [97]. Livestock-grazed sites in Canyonlands National Park, Utah, had 20% less plant cover and 100% less soil carbon and nitrogen than areas grazed only by native herbivores [98]. Declines in soil carbon and nitrogen were found in grazed areas compared to ungrazed areas in sage steppe habitats in northeastern Utah [84]. As grazing intensity increased, mycorrhizal fungi at the litter/soil interface were destroyed by trampling, while ground cover, plant litter, and soil organic carbon and nitrogen decreased [84]. A review by Beschta et al. [20] determined that livestock grazing and trampling in the western USA led to a reduction in the ability of vegetation and soils to sequester carbon and also led to losses in stored carbon.

Conclusion. Livestock are a major source of greenhouse gas emissions. Livestock removal of plant biomass and altering of soil properties by trampling and erosion causes loss of carbon storage and nutrients as evidenced by studies in grazed and ungrazed areas.

7. What Is the Evidence That Holistic Management Does Not Produce the Claimed Effects?

HM is a management system that includes setting goals, monitoring, and adapting in order to continually move towards the goals established by the producer [6]. Thi more goal-oriented and adaptive management aspect of the HM system, its promise of environmental benefits, and increased production make it attractive to many ranchers [99]. However, researchers who have studied HM in South Africa and Zimbabwe, where Savory originated his theories, have rejected many of HM's underlying assumptions and found that HM approaches result in reduced water infiltration into the soil, increased erosion, reduced forage production, reduced soil organic matter and nitrogen, reduced mineral cycling, and increased soil bulk density [82, 100, 101].

In a recent evaluation of HM by Briske et al. [102], three of its principle claims were addressed: (1) all nonforested lands are degraded; (2) these lands can store all fossil fuel carbon in the atmosphere; and (3) intensive grazing is necessary to prevent the degradation. The authors pointed out that there are well managed lands that are not degraded; deserts are a consequence of climate and soils as well as improper management; and degradation is largely a function of growing populations of humans and livestock, land fragmentation, and other societal issues. As to the claim that these nonforested lands could store all the carbon emissions that humans produce, the researchers show that the potential carbon sequestration of these lands is only about one to two billion metric tons per year (mtpy), a small fraction of global carbon emissions of 50 billion mtpy. They further point out that these lands would have to produce much larger vegetation biomass than they are capable of producing in order to sequester human-caused carbon emissions and that much of the carbon is released back to the atmosphere through respiration as CO_2. They note that grass cover increases dramatically with rest and intensive grazing delays this recovery; many desert grassland soils are sandy, so hoof action does not increase infiltration; and biological crusts stabilize these soils and protect them from wind erosion and carbon loss.

A review of short-duration grazing studies in the western USA by Holechek et al. [83] included locations in the more arid western states as well as prairie types. Th researchers found that this grazing system, which is equated with HM, resulted in decreased infi tration, increased erosion, and reduced soil organic matter and nitrogen. Forage production and range condition were similar under short-duration and continuous grazing with the same stocking rates. Under short-duration grazing, standing crop of forage declined as stocking rates increased, while bare ground and vegetation composition were a function of stocking rate as opposed to grazing system. Grazing distribution was not improved over continuous grazing and the claims for hoof action and improved range condition under increased stocking rates and densities were not realized [83]. Another review of grazing systems by Briske et al. [103], including HM, versus continuous grazing concluded that plant and animal production were equal or greater in continuous grazing than in rotational grazing systems.

Even though the ecosystems of the Great Plains states evolved with the pressure of bison, Holechek et al. [83] and Briske et al. [103] found that HM did not diff r from traditional, season-long grazing for most dependent variables compared. Studies commonly held up as supporting HM [104–108] used HM paddocks that were grazed with light to moderate grazing, not the heavy grazing that Savory recommends. Further, long-term range studies have shown that it is reductions in stocking rate that lead to increased forage production and improvements in range condition, not grazing system [33, 109, 110]. While HM advocates allowing recovery to take place following grazing, recovery can take many years to decades even under total rest from livestock, but it does occur [43, 111]. Native, western USA bunchgrass species such as bluebunch wheatgrass (*Pseudoroegneria spicata*) and Idaho fescue (*Festuca idahoensis*) are sensitive to defoliation and can require long periods (years) of rest following each period of grazing in order to restore their vigor and productivity [34, 35].

Conclusion. Studies in Africa and the western USA, including the prairies which evolved in the presence of bison, show that HM, like conventional grazing systems, does not compensate for overstocking of livestock. As in conventional grazing systems, livestock managed under HM reduce water infiltration into the soil, increase soil erosion, reduce forage production, reduce range condition, reduce soil organic matter and nutrients, and increase soil bulk density. Application of HM cannot sequester much, let alone all the greenhouse gas emissions from human activities because the sequestration capacity of grazed lands is much less than annual greenhouse gas emissions.

8. What about Riparian Areas and Biodiversity?

In the western USA, riparian areas are rare and valuable ecological systems supporting a disproportionate number of species and providing many ecosystem services [112]. How does HM, with its emphasis on high stocking rates and trampling, aff ct these systems? Soil compaction from livestock is a common and widespread problem in grazed riparian areas, reducing infiltration rates and water storage and increasing surface runoff and soil erosion during storm events [78, 112]. Soil compaction from livestock increases with increased numbers, as in an HM application [70]. Livestock grazing in riparian areas reduces willow and herbaceous production and canopy cover of shrubs and grasses compared to ungrazed controls [113]. Th most effective way to restore damaged riparian areas is to remove livestock [110, 112].

We found very little information about total number of plant, animal, or invertebrate species present when HM is compared to other grazing methods or nongrazed areas, and, further, what proportion of total plant species or total cover of plant species was native or nonnative. Moreover, we did not see other biodiversity considerations addressed in any of the published studies investigating HM. Rotational grazing systems do not improve range condition and plant production over conventional grazing systems [83, 103], while stocking rate is considered the most important variable affecting vegetation production and range condition [33]. Range condition is determined based on the current plant community composition and production as compared to the potential natural community [114]. The relative composition of Increasers, those plants with tolerance for grazing, Decreasers, those plants with low tolerance for grazing, and Invasives, those plants occupying a site that are grazing tolerant and nonnative, forms the basis of the determination of condition. Higher range condition ratings reflect greater similarity to the native plant community for a site [115]. This basic concept reflects the biodiversity of the native plant community, which necessarily declines as range condition declines. Application of HM with its large herd size and density of use, like other grazing systems with high stocking rates, must necessarily decrease native plant diversity and productivity. This affects the animal communities accordingly as habitat structure and production are altered.

A review, by Fleischner [76], of the effects of livestock grazing on plant and animal communities in the western USA found that livestock grazing reduced species richness and abundance of plants, small mammals, birds, reptiles, insects, and fish compared to conditions following removal of livestock. A quantitative review by Jones [77] of published studies of ecosystem attributes in North American arid ecosystems affected by livestock grazing, compared to ungrazed conditions, found decreases in rodent species richness and diversity and vegetation diversity in the grazed areas. Livestock grazing-induced simplified plant communities in western USA arid and semiarid lands have negative effects on pollinators, birds, small mammals, amphibians, wild ungulates, and other native wildlife [20]. Riparian songbird abundance increases as riparian systems recover after livestock exclusion [116, 117], while overall biodiversity increases under long term rest from livestock grazing [46, 47, 118]. Invasives such as cheatgrass (*Bromus tectorum*) are favored and increase in abundance in the presence of livestock grazing [23, 65] and are inversely related to abundance of native perennial grasses [43].

Conclusion. HM does not address riparian areas and biodiversity with its focus on livestock production, although operators could choose these as goals. We have seen no studies of HM impacts on riparian areas and biodiversity, although livestock grazing impacts on riparian areas and biodiversity have been well documented. Livestock degrade riparian areas by removal of streamside vegetation, reduction of cover and food for fish and wildlife, and soil compaction, erosion, and sedimentation. These impacts lead to loss of native fish and wildlife populations. Studies in areas from which livestock have been removed demonstrate increases in diversity and abundance of birds, mammals, insects, and fish

9. Is Scientific Evidence Important?

Effectiveness studies of HM have been undertaken by ranchers and farmers who were selected because of their commitment to HM [119]. In other words, such studies were neither experimental nor were the participants randomly selected. Livestock producers who may have had negative experiences with HM were not included in the studies. Nearly all of the support and confirmation for HM come from articles developed at the Savory Institute or testimonials by practitioners. Most of the published literature that attempts to rigorously test HM in any scientific fashion does not support its principal assumptions.

Holechek et al. [83] stated that "No grazing approach, including that of Savory, will overcome the adverse effects of drought and/or chronic heavy stocking on forage production." These researchers were also critical of government agencies for adopting these unproven theories rather than basing management on "scientifically proven range management practices and principles" [83] (page 25). Briske et al. [103] stated,

> the rangeland profession has become mired in confusion, misinterpretation, and uncertainty with respect to the evaluation of grazing systems and the development of grazing recommendations and policy decisions. We contend that this has occurred because recommendations have traditionally been based on perception, personal experience, and anecdotal interpretations of management practices, rather than evidence-based assessments of ecosystem responses. [103] (page 11).

Briske et al. [102] state, "Mr. Savory's attempts to divide science and management perspectives and his aggressive promotion of a narrowly focused and widely challenged grazing method only serve to weaken global efforts to promote rangeland restoration and C sequestration." [102] (page 74).

Conclusion. Studies supporting HM have generally come from the Savory Institute or anecdotal accounts of HM practitioners. Leading range scientists have refuted the system and indicated that its adoption by land management agencies is based on these anecdotes and unproven principles rather than scientific evidence. When addressing the application of HM or any other grazing systems, practitioners, including agencies managing public lands, private livestock operations, and scientists, should (1) consider inclusion of watershed-scale ungrazed reference areas of suitable size to encompass the plant and soil communities found in the grazed area, (2) define ecological (plant, soil, and animal community) and production (livestock) criteria on which to base quantitative comparisons, (3) use sufficient replication in studies, (4) and include adequate quality control of methods. Economic analysis of grazing systems should compare all expenditures with income, including externalized costs such as soil loss, water pollution, reduction of water infiltration, and carbon emissions and capture.

10. Management Implications

This review shows that the underlying assumptions of HM regarding the evolutionary adaptation of western North

American landscapes to large herds of hooved animals only applies to prairie grasslands and that most arid and semiarid areas of western North America are not adapted to their impacts. Th premise that rest results in degradation of grassland ecosystems by allowing biological crusts to persist and grasses to senesce and die has been disproven by a large body of research. Reliance on hoof action to promote recovery by trampling seeds and organic matter into the soil and breaking up soil crusts needs to be considered in the context of increased soil compaction, lower infiltration rates, and the destruction of biological crusts that normally provide long-term stability to soil surfaces, enhance water retention, and promote nutrient cycling. The use of HM in an attempt to capture atmospheric greenhouse gases and incorporate them into soils and plant communities, thereby reducing climate change effects, is demonstrably impossible because the nonforested grazed lands of the world do not have the capacity to sequester this amount of emissions. Even in the prairie regions of the United States, which are evolutionarily adapted to large herbivores such as bison, research indicates that not only does HM not produce results superior to conventional season-long grazing, but also that stocking rate, rest, and livestock exclusion represent the best mechanisms for restoring grassland productivity, ecological condition, and sustainability. Various studies indicate livestock grazing reduces biodiversity of native species and degrades riparian areas, with nearly all studies findi g livestock exclusion to be the most effective, reliable means to restore degraded riparian areas. Claims of the benefits of HM or other grazing systems should be validated by quantitative, scientifically valid studies.

Conflict of Interests

The authors declare that there is no conflict of interests regarding the publication of this paper.

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Exhibit 6

50 YEARS WITH IMPACT Journal of Applied Ecology 1913 2013 British Ecological Society CELEBRATING 100 YEARS

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Conditions favouring *Bromus tectorum* dominance of endangered sagebrush steppe ecosystems

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Summary

1. Ecosystem invasibility is determined by combinations of environmental variables, invader attributes, disturbance regimes, competitive abilities of resident species and evolutionary history between residents and disturbance regimes. Understanding the relative importance of each factor is critical to limiting future invasions and restoring ecosystems.

2. We investigated factors potentially controlling *Bromus tectorum* invasions into *Artemisia tridentata* ssp. *wyomingensis* communities across 75 sites in the Great Basin. We measured soil texture, cattle grazing intensity, gaps among perennial plants and plant cover including *B. tectorum*, biological soil crusts (BSCs) and bare soil. Using *a priori* knowledge, we developed a multivariate hypothesis of the susceptibility of *Artemisia* ecosystems to *B. tectorum* invasion and used the model to assess the relative importance of the factors driving the magnitude of such invasions.

3. Model results imply that bunchgrass community structure, abundance and composition, along with BSC cover, play important roles in controlling *B. tectorum* dominance. Evidence suggests abundant bunchgrasses limit invasions by limiting the size and connectivity of gaps between vegetation, and BSCs appear to limit invasions within gaps. Results also suggest that cattle grazing reduces invasion resistance by decreasing bunchgrass abundance, shifting bunchgrass composition, and thereby increasing connectivity of gaps between perennial plants while trampling further reduces resistance by reducing BSC.

4. *Synthesis and applications*. Grazing exacerbates *Bromus tectorum* dominance in one of North America's most endangered ecosystems by adversely impacting key mechanisms mediating resistance to invasion. If the goal is to conserve and restore resistance of these systems, managers should consider maintaining or restoring: (i) high bunchgrass cover and structure characterized by spatially dispersed bunchgrasses and small gaps between them; (ii) a diverse assemblage of bunchgrass species to maximize competitive interactions with *B. tectorum* in time and space; and (iii) biological soil crusts to limit *B. tectorum* establishment. Passive restoration by reducing cumulative cattle grazing may be one of the most effective means of achieving these three goals.

Key-words: bare ground, biological soil crusts, cattle grazing, disturbance, diversity, invasion, plant gaps

Introduction

Ecosystem invasibility is governed by a complex collection of biotic and abiotic factors including environmental conditions, disturbance regimes and responses of native species to those regimes, as well as the biotic resistance provided by the resident community (Lonsdale 1999; Richardson & Pysek 2006). Biotic resistance is especially important in limiting the magnitude of invasive species after they have established (Levine, Adler & Yelenik 2004). Changes that increase resource availability are likely to increase susceptibility to invasion (Davis, Grime & Thompson 2000). Further, the introduction of an exotic herbivore with which resident species have no

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evolutionary history may exacerbate the magnitude of non-native plant invasions if it reduces the competitive abilities of native plants and increases resource availability (Parker, Burkepile & Hay 2006). Developing a predictive understanding of invasibility requires that we develop an understanding of how the various factors work together to limit invasion (Agrawal *et al*. 2007).

Artemisia tridentata big sagebrush ecosystems of the Intermountain West, USA, evolved with little herbivore pressure until the introduction of livestock (Mack & Thompson 1982). Within these ecosystems, lower elevation, more arid, *A. tridentata* ssp. *wyomingensis* (henceforth *Artemisia*) communities are the most common, but least resistant to invasion by exotic annual plants and least resilient to disturbance (Miller *et al*. 2011). Even in the absence of fire, these communities are especially vulnerable to invasions by *Bromus tectorum* L, and under some circumstances, *B. tectorum* can dominate the herbaceous understorey community (Miller *et al*. 2011). Previous studies have demonstrated the importance of several factors in the invasion process (soil texture, landscape orientation, competition-driven biotic resistance from native bunchgrasses and biological soil crust (BSC) communities (Table 1). Livestock grazing has been implicated in the spread and dominance of *B. tectorum* via several mechanisms (Mack & Thompson 1982; Table 1). Nonetheless, we have a poor understanding of how these factors work together and their relative importance in determining the magnitude of *B. tectorum* invasions (Miller *et al*. 2011).

Once *B. tectorum* sufficiently dominates the understorey and fills interspaces among plants, it creates a continuous, highly flammable fuel that significantly increases the risk of fire (Pyke 2011). Once a fire occurs, *B. tectorum* increases the frequency of fires. This change in fire regime may lead to a 'catastrophic regime shifts' (Scheffer *et al*. 2009), whereby native shrub–steppe communities are transformed into annual grasslands dominated by *B. tectorum* and other invasives (Miller *et al*. 2011). For practical purposes, these shifts are irreversible because of the significant investments necessary to restore these systems (Pyke 2011).

Preventing such regime shifts will require a better understanding of the simultaneous interacting factors that determine the magnitude of *B. tectorum* invasion once it has established in pre-fire *Artemisia* communities (Miller *et al*. 2011). Managers would benefit from understanding the causal network of mechanisms by which these factors interact with each other and how they collectively influence *B. tectorum* dominance. They would also benefit from an early warning indicator that the cumulative resistance of the resident community has been compromised to the point that *B. tectorum* likely dominates the understorey and thereby potentially setting the stage for a regime shift with the next fire.

Using *a priori* knowledge, we developed a multivariate hypothesis of the invasibility of *Artemisia* ecosystems to *B. tectorum* invasion in the absence of fire based upon the findings of previous studies in this system. The model included abiotic (soil physical properties, landscape orientation), cattle grazing disturbance and biotic factors (resident community abundance, composition and structure), predicted to be important determinants of *B. tectorum* dominance (Fig. 1, Table 1). Our analyses addressed the following questions: (i) What combination of abiotic and biotic conditions limit the magnitude of *B. tectorum* dominance? (ii) Can shifts in community structure, measured by the size and connectivity of gaps between native plants, serve as an indicator of susceptibility to *B. tectorum*

Table 1. Components of hypothesis represented by initial metamodel (Fig. 1)

Path	Hypothesized mechanism
1	(−) Cattle herbivory decreases *Bromus tectorum* abundance (Hempy-Mayer & Pyke 2009). (+) Cattle increase abundance by dispersing seeds and increasing propagule pressure (Schiffman 1997)
2	Cattle trampling decreases biological soil crusts cover and increases safe sites for *B. tectorum* establishment (Ponzetti, McCune & Pyke 2007)
3	Cattle herbivory decreases bunchgrass abundance (Briske & Richards 1995)
4	Cattle herbivory alters bunchgrass community composition by favouring more grazing-resistant species (Briske & Richards 1995)
5	Higher heat loads and spring insolation increase *B. tectorum* abundance (Stewart & Hull 1949; Chambers *et al*. 2007)
6	Lower heat loads increase bunchgrass productivity (Davies, Bates & Miller 2007)
7	Deeper, coarser-textured soils increase *B. tectorum* abundance (Stewart & Hull 1949)
8	Changes in bunchgrass composition influence community structure because species have different life forms (Grime 1977; James *et al*. 2008)
9	Changes in bunchgrass composition influence invasibility because species have different competitive abilities (Goldberg & Barton 1992) and patterns of resource use (James *et al*. 2008)
10	Bunchgrass abundance is inversely related to the size of and connectivity between gaps in perennial vegetation (Herrick *et al*. 2005)
11	Native bunchgrass abundance decreases *B. tectorum* abundance by reducing resource availability (Chambers *et al*. 2007)
12	Safe sites increase *B. tectorum* establishment rates (Fowler 1988).
13	Sagebrush abundance may increase *B. tectorum* abundance via facilitation (Griffith 2010) or decrease abundance via competition (Reichenberger & Pyke 1990).
14	Increases in the size of and connectivity between gaps in perennial vegetation increase *B. tectorum* abundance by increasing general resource availability (James *et al*. 2008; Okin *et al*. 2009)

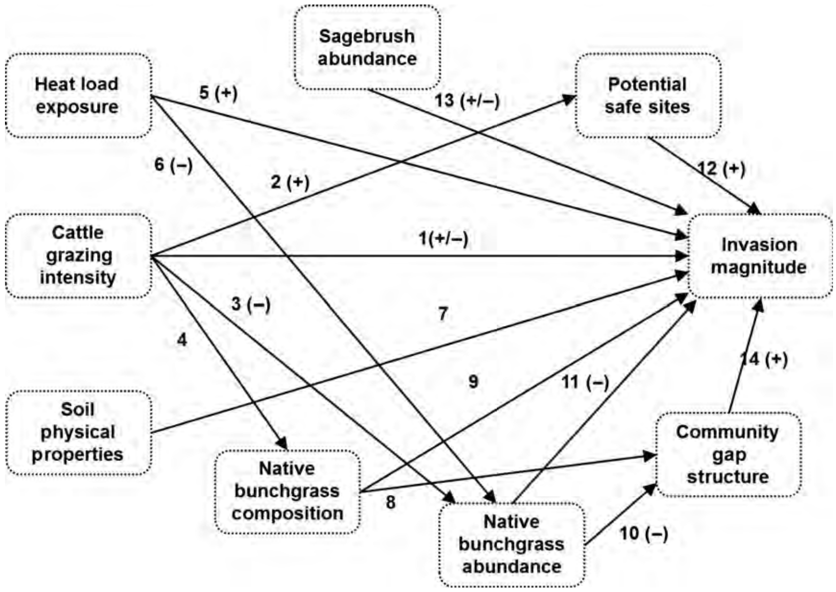

Fig. 1. Conceptual *a priori* multivariate model of *Artemisia* ecosystem susceptibility to *Bromus tectorum* dominance in the absence of fire. Dotted-line boxes represent conceptual variables hypothesized to influence invasibility. Components of the overall hypothesis are described in Table 1.

dominance and thereby vulnerability to a regime shift with the next fire?

We used structural equation modelling (SEM) to evaluate a multivariate hypothesis across 75 sites already invaded by *B. tectorum*. SEM provides a means of representing complex hypotheses about causal networks and testing for model data consistency (Grace 2006). It represents an advance over classical regression approaches (e.g. multiple regression) when used with observational data (Grace, Youngblood & Scheiner 2009). The advance provided by SEM comes partially from incorporating the associations among predictors into the overall hypothesis rather than to simply ignore or control for them. This is accomplished by extending the univariate model ($y = a + \beta x + \varepsilon$) to allow ys to depend on other ys and thereby represent networks of relationships in SEMs ($y = a + \beta x + \gamma y + \varepsilon$).

Because of this capability, SEM models can be used to specify hypotheses about mediating pathways and address questions, such as Can an association between A and C be explained by the factor B? This is achieved by evaluating a model such as $A \rightarrow B \rightarrow C$ and determining whether or not $r_{AC} = r_{AB} \cdot r_{BC}$. By assuming (and justifying based on prior information) that if A were manipulated, B could show a response, and similarly, if B were manipulated, C could show a response, then a test of the conditional independence of A and C in our example ($A \perp C | B$) could permit a result leading us to reject that possibility (A not independent of C when conditioned on B). SEMs thus build on causal assumptions to yield testable implications that can be evaluated with data. Estimated parameters obtained for a selected model then represent a set of predictions for further testing. The ultimate test of an SEM model is its ability to correctly predict future samples. For individual applications, the plausibility of causal assumptions (e.g. previous demonstrations that varying A can lead to response in B or known mechanisms whereby A can influence B) is often sufficient for reasonable inferences to be made.

Our results suggest that bunchgrass community structure, abundance and composition, and BSCs all play critical roles in limiting the magnitude of *B. tectorum* dominance. Cattle grazing may exacerbate the magnitude of invasion by reducing biotic resistance. Model evaluations imply that cattle grazing can reduce bunchgrass cover and shift bunchgrass community composition towards grazing-tolerant species and thereby increase the size and connectivity of gaps among perennial vegetation exacerbating invasion. Cattle trampling may also exacerbate invasion by reducing BSC cover. Ultimately, increases in the size and connectivity of gaps among native perennial vegetation may provide managers with an early warning indicator of increased susceptibility to *B. tectorum* dominance of *Artemisia* communities and thereby increased vulnerability to regime shifts in one of North America's most widespread but endangered ecosystems.

Materials and methods

STUDY AREA AND SAMPLING DESIGN

The study examined 75 *Artemisia* sites scattered across 4700 km^2 (roughly the size of state of Rhode Island) with elevations between 1265 and 1580 m across five *Artemisia*-dominated plant associations of the northern Great Basin floristic province of Oregon, USA. Natural Resource Conservation Service (NRCS) Ecological Site Descriptions and digital soil maps (http://websoilsurvey.nrcs.usda.gov) were used to ensure coverage of spatial variation in water stress driven by soil texture. Plant communities varied in dominant perennial tussock grasses and included the following ecological sites (ES): (i) loamy 254–308 mm precipitation zone (PZ) with *Pseudoroegneria spicata* and *Achnatherum thurberianum*; (ii) sandy loam 203–254 mm PZ with *Hesperostipa comata* and *P. spicata*; (iii) clayey 254–308 mm PZ with *A. thurberianum* and *Poa secunda*; and (iv and v) north slopes and south slopes 152–254 mm PZ with *P. spicata* and *A. thurberianum* co-dominating north and south slopes, respectively.

Each ES was delineated into three landscape substrata using 10-m resolution US Geological Survey Digital Elevation Models (DEM) to ensure variation in heat loads and water stress associated with changes in landscape orientation: (i) northerly aspects (0–90°, 270–360°), (ii) southerly aspects (90–270°) or (iii) flat. Study plots were located at different distances from the nearest livestock watering locations to capture variation in cattle grazing intensity. Random points were selected and field verified to ensure that plots were located: (i) every 200–400 m, starting at 100 m and extending to >3200 m from the nearest water; (ii) in as many soil–landscape strata combinations as possible; and (iii) >200 m from the nearest road to minimize related effects. To reduce potential confounding effects of time since fire, all sites burned since 1930 were excluded using a fire perimeter data base (http://sagemap.wr.usgs.gov accessed 17/3/2008).

SAMPLING

Thirty of the 0·39-ha study plots were sampled in 2008 and another 45 in 2009. Six 25-m transects were established in each plot using a spoke design, and herbaceous, shrub and BSC cover measured using line–point intercept (Herrick *et al.* 2005). All sampling occurred between 10 May and 15 July in both years to capture peak herbaceous biomass. Aspect and slope of each plot were calculated from DEM using Arc-GIS 13·0 and, with latitude, used to calculate potential heat loads for each plot (McCune 2007).

Potential variation in water stress was inferred by the following measurements: (i) soil texture at 0–15 cm soil depth; (ii) potential effective rooting depth, which was measured by digging a soil pit until bedrock, a restrictive layer (clay accumulation layer) or 2 m depth was reached; and (iii) amount and timing of precipitation for each study site derived from PRISM at 2-km^2 cell resolution (Daly *et al.* 2008). Sampling-year precipitation for all study plots was estimated for three seasons: 1 August to 31 October (fall), 1 November to 31 March (winter) and 1 April to 31 July (spring–summer).

Cattle grazing intensity was quantified by four measurements: field-verified distance from the nearest water; dung frequency and dung density from 12, 1 × 25 m belt transects; and bunchgrass (tussock) basal area. Basal circumference (C) of 30 randomly selected bunchgrasses was measured in each plot and used to calculate bunchgrass basal area (cm^2) using the following formula: Area = $\pi\ (C/2\pi)^2$.

Bare soil cover was calculated using line–point intercept data to represent exposed soil surface not covered by vegetation, visible BSC, dead vegetation, litter or rocks (Herrick *et al.* 2005). Soil surface aggregate stability was assessed in interspace microsites at 18 random sampling points along transects using soil from the upper 0–4 mm (Herrick *et al.* 2005). Two indicators of soil erosion resistance were calculated: mean soil stability and proportion of samples rated as extremely stable (Beever, Huso & Pyke 2006).

We assessed the structure of the native perennial community by quantifying the size of and connectivity of gaps between such vegetation using the basal gap intercept method (Herrick *et al.* 2005). We calculated mean gap length and the proportion of transects covered by large gaps (>200 cm in length).

MULTIVARIATE ANALYSIS

Species cover, distance from nearest water, dung density, bunchgrass basal area, heat loads, soil depth, precipitation, gap size and herbaceous biomass data were log-transformed to improve distributional properties, correlations with ordination axes and variation explained by ordinations (McCune & Grace 2002). Other variables were not transformed.

Non-metric multidimensional scaling (NMS) ordination was used to relate patterns in community composition to environmental gradients (PC-Ord™; McCune & Grace 2002). Joint plots and Pierson's correlations were used to describe relationships between environmental gradients and the strongest patterns of community composition.

We used nonparametric multiplicative regression (NPMR) in HyperNiche™ to quantify the relationship between species' cover and environmental gradients (McCune 2009). Predictors were scores of the three ordination axes. These scores represented an integrated measure of complex environmental gradients associated with dominant patterns of herbaceous community composition. Response variables were the cover of each species using a local mean estimator and Gaussian kernel function. To control for potential interactions between axes, response curves were generated using partial models and focal variables (McCune 2009). A final NPMR model was run using the three axes' scores as predictors. Final model fit was assessed with a cross-validated R^2 (McCune 2009).

Hierarchical agglomerative cluster analysis was used to identify groups of sites differing in community composition (McCune & Grace 2002). Multivariate differences in community composition between identified groups were tested using multiresponse permutation procedures (MRPP) ($\alpha = 0{\cdot}05$). Identified groups were overlaid onto ordinations to accentuate relationships between groups and environmental gradients. Multivariate differences in relativized environmental variables between groups were tested with MRPP. Differences in individual environmental variables between groups were assessed with ANOVA ($\alpha = 0{\cdot}10$), and Bonferroni-adjusted 90% confidence intervals were used to quantify differences between groups.

STRUCTURAL EQUATION MODELLING

In our study, an initial conceptual model was used as a SEM meta-model, representing a family of possible models (Fig. 1, Table 1). Our modelling process considered the available observed variables to identify 'indicator variables' (the observed variables that will serve as proxies for conceptual variables in the meta-SEM) using procedures described in Grace *et al.* (2012). Except for 'Potential Safe Sites', all model constructs were represented using single indicator variables. *Bromus tectorum* cover was selected as the indicator to measure 'Invasion Magnitude'. Bunchgrass and sagebrush cover were selected to measure their abundances. The three NMS ordination axes of bunchgrass species' cover data were used to develop an indicator of 'Bunchgrass Community Composition'. Distance from nearest water was selected as the indicator to measure cumulative 'Cattle Grazing Intensity'. Heat load was selected to measure 'Heat load Exposure', and percent sand content at 0–15 soil depth was selected to measure 'Soil Physical Properties'. The proportion of transects covered by large gaps (>200 cm in length) was selected as the measure of 'Community Gap Structure'. Two indicators were selected to represent 'Potential Safe Sites' – BSC and cover of bare soil.

All SEM analyses were conducted using AMOS 18.0 software (SPSS 2010). Maximum likelihood procedures were used for model evaluation and parameter estimation. Model fit was evaluated by sequentially evaluating likelihood ratios by using the single-degree-of-freedom chi-squared goodness-of-fit statistic. Modification indices were used to evaluate the need to include links or error correlations not in the original model. This process produced a final inferential model. The stability of the final model was evaluated by introducing other available indicators to determine whether they represented additional contributing information. For example, our initial indicator for cattle grazing intensity was 'distance from nearest water'. The three alternative potential indicators (cow pie frequency, cow pie density and bunchgrass basal area) for this construct did not improve model fit or amount of variation in cheatgrass dominance explained and were no longer included.

Results

PATTERNS OF INVASIBILITY – CONVENTIONAL MULTIVARIATE RESULTS

Nearly 92% of variation in community composition was explained by the final ordination (Fig. 2). Axis 1 was the dominant axis explaining 60·9% of variation in composition data. Axis 1 was a strong gradient of decreasing cattle grazing disturbance and heat stress (Fig. 2): dung density (r = −0·35); dung frequency (r = −0·36); distance from water (r = 0·41); deep-rooted bunchgrass basal area (r = 0·71); and heat loads (r = −0·44). In addition, BSC cover, soil aggregate stability and proportion of soil aggregate stability values rated as highly stable increased along Axis 1 (Fig. 2). The size of and connectivity between gaps and amount of bare soil decreased strongly

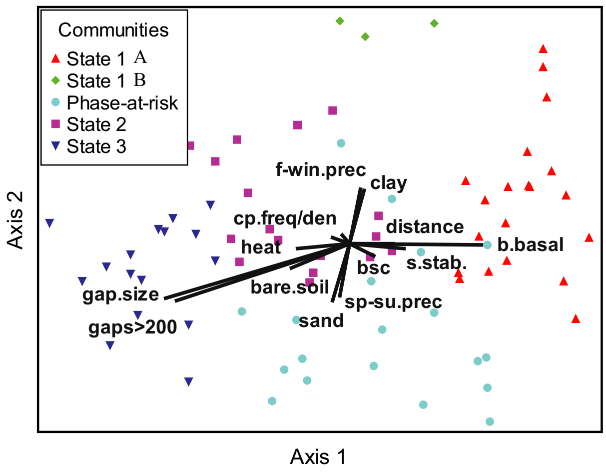

Fig. 2. Ordination of plots in community composition space. Non-metric multidimensional scaling ordination with final stress of 9·92; final instability of <0·01; Monte Carlo test P-value < 0·05. Vectors show the strength and direction of correlations between environmental variables and axes. Only variables with significant R^2 (>0·20) are shown. Different plot symbols show groups derived from cluster analysis that differ in composition and environmental factors. State 1A and 1B communities have understoreys dominated by native bunchgrasses; phase-at-risk communities are co-dominated by bunchgrasses and non-natives, and State 2 and State 3 communities are dominated by non-native species.

along Axes 1 and 2 (Fig. 2) (see Table S1, Supporting Information).

Axes 2 and 3 represented weaker relationships explaining 19·3% and 11·6% of the variation, respectively. Axis 2 showed a strong gradient of decreasing sand, increasing clay and increasing fall and winter precipitation (Fig. 2). Axis 3 demonstrated a weaker gradient of decreasing cattle grazing associated with decreasing dung density and frequency and increasing deep-rooted bunchgrass basal area (see Table S1, Supporting Information).

Nonparametric multiplicative regression model sensitivities indicate that Axis 1 was the best predictor of non-native species. The strength of the relationship between cover of native species and these three axes varied considerably (Fig. 3; see Table S2, Supporting Information). *P. spicata*, *A. thurberianum*, *P. secunda* and forbs had strong positive relationships with Axis 1, *P. secunda* and forbs had strong positive relationships with Axis 2, and *Elymus elymoides* had a strong positive relationship with Axis 3 (Fig. 3).

Cluster analysis identified five distinct groups of communities with 0% of the information remaining (MRPP using species data: A = 0·33, P < 0·01; Fig. 4; See Tables S3 and S4, Supporting Information). Several species were uniquely associated with one or more groups (Fig. 4; See Table S3, Supporting Information). Combined heat loads, soil physical properties, BSC cover, bare soil cover, soil stability, community gap structure and cattle grazing intensity differed significantly among groups (MRPP using environmental data: A = 0·59, P < 0·0001; Fig. 5; See Table S4, Supporting Information).

State 1 consisted of two groups (1A and 1B) of communities with an intact herbaceous understorey dominated by native bunchgrasses and forbs (Fig. 4). Thirty-one percentage of study plots were in one of these groups. State 1 also contained phase-at-risk communities (communities at risk of crossing a biological threshold to being dominated by *B. tectorum*; 25% of study plots) with an understorey co-dominated by native species and *B. tectorum*. States 2 (23% of study plots) and 3 (21% of study plots) consisted of communities that have crossed a biological threshold and had understories dominated by *B. tectorum* and the non-native annual forb, *Lepidium perfoliatum*.

Communities in Groups 1A and 1B had the lowest levels of cattle grazing combined with the smallest and least connected gaps between perennial vegetation (Fig. 5). Group 1B communities had higher heat loads and finer-textured soils compared to those of Group 1A. Communities comprising phase-at-risk communities were characterized by intermediate levels of cattle grazing, heat loads, water stress and size of and connectivity between gaps (Fig. 5).

State 2 communities were characterized by intermediate to high levels of cattle grazing and intermediate levels of heat loads and water stress. State 3 communities had the highest levels of cattle grazing and bare soil cover, largest and most connected gaps and lowest soil aggregate stability (Fig. 5).

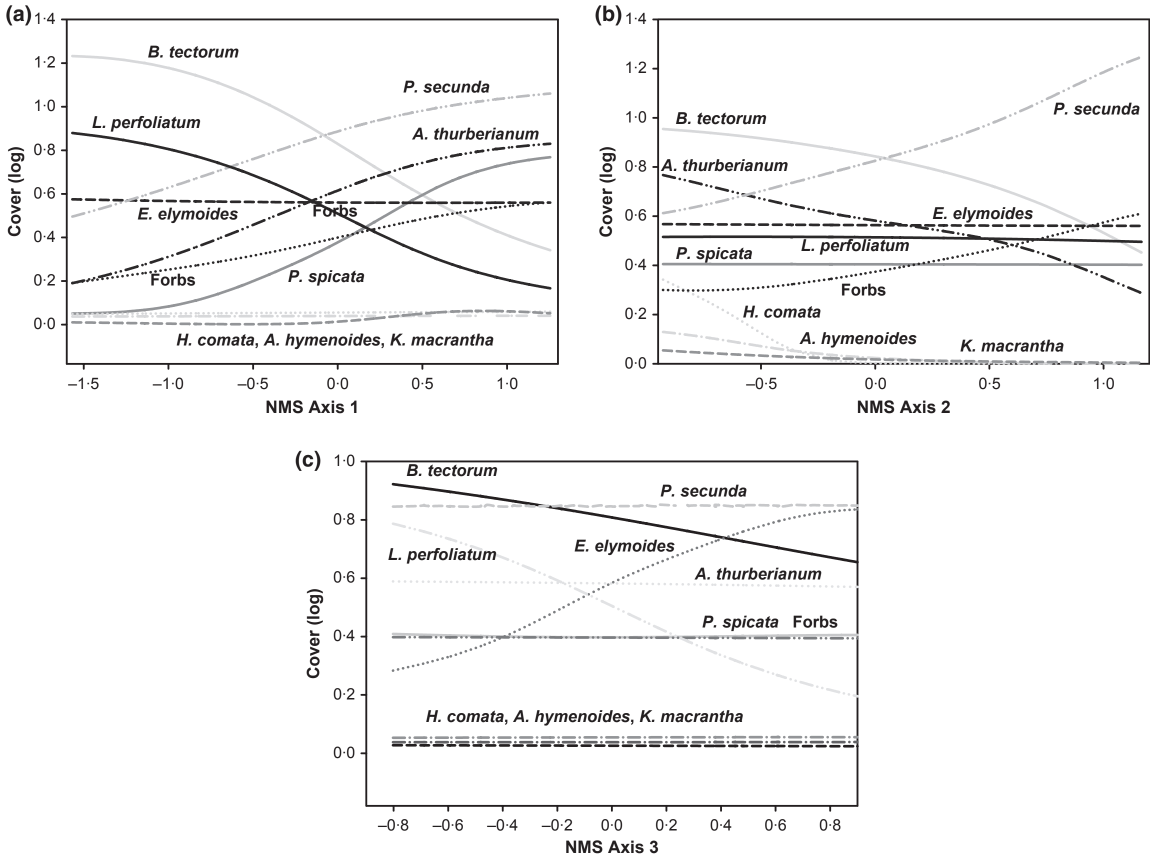

Fig. 3. Nonparametric multiplicative regression response curves showing relationship between species cover and gradients represented by non-metric multidimensional scaling ordination axes. Axis 1 is a gradient of decreasing cattle grazing intensity and heat load exposure (a), Axis 2 is a gradient of decreasing water stress (b), and Axis 3 is a gradient of decreasing cattle intensity (c).

STRUCTURAL EQUATION MODELLING RESULTS

The final SEM model ($\chi^2 = 18{\cdot}88$; $P = 0{\cdot}54$; 20 d.f.) showed very close fit between model and data. A number of the initially hypothesized relationships (Table 1) were not supported by data. Sagebrush abundance did not help explain invasion magnitude, either directly or indirectly. As a result, that variable was removed from the final model. Heat load exposure, cattle grazing intensity and native bunchgrass cover were indirect predictors of invasion magnitude in the final model (Fig. 6). Unanticipated in the initial model was dependence of safe sites on heat loads and sand, and dependence of bunchgrass composition on sand content. The final model explained 72% of the variation in the magnitude of invasions among sites.

Concerning strengths of linkages in the final model, changes in community gap structure, that is, increases in the size of and connectivity between gaps among native plants, were predictive of higher levels of *B. tectorum* cover ($r = 0{\cdot}83$). Native bunchgrass cover and composition were not direct predictors of *B. tectorum* cover, rather they were indirect predictors through their relationship to gaps. Gaps characterized by bare soil had a strong positive association with *B. tectorum* cover ($r = 0{\cdot}38$), whereas gaps characterized by BSC cover had a strong negative association with *B. tectorum* cover ($r = -0{\cdot}26$).

Cattle grazing intensity was positively associated with *B. tectorum* cover through three independent pathways. Because distance from water is inversely related to cattle grazing levels, positive path coefficients indicate a negative relationship between cattle grazing and the response variable in the model (Fig. 6). Thus, model results imply that pathways from cattle grazing to *B. tectorum* cover propagate through (i) negative influences on bunchgrass abundance (0·34), (ii) negative influences on BSC abundance (0·29) and (iii) impacts on bunchgrass community composition (Axis 2) (0·22). There was no evidence that cattle grazing directly decreased or increased *B. tectorum* cover independent of these stated routes.

High levels of heat load exposure were associated with lower levels of bunchgrass (−0·46) and BSC (−0·36) abundance. Coarser-textured soils were more likely to have higher levels of *B. tectorum*, regardless of the other factors (i.e. a direct linkage of 0·48). Coarser-textured soils also had an indirect path through effects on bare soil cover and bunchgrass community composition (Axis 3) that increased *B. tectorum* cover.

By adding up the path strengths, it is possible to compute what is referred to as 'total effects' of predictors on

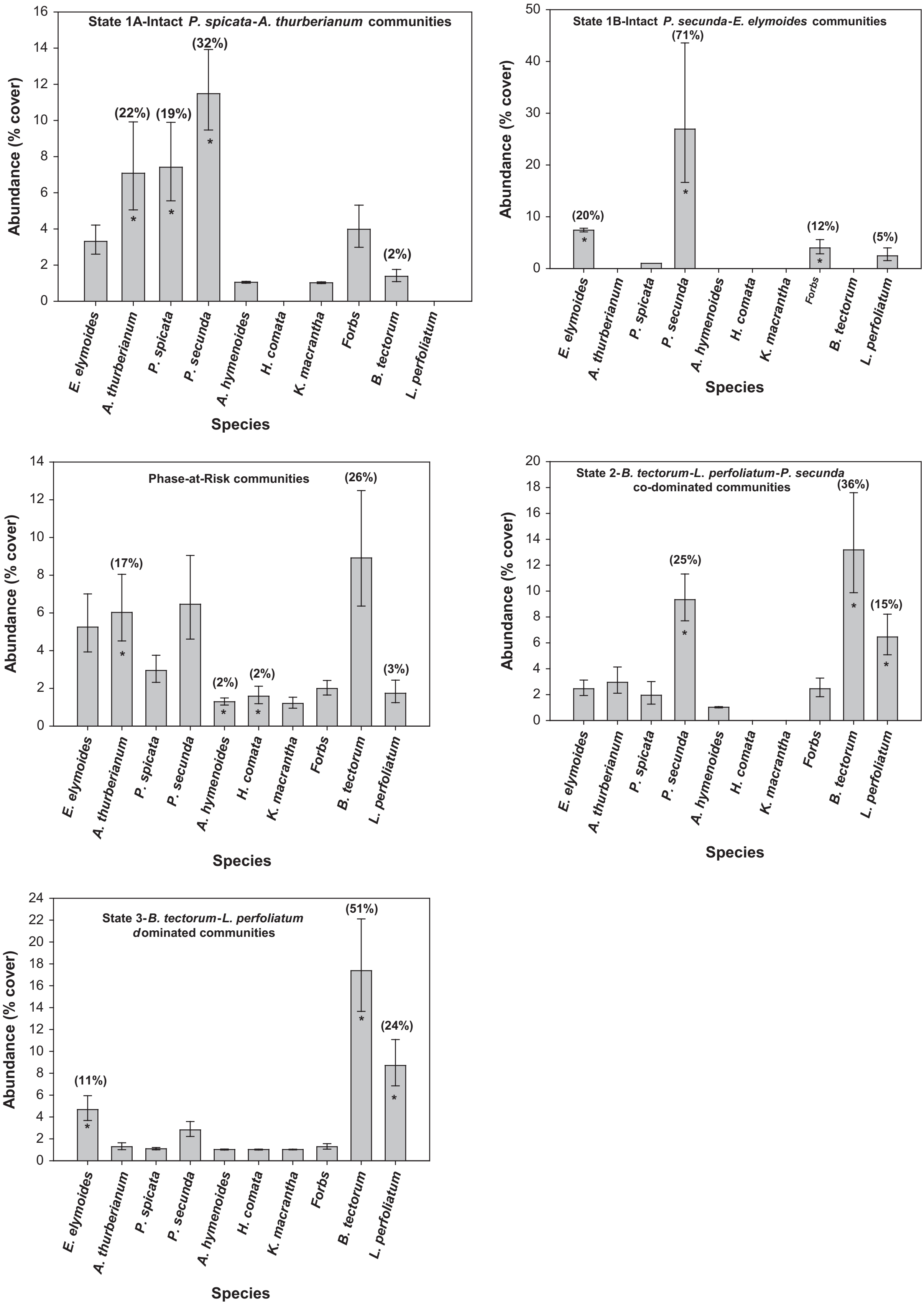

Fig. 4. Community composition of five groups derived from cluster analysis. *Denotes species with highest three indicator values for the group from indicator species analysis. Reported values are back-transformed means, and error bars are 90% Bonferroni-adjusted confidence intervals. (%) is the relative abundance of the species calculated as the proportion of total cover of the group.

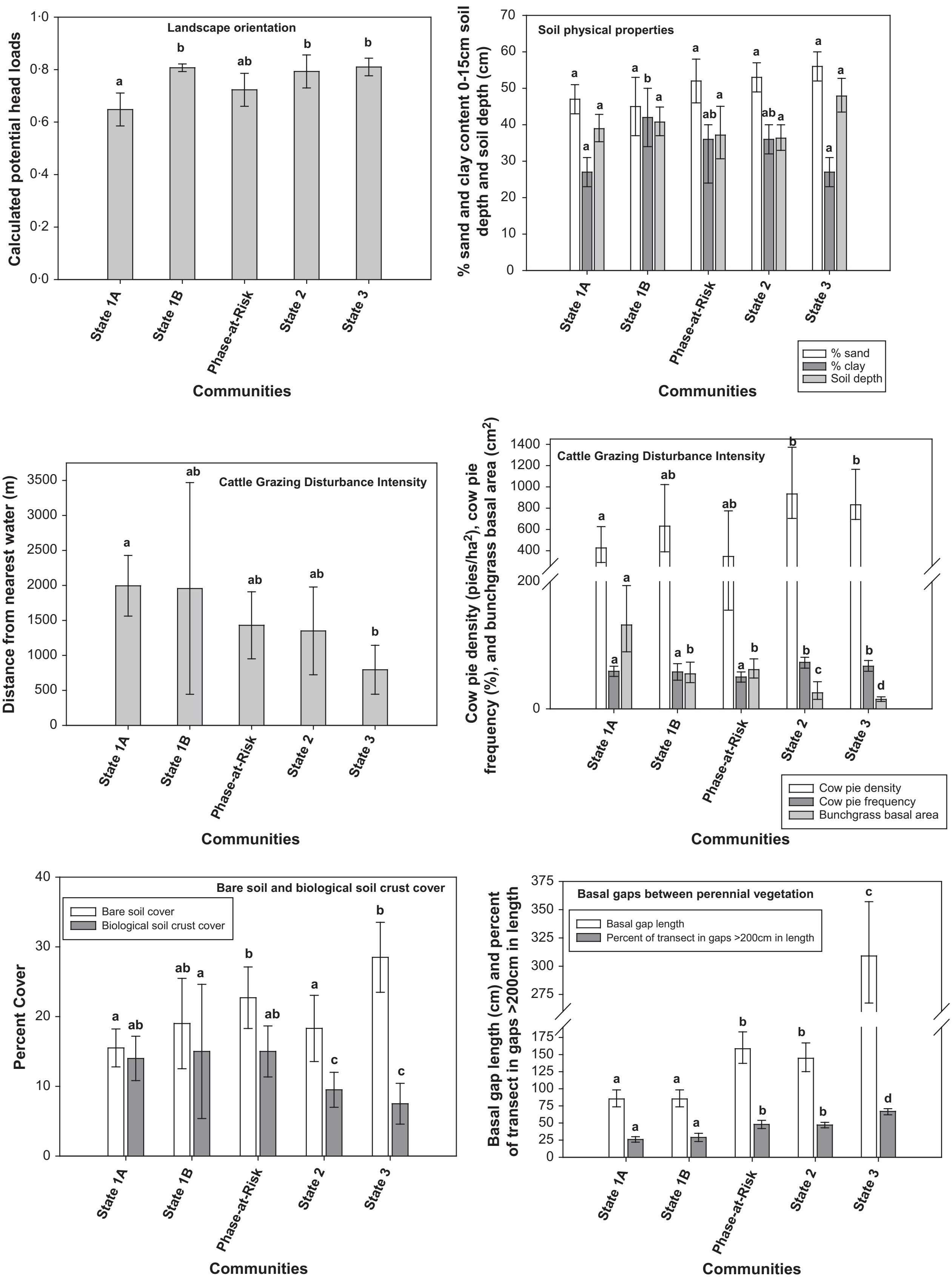

Fig. 5. Differences in heat loads, soil physical properties, biological soil crusts, bare soil cover, soil stability, community gap structure and cattle grazing intensity of five groups identified by cluster analysis. Error bars represent Bonferroni-adjusted 90% confidence intervals. Different lower-case letters above bars indicate significant differences between groups (α = 0·10).

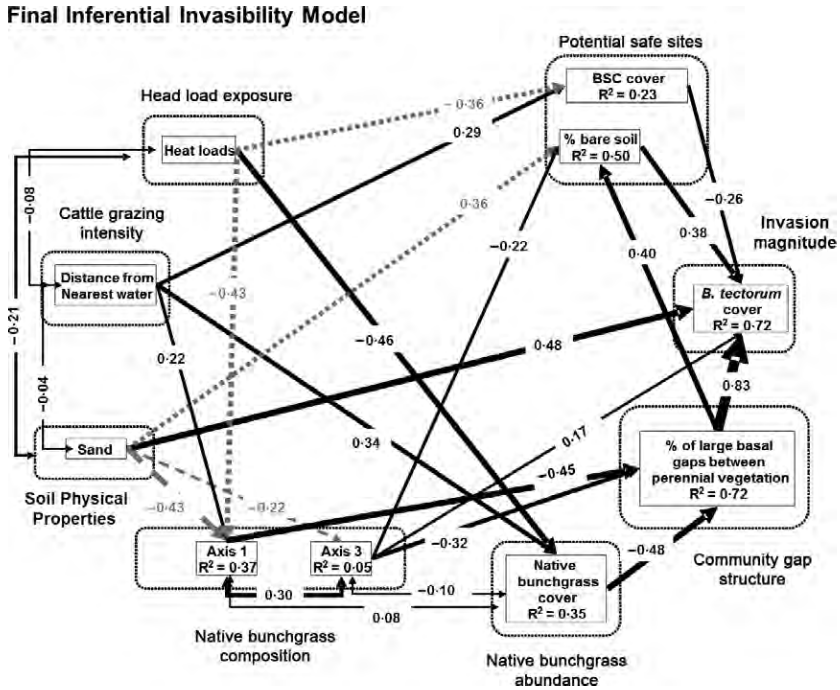

Fig. 6. Final inferential model of *Artemisia* ecosystem invasibility. Single-headed arrows represent significant linkages (α = 0·05). Double-headed arrows indicate significant correlations between variables and their residuals. The magnitudes of standardized path coefficients are represented by line thicknesses. Dotted grey lines are unanticipated significant paths. Because distance from water is inversely related to cattle grazing intensity (i.e. grazing intensity increases with decreasing distance), positive path coefficients and correlations between grazing intensity and variables indicate an inverse relationship (i.e. increasing cattle grazing intensity decreases bunchgrass abundance). R^2 values depict the proportion of variation of endogenous variables explained by the model. The dotted boxes depict conceptual variables of the meta-structural equation modelling (Fig. 1).

downstream responses. Computed total effects of predictor variables on *B. tectorum* in order of importance were as follows: (i) community gap structure (0·68), (ii) soil physical properties (0·42), (iii) safe sites (bare soil cover, 0·38), (iv) heat load exposure (0·37), bunchgrass community composition (NMS#1, −0·31), (v) safe sites (BSC cover, −0·26), (vi) cattle grazing disturbance (−0·26), (vii) bunchgrass abundance (−0·24) and (viii) bunchgrass community composition (NMS#3, 0·04).

Discussion

By combining SEM with an observational approach, we were able to gain important new insight into the relative importance of the numerous factors determining the magnitude of *B. tectorum* invasions of *Artemisia* ecosystems and gain valuable insight into potential underlying mechanisms. Our results provide strong support for some *a priori* hypothesized mechanisms (i.e. cattle trampling reduces bunchgrass and BSC abundance) and no support for others (i.e. cattle reduce invasions by grazing *B. tectorum*). Our SEM findings should help prioritize future experiments to test our inferences regarding underlying mechanisms and more landscape-scale observational studies to further evaluate and refine the model and construct predictive models (Grace 2006).

In this study, model results support the idea that a complex causal network of simultaneously operating factors and mechanisms are driving invasion of *B. tectorum* in *Artemisia* ecosystems. Based upon the SEM results, shifts in community structure, as measured by the size of and connectivity of gaps between native plants, exert a strong positive effect on the magnitude of *B. tectorum* invasion. This finding is consistent with growing evidence in semi-arid and arid ecosystems showing that increases in gap connectivity (Busso & Bonvissuto 2009; Okin *et al.* 2009) and changes in how species abundance is distributed in a community (James *et al.* 2008) are associated with a loss of ecosystem resistance to invasion (Scheffer *et al.* 2009). We define resistance as the collective ability of the resident sagebrush community to limit *B. tectorum* dominance in the face of invasion (Chambers *et al.* 2007). Increases in the connectivity of these gaps were associated with a dramatic increase in the magnitude of such invasions (Okin *et al.* 2009). This loss of resistance to invasion probably increases the magnitude of *B. tectorum* dominance after subsequent disturbances and may set the stage for a regime shift to *B. tectorum*-dominated grasslands with the next fire (Scheffer *et al.* 2009).

Our research suggests that two environmental factors influence the inherent resistance of *Artemisia* ecosystems to *B. tectorum* invasion. Communities located on coarser-textured soils or characterized by higher potential heat loads (Stewart & Hull 1949) were inherently least resistant to *B. tectorum* invasion. These communities are characterized by higher levels of water stress and lower productivity. The inherent structure of these communities that consists of larger and more connected gaps among perennial vegetation and higher amounts of bare soil may make them vulnerable to other disturbances that increase the size of gaps.

Consistent with other studies, biotic resistance from resident bunchgrass and BSC communities played pivotal roles and appears to limit the magnitude of *B. tectorum* invasion (Richardson & Pysek 2006). Water availability is the primary controlling factor of seedling establishment in these ecosystems (Schupp 1995). Several studies have found a strong negative association between BSC community integrity and *B. tectorum* abundance (Ponzetti, McCune & Pyke

2007; Ponzetti & McCune 2008) and showed that BSCs reduce *B. tectorum* germination and establishment rates by impeding root penetration and growth (Serpe *et al.* 2008). Our findings suggest that BSC communities are especially important in limiting the magnitude of *B. tectorum* invasions in gaps between perennial vegetation by minimizing potential safe sites for establishment.

Consistent with other studies, we found that bunchgrasses reduced the magnitude of *B. tectorum* invasions most likely by reducing water and nutrient availability (Chambers *et al.* 2007; Prevéy *et al.* 2010). Our findings provide important insight into this mechanism. Nearly all the biotic resistance effect was indirect through a strong direct effect of bunchgrass abundance and composition on community structure. By limiting the size and connectivity of gaps, bunchgrasses likely minimize resources available to *B. tectorum* spatially. Further, three species, *P. spicata*, *A. thurberianum* and *P. secunda,* appear to be especially important determinants of such resistance. *P. spicata* and *A. thurberianum* are dominant deep-rooted bunchgrasses with most active growth in later spring, whereas *P. secunda* is a shallow-rooted bunchgrass that is active in late winter and early spring. This combination of differing structure and phenology reflects their differing abilities to acquire resources at different soil depths (James *et al.* 2008) and seasons and thereby provide continuous interaction with *B. tectorum* and collectively limit available resources temporally and at different soil depths.

By controlling for several potentially confounding factors (Knick *et al.* 2011), we gained important insights into the role of cattle grazing as a determinant of ecosystem resistance to *B. tectorum* invasion. We found no evidence that cattle grazing, even at the high intensities 100 m from the nearest water development, reduced *B. tectorum* cover. To the contrary, we found strong evidence that increasing cattle grazing intensity indirectly promotes an increase in the magnitude of *B. tectorum* dominance. Cattle herbivory was found to be associated with reduced native bunchgrass abundance, shifts in bunchgrass composition to only the most grazing-tolerant species and aggregated bunchgrasses beneath protective sagebrush canopies (Reisner 2010). These collective cattle-induced changes thus appear to ripple through the community by increasing the size and connectivity of gaps between perennial vegetation. As gaps get bigger and more connected, both live and dead (litter) herbaceous soil cover decreases and the amount of bare soil increases. Cattle trampling reduced resistance within these larger gaps by reducing BSC cover.

Changes in community structure and how species' abundance is distributed in the community may increase general resource availability (James *et al.* 2008). As cattle grazing increased, *P. spicata*, *A. thurberianum* and *P. secunda* cover decreased, *E. elymoides* cover did not change, and *B. tectorum* cover increased. These shifts parallel the relative differences in grazing avoidance and tolerance mechanisms among these species. Cattle grazing introduced a novel disturbance regime into this system where most bunchgrasses are highly sensitive to herbivory (Mack & Thompson 1982). To the contrary, *B. tectorum* exhibits a collection of grazing avoidance and tolerance attributes that makes it extremely tolerant of even highly intensive grazing (Vallentine & Stevens 1994; Hempy-Mayer & Pyke 2009). Because of its attributes (Chambers *et al.* 2007), *B. tectorum* is well positioned to take maximum advantage of this window of invasion opportunity by exploiting larger and more connected gaps.

If the goal is to conserve and restore resistance of these systems to invasion, managers should consider focusing their efforts on maximizing the pre-emption of resources provided by BSC and bunchgrasses. We suggest three priorities: first, maintain and/or restore high overall bunchgrass cover and community structure characterized by spatially dispersed bunchgrasses in interspaces and small gaps between such individuals to maximize the capture of resources; second, maintain and/or restore a diverse assemblage of bunchgrass species with different spatial and temporal patterns of resource use to maximize capture of resources at different soil depths and times; third, maintain and/or restore a BSC community to limit safe sites for *B. tectorum* establishment within gaps.

Our findings suggest that multiple factors (bunchgrass cover, BSC cover, cattle grazing, etc.) may influence the susceptibility of these ecosystems to *B. tectorum* invasion. Importantly, many of these influences are mediated by the size and connectivity of gaps, as well as the conditions of gaps. Thus, gaps in perennial vegetation may serve as an important early warning indicator of when cattle grazing or other stressors are compromising resistance of these systems to *B. tectorum* invasion. Our findings raise serious concerns regarding proposals to use cattle grazing to control *B. tectorum* in these systems where remnant bunchgrass communities persist (Vallentine & Stevens 1994). In contrast, our findings support recent guidance for passively restoring resistance of these systems by reducing grazing levels (Pyke 2011). Future research should focus on gathering information concerning the size of and connectively of such gaps across a range of ES consistent with maintaining resistance. These data could be used to develop indicators for adaptive management frameworks to conserve and restore these endangered systems.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Table S1. Relationships between environmental variables and ordination axes.

Table S2. Relationships between species abundance and ordination axes.

Table S3. Indicator species analysis of groups.

Table S4. Pairwise MRPP comparisons of groups.

Exhibit 7

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Livestock Grazing Effects on Fuel Loads for Wildland Fire in Sagebrush Dominated Ecosystems

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Abstract

Herbivory and fire are natural interacting forces contributing to the maintenance of rangeland ecosystems. Wildfires in the sagebrush dominated ecosystems of the Great Basin are becoming larger and more frequent, and may dramatically alter plant communities and habitat. This synthesis describes what is currently known about the cumulative impacts of historic livestock grazing patterns and short-term effects of livestock grazing on fuels and fire in sagebrush ecosystems. Over years and decades grazing can alter fuel characteristics of ecosystems. On a yearly basis, grazing can reduce the amount and alter the continuity of fine fuels, potentially changing wildlife fire spread and intensity. However, how grazing-induced fuel alterations affect wildland fire depends on weather conditions and plant community characteristics. As weather conditions become extreme, the influence of grazing on fire behavior is limited, especially in communities dominated by woody plants.

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Contents

Key Points

- *Cover and biomass of perennial herbaceous plants in sagebrush communities can be reduced by heavy (or severe) grazing repeatedly in the spring before the perennial grasses initiate bolting.*
- *High severity grazing (i.e. >50% utilization), especially in the spring during initiation of bolting of perennial grasses, can suppress competition from native herbaceous plants and cause soil disturbance that can favor annual invasive grasses including cheatgrass.*
- *Livestock grazing at low/moderate severity (i.e., <50% utilization) generally has little influence on the cover of perennial grasses and forbs.*
- *Areas grazed by livestock can have more, less, or the same density and cover of sagebrush compared to non-grazed areas. Determining factors include the season and intensity of grazing, species of livestock, ecological site, and site conditions at the time of grazing.*
- *A window of opportunity may exist for targeted grazing to reduce annual grasses before perennial grasses initiate bolting or during dormancy of perennial grasses.*
- *Targeted grazing with sheep or goats can reduce the fuel load of shrublands in the short term by reducing woody fuels.*
- *Livestock grazing can reduce the standing crop of perennial and annual grasses to levels that can reduce fuel loads, fire ignition potential, and spread.*

- *Grazing after perennial grasses produce seed and enter a dormant state can reduce the residual biomass left on the site, thereby decreasing the fire hazard the following spring and summer.*
- *Grazing can reduce the continuity of fuels, including the amount of herbaceous biomass between shrubs, in sagebrush ecosystems.*
- *Economic analyses reveal that fuel treatments in sagebrush ecosystems have the highest benefit/cost ratio when the perennial grasses comprise the dominant vegetation, i.e. prior to annual grass invasion and shrub dominance.*
- *Extreme fire weather conditions, characterized by low fuel moisture and relative humidity, and high temperature and wind speed, affect wildland fires more than do fuel characteristics, and the potential role of grazing to alter fire behavior is limited.*

Introduction

Sagebrush steppe and semi-desert ecosystems cover vast areas in western North America and dominate landscapes of the Great Basin and Colorado Plateau (Miller et al., 1994). In this review we focus on the sagebrush steppe and semi-desert ecosystems within the Great Basin. Despite their immense extent, several forces threaten the persistence and distribution of these ecosystems. Climatic conditions, grazing, exotic plant invasion, habitat fragmentation, and fire all can alter the extent and composition of sagebrush-dominated landscapes (Miller et al., 1994; Miller & Eddleman, 2000; Davies et al., 2011). A significant concern in recent years is increasingly large and severe wildfires occurring across the arid regions in the west; these fires remove sagebrush and favor more tolerant annual and perennial grasses (Knick & Rotenberry, 199? . Invasive early-curing annual grasses such as cheatgrass, red brome, and medusahead are filling the interspaces between shrubs on many arid sites, or are becoming the ecologically dominant species after a fire. Both situations create a continuous fuel bed, allowing fire to spread more readily across the landscape (stewart & Hull, 1949; D'Antonio & Vitousek, 1992 . A warming climate with earlier snowmelts contributes to a prolonged fire season with larger and more severe fires (chambers and Reliant, 2008).

Weather, fuel characteristics, and landscape features all affect fire spread, severity, and intensity (Figure 1). Efforts to reduce the risk of extensive fires in sagebrush-dominated ecosystems have focused considerable attention on how livestock grazing affects fuels, fire behavior and fire effects. Livestock grazing influences factors related to fuel characteristics, including the proportions of herbaceous and woody fuel, amount of herbaceous biomass, live/dead fuel mix, and continuity of fuel at a patch and landscape scale (Figure 1). Fire behavior and effects are also influenced by weather and

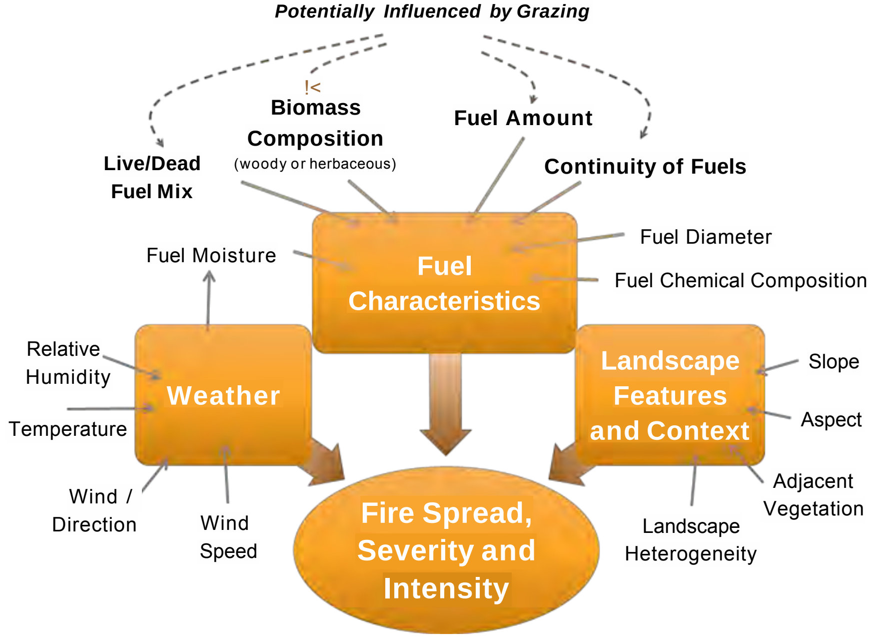


Figure 1. Factors that affect rate of spread, intensity, and severity of wildland fires can be separated into factors related to weather, fuel characteristics, and landscape features and context. Grazing can potentially influence factors related to fuel characteristics.

landscape features that are independent from grazing (Figure 1). This paper provides a comprehensive overview and scientific synthesis of published research on: 1) how livestock grazing can modify plant community composition and alter fuel characteristics of sagebrush-dominated plant communities; 2) how yearly grazing patterns affect fuel loads and wildland fire behavior; and, 3) the comparative economics of grazing as a fuel reduction treatment.

Historic Livestock Grazing Patterns

The introduction of domestic livestock to the Great Basin in the 1860s initiated an era of broad-scale ranching and significant changes in rangeland ecosystems (Miller et al., 1994). Early grazing practices during settlement and homesteading were by all accounts ill-informed and poorly managed (Belsky & Blumenthal, 1997; Miller & Eddleman, 2000 . After several decades of heavy stocking and season-long use, the perennial grass and forb understory was considerably depleted across much of the sagebrush steppe and semi-desert (Vale, 1974 . One of the greatest effects of this excessive grazing pressure was a reduction of the fine fuels that had previously carried wildfires (Miller et al., 1994; Miller & Eddleman, 200C . Concomitant with excessive grazing pressures was the reduction and relocation of Native American populations which reduced the presence of rangeland fire in sagebrush systems (McAdoo et al. 2013).

With a reduced frequency of wildfires, the woody plant cover increased, and shrublands and woodlands expanded (Miller et al., 1994; Miller & Eddleman, 2000). Livestock grazing also promoted woody plant growth by suppressing competition from herbaceous plants through preferential grazing of grasses and forbs (Miller et al., 1994; Wilcox et al., 2012).

Grazing management programs designed to improve native perennial grass communities were first implemented in the 1940s. Managed grazing, including periods of rest, seasonal deferment, and reduced stocking rates was widely implemented in the latter half of the 20th century (Krueger et al., 2002). As grazing management practices were implemented, herbaceous fuel loads generally increased, and wildfires became more common. This reduced the abundance of non-sprouting shrubs, including most sagebrush species, across vast areas (Young & Blank, 1995; Davies et al., 2009).

Introduction of Exotic Annuals Grasses

Cheatgrass, an invasive annual grass introduced to North America in the 18th century (Mack, 1981), has vastly changed the fire regimes across the Great Basin and western North America (Brooks et al., 2004). Medusahead is another annual grass, introduced from the Mediterranean region in the late 1800s and spread rapidly across the Great Basin (DiTomaso et al., 200s . These and other invasive annual grasses, including red brome, have changed fuel characteristics and fire regimes of the ecosystems they invaded (D'Antonio & Vitousek, 1992; Brooks et al., 200 . These fine-textured, flammable, and early maturing grasses have lengthened the annual fire season and shortened the return interval of wildfires across the Great Basin (Hull & Pechanec, 1947; Stewart & Hull, 1949; Davison, 1996; Bradford & Lauenroth, 2006; Balch et al., 2013). Their rapid spread was exacerbated by excessive stocking rates and inappropriate grazing practices (Knapp, 1996; Young & Sparks, 2002; Chambers et al., 200; . Thus a discussion of the effects grazing has on fire patterns in sagebrush-dominated ecosystems necessitates a discussion of how grazing affects annual grass abundance.

Fire is a widespread disturbance type in sagebrush ecosystems, but when cheatgrass and other annual grasses become established, they change fuel characteristics and shorten the fire return interval in these ecosystems (Stewart & Hull, 1949; Brooks et al., 2004; Balch et al., 2013 . Fires can occur more frequently because it only takes a few years post-fire (i.e., three to six) to develop a sufficient fuel continuity to facilitate another fire (Peters & Bunting, 1994). The abundance of cheatgrass also increases the likelihood Of fire ignition and spread (Bunting et al., 1987; Link et al., 2006; Balch et al., 2013 . For example, the estimated fire ignition risk more than doubled (i.e., from 46% to 100%) in bunchgrass communities in southwestern Washington when the cover of cheatgrass increased from 12% to 45% (Link et al., 2006). The continuity and flammability of cheatgrass contribute to a highly connected fuel bed which facilitates rapid spread across the landscape (Figure 2). The fire return interval can be halved and the fire size greatly increased on rangelands dominated by cheatgrass as compared to fires in vegetation communities without cheatgrass (Balch et al., 2013). As wildfires become more frequent, perennial grasses and native shrubs are generally lost from the plant community (Peters & Bunting 1994). With repeated fires, the seedbank of perennial herbaceous species eventually becomes depleted, permanently altering vegetation composition in sagebrush communities (Knapp, 1996; Humphrey & Schupp, 2001).

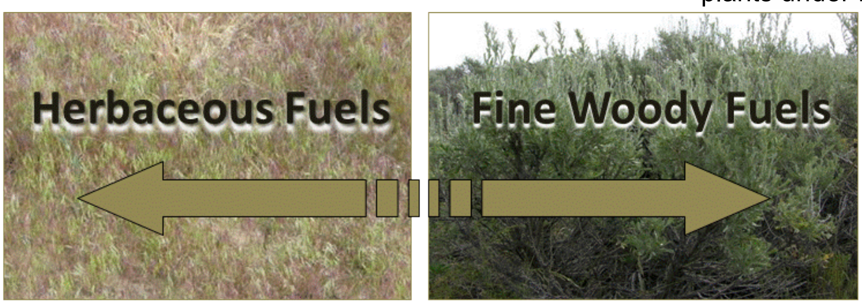
How Grazing Alters Plant Community Composition in Sagebrush Ecosystems

How grazing affects the plant composition of sagebrush ecosystems depends on several factors: precipitation is key, followed by soil characteristics, season and intensity of grazing, and species of grazing herbivore. Plant community composition also has important implications for fire regimes and potential fire behavior. Different types of plants exhibit very different fuel characteristics that affect fire ignition, fire behavior, and fire effects (Figure 2). Fine herbaceous fuels cure over the summer, rapidly equilibrate with the ambient relative humidity, and facilitate easy ignition in the summer and early fall. Fire spread through these fuels is usually low intensity because of the lower amount of biomass per unit area (Scott & Burgan, 2005).

Sagebrush-dominated ecosystems support an overstory of shrubs composed of fine woody fuels (i.e., less than 7.6 cm [3.0 inches] diameter). Fine woody fuels are more difficult to ignite but typically burn longer and hotter than the herbaceous grass and forb fuels in the understory. Fine woody vegetation increases flame length and fire intensity. Increasingly greater shrub biomass and fuel loads lead to more severe fire effects, e.g. plant mortality, smoke emissions, soil heating, and biomass consumption (Sikkink et al., 2009). Woody plants such as sagebrush can also contain volatile oils that can create highly flammable fuel loads and increase both flame lengths and fire spread (Buttkus & Bose, 1977).

Livestock grazing effects on shrub cover/densities

An examination of a variety of grazing studies and comparisons reveals no clear and consistent effect of grazing on cover, density, or biomass production of shrubs. For example, researchers in eastern Oregon recorded increased density of juvenile sagebrush plants under high stocking rates (1.2 AUM/ha or .48 AUM/ac) compared to no grazing or a low stocking rate (0.6 AUM/ha or 0.24 AUM/ac) in Wyoming big sagebrush with a crested wheatgrass understory (Angell, 1997). Likewise, sagebrush density increased in response to early season grazing, before perennial grasses flower and set seed, in a threetip sagebrush community (Laycock, 1967; Bork et al., 1998).



- Primary fuel for ignition
- Equilibrates quickly to ambient relative humidity & temperature
- Rapid fire spread
- Low fire intensity & severity

- Act as ladder fuel
- Higher flame lengths
- High intensity
- High burn severity
- Smoldering

Figure 2. The fuels of sagebrush-dominated ecosystems can be categorized and described as herbaceous (i.e., grasses and forbs) and fine woody fuels (i.e., < 7.6 cm [3.0 inches] diameter woody stems). The fuels vary in how they contribute to fire behavior and effects.

The variable effect of grazing on shrubs can also be assessed by comparing the plant community in areas where grazing has been excluded to adjacent similar areas where grazing has continued. We examined eleven exclosure studies in sagebrush ecosystems where grazing had been excluded for ten years or more. In these comparison studies, sagebrush and other shrubs' response to the removal of grazing varied depending on the species of shrub, soil type, community condition at the time of exclosure, species of herbivore, and season and intensity of grazing. In seven of the eleven studies; there was no consistent or discernible difference in shrub cover or density between grazed and ungrazed sites (Rice & Westoby, 1978; Daddy at al., 1988; Courtois et al., 2004; Yeo, 2005; Davies et al., 201c . For example, Davies et al. (2010) found no difference in Wyoming big sagebrush cover in areas grazed at moderate intensity (30-50% utilization and a deferred rotation grazing system) over the past 70+ years compared to areas that had been excluded from grazing in Wyoming big sagebrush steppe. A similar comparison of rangeland vegetation in fourteen grazing exclosures with mountain and Wyoming big sagebrush in southeastern Idaho revealed no difference in shrub cover inside and outside the exclosures in areas available for grazing by wildlife and livestock (primarily cattle) under a variety of grazing systems. (Yeo, 2005).

Several exclosure studies revealed that grazing affected shrub cover or density, but the effect was not consistent. Laycock (1967) described greater production of threetip sagebrush in areas grazed (at levels describe as "heavy") in the spring by sheep compared to those areas excluded from grazing (for 25 years) or areas grazed in the fall. Holechek and Stephenson (1983) similarly showed greater cover of basin big sagebrush on grazed lowland sites in an exclosure study in northern New Mexico. However, on upland sites Holechek and Stephenson (1983) reported greater cover of sagebrush in the exclosure compared to the adjacent grazed area with 30 to 50% utilization levels. Manier and Hobbs (2006) showed greater cover of mountain big sagebrush in exclosures than on adjacent grazed areas at 17 exclosure sites in western Colorado. Similarly, Whisenant and Wagstaff (1991) reported greater relative cover of bud sage in exclosures without grazing for 53 years compared to adjacent grazed areas. Exclosure studies may be valuable in discerning the effects of the recent grazing regimes on specific areas. However, exclosure studies, collectively, do not reveal global trends due to grazing. The results of these studies suggest that the effects of grazing on shrub cover and production are site-specific, and depend on the site conditions; the historic grazing regimes; plant community composition at the time the exclosures were constructed; and, the specific grazing regime after the exclosures were established.

Several researchers also attributed plant community change to the removal or reduction of grazing by comparing observations before and after changes in a grazing regime. For example, Yorks et al. (1992) found an increase in basin big sagebrush cover (0.5% to 13% from 1933 to 1989) along a 37-km (30-mile) transect in sagebrush semi-desert in Utah. During this 56-year period, grazing pressure was reduced as a result of the Taylor Grazing Act of 1934; however, a general increase in annual average precipitation may have had a greater influence on shrub cover than did the reduction in grazing. Similarly, Wyoming big sagebrush cover on a site in south-central Idaho increased from 18% in 1950 to 25% in 1975 after the removal of grazing (Anderson & Holte, 1981). In this study, the increase can be attributed to succession and adequate precipitation. In a subsequent study on the same site, sagebrush cover declined from 25% in 1975 to 13% in 1995 because of wide-spread die off of sagebrush likely related to drought, insect and rodent damage, and/or fungal pathogens (Anderson & inouye, 2001). Increased shrub cover was also observed ten to fifteen years after removal of grazing by free-roaming horses in sagebrush ecosystems across the Great Basin (Beever et al., 2008 . On the other hand, big sagebrush cover decreased on a site in northern Utah eleven years after livestock grazing was removed (Austin & Urness, 1998;. This decrease was largely attributed to increased grazing pressure by mule deer and more competition with sagebrush from perennial grasses that were not grazed after the removal of cattle.

Livestock grazing effects on perennial grass cover

The effects of grazing on perennial grass cover in sagebrush communities depends on factors similar to those affecting sagebrush cover, including precipitation, soil characteristics, season of grazing, grazing intensity, and type of herbivore. Severe grazing that occurs repeatedly in the spring, before plants produce seeds, has been shown to reduce the cover of perennial grasses and forbs (Vale, 1974; Bork et al., 1998); the effect of light to moderate intensity livestock grazing on vegetation is more obscure.

It is difficult to discern grazing effects from other biotic effects and abiotic environmental conditions (Miller et al., 1994, Holechek et al., 2006). When grazing was removed from a Wyoming big sagebrush site, increases in perennial grass cover occurred sometimes (Robertson, 1971), but not always (Rice & Westoby, 1978; West et al., 1984 . Yorks et al. (1992) observed a ten-fold increase in perennial grass cover from 1933 to 1989 in Utah semi-desert where grazing pressure by livestock was reduced. On the other hand, Davies et al. (2010) found no difference in current year's herbaceous production when comparing long-term (i.e., 70 years) moderately grazed rangeland (30-50% utilization) with areas excluded from grazing in Wyoming sagebrush steppe communities in eastern Oregon.

Livestock grazing effects on annual grass abundance

Livestock grazing and annual grasses are interacting factors that affect fuel characteristics and wildland fire occurrence and behavior throughout sagebrush ecosystems. Intense (high stocking rate), severe (high utilization levels), and repeated (multiple defoliation events in the same season) grazing can suppress competition from native plants and cause soil disturbance that can favor annual invasive grasses including cheatgrass (Klemmedson & Smith, 1964; Mack, 1981; D'Antonio & Vitousek, 1992; Knapp, 1996; Bradford & Lauenroth, 2006; Chambers et al., 2007; Loeser et al. 2007). Perennial grasses are strong competitors with cheatgrass (Booth et al., 2003; Chambers et al., 2007; Blank & Morgan, 2012), so grazing that adversely affects perennial grasses can actually increase annual grasses.

Exclusion of livestock does not necessarily slow invasion or reduce abundance of annual grasses (Cottam & Evans, 1945; West et al., 1984; Young & Allen, 1997; Anderson & Inouye, 2001; Courtois et al., 2004; Young & Sparks, 2002). A comparison of grazed and ungrazed canyon vegetation in Utah showed that cheatgrass was 1.5 times more frequent in an ungrazed than a grazed canyon (Cottam & Evans, 1945). Substantial invasion by cheatgrass and other exotic annual grasses can also occur on sites that have never been grazed by livestock but where there is a seed source (Daubenmire, 1940; Tisdale et al., 1965; Svejcar & Tausch, 1991; Goodwin et al., 199E . However, caution should be applied to site comparisons aimed at ascertaining the effects of grazing because the spread of cheatgrass across an area depends on the level of site degradation when the annual grasses were introduced (Young & Sparks, 2002), frequency Of wildfire (Cottam and Evans, 1945) and on the relative resistance of different ecological sites to cheatgrass invasion (Chambers et al., 2007).

Though severe and poorly timed grazing can promote annual grasses, in some situations livestock grazing can suppress annual grasses, including cheatgrass (Daubenmire, 1940; Mosley, 1994; Vallentine & Stevens, 1994; Mosley & Roselle, 2006; Loeser et al., 2007) and medusahead (DiTomaso et al., 2008). The intensity of grazing can influence whether annual grasses are suppressed or promoted. For example, in northern Arizona high elevation semi-arid grasslands, sites with moderate grazing intensity (about 50% utilization) in the summer grazing season had lower cheatgrass abundance than either intensely grazed (stocked to accomplish high utilization >70% in a 12-hour grazing period) or ungrazed treatments (Loeser et al., 2007). In southeastern Washington bluebunch wheatgrass communities, high-intensity sheep grazing pressure during winter dormancy and the spring grazing season eliminated cheatgrass from a site within a few years. However, a reduction in perennial grasses also occurred and the rapid reinvasion by annual grasses was observed after cessation of grazing (Daubenmire, 1940).

The impact of grazing on invasive annual grasses is highly variable and site specific, which gives rise to opposing research and field observations that either implicate grazing in the spread and abundance of annual grasses, or describe the suppression of annual grasses by livestock grazing. Important factors contributing to these conflicting results include resistance to cheatgrass as determined by soil temperature, the timing and amount of available soil moisture, the relative abundance of perennial herbaceous species, and the season and intensity of grazing (Chambers et al., 2013).

The timing and amount of precipitation and winter or spring temperatures strongly affect the germination, survival and growth of annual grasses such as cheatgrass (Mack & Pyke, 1983; Chambers et al., 2007). Cheatgrass also is favored after fire or other disturbances when the community of perennial herbaceous plants has been depleted (chambers etal., 2007; Hoover & Germino, 2012 . Precipitation timing and amount are immensely important factors determining the response of cheatgrass to grazing (Young et al., 1987). Because cheatgrass responds quickly to early season rains, grazed cheatgrass plants may exceed the growth of ungrazed plants if moisture is available following spring grazing (Vallentine & Stevens, 1994).

However, grazing can also increase annual grass abundance in dry years. A study in a high-elevation, Great Basin grassland in Arizona revealed similar levels of cheatgrass in high-intensity cattle grazing with high stocking rates compared to ungrazed pastures until a drought year occurred (Loeser et al., 2007). In the two years after the drought year, the high-intensity grazing treatment resulted in an 80% increase of cheatgrass cover and a frequency of occurrence of nearly 100% compared to about 40% on ungrazed sites (Loeser et al., 2007).

The effects of grazing on annual grass abundance also varies by season in which grazing occurs. There appears to be a window of opportunity for grazing to reduce annual grasses if grazing occurs when annuals begin to produce seeds but before native perennial grasses initiate bolting (Figure 3; Mosley, 1994; Vallentine & Stevens, 1994; Mosley & Roselle, 2006; Smith et al., 2012). Cheatgrass is very palatable to livestock and has high nutritional value in the vegetative stage and is preferentially selected over many perennial grasses in early spring throughout the Great Basin (Young & Clements, 2007 . Early and late spring clipping that simulated grazing reduced the biomass of cheatgrass compared to an unclipped control, though density of cheatgrass was unaffected (Tausch et al., 199 . A similar clipping study found no effect of clipping on cheatgrass seed density except when plants were clipped in the boot stage and then clipped again two weeks later, resulting in reduced seed density (Hempy-Mayer & Pyke, 2008).

The timing of grazing is critical because annual grasses may flourish if perennial plants are grazed preferentially at times when the perennial grasses are sensitive to damage by grazing (Pyke, 1986; Ganskopp, 1985 . If bunchgrasses are routinely heavily grazed (exceeding 50% utilization) in the period from bolting through seed-set, and particularly if multiple defoliation events in the same season occur, the competitive advantage can be shifted toward cheatgrass (Daubenmire, 1940; Young et al., 1987). Late season grazing, after perennial grasses have produced seed and begin to senesce, has minimal

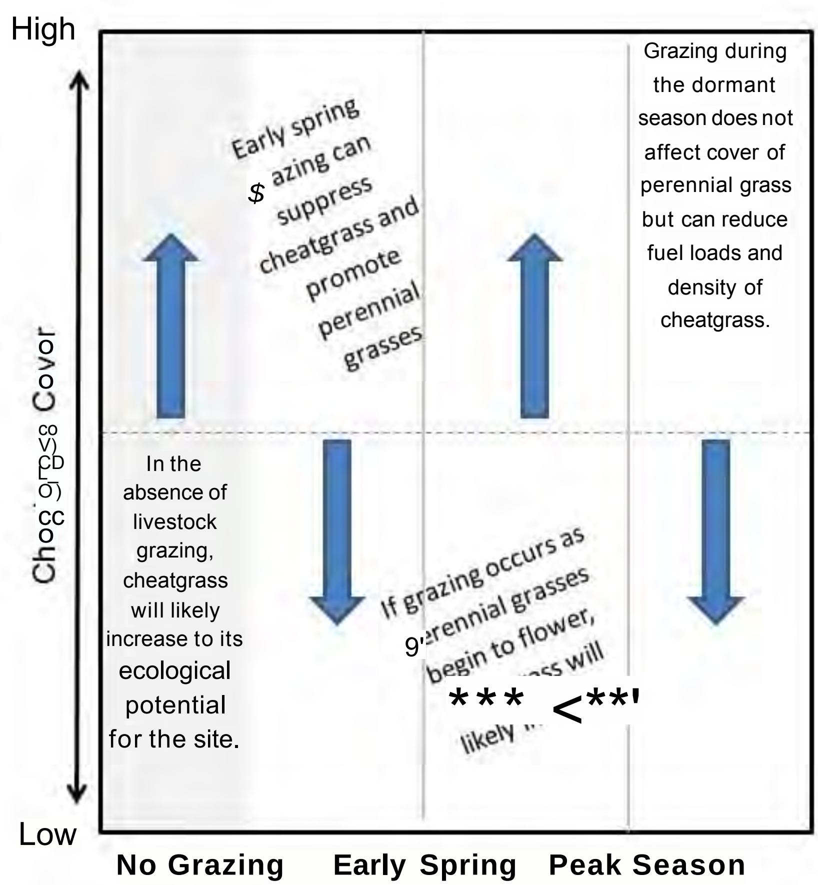


Figure 3. Conceptual depiction of how livestock grazing can influence cheatgrass abundance in sagebrush-dominated ecosystems with a significant component of perennial grasses. Grazing can suppress or promote cheatgrass depending primarily on the season of grazing. Grazing suppresses cheatgrass when applied in early spring when annuals begin to produce seeds and before native perennial grasses initiate bolting; and when applied during the dormant season.

impact on these grasses (Ganskopp, 1988; Hempy-Mayer & Pyke, 2008). Several years of fall grazing by cattle on semi-desert (27 cm [10.6 inches] annual precipitation) sites in Nevada dominated by Wyoming big sagebrush and salt desert shrub plant communities has been shown to reduce cheatgrass density and cover and increase cover of perennial grasses compared to sites without fall or winter grazing (Schmelzer et al., in press).

Well-timed and closely managed spring grazing can be an effective tool to suppress annual grasses including cheatgrass and medusahead (Mosley, 1994; Vallentine & Stevens, 1994; Mosley & Roselle, 2006; DiTomaso et al., 2008; Smith et al., 2012). One of the best opportunities to reduce the abundance and cover of cheatgrass is before most perennial grasses begin active growth (Vallentine & Stevens, 1994; Young & Allen, 1997; Mosley & Roselle, 2006; Smith et al., 2012). The challenge is to remove livestock before perennial plants begin active growth in order to avoid reduced vigor in the perennial grasses (Laycock, 1967; Miller et al., 1994; Loeser et al., 2007). Regardless, perennial grasses with similar phenologies as annual grasses, like bottlebrush squirreltail, may be reduced (Booth et al., 2003). On cheatgrass-dominated sites, high grazing intensity and annual use must be maintained or annual grasses will quickly re-invade and dominate an area (Daubenmire, 1940; Klemmedson & Smith, 1964; Pyke, 1986).

How Livestock Grazing Can Modify Fuel Loads

Management practices can greatly affect a landscape's fuel amount and distribution. Fuel load, or biomass, is one of the most influential and easily manipulated fuel variables affecting fire intensity (Figure 4). Fuel load is the portion of the biomass that will actually burn in a wildfire or prescribed fire, and is closely related to vegetation biomass. Fuel loads are the primary drivers of heat, and all measures of heat increase with increasing fuel loads (Vermeire & Roth, 2011). The likelihood of fire-induced bunchgrass mortality depends upon the amount of heat received and the type of plant tissue exposed to lethal heat (Miller, 2000; Wright, 1971). Livestock grazing is one management technique that has been shown to decrease fine fuel loading and subsequent wildfire severity (Archibald et al., 2005; Davies et al., 2010).

Fuel management objectives aimed at reducing flame lengths and fire spread in grassland and shrubland ecosystems could be accomplished by altering the fuel bed depth, fine fuel loading, cover, and continuity such that the flame length never reaches 1.2 meters (3.9 feet; Nader et al., 2007). Livestock (i.e., cattle horses, sheep and goats) grazing primarily impacts small diameter fuels (< 0.51 cm [0.2 inch] diameter), including grass and small woody stems that equilibrate with the ambient humidity and temperature within 1 hour (i.e., the 1-hour time lag

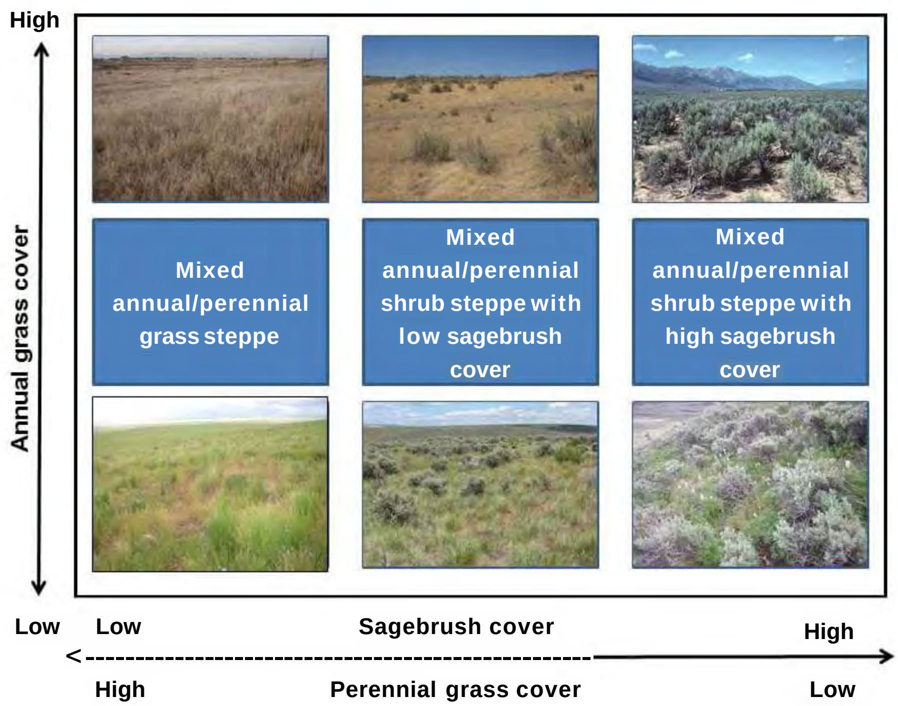


Figure 4. The plant composition in sagebrush-dominated ecosystems is variable across the landscape; this has important implications for fire behavior because different types of plants exhibit different fuel characteristics affecting fire ignition, behavior, and effects. Fire spreads quickly through cured grass usually at low intensity because of the low amount of biomass per unit area. Fine woody vegetation increases flame length and fire intensity. Higher shrub loads lead to more severe fire effects when the area burns. The continuity and flammability of cheatgrass contributes to fire connectivity and spread.

[htl] fuels). Livestock can also impact larger fuels (0.51-2.54 cm [0.2 - 1 inch] diameter or 10-htl fuels) through browsing and trampling as suggested in a review by Davison (1996). Hence, grazing could be a useful management tool for reduction of grass and shrub biomass (1-htl and 10-htl fuels).

Shrub fuel loads

Targeted grazing can be applied to reduce the fuel load of shrublands through "brush-clearing" strategies (Nader et al., 2007. Strategies to reduce shrub abundance generally rely on goats or sheep because these species generally consume greater quantities of shrubs than cattle (Taylor, 2006; Papanastasis, 2009). Grazing by cattle would not be expected to affect sagebrush cover through direct consumption of sagebrush.

Perennial grass fuel loads

The effect of grazing on fire behavior and extent is predictably less pronounced on sites dominated by woody plants compared to those with more herbaceous biomass. However, reduced fire frequency and spread in grazed shrublands and forests have been observed because the herbivores remove the fine herbaceous fuels that are most likely to ignite and initiate fire spread (Zimmerman & Neuenschwander, 1984; Hobbs, 1996).

Grazing with the goal of reducing herbaceous fuel loads generally is more effective if it occurs right before the season of greatest fire risk, which generally coincides with peak biomass and the initiation of dormancy (Taylor, 2ooe . If grazing occurs early in the growing season, grasses can regrow and biomass can be reestablished to levels similar to ungrazed areas (Anderson & Frank, 2003). Grazing or mowing after plants have initiated seed formation and reached peak biomass can reduce biomass levels below those of ungrazed plants and paddocks (Miller et al., 1990; Anderson & Frank, 2003). Grazing late in the growing season (after seed set) and in the dormant season can thereby reduce the residual biomass carried over to the following spring and summer (Launchbaugh et al., 2008). Furthermore, grazing after seed production has lower impact on plant vigor and survival than grazing before floral initiation (Adler et al., 2001).

Ungulate grazing reduces the standing herbaceous plant material available for burning; this in turn can potentially reduce the frequency, extent and intensity of fires in grass, shrub, and forest understory fuel types (Vale, 1974; Zimmerman & Neuenschwander, 1984; Tausch et al., 1994; Hobbs, 1996; Belsky & Blumenthal, 1997; Blackmore & Vitousek, 200C . In relatively moist Wyoming big sagebrush steppe (30 cm [11.8 inches] annual precipitation), Davies et al. (2010) found that grazing reduced the amount of herbaceous fuel. The fine flammable grassy fuel load, including dead standing crop, was two-fold greater in plots that had not been grazed for 70 years compared to adjacent areas that had been grazed long-term at moderate grazing intensity (30-50% utilization). In grasslands without shrubs, fire intensity is inversely related to standing crop biomass (Stronach & McNaughton, 1989; Hobbs, 1996), and grazed patches burn less completely and intensely than ungrazed patches (Hobbs et al., 1991); however, these relationships have not been well researched in shrubland systems of the Great Basin.

Beyond the amount of residual fuel remaining after grazing, the proportion of live versus dead herbaceous biomass may be an important factor affecting a fire's ability to spread in grasslands. Grazing can in some instances increase the propensity for fire to spread because herbivores selectively remove green biomass and thereby increase the proportion of dead to live biomass (Leonard et al., 2010). Though alteration of the live-dead ratio of herbaceous biomass is possible through grazing, it is unlikely to be important in late season wildfires in the Great Basin when most vegetation is dormant.

Annual grass fuel loads

The effects of livestock grazing on fuel characteristics of communities with significant amounts of annual grasses can be viewed in two ways. First, as noted, grazing can promote or suppress annual grasses over years or decades. Second, livestock grazing can reduce the standing biomass of annual grasses within a year to reduce fuel loads and alter fuel continuity. Only a few studies have addressed the potential effect of livestock grazing on fuel loads in annual grasslands. Diamond et al. (2009) examined the effect of grazing by cattle on fire behavior on a cheatgrass-dominated site in Nevada. Targeted grazing when cheatgrass was in the boot stage was applied to reduce 80 to 90% of the herbaceous biomass. This treatment resulted in reduced flame length and rate of fire spread in a prescribed burn conducted in October: fuel characteristics were so greatly reduced that the fire would not spread across the grazed plots (Diamond et al., 2009). Note, however, this study was conducted in a confined area that may not be easily replicated on landscape scales.

Cheatgrass is most palatable and nutritious before the seeds mature and plants turn purple (Hull & Pechanec, 1947; Young & Allen, 1997). However, livestock will eat cheatgrass throughout the season and it has been considered by some as important winter forage in the Great Basin (Emmerich et al., 1993; Vallentine & Stevens, 1994). This may create an opportunity to graze cheatgrass almost year-round to manage fuel loads. Research in Nevada examined the potential value of winter grazing by cattle to reduce cheatgrass fuel loads (Schmelzer et al., in press). In this study, cheatgrass fuel loads were reduced 70 to 80% by winter cattle grazing. The cattle favored cheatgrass over perennial grasses, and with a protein supplement were able to maintain their weight. This study suggests that winter livestock grazing could be accomplished on landscape scales as a part of regular grazing practices to manage fuel loads of cheatgrass. Winter (dormant season) grazing reduces fuel carryover to the next summer (Figure 5), can reduce the thick litter layer known to facilitate the germination of medusahead and cheatgrass seed, may decrease the size of the annual grass seedbank, and has few if any adverse effects on the dormant, desired perennial grasses.

A significant challenge to managing fuel loads of annual grasses with livestock grazing is the highly variable biomass production related to rainfall patterns (Young et al., 1987). One study in southern Idaho showed that cheatgrass biomass varied ten-fold depending on annual precipitation; from 404 to 3879 kg/hectare [452 to 4344 lbs/acre] in a dry compared to a wet year (Hull & Pechanec, 1947). Thus, a program using livestock grazing to manage fuel loads created by annual grasses will need to be flexible, and responsive to annual moisture regimes that will alter plant growth and biomass. Winter grazing of cheatgrass has one distinct advantage for livestock production: the amount of potential forage is known months in advance, so the livestock numbers needed to achieve desired utilization levels can be easily determined. For spring grazing of cheatgrass, biomass production can fluctuate dramatically in a short period due to sudden and unpredictable changes in precipitation and temperature. Matching livestock numbers with that forage base is much more difficult.

Continuity of fuels

Fuel continuity describes the spatial arrangement or distribution of fuel and is a major factor affecting the spread of fire across a landscape (Cheney & Sullivan, 2008). Greater fuel continuity leads to faster rates of spread, and spread with lower fireline intensity (NWCG, 1994). Horizontal continuity is the relationship of the horizontal distance between fuel particles and is related to percent cover of vegetation. Management actions that alter vegetation species composition and abundance can strongly affect the fuel continuity (Brooks et al., 2004). Fuel continuity generally increases as fuel load increases. However, if major shifts in vegetation composition occur, then fuel load can decrease while fuel continuity increases. Invasion of the sagebrush ecosystem by annual grasses is a classic example of this phenomenon: bunchgrasses

Figure 5. Sagebrush steppe where cheatgrass dominates the herbaceous vegetation. Winter grazing has been applied in the image to the left while the image to the right has been excluded from grazing for the past 20 years. Both photos were taken in late May in neighboring pastures.

and shrubs, with abundant open space between plants, are being replaced with smaller-statured grasses with less space between individual plants. A major factor of larger wildfires in recent years is the increased fuel continuity across the landscape (Davison, 1996).

Livestock grazing can alter the spatial pattern of vegetation which in turn can have important consequences for fire occurrence and spread (Adler et al., 2001). Grazing can increase or decrease the spatial heterogeneity of vegetation depending on the existing plant community's grazing animal distribution, and the scale of observation (Adler et al., 2001). Typically, grazing increases patchiness when the grazing pattern is stronger than the vegetation pattern and when grazing increases the contrast among vegetation types. In grassland systems, grazed patches may be more likely to be re-grazed in subsequent years because they typically contain a greater proportion of new growth (Hobbs et al., 1991).

In grasslands, landscape mosaics created by variable grazing intensity can provide "firebreaks" and prevent fires from becoming larger (McNaughton, 1992). However, these relationships may not apply to shrublands because fire can be carried by the shrubs. Davies et al. (2010) observed larger fuel gaps in moderately grazed areas compared to ungrazed areas in a Wyoming big sagebrush community, and the continuous perennial grass patches were larger in ungrazed areas. Furthermore, herbaceous fuel between shrubs in sagebrush ecosystems may be effectively reduced by livestock grazing. France and colleagues (2008) documented grazing by cattle in Wyoming big sagebrush ecosystems was focused on bunchgrasses in interspaces between shrubs with only negligible removal of grasses under shrub canopies at moderate (i.e., 40%) utilization levels. A case study in sagebrush steppe in northern Nevada demonstrated success keeping landscape scale fire at a minimum using livestock to reduce fuels and implementing range improvement projects, such as flanking existing roads with green-strip seedings, managing brush, seeding projects, and improving riparian areas to function as green-strips (Freese et al., 2013).

Although high-intensity grazing can reduce biomass and fine fuel loads, light grazing can produce patchy burn patterns in continuous sagebrush steppe fuels (Bunting et al., 1987). Low- to moderate-intensity grazing can remove sufficient fuel and break up fuel continuity to significantly reduce fire spread (Bunting et al., 1987). Patchy burn patterns are particularly important in sagebrush regions where maintenance of sagebrush cover (e.g., for wildlife habitat) is a management objective. Patchy burns leave islands of unburned sagebrush, thereby creating a seed source for reestablishment of sagebrush plants across the affected area (Colket, 2003).

Much of the evidence on fire behavior in herbaceous fuels is extrapolated from grassland ecosystems in both North America and Africa. However, because fire itself is a physical process driven by fuel, it is largely unaffected by the specific plant species, but rather the amount and structure of the fuel source. For example, one classification for wildland fuels is the time required for dead fuel to equilibrate to changes in relative humidity (largely a function of fuel diameter). Additionally, fuel loading and fuel continuity are used in fire spread models, whereas vegetation species is generally not included (Scott and Burgan, 2005). This allows for comparisons of fire behavior among ecosystems with similar fuel classes, but completely different species composition.

Grazing to Manage Fuels Depends On Weather, Topography, and Vegetation Composition

Carefully targeted grazing can be used as a tool to reduce fine fuel loads, the rate of spread and final extent of fires, and ultimately fire frequency, in sagebrush-dominated ecosystems. However, the level to which grazing affects fire behavior depends on a number of physical and environmental conditions, such as ambient temperature, wind speed, humidity, fuel composition, fine-scale continuity (tuft-scale), spatial distribution, and topography (Figure 1). Fuel loading and fuel moisture directly affect the fire behavior and consumption rates in sagebrush ecosystems under most environmental conditions. In the absence of sagebrush cover, if fine fuel loading is less than 560 to 650 kg/hectare (627 to 728 lbs/acre), fires will sustain only under environmental conditions characterized by less than 15% relative humidity, temperatures exceeding 29°C (84.2 °F), dead fuel moisture less than 12%, and wind speeds greater than 16 km/hour (9.9 miles/hour (Britton et al., 1981; Bunting et al., 1987; Launchbaugh et al., 2008 . However, when fine fuel loading is above 1700 kg/hectare (1904 lbs/acre), fire will spread under a wide array of environmental conditions (Bunting et al., 1987). Given these estimates, based on models, livestock grazing could remove sufficient fine fuel to reduce the risk for fire ignition and spread throughout most of the year. Consequently, areas that are selected for a prescribed fire should not be grazed the season before the planned fire to allow fine fuel accumulation (Bunting et al., 1987).

In sagebrush steppe and semi-desert, the shrub component adds vertical structure to an understory of herbaceous forbs and grasses. Brown (1982) suggested that at 20% sagebrush canopy cover, a cured herbaceous fuel load of at least 340 kg/hectare (381 lbs/acre) would be required to sustain a fire with a 16 km/hour (9.9 mile/hour) wind. Areas with greater sagebrush cover may burn at lower herbaceous fuel loads. Lower fuel moistures, typical in the fall, increase the rate of spread, flame lengths and fire intensity when compared to spring burns (Sapsis & Kauffman, 1991). Consumption rate of 10-htl and 100-htl woody fuel also increases with lower fuel moisture content (Sapsis & Kauffman, 1991). In addition to fuel moisture and weather, topography also affects fire behavior. At 30% slope the fire rate of spread is two to three times greater than a flat area, while at 50% slope the rate of spread increases four to seven times (Brown, 1982). Thus, fuel reduction by grazing will have the most pronounced effects and potential to benefit suppression activities on more level parts of the landscape.

Reducing levels of fine fuels, as could be accomplished with livestock grazing, reduced the modeled surface rate of spread and fire intensity in simulated shrub and grassland communities (Launchbaugh et al., 2008). Model assumptions using Behave Plus software include uniform fuel continuity, weather, and slope (Andrews, 2008). In addition, the model does not include potential spotting to advance the fire ahead of the containment line. The effects of reduced fuel load on fire behavior were more pronounced at low wind speeds and high fuel moisture. When burning conditions became extreme, changes in the amount of herbaceous fuels (1-htl fuel classes) had little effect on fire behavior variables. Under less extreme fire weather conditions, livestock grazing to reduce herbaceous fuel loads could influence fire behavior, making fire in these sagebrush communities easier to contain.

A similar study with similar fire model assumptions and results was conducted at study sites near Las Cruces, New Mexico and Tucson, Arizona (Varelas, 201: . This study confirmed that with moderate fuel moisture and light winds the reduction of fine fuels by grazing could reduce flame lengths below a 1.2-meter (4-feet) level, permitting direct attack by hand crews. However, the grazing treatments were not effective under more extreme burning conditions and the cattle grazing treatment had limited potential to alter fire behavior when a significant shrub component was present.

Important factors driving the behavior and effects of fire in sagebrush steppe and semi-desert systems are fuel characteristics and fire weather (Figure 6). Livestock grazing has the highest potential to reduce fire spread and intensity in areas dominated by herbaceous fuels with low sagebrush cover under moderate or better weather conditions, (i.e., conditions represented in the upper left region of Figure 6). Grazing by cattle is generally focused on grasses and other herbaceous forage, therefore cattle grazing would have limited potential to alter fire behavior that is driven primarily by sagebrush cover (i.e., conditions represented in the lower left region of Figure 6). However, under moist and cool conditions, grazing can influence fires that move through sagebrush communities by slowing the movement of fire along the herbaceous understory between shrubs. Under extreme burning conditions, characterized by low fuel moisture and relative humidity, and high temperature and wind speed, wildland fires are driven more by weather conditions than by fuel characteristics. Therefore, as fire weather conditions become extreme, the potential role of grazing on fire behavior decreases and may

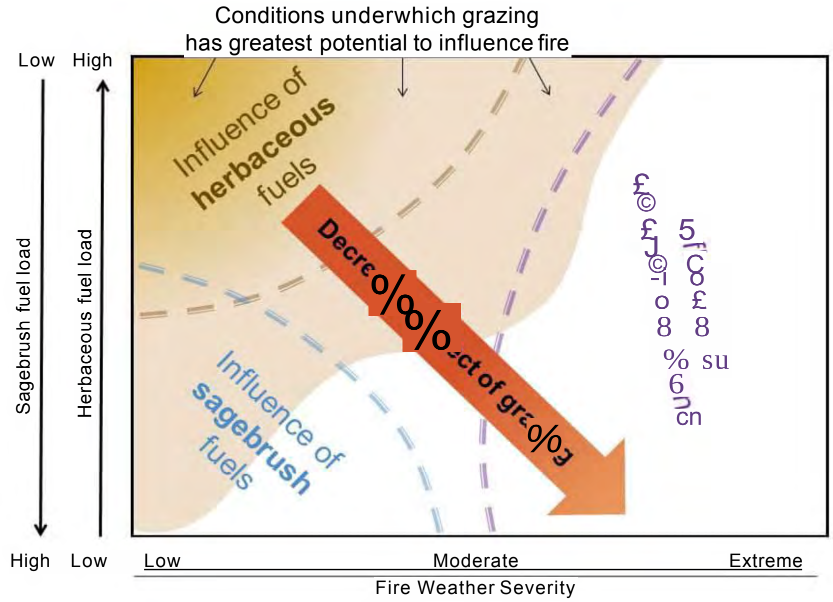


Figure 6. The potential for grazing to influence fire behavior occurs along continuums of fuel and weather conditions. In this conceptual model, fuel composition is displayed on the y-axis and fire weather condition is displayed on the x-axis. Low fire weather severity is characterized by high fuel moistures, high relative humidity, low temperature, and low wind speeds, while extreme fire weather is characterized by the opposite conditions. The potential for grazing to be effective in reducing the risk of fire initiation and spread is greatest when the sagebrush cover is low and the fire weather severity is low to moderate.

become meaningless (e.g., conditions represented on the right side of Figure 6).

Economics of Fuel Treatments

Fuel treatments are designed to alter fuel conditions so that wildfire is easier to control and less destructive (Reinhardt et al., 2008). As noted above, cattle grazing primarily alters fuel conditions by reducing the amount of herbaceous fine fuels, whereas goat and sheep grazing can potentially also reduce the shrub component. Other fuel treatments that can be used to accomplish these same objectives include, herbicides, mechanical treatments such as mowing, prescribed/controlled fires, or a combination of these treatments (Nadar et al., 2007; Diamond et al., 2009).

The costs of fuel treatments vary widely, yet the relative costs and success of alternative treatments is an obvious concern and must be considered when evaluating fuel management options. Several studies review and describe the many factors affecting fuel treatment costs on forested areas where management of the woody overstory is of key concern (Cleaves et al., 2000; Hesseln, 2000; Kline, 2004) similar cost estimates on rangelands are limited. Least cost fuel treatments will vary with conditions and objectives, but grazing alternatives appear to be cost-competitive especially if the objective is reduced fine fuel loads where mowing or a prescribed burn are potential alternatives.

As described by Mercer et al. 2007) and Kline (2004), expanding beyond costs to consider net economic benefits of fuel treatments is a complex analysis. The most important unanswered economic question is whether the resource expended to reduce wildfire risk and damages result in net economic gains. Tradeoffs also exist between increased expenditures on fire suppression versus fuels management (Mercer et al., 2007). The benefit/cost (B/C) assessment requires definition of a wildfire production function that defines the relationship between size and intensity of wildfires as it relates to alternative fuel management treatments, climate variables, and site-specific characteristics. Potential benefits of fuel treatments such as reduced wildfire risk, reduced fire suppression costs, and reduced structural losses will be site-specific. Thus, the site-specific analysis must account for the cumulative cost of fuel treatments, the likelihood of wildfire events with and without treatments, the effects and costs of fire suppression and post-fire restoration, and the effect of management actions and wildfires on resource conditions, structural damages, and saleable products over time (Kline, 2004). Given these complexities, only a few studies have estimated net economic benefits of fuel treatments in forested areas (Mercer et al., 2007; Prestemon et al., 2012), and only one recent study considered net economic benefits of fuel treatments on rangelands (Taylor et al. 2013).

Table 1. Estimated costs for alternative fuel management options on rangeland. a/Personal communication, Feb. 4, 2013, Dan Macon, Flying Mule Farm (http://flyingmulefarm.com/), custom land and vegetation services

Treatment	Source	Location	Description	Cost $/acre
Herbicide	Wolcott et al. 2007	Florida	Grass/Shrub/Tree Intermix	$68-$l,000+
	Torell et al. 2005	New Mexico	Closed canopy of sagebrush, treat with tebuthiuron, aerial	$21
	Taylor et al. 2013	Utah	Closed canopy of sagebrush, treat with tebuthiuron, aerial	$31
	Taylor et al. 2013	Utah	Closed canopy of sagebrush, treat with tebuthiuron, ground application	$52
	Nader et al. 2007	California	Grass/Shrub	$25-5250
Hand Crews	Dan Macon[3]	California	Brush removal	$350 - $600
Mechanical	Nader et al. 2007	California	Mowing on grassland	$25-$40
	Wolcott et al. 2007		Mowing on grass/shrubland	$35-5500
Prescribed Fire	Nader et al. 2007	California	Brush, range, and grassland burns	<$150
	Cleaves et al. 2000		Brush, range, and grassland burns	$57
	Mercer et al. 2007		Southeast U.S.	$11- $344
	Taylor et al. 2013	Utah	Healthy Sagebrush, perennial understory	$20
	Taylor et al. 2013	Utah	Pinyon-Juniper with mature shrubs	$46
Combined Treatments	Taylor et al. 2013	Utah	Closed-canopy pinyon-Juniper; brush management, herbicide, and reseeding	$205
	Taylor et al. 2013	Utah	Annual grass dominated; prescribed fire, herbicide, and reseeding	$165
Targeted Grazing	Nader et al. 2007	California	Targeted grazing with goats	$60-$70
	Dan Macon[3]	California	California sheep and goat grazing contractor	
			< 20 acres	$300
			> 20 acres	$150- $200
	Varelas (2012)	New Mexico	Targeted cattle grazing	$45-$65

The net economic benefits of selected fuel treatments in the sagebrush ecosystems of the Great Basin were estimated in a study by Taylor et al. (2013). State-and-transition models were used to define vegetative characteristic changes expected to occur based on natural succession and disturbance interactions for sites dominated by Wyoming big sagebrush and mountain big sagebrush. The analysis was a probabilistic benefit/cost assessment where the benefit of the treatment was considered to be fire suppression costs averted over the next two hundred years because alternative fuel management treatments were undertaken. The success of fuel treatments (movement to a state with less shrubs and invasive annual grasses) was considered to be uncertain with re-treatment required when the simulation projected a treatment failure.

Because healthier ecological states with relatively high perennial grass cover and without an overgrown sagebrush canopy (sagebrush present but not ecologically dominant) were considered to be resilient and responsive to treatment, and with a marked reduction in fire frequency following relatively low-cost fuel treatments, these healthy areas were found to be more economical to treat than were mature sagebrush areas with a depleted perennial herbaceous understory or areas invaded with annual grasses. The estimated B/C ratio ranged from a high of 13.3 for productive Wyoming sagebrush steppe sites (a relatively low-cost controlled burn treatment) to less than one (i.e., not economically efficient) for high shrub densities and levels of annual grass invasion (requiring expensive and often unsuccessful treatments). Similarly, the estimated B/C ratio for mountain big sagebrush sites decreased with a rise in brush canopy and annual grass invasion. The implication is that the desired time for fuel treatments is before a decadent shrub canopy (one with considerable dead standing biomass) occupies the area and annual grass invasion occurs.

No known studies have quantified the net economic benefits of grazing treatments for fuels management. The analysis would be quite different from that of Taylor et al. (2013) because the effects of grazing treatments would generally only last one or two years: the herbaceous understory regrows and the treatment must be reapplied. However, several broad conclusions might be drawn. First, grazing treatments would potentially be an economical alternative to the prescribed burn treatment suggested by Taylor et al. (2013) specifically for areas with relatively low shrub cover where perennial grasses dominate. Areas where sagebrush fuel loads are low and herbaceous fuel loads are high are the conditions most favorable for grazing treatments (Figure 6). Second, given the relatively short benefit period for the grazing treatment, unless the brush canopy is significantly altered, the cost of the treatment must remain relatively low. The harvested forage would contribute an additional grazing benefit to livestock production if previously unused forage were harvested.

Negative potential ecological impacts from grazing treatments are of concern, as are treatment costs (Table] . Varelas (2012) estimated the cost of targeted cattle grazing treatments increased by about $18/hectare ($7.30/acre) for each 89 kg/hectare (100 lbs/acre) of herbaceous material removed by grazing animals. Targeted cattle grazing treatments using herding and low moisture blocks to hold cattle on targeted areas were found to be effective and cost competitive ($123/hectare [$50/acre]) when the standing herbaceous materials were reduced by about 45% (from 1,161 kg/hectare [1,300 lbs/acre] to about 536 kg/hectare [600 lbs/acre]).

Summary and Remaining Knowledge Gaps

The legacy of early post-settlement livestock grazing has played an important role in shaping vegetation dynamics in sagebrush ecosystems. High intensity and severe grazing in the late 1800s contributed to a dramatic reduction in both fine herbaceous fuels and fire frequency, and provided a competitive advantage for, and consequent increase in, woody plants. The introduction of exotic annual grasses in the late 1800s to early 1900s radically altered the fuel characteristics of many sites in the Great Basin. Over the last several decades, reduced grazing pressure, increased cover of flammable exotic annuals, increased human activity, and more recently, a longer climate-induced fire season (Chambers & Reliant, 2008), have all led to the current situation in the Great Basin where fires are larger and more frequent than 25+ years ago. Wildfires burn frequently enough to prevent establishment of sagebrush and cause a change in vegetation types across this vast region.

There are several ways contemporary livestock grazing practices can affect the extent and behavior of fires in sagebrush-dominated ecosystems. These include cumulative effects that occur on decadal time scales and that alter plant community composition (i.e., woody versus herbaceous) and those influenced annually through changes in fuel loads. Over decades, livestock grazing can change the relative proportions of shrubs, perennial grasses, and annual grasses, altering the fuel composition. On an annual basis, grazing can reduce the amount of herbaceous fine fuels, including cheatgrass, forbs and small twigs of woody plants. Grazing can reduce fire spread and intensity by removing understory vegetation, reducing the amount of fuel, and accelerating the decay of litter through trampling. This altering of fuels continuity can create patchy burns that result in unburned islands of vegetation providing seed sources for re-establishment of plants after the burn.

The effects of grazing could result in fires that burn at lower intensity, increased patchiness, decreased rate of spread, and increased subsequent survival of plants after fire. The specific outcome will depend on the fire weather conditions and the structural composition of the plant community when a fire occurs. As fire weather conditions become extreme, the potential role of grazing on fire behavior is limited.

Fuels management programs that incorporate grazing treatments must consider the long-term effects of such treatments on both desired and undesired plant species, with desirability defined by site-specific management goals and objectives. Grazing practices can alter plant communities such that shrub density increases, perennial grasses decrease, and exotic annual grasses and other invasive species gain a foothold, an outcome that would decrease resistance to and resilience from fire. Sound grazing practices and targeted grazing efforts aimed at wildland fuel reduction, however, have a strong potential to decrease undesirable fire behavior. Reductions of fine fuels and the desirable alterations to wildfire behavior are often overlooked benefits from including sound grazing practices on the landscape.

We identified four main research gaps in understanding how grazing influences fire behavior in sagebrush ecosystems and the related fuel treatment economics. First, research and observations clearly support the statement that grazing can influence wildland fuels and thereby fire behavior. However, residual herbaceous biomass level thresholds required to stop or carry the spread of fire under various weather conditions are largely unknown. Second, it is not known how shrub properties (cover, height, structure, etc.) influence the probability that an area will burn under different weather conditions. Third, further research is needed to discern the effects of landscape scale grazing patterns on fire behavior. Fence-line contrasts suggest that uneven utilization or spatial variation in grazing systems at the pasture scale can contribute to stopping or carrying fires, thereby reducing the area burned. However, this hypothesis has not been tested at meaningful scales. Fourth, an important economic question is whether the resources expended to reduce wildfire risk result in net economic gains.

From an ecological point of view, many questions remain unanswered. Sagebrush ecosystems evolved with fire. However, invasive annual grasses have altered the nature and impact of fire in these systems. Fire will always play an important role in sagebrush steppe and semi-desert, with effects ranging from rejuvenation to destruction. Grazing is one of the tools rangeland managers can apply to moderate these effects.

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Common and Scientific Names of Plants Listed in Text According to the USDA PLANTS Database (http://www.plants.usda.gov/).

Common Name	Scientific Name
Basin big sagebrush	*Artemisia tridentate Nutt. ssp. tridentata*
Bluebunch wheatgrass	*Pseudoroegneria spicata* (Pursh) A. Love
Bottlebrush squirreltail	*Elymus elymoides* (Raf.) Swezey
Cheatgrass	*Bromus tectorum* L.
Crested wheatgrass	*Agropyron cristatum* (L.) Gaertn.
Medusahead	*Taeniatherum caput-medusae* (L.) Nevski
Mountain big sagebrush	*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle
Red brome	*Bromus rubens* L.
Sagebrush	*Artemisia* spp.
Threetip sagebrush	*Artemisia tripartita* Rydb.
Wyoming big sagebrush	*Artemisia tridentate Nutt. ssp. wyomingensis* Beetle & Young

Common and Scientific Names of Animals Listed in Text According to the Integrated Taxonomic Information System (www.itis.gov).

Common Name	Scientific Name
Goat	*Capra hireus*
Horses	*Equus caballus*
Sheep	*Ovis aries*

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