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MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Dena N. Bergman

A Project Presented in Partial Fulfillment
of the Requirements for the Degree
Masters of Science
in Environmental Biology

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May, 2020

MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

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CHAPTER 1. LITERATURE REVIEW

Birds and Butterflies as Surrogate Indicators for Urban Ecological Health

Introduction

Urbanization is an influential force that replaces natural ecological communities with human-constructed ecosystems (Gómez-Baggethun et al., 2013). These urban ecosystems function to support the health and well-being of their human residents (Gómez-Baggethun et al., 2013), but often have negative impacts on local and surrounding ecosystems. Monitoring urban ecosystem health is important for populations that live within these systems as well as those in surrounding systems that are impacted by urbanization. While urban ecological assessment and monitoring is difficult and expensive for growing urban centers, understanding urban ecosystem dynamics is vitally important for planning sustainable urban growth that minimizes negative impacts to surrounding ecosystems and enhances benefits to urban communities (Niemelä, 1999). When municipalities are faced with limitations to assessing and monitoring urban ecosystems, birds and butterflies can serve as surrogate indicators for urban ecological health.

Urban ecosystems are defined by a complex system of natural, societal, and economic components with a marked dominance of human influence and significant impacts on other ecosystem types (Maes et al., 2015; Su et al., 2010). This includes a range of urban areas from densely populated city centers to exurbs and peri-urban spaces (Maes et al., 2015) which make up between 3-5% of land area on earth (Schneider et al., 2009). Despite this seemingly small extent, urbanization has far-reaching and lasting impacts on ecosystems. Urban areas reduce surrounding water quality, causing eutrophication, harmful algal blooms, and reduction in downstream fisheries and shellfish (Freeman et al., 2019). Air quality is reduced in cities and is especially degraded during rapid urban expansion (Xia et al., 2014). Urbanization changes bird migratory habits (Bonnet-Lebrun et al., 2020) and negatively affects wildlife health (Murray et al., 2019). While urban centers have clear negative impacts on ecosystems, they also create novel, human-centered ecosystems that are important to study.

Urban ecosystems are home to diverse and complex ecological communities that often go overlooked by urban policy and developers (Beatley & Bekoff, 2013; Chan et al., 2014). Urban ecological communities can provide untapped opportunities for greater conservation efforts (Chan et al., 2014). Cities are hot spots for vegetative species richness, with higher native and introduced species numbers. While urbanization is generally associated with rare species declines, in some cases, more rare native plant species are found in urban areas when compared with their rural counterparts (Kowarik, 2011). Novel ecosystems arising from mixing native and introduced species under urban pressures cannot replace natural systems; however, they can support self-sustaining rare and endangered species populations (Kowarik, 2011). In some cases, urban areas serve as refuges for non-native endangered species (Shaffer, 2018).

Over the last ~200 years, urbanization has dramatically increased; in 1800, 10% of the global population lived in urban centers, as of 2017, this increased to over 55% (Ritchie & Roser, 2018) and by the year 2050, 70% of the global population is projected to live in urban areas (Chan et al., 2014). As the human population and urbanization inevitably expand, these human-constructed ecosystems will increase their ecological footprint which will amplify their influence on surrounding and distant environments (Allen, 2009). How do we ensure that this expansion happens in a sustainable and conscientious way in regard to our surrounding ecological communities and the newly assembled communities that exist in urban environments? Understanding ecological community dynamics in the human-built environment will enable planning for ecologically sustainable urban systems as cities grow (Niemelä, 1999). Through careful study of urban ecosystems, we can learn how to develop urban areas in sustainable ways. Assessing and monitoring urban ecosystem health is important to ensure sustainable urban growth while minimizing impacts on surrounding habitat.

Diverse species assemblages in cities play important functional roles in the urban ecosystem and offer supportive ecosystem services to their human counterparts. For example, urban centers generate local air pollutants that negatively affect human health manifesting as increased respiratory illness and cardiovascular disease (Gómez-Baggethun et al., 2013). Urban vegetation, specifically trees and shrubs, reduce these pollutants through filtration (Gómez-Baggethun et al., 2013). Urban centers create islands of heat generated from anthropogenic sources and heat stored in built structures which degrades the environment and raises energy use and ground-level ozone (Rizwan et al., 2008). Urban vegetation reduces these effects through evaporative transpiration, solar radiation reflection, and shade provision (Gómez-Baggethun et al., 2013). Ecological infrastructure such as waterbodies can also help to regulate heat (Gómez-Baggethun et al., 2013). Effective urban ecosystems can thus positively impact human health.

Humans directly benefit from interaction with nature. People self-report greater well-being when real or perceived urban diversity is higher in urban greenspaces (Schebella et al., 2019; Shwartz et al., 2014; Wood et al., 2018). Native diversity can contribute to an individual's 'sense of place,' a measure of human connection with and attachment to locations (Cross, 2001). Increased sense of place can enhance an individual's propensity toward environmental stewardship (Chapin et al., 2011). As people migrate away from rural areas and congregate in urban centers, many people's primary interaction with nature will be in these urban communities (Miller, 2005; Soga et al., 2016). Fostering diverse urban ecological communities and highlighting the direct services they provide connects city-dwellers to nature, helping to create emotional connections and bridge the gap between the perceived human-nature divide (Miller, 2005; Soga et al., 2016). Connecting our human communities to nature is essential to minimizing the negative impacts that urban expansion has on the surrounding ecological community.

Assessing and Monitoring Urban Ecosystems

A healthy ecosystem is defined as “a biological system… [whose] inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed” (Karr et al., 1986, p. 6). Urban ecosystems may have a much different ‘inherent potential’ than what Karr envisioned when forming this definition. Nonetheless, this definition sets goals for even the most human-dominated systems: stability, resiliency, and independence. These goals provide clear direction for measuring and monitoring ecosystem health in urban settings.

Biological diversity is an overarching measure for ecosystem stability and resiliency. High biodiversity leads to overlapping functional roles of species, which results in more stable ecosystems than those with less diversity (Purvis & Hector, 2000). 95% of diversity studies found a positive relationship between diversity and ecosystem functioning (Purvis & Hector, 2000). Biodiversity directly supports ecosystem stability because organisms’ traits influence ecosystem processes (Chapin et al., 2000). This influence is easiest to understand in the context of invasive species. For example, the introduction of Fire Tree (*Myrica faya*) to Hawaiian ecosystems increased system nitrogen input fivefold due to its enhanced nitrogen-fixing trait relative to native species; this introduction changed Hawaiian forest structure and function (Vitousek et al., 1987). When biodiversity changes, ecosystem processes can suffer. Thus, biodiversity is a key measure of ecological community health.

A second measure of ecosystem health, particularly in urbanized areas, is degree of habitat connectivity, also measured as its antithesis: habitat fragmentation. Fragmentation and connectivity loss are caused when floral and faunal movement corridors are cut off, or large habitat patches are split so no or reduced movement is allowed between them, negatively affecting reproduction and species survival (Franklin et al., 2002). This connectivity loss can result in a decline of biodiversity and resiliency as plants and wildlife get stranded on urban islands or lose connection to vital resources. Measuring the degree of fragmentation in urban areas can be a key community health and integrity indicator.

A final measure of ecological community health and resiliency is invasive and non-native species presence. This is of particular concern in urban environments because cultivated invasive and non-native species often escape urban areas and contribute to native species displacement outside the urban grid (Gaertner et al., 2017). In recent years, concern has grown within the scientific community regarding global ecological homogenization due to similar species occupying urban habitats (Gaertner et al., 2017). Non-native urban species contribute to the high biodiversity we see in cities, but they can also undermine ecological functioning and community resiliency (Gaertner et al., 2017) and thus must be monitored in tandem with overall biodiversity.

Unfortunately, measuring these ecological health factors is complex and is often limited by funding, time, and resources. While many cities have created individualized biodiversity and ecosystem sustainability goals with varying degrees of success (Pierce et al., 2020), there is no universally accepted index or protocol for measurement. One widely applied indicator, City Biodiversity Index (CBI), has been developed by the Singapore National Parks Board (Chan et al., 2014). This comprehensive protocol uses 23 urban biodiversity indicators across three categories: native biodiversity, ecosystem services, and governance and management. When assessed, each index receives a score between 0-4, giving a total possible score of 92. CBI has been implemented in dozens of cities and has seen some success in monitoring urban ecosystem health and setting diversity goals (Uchiyama & Kohsaka, 2019). However, implementing CBI is not always successful due to budgetary and resource constraints (Uchiyama & Kohsaka, 2019).

Indicator species have long been used as surrogate measures for ecosystem attributes that are difficult to directly measure due to inherent complexity and time and resource restraints (Landres et al., 1988; Lindenmayer & Likens, 2011). The use of indicator species is based on the premise that changes in the ecological community and surrounding environment are cumulatively reflected in specific, measurable traits of a present species or group of species (Lindenmayer & Likens, 2011). Through monitoring indicator species, ecological managers can detect changes in the environment and can immediately work to mitigate negative impacts.

Indicator species successfully direct management in natural areas. Yellowstone National Park currently monitors their bat population as an indicator of ecosystem health and community and habitat changes (Treanor et al., n.d.) and the Northern spotted owl (*Strix occidentalis caurina*) directs old-growth forest management in the Pacific Northwest (Caro & O’Doherty, 1999). The National Park Service monitors bird assemblages and invasive plant species as indicators for ecological integrity and uses these communities to direct land management (Wurtzebach & Schultz, 2016). Municipalities manage large portions of urban areas and indicator species such as those that direct management in natural areas can direct municipal management of urban ecological communities.

The use of indicator species is not without its limitations. Several literature reviews have assessed the criteria used across studies to choose indicator species; across these reviews, a clear trend of pitfalls in the use and determination of indicator species emerges. Many selected indicator species have poor justification for selection, lack transferability to other landscapes and ecosystems, and are not thoroughly tested to show correlation with what they are purported to indicate (Heink & Kowarik, 2010; Lindenmayer & Likens, 2011; Siddig et al., 2016). Relying on a single species to speak for an ecosystem is a bit of a dream for managers; no one species can be an all-encompassing indicator (Heink & Kowarik, 2010), however, specific species may tell enough to inform management decisions. When time and resources are limited, specific, urban-sensitive indicator taxa can be used to measure ecosystem health: birds and butterflies.

Indicator Taxa for Urban Ecosystems

I propose two taxa be used in tandem to assess and monitor the previously identified three crucial measures of urban ecosystem health: biodiversity, fragmentation, and invasive species. Managers can use birds and butterflies to monitor these measures when municipalities have limited resources to implement larger assessment plans like CBI. The joint use of butterflies and birds to assess these measures can give sufficient information to effectively inform management decisions and set goals for the ecological communities in urban centers.

Butterflies are particularly sensitive species that are found within the urban grid. They are specifically recommended by CBI as a ‘core indicator’ for measuring biodiversity based on their universal presence in cities and the depth of knowledge on the taxa (Chan et al., 2014). Butterflies have been extensively studied with regard to urban fragmentation because they vary in dispersal ability and quickly react to environmental changes (Bergerot et al., 2011; Clark et al., 2007; Lizée et al., 2011). Butterfly species richness and abundance are linked to native plant cover (Burghardt et al., 2009). These traits indicate that managers could use butterfly populations to assess and monitor an urban ecosystem’s biodiversity, degree of fragmentation, and native plant cover. After initial assessment, changes in the abundance or presence of sensitive butterfly species could alert managers to negative urbanization effects allowing managers and city planners to quickly address issues.

The avian community is widespread throughout the urban grid and comprises varying species dependent on habitat type and resource availability. Birds are included in CBI as a ‘core indicator’ for biodiversity assessment and monitoring, like butterflies, based on their universal presence in cities and the fact that they are well researched and commonly surveyed (Chan et al., 2014). Bird community assemblages, abundance, and species richness are correlated with urban fragment size and degree of urbanization (Crooks et al., 2004). Bird species fall on a gradient of urban-sensitive to urban-enhanced (species which thrive in urban environments) (Crooks et al., 2004), the ratio of which could measure effects of urban change on the ecological community. Birds are generally more mobile than butterflies, thus they could serve as urban fragmentation indicators at a larger scale, making the joint use of birds and butterflies as indicators for urban fragmentation more robust. Bird abundance and species richness correlate with urban forest and woody plant diversity (Kati et al., 2004; Sanesi et al., 2009), which makes birds good urban plant diversity indicators. Bird community changes across the urban gradient, like butterflies, could alert managers to biodiversity and tree diversity loss and increasing fragmentation, enabling quick management changes to mitigate the effects urbanization has on urban ecosystem health.

Conclusion

While indicator species have been used to detect impacts of urbanization and urban sprawl on surrounding ecological communities, there is a lack of identified indicator species for measuring the health of the urban ecosystem itself. A healthy urban ecosystem directly benefits the health and well-being of its human and non-human inhabitants and minimizes negative impacts on surrounding ecosystems. Butterfly and bird communities are both sensitive to key measures of ecological health: biodiversity, habitat fragmentation, and invasive species, making them jointly a robust indicator for the impacts of urban change on the greater community. When faced with resource constraints for assessment and monitoring, municipalities can use these joint indicators as surrogate measures for the overall health of their urban ecological communities.

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CHAPTER 2. GRANT PROPOSAL

Effectiveness of IUCN Protected Area Categories for the Conservation of North American Alpine Plant Communities

Abstract

Ecological communities in protected areas that also allow recreation are particularly vulnerable to human disturbance. IUCN created widely used categories encompassing different protection levels based on an area's management plan. Evidence points toward increased recreational desirability and human visitation when an area falls under one of these categories. Little work has been done to determine if IUCN protected area categories accurately reflect the effective level of protection to plant communities. The following study will empirically test the effective level of protection for sensitive alpine plant communities in IUCN protected area categories and will determine if significant differences exist between categories. Additionally, the study will reveal which categories would benefit the most from increased protection and conservation efforts. Alpine plant communities in the Colorado Rocky Mountains will be sampled for native and invasive species abundances and functional group-based community structure to measure differences in communities in distinct protection categories. Results will be useful to conservation efforts in alpine plant communities and could be applicable to other ecosystem types. With limited funding, results can be used to direct conservation efforts to truly vulnerable ecological communities.

Objectives

This study will evaluate the effectiveness of the International Union for Conservation of Nature's (IUCN) Protected Area Categories (PAC) for preserving sensitive ecological communities. When an area is designated a PAC, anthropogenic disturbance may increase because the area is deemed desirable for recreation. Ecological communities in these areas are therefore particularly vulnerable to disturbance, despite their protected status. Determining the realized protection of ecological communities in IUCN PACs through a case study of sensitive alpine plant communities will provide direct evidence for the effectiveness of PAC protection. This study aims to assess the impact of a human disturbance gradient on sensitive alpine plant communities protected under IUCN PACs through measurement of native and invasive plant species abundances and functional group-based community structure within different PAC levels. The results can be used to direct limited funding to PACs that are identified as the most vulnerable.

Anticipated Value

Ecological communities worldwide are threatened by human encroachment and anthropogenic disturbance. These communities represent important reserves of biodiversity and contribute to greater conservation efforts (Dudley, 2008). Reduction of diversity in alpine plant communities can make them less resilient to disturbance over time (Chardon et al., 2018). Alpine plant communities are particularly sensitive; they are increasingly at risk due to their limited ability to expand the upper limits of their ranges under climate change scenarios (Wipf et al., 2009). Additionally, alpine areas are increasingly valued as recreation sites as people seek to escape from summer heat (Forbes et al., 2004). Alpine communities provide a perfect case study to determine the effectiveness of IUCN protection categories. While this study will specifically examine alpine plant communities, the findings could be applied to other ecosystem types protected under IUCN PACs. The results of this study will be useful for planning and directing future conservation efforts by enabling pointed allocation of limited funding to areas that will benefit the most.

Questions & Hypotheses

Q1. How do differing levels of human disturbance, as defined by IUCN PACs, impact alpine plant communities?

- **H1.** Increased human disturbance in alpine plant communities, as defined by IUCN PACs, will change community structure, decrease species richness, evenness, and cover, and increase dominance of non-native species because disturbance changes competitive variables in plant communities.

Q2. Are there significant differences in alpine community response between IUCN PACs and which categories would benefit the most from shifting to a new category?

- **H2a.** Alpine communities in low IUCN PACs will have greater amounts of bare ground, invasive species and distinct community structure when compared to higher IUCN PACs because different PACs represent different levels of allowed human disturbance.
- **H2b.** Communities under the lowest level of protection will benefit the most from an increase in protection level because they have the least protection from human disturbance.

Literature Review

Landscapes in North America have been set aside and protected from development and human activity for over 100 years, starting with the founding of Yellowstone National Park in 1872 (Djossa, 2018). As the human population grew and anthropogenic forces such as urbanization and land conversion transformed ecosystems, concern also grew regarding the protection of natural landscapes for their inherent value, ecosystem integrity, and human enjoyment (Nash & Miller, 2014). In North America today, 12% of landcover falls under some form of formal protection as defined by IUCN (NAWPA Committee, 2016). Achieving a balance between offering places for humans to experience the outdoors and offering sensitive ecosystems protection from human disturbance is complicated. Government funding is increasingly inadequate for managing protected areas and supplementary income from eco-tourism is often needed (Whitelaw et al., 2014), which places managers in a difficult position. This raises the question: how effective is the sanctuary offered to sensitive communities in protected areas where human recreation is a joint priority?

Protected areas imply a level of preservation from anthropogenic disturbances, but often-times the act of designating a protected area draws public attention and attracts human disturbance in the form of eco-tourism and recreation. In fact, a national designation accelerates degradation in natural areas due to increased visitation, undermining the very goal of preservation (Becker, 1981), as is the case in national parks, monuments and other wilderness areas. Furthermore, wilderness visitation is on the rise; Over a 40-year span, total wilderness recreation use increased 10-fold (Cole & Landres, 1996). Increased recreation causes ecological disturbance through trampling, redistribution and removal of materials through livestock grazing and firewood gathering, general disruption of native fauna, and pollution of waters from human waste and foreign substances (Cole & Landres, 1996). Conservation of biodiversity is a primary concern for protected areas (Dudley, 2008), yet places with high biodiversity attract more human visitors (Siikamäki et al., 2015) thereby compromising protection at these sites.

IUCN developed a globally recognized categorical ranking system for protected areas ranging from most protected (I - nature reserves) to least protected (VI - protected area with sustainable use of natural resources) as determined by an area's management objectives (Dudley, 2008). The definitions of these categories imply a gradient of "naturalness," human disturbance, and protection level (Leroux et al., 2010). IUCN PAC categories are commonly used in the context of conservation planning and are outlined for use as a measure of effective protection in the newly published North American Strategy for Alpine Plant Conservation (Ripley et al., 2020). However, questions have been raised about the effectiveness of PACs in preserving ecosystems and whether effectiveness differs between categories (Jones et al., 2018; Leberger et al., 2020). Detected forest loss was most pronounced in the highest protection categories (Leberger et al., 2020) and human influence was not negatively correlated with increasingly protected categories as would be expected (Leroux et al., 2010).

Alpine communities are some of the most sensitive and threatened assemblages on earth (Crisfield et al., 2012). Current threats to these ecosystems include increasing recreational visitors (Forbes et al., 2004) and abiotic changes associated with climate change (Wipf et al., 2009). Because alpine ecosystems are inherently sensitive to disturbance and have slow recovery rates (Capers & Taylor, 2014), these communities are ideal places for uncovering the realized protection of sensitive communities in IUCN PACs. Thus, the proposed study will not only benefit the future conservation of alpine communities in North America but will also offer support to other conservation efforts by revealing the realized protection of IUCN PACs.

Despite global acceptance of IUCN PACs, few studies have tested the effectiveness of these categories with empirical ecological data, and fewer still focus on alpine communities. Alpine communities offer a distinct opportunity for testing these categories and will benefit immensely from determining the realized level of threat within these categories. Identifying the realized protection of communities in IUCN PACs will enable pointed direction of limited conservation and recreation management funding to areas that stand to benefit the most.

Methods

Study Site

To test the effectiveness of IUCN PACs in North American alpine plant communities, I will sample plant communities in the Colorado, Rocky Mountain, alpine tundra. IUCN PACs have not yet been determined for alpine plant communities in North America, therefore I will overlay management data layers on an existing map of sensitive alpine plant community sites in the Colorado Rocky Mountain region (Ripley et al., 2020). After identification of PAC for each alpine plant community site, I will use ArcGIS to randomly generate 30 sampling points in each PAC present, points will be distributed randomly across all identified sites classified under a single PAC. I will replace points that are within 10m of structures, roads, trails, or campgrounds with a subsequent random point. At each point, I will randomly place and sample three quadrats. I will average values measured to better represent the vegetative community around the sampling point. I will complete all sampling during the alpine growing season in July and August 2022.

Data Collection

To test whether increased human disturbance, as designated by IUCN PACs, impacts alpine plant communities, I will collect the following data in each quadrat: species present, percent cover of each species, bare ground, and litter cover. I will take representative photographs of each species for future reference and for identification verification. I will use these data to calculate species richness, dominance, non-native cover and community structure based on functional groups previously identified by International Tundra Experiment (evergreen shrubs, deciduous shrubs, graminoids and forbs) (Elmendorf et al., 2012). I will gather additional climate data including mean annual temperature and precipitation for each point from the nearest weather station.

Data Analysis

To address H1, I will first compare community composition among alpine plant communities by running a NMDS ordination based on species bray-curtis distances to visually assess differences among IUCN PACs, followed by a permanova using vegan (Oksanen et al., 2020). To test disturbance effect on species richness, I will fit a generalized linear model (GLM) with Poisson distribution, species richness will be the response variable and IUCN PAC category along with climate variables will be predictors. Additional GLMs will be fit with the same predictors and percent bare ground, percent cover and percent non-native species as response variables.

To address H2a, I will use ANOVA to assess mean differences among IUCN PACs' percent cover of functional groups, percent bare ground, overall percent cover and percent non-native species. I will complete post-hoc testing with Bonferroni correction if distinct differences are found.

To address H2b, I will calculate Shannon's diversity index for each PAC and compare values. Difference in means between each category determined through previous ANOVA and differences in Shannon diversity index values will be assessed to determine which categories will benefit the most from an increase in IUCN protection level. All calculations will be performed in R v1.3 (R Core Team, 2020).

Potential Negative Impacts

Potential negative impacts for this study will be minimal. To reach sampling locations, we will use designated roads and hiking trails as much as possible. We will not collect any plants during this study. Some plant trampling and wildlife disturbance is expected but we will take precautions, such as remaining on trails whenever possible and enforcing low noise levels, to keep these disturbances to a minimum.

Project Timeline

Date	Goals	Deliverables
July 17-August 13, 2022	Field sampling	Data
August 15-31, 2022	Data entry	Data sheet
September 1-14, 2022	Data analysis	Statistical results
September 15-October 31, 2022	Manuscript preparation	Manuscript

Budget

Item	Justification	Cost/unit	Quantity	Total
Travel				
4x4 rental vehicle	Travel to sites	$35/day	28 days	$980
Rental taxes & fees		$300		$300
Vehicle insurance		$23/day	28 days	$644
Gas	Travel to sites	$3.4/gal	~2000mi/22mpg = 90.9gal	$309
Per diems				
Food & incidentals	Food for duration of sampling	$59/day x 2	28 days	$3,304
Lodging	Accommodations for duration of sampling	$96/day	28 days	$2,688
Student assistant	To assist with data collection	$80/working day	21 days	$1,680
Sampling supplies				
Quadrats	For vegetation sampling	$35	2	$70
All weather notebooks	For data recording	$8	2	$16
Total				$9,991

Qualifications of Researcher

Dena Bergman denabergman@gmail.com

Education

- **Master of Science in Environmental Biology** *(in progress)*

Regis College, Regis University, Denver, CO GPA TBD

- **Bachelor of Science in Conservation Biology and Ecology** December 2018

School of Liberal Arts and Sciences, Arizona State University, Tempe, AZ summa cum laude, GPA 4.0
New American University Scholar Award

Highlighted Training, Research & Volunteer Work

- **Denver Mountain Parks (DMP) Ecological Community Research** August 2020 – April 2021
  - Worked as part of a team to design and execute a sampling protocol for sampling the plant, dung beetle and bison communities, and soil characteristics and nutrients in Daniels Park.
  - Analyzed data using R and communicated results to DMP managers to inform management of the ecosystem.
- **Croke Reservoir Management Plan** August 2020 - January 2021
  - As part of a team, designed improvements for Croke Reservoir to mitigate damage caused by human misuse.
  - Followed NEPA protocol to assess environmental impacts of proposed plan.
  - Plan is currently in use by the Northglenn Community Association.
- **Clear Creek Macroinvertebrate Sampling** September 2020
  - Sampled macroinvertebrate communities at upper and lower Clear Creek.
- **Self-Directed Research and Poster Design** August 2018 - January 2019
  - Performed research focusing on native plants in residential lawns. Required design of a research question, data collection, statistical data analysis, literature review, poster design and presentation at the Central Arizona Phoenix Long Term Ecological Research (CAP LTER) all scientists meeting.
- **R Studio Workshop** November 2018
  - Learned methods for data analysis and visualization using R.
- **Wetland Ecosystem Ecology Lab Volunteer** January - May 2016
  - Measured growth of wetland plant species in the Tres Rios Wetlands in Phoenix AZ.
  - Collected and prepared wetland water samples for various lab tests.

Professional Work History

- **GIS Project Intern** January 2022 - Present

Betty Ford Alpine Gardens, Vail, CO
  - Created maps of IUCN alpine areas to fulfill goal of North American Alpine Conservation Strategy.

- **Seasonal Member & Visitor Services Representative** June 2021 - January 2022

Denver Botanic Gardens, Denver, CO
  - Greeted and directed guests and managed gardens memberships.

- **Customer Experience Agent** *(Temporary pandemic appointment)* June 2020 - March 2021

Slack Technologies, Denver, CO
  - Assisted users with software troubleshooting via email.

- **Food and Beverage Coordinator** October 2019 - June 2020

Slack Technologies, Denver, CO
  - Managed internal kitchen and coffee bar and planned food service and events for an office of 250+ employees.
  - Maintained existing and built new relationships with over 50 vendors.

- **Research Aide** May - December 2018

Sharon J. Hall Research Lab, Tempe, AZ
  - Pinned, sorted and labeled thousands of bees and wasps in preparation for the Hasbrouck Insect Collection.
  - Completed 200+ yard surveys focusing on identifying plants in residential lawns.
  - Aided in seed collection, tissue sampling, and transpiration measurements of urban plant species.
  - Performed data entry and data quality verification tasks with impeccable accuracy and wrote technical reports.

- **ASU Vascular Plant Herbarium Intern** January - May 2018

Arizona State University Vascular Plant Herbarium, Tempe, AZ
  - Mounted, labeled and archived hundreds of plant specimens for the herbarium collection.
  - Entered collection data into an online database and merged data with hundreds of specimen photographs.

Skills

ArcGIS, Google Suite, Microsoft Suite, Slack

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CHAPTER 3. JOURNAL MANUSCRIPT

Rocky Mountain Alpine Plant Communities Respond Dynamically to Environmental Change

Abstract

Alpine ecosystems harbor high endemic species richness and diversity which are threatened by changes in abiotic climate conditions that promote invasion by nonnative species. Invasive species introduction is correlated with human disturbance, which is increasing in protected areas like national parks where visitation numbers sky-rocketed in the past eleven years. Local increases in species richness and diversity due to invasive species may be accompanied by regional community homogenization as ubiquitous invasive and generalist species replace alpine specialists. I used eleven years of alpine vegetative data gathered using GLORIA protocol to assess local and regional alpine plant community changes in three national parks located in the Rocky Mountains, U.S.A and whether invasive species are expanding into alpine habitat. Locally, nonnative species proportion did not change over the course of the study, implying inherent resistance to invasion, however, species richness and diversity responded differently within each park, indicating high levels of turnover. Changes in plant communities were more consistently predicted by precipitation than by time. Regionally, communities are not becoming more homogenous over time and show relative stability. Sighting of a new invasive species in the lowest elevational ecotone of Rocky Mountain National Park may indicate that invasive species are expanding into alpine habitat on longer timescales and that alpine resistance to invasion may be waning. Alpine managers have a window of time to implement protection

measures in national park alpine habitat before invasive species begin to significantly impact alpine ecosystems.

Introduction

Alpine ecosystems, ecological communities above treeline, are particularly vulnerable to climate change, which contributes to biodiversity loss, shifts in species' ranges and mismatched timing of key species interactions (Halloy & Mark, 2003; Parmesan, 2006). These threats are especially severe in mountainous habitats because alpine ecosystems are often isolated and surrounded by dispersal barriers such as ridges, valleys and large inhospitable geographic areas (Jurasinski & Kreyling, 2007). Additionally, alpine ecosystems suffer shrinking habitat caused by encroachment of subalpine vegetation (Inouye, 2020). Loss of alpine ecosystems can result in reduced ecosystem services such as water resources, biodiversity, as well as aesthetic and recreational opportunities (Malanson & Fagre, 2013). Understanding how alpine vegetative communities respond to a changing climate is important if we are to sufficiently protect these shrinking islands of biodiversity.

Alpine communities are characterized by a high proportion of endemic species because the isolated nature of mountain communities fosters speciation (Cox et al., 2016). Endemic alpine species are threatened by invasive and nonnative species introduced at lower elevations, which expand their altitudinal range as climate warms (Alexander et al., 2011). Invasive species impair interactions between native plants and pollinators (Muñoz & Cavieres, 2008), replace native flora, and change ecosystem functioning (Pauchard et al., 2009). There are conflicting schools of thought on the invasibility of alpine ecosystems with some evidence showing upward expansion of climatic generalists (Alexander et al., 2011) and others claiming alpine ecosystems are resistant to invasion due to abiotic factors such as shortened growing season, limits on seed dispersal, and low temperature (Inouye, 2020). However, invasive plants are associated with changes in temperature and increasing human disturbance (Dainese et al., 2017) which points to an increase in invasion as time progresses and these forces inevitably escalate. Increasing human use of alpine areas can increase spread of nonnative species on local, regional and global scales (Pauchard et al., 2009). Initial increases in local species richness due to migrating species may be accompanied by a decrease in regional diversity as ubiquitous species replace alpine specialists (Britton et al., 2009; Jurasinski & Kreyling, 2007; Ross et al., 2012).

While the upward expansion of subalpine vegetation into alpine ecosystems has been well studied, evaluation of alpine regional diversity has received little attention (Jurasinski & Kreyling, 2007). Additionally, few studies have focused on North American alpine ecosystems with most research conducted in Europe at smaller regional scales (Britton et al., 2009; Jurasinski & Kreyling, 2007; Ross et al., 2012). One reason for the lack of studies in North American alpine systems is the absence of long-term data that follow standardized protocols.

In response to the looming threat of alpine ecosystem loss, Grabherr et al. (2000) established The Global Observation Research Initiative in Alpine Environments (GLORIA), a global protocol for gathering long-term data on alpine vegetation, which is now in place in over 137 target regions. Between 2009 and 2011, three GLORIA sites were established in the U.S. Rocky Mountains in Rocky Mountain National Park, Great Sand Dunes National Park and Preserve and Yellowstone National Park and their vegetative communities were sampled regularly until 2020. Sampled summit elevations range from 3,122 to 4,000m and span 8.5099 degrees of latitude. All sites are located within U.S. national parks that allow recreational activities, drawing increasingly larger crowds from around the world as people seek outdoor opportunities for relaxation, solitude, and sport (McGivney, 2022; National Park Service, 2022).

Initial observations at these three sites noted the presence of a ubiquitous invasive species *Taraxacum officianale*, the common dandelion (Ashton, 2011; Ashton et al., 2010; Kuhn et al., 2015). Changes in U.S. national parks' alpine communities have not been fully explored in light of increasing human disturbance caused by recreational visitors.

In this study, I assess how alpine vegetative communities in the U.S. Rocky Mountains are changing over time at both local and regional scales and to what degree invasive species are contributing to community changes. To evaluate how alpine ecosystems are changing, I explored several hypotheses: *H1*) At the local scale alpine plant communities will have more invasive species, greater diversity, and greater heterogeneity over time due to the migration of nonnative species into alpine environments as average annual temperature and human disturbance increase; *H2*) At the regional scale alpine plant communities will be more homogeneous over time due to ubiquitous invasive and subalpine species expanding their ranges into alpine ecosystems. At the local scale, I predict that there will be greater plant species richness with more nonnative species and greater vegetative diversity in plots sampled in 2020 than in plots sampled in 2009. On the regional scale, I predict that there will be decreased species richness and vegetative diversity among sites sampled in 2020 than in plots sampled in 2009. As one of the first studies to explore broad regional changes in alpine vegetative communities of the Rocky Mountains, the results of this analysis will provide critical information for the protection of these fragile ecosystems.

Methods

Dataset

The dataset used in this analysis was gathered, built, and published for research use as a collaborative effort between the Rocky Mountain Inventory & Monitoring Network, the National Park Service Inventory & Monitoring Project, and the Colorado Natural Heritage Program

(Rocky Mountain Network Inventory and Monitoring & Greater Yellowstone Network Inventory and Monitoring, 2021). All three study sites used GLORIA long-term monitoring protocol to select sites and collect data.

Study areas and site selection

Great Sand Dunes National Park and Preserve (GSD) is located in south-central Colorado and represents to southernmost study area in the Rocky Mountains, Rocky Mountain National Park (RMN) is located in north-central Colorado and represents the center study area, and Yellowstone National Park (YNP) is located in northwestern Wyoming and represents the northernmost study area (Fig. 1). Between establishment and last sampling year of long-term monitoring plots, the three parks have seen a dramatic increase in recreational visitation and annual temperature and a decrease in annual precipitation (Fig. 2). Each site is characterized by tundra vegetation located above treeline.

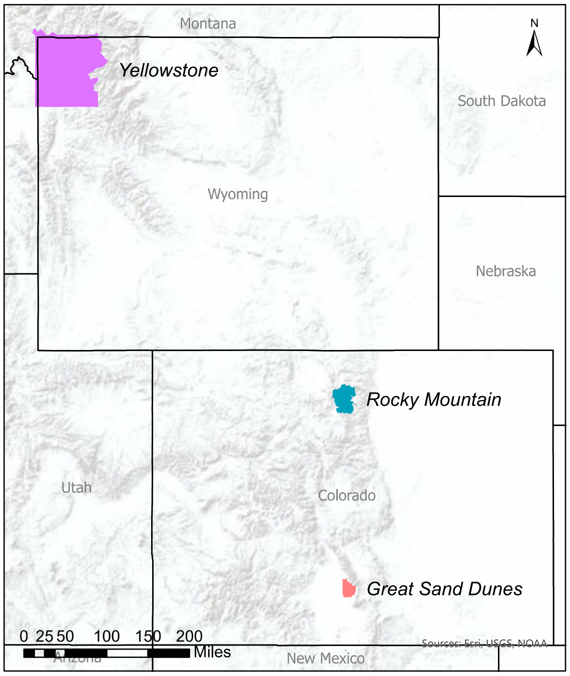

Figure 1. The three National Parks in the Rocky Mountains where study sites are located. Sites stretch from South-central Colorado to North-Western Wyoming.

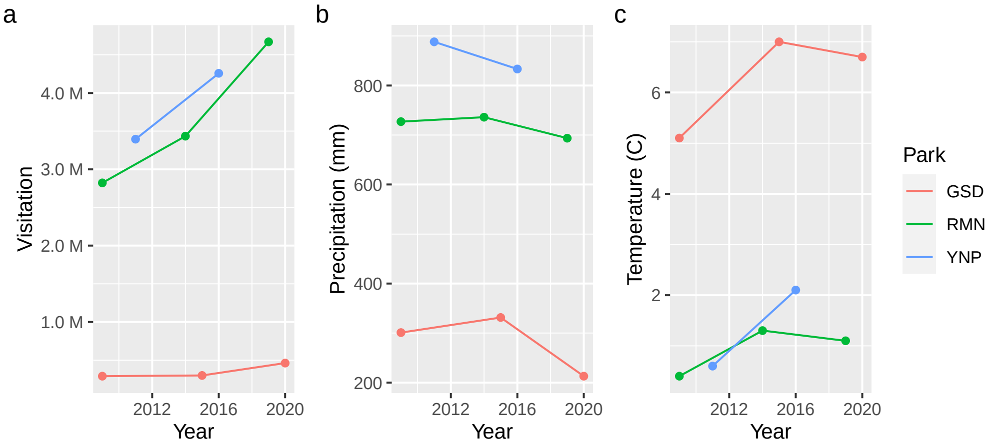

Figure 2. Changes in annual visitation, mean annual precipitation and mean annual temperature over sampling period (Ashton, 2011; Ashton et al., 2010; Kuhn et al., 2015; Meteoblue, 2022; National Park Service, 2022).

GLORIA protocol for site selection consists of four summits arranged along an elevational gradient as best as possible, with summits representing four ecological transition zones: nival/subnival, subnival/upper alpine, upper/lower alpine and lower alpine/subalpine (Pauli et al., 2015). All summits groupings are required to be within the same climate region and are chosen based on vegetation patterns characteristic of that region (Pauli et al., 2015). The four peaks selected in RMN range from subalpine to upper alpine and include Shoulder of Mount Ida, Jackstraw and two unnamed peaks (Ashton, 2011; Table 1). The peaks selected at GSD range from subalpine to nival and include four unnamed peaks (Ashton et al., 2010; Table 1). The four peaks selected in YNP are all in the lower alpine ecotone and include Boundary Line Peak, Stone Crop Peak, Snow Pipit Peak and Wolf Scat Peak (Kuhn et al., 2015; Table 1).

Table 1. Selected sampling peaks, vegetation zones and sampling years.

Park	Summit code	Summit name	Elevation (m)	Vegetation zone	Sampling years
YNP	BLP	Boundary Line Peak	3195	Lower alpine	2011, 2016
YNP	SCP	Stone Crop Peak	3122	Lower alpine	2011, 2016
YNP	SPP	Snow Pipit Peak	3169	Lower alpine	2011, 2016
YNP	WSP	Wolf Scat Peak	3124	Lower alpine	2011, 2016
RMN	GLA	Shoulder of Mount Ida	3862	Upper alpine	2009, 2014, 2019
RMN	PIK	Unnamed	3715	Lower/upper alpine	2009, 2014, 2019
RMN	VQS	Unnamed	3623	Lower alpine	2009, 2014, 2019
RMN	JSM	Jackstraw	3520	Lower alpine/subalpine	2009, 2014, 2019
GSD	SIX	Unnamed	4000	Alpine/nival	2009, 2015, 2020
GSD	PAD	Unnamed	3870	Upper alpine	2009, 2015, 2020
GSD	MUD	Unnamed	3700	Lower alpine	2009, 2015, 2020
GSD	HUK	Unnamed	3550	Lower alpine/subalpine	2009, 2015, 2020

Sampling Methods

Following standardized GLORIA sampling protocols, each summit is divided into eight summit area sections, four extending from the top to 5m below the summit, each facing one cardinal direction, and four extending from 5m-10m below summit elevation, each facing one cardinal direction (Pauli et al., 2015; Fig. 3). A permanent 3x3m quadrat cluster is placed facing each cardinal direction ~5m below the summit, targeting ecological transition zones (Pauli et al., 2015). A quadrat cluster consists of a 3x3m square divided into a 1x1m grid, in each 1x1m corner grid, surface type percent cover is estimated and all species are identified and their percent cover estimated (Pauli et al., 2015). Additionally, the eight summit area sections are surveyed to build a complete species list and the abundance of each species in five abundance categories (very rare, rare, scattered, common and dominant) is estimated (Pauli et al., 2015). Lastly, each species is categorized as native or nonnative (Ackerfield, 2015; http://www.bonap.org/).

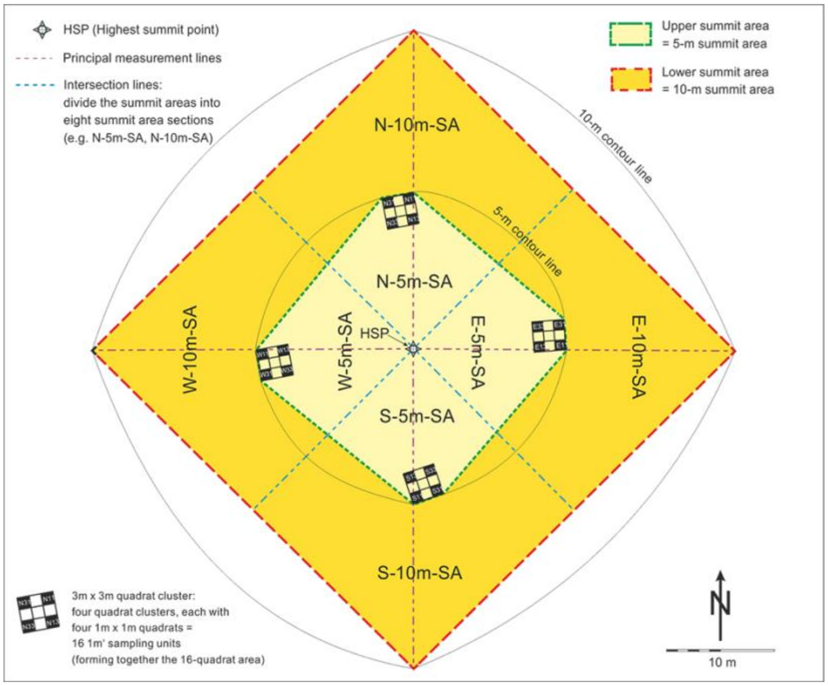

Figure 3. GLORIA sampling design (Pauli et al., 2015)

Local scale statistical analysis (H1)

To test local vegetation community change in the three parks, I fit a series of generalized linear mixed effect models at each of the parks with sampling year, annual visitation, mean annual temperature, and mean annual precipitation as fixed effect predictors, I accounted for elevational ecotone by using peak as random effect and selected the best model based on AIC score. To test if species richness is increasing, I used species richness of summit area sections as a response variable with a Poisson regression. To test if nonnative species are increasing, I used the proportion of nonnative species in sample plots as a response variable with a logistic distribution. Finally, to test if diversity and evenness are changing, I calculated Shannon's diversity index (SDI) in summit area sections for all sampling years and to better fit assumptions of the statistical analysis, I log-transformed the index to use as a response variable.

Regional scale statistical analysis (H2)

To test if species richness is decreasing on the regional scale, I fit a series of generalized linear mixed effect models with a Poisson regression. I used park, sampling year, annual visitation, mean annual temperature, mean annual precipitation as fixed effect predictors, summit as a random effect, and species richness of summit area sections as a response variable, and chose the best model based on AIC score. To test if diversity is decreasing on the regional scale, I calculated SDI for all years across the three parks and used the log-transformed index as a response variable. To test regional alpine vegetation community change among the three parks, I calculated the relative abundance of species in each 1x1m plot and then calculated Bray-Curtis distances for all sampling years. I then performed an NMDS to find axes of explanation for the ecological distance between plots using commands in the R package vegan (Oksanen et al., 2020). I assessed grouping of parks differentiated by sampling year visually. Finally, I grouped each park into first and last sampling categories and calculated the mean Bray-Curtis dissimilarity scores for each plot. I then performed a t-test to quantify whether mean dissimilarity between first and last sampling dates differed across parks. I conducted all statistical analyses in R (R Core Team, 2020).

Results

Across all parks and sampling dates, a total of 416 species were recorded with *Minuartia obtusiloba* and *Poa glauca* being the most commonly occurring species. Within parks, dominant and commonly occurring species changed between first and last sampling dates (Table 2). The ubiquitous invasive species *Taraxacum officinale* was found at all parks and an additional invasive species, *Hieracium aurantiacum*, appeared on the lowest elevational ecotone in RMN during the 2019 sampling.

Table 2. Summary of most commonly occurring and dominant species in first and last sampling years at each park. Asterisk indicates graminoid species, all others are forbs.

Park	**GSD**		**RMN**		**YNP**	
Year	**2009**	**2020**	**2009**	**2019**	**2011**	**2016**
Total	84	106	176	135	102	130
Common	*Arenaria fendleri,* *Kobresia myosuroides** *Minuartia obtusiloba* *Oreoxis alpina* *Selaginella densa*	*Carex rupestris** *Oreoxis alpina*	*Minuartia obtusiloba* *Poa glauca**	*Carex rupestris** *Minuartia obtusiloba* *Oreoxis alpina*	*Geum rossii* *Lupinus depressus*	*Androsace septentrionalis* *Cerastium arvense* *Poa glauca** *Sedum lanceolatum*
Dominant	*Kobresia myosuroides**	*Oreoxis alpina* *Carex rupestris**	*Minuartia obtusiloba*	*Carex rupestris** *Minuartia obtusiloba*	*Carex phaeocephala** *Polygonum bistortoides*	*Carex elynoides** *Poa glauca**

Great Sand Dunes National Park & Preserve

At GSD, time had a positive effect on species richness, diversity, and proportion of nonnative species but only diversity increased significantly. After accounting for precipitation and elevational ecotone, between 2009 and 2020, GSD species richness increased by 0.89% every year with marginal significance (95% CI = 0.06% decrease to 1.86% increase, $p = 0.07$, $R^2 = 0.224$; Fig. 4a, Table 3) and proportion of invasive species did not significantly change. For every year, the odds of a nonnative species occurring in a sample plot increased 0.13% (95% CI = 24.80% decrease to 51.48% increase; $p = 0.993$, $R^2 = 0.422$; Table 4). However, SDI increased

0.42% every year (95% CI = 0.17% to 0.68%, p = 0.017, R^2 = 0.770; Fig. 5a, Table 5). For every 1cm increase in precipitation, species richness increased 1.3% (95%CI = 0.51% to 2.22%; p = 0.0016; Table 3) and SDI increased 0.32% (95%CI = 0.09% to 0.55%; p = 0.033; Table 5). Visitation and temperature were removed from the models due to over-correlation and the fixed effect model was the best fit in all cases.

Rocky Mountain National Park

At RMN, time had a negative effect on species richness, diversity, and proportion of nonnative species but only diversity decreased significantly. After accounting for precipitation, temperature and elevational ecotone, between 2009 and 2019 RMN species richness decreased 0.29% every year, but not significantly (95% CI = 2.03% decrease to 1.46% increase, p = 0.749, R^2 = 0.144; Fig. 4b, Table 3) and proportion of invasive species did not significantly change, for every year the odds of a nonnative species occurring in a sample plot decreased 0.36% (95%CI = 35.69% decrease to 38.01% increase; p = 0.983, R^2 = 0.339, Table 4). SDI decreased 0.63% every year (95%CI: 0.15% to 1.10%, p = 0.030, R^2 = 0.622; Fig. 5b, Table 5). For every 1cm increase in precipitation, species richness increased 1.46% (95%CI= 0.15% to 2.82%; p = 0.032; Table 3) and SDI increased 1.05% (95%CI= 0.60% to 1.50%; p = 0.0009; Table 5). Visitation was removed from the models due to over-correlation. The fixed effect model was the best fit for species richness and the model that included a random effect for the intercept of elevational ecotone was the best fit for SDI.

Yellowstone National Park

At YNP, time had a positive effect on species richness, diversity, and the proportion of nonnative species, but only species richness and diversity increased significantly. After accounting for elevational ecotone, between 2011 and 2016, YNP species richness increased

2.25% every year (95% CI = 0.78% to 3.74%, p = 0.003, R^2 = 0.313; Fig. 4c, Table 3). However, proportion of invasive species did not significantly change, for every year the odds of a nonnative species occurring in a sample plot increased 0.80% (95%CI = 47.21% decrease to 92.49% increase; p = 0.978, R^2 = 0.088; Table 4). SDI increased 0.78% every year (95%CI = 0.59% to 0.97%, p = 0.003, R^2 = 0.962; Fig. 5c, Table 5). Visitation, temperature, and precipitation were removed from the models due to over-correlation. The fixed effect model was the best fit for species richness while the model that included a random effect for the intercept of elevational ecotone was the best for SDI.

Community patterns

Regionally, time had a negative effect on species richness but a positive effect on diversity. After accounting for elevational ecotone, visitation, precipitation, and temperature, between 2009 and 2020 species richness decreased 0.67% every year, but not significantly (95%CI = 2.45% decrease to 1.14% increase, p = 0.465, R^2 = 0.158; Fig. 4d; Table 3) and SDI increased 0.21% every year across parks, but not significantly (95%CI = 0.68% decrease to 1.10% increase, p = 0.124, R^2 = 0.696; Fig. 5d, Table 5). For every 1cm increase in precipitation, species richness increased 0.79% (95%CI= 0.03% to 1.56%; p = 0.043; Table 3) and SDI increased 0.60% (95%CI= 0.03% to 1.18%; p = 0.053; Table 5). The best fitting model in all cases was the fixed effect model. NMDS ordination shows differential change through time for parks: YNP appears relatively stable while RMN and GSD show movement over time and greater overlap in composition in this ordinated species community space (Fig. 6). Mean dissimilarity values between first and second sampling time steps do not significantly differ (p = 0.482, T = -0.704; Fig. 7).

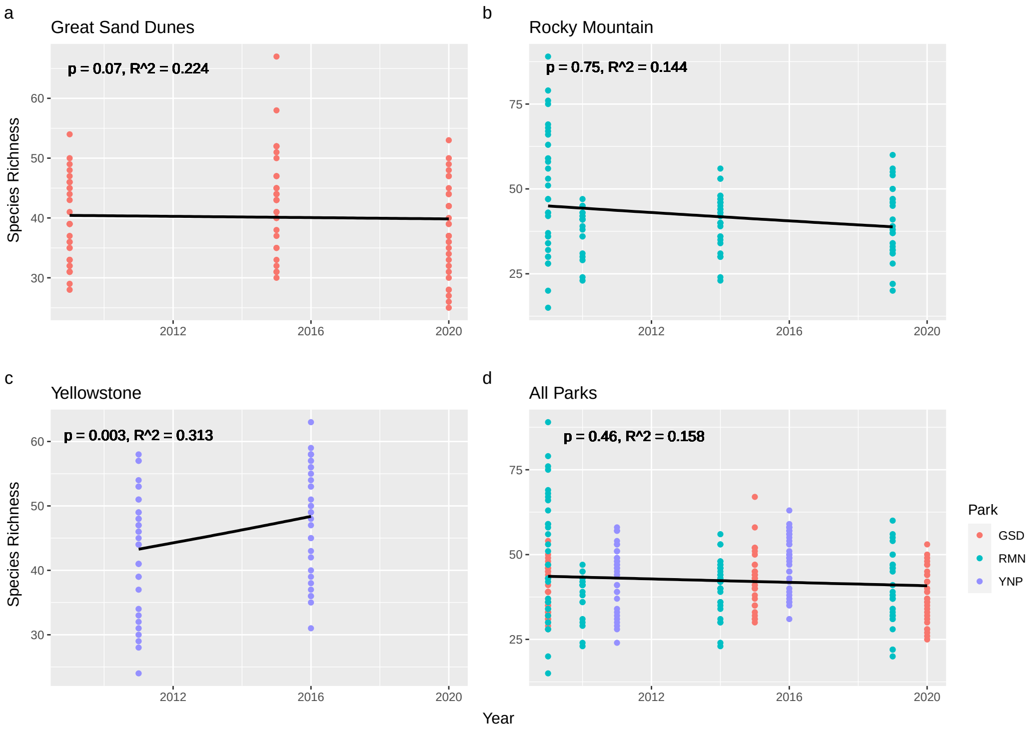

Figure 4. Alpine plant species richness did not change over time at GSD (a) and RMN (b). Species richness increased at YNP (c) and did not change across all parks (d). Each point represents one sampling point, the line shows the relationship between species richness and time and the shaded area represents the 95% confidence interval of the line of best fit.

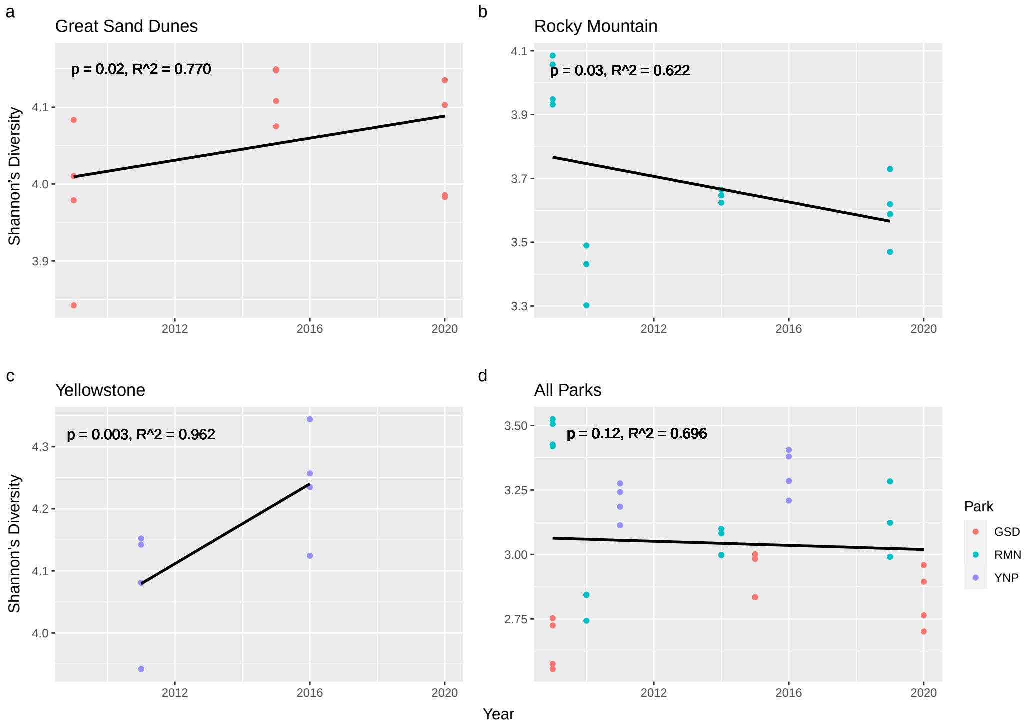

Figure 5. Alpine plant diversity increased over time at GSD (a) but not RMN (b). Diversity increased at YNP (c) and did not change across all parks (d). Each point represents SDI calculated on for peak, the line shows the relationship between diversity and time and the shaded area represents the 95% confidence interval of the line of best fit.

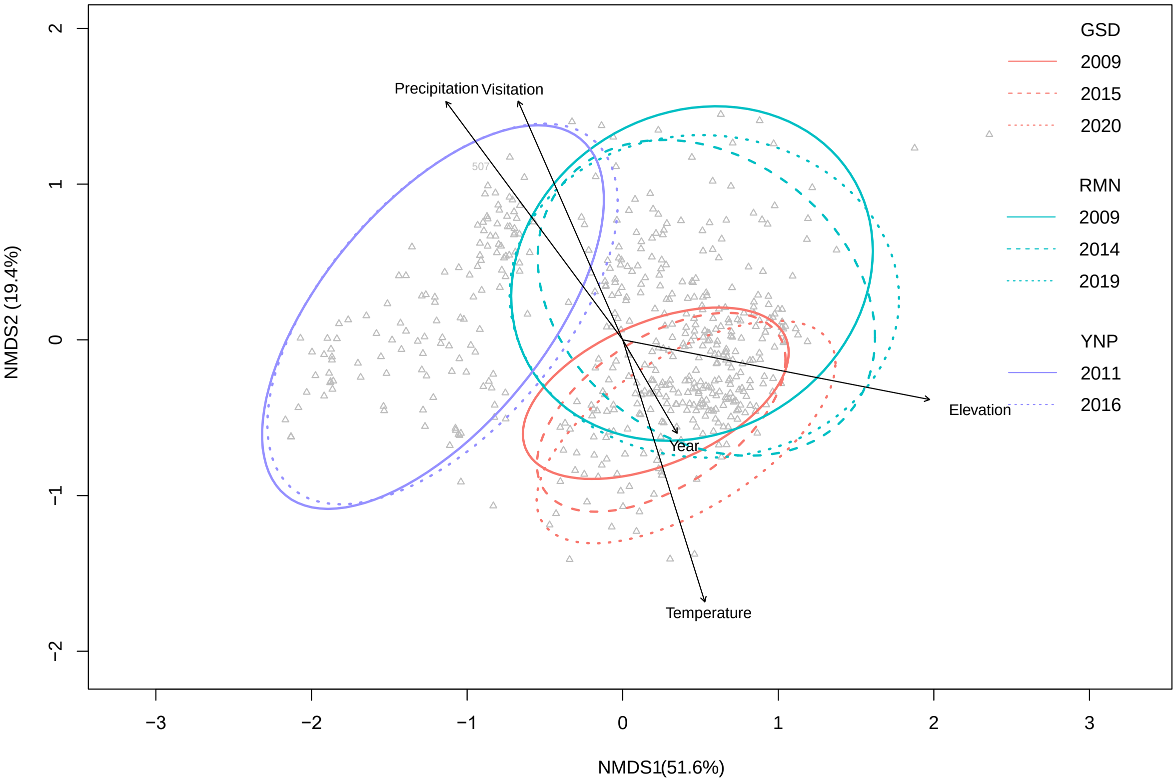

Figure 6. Alpine plant community composition changed across three sampling periods spanning 11 years at GSD and 10 years at RMN but remains relatively stable at YNP over five years. Ellipses represent 95% confidence area and line types represent consecutive sampling years, triangles represent sampling points and overlain arrows show strength of the variable’s association with NMDS axes.

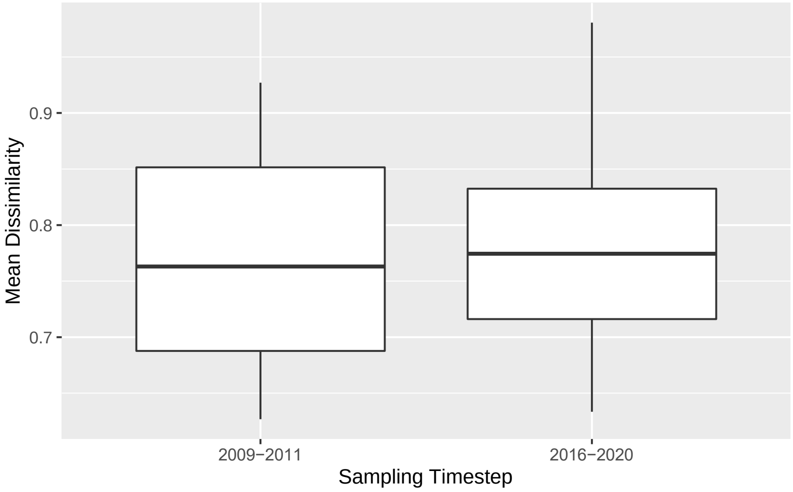

Figure 7. Mean alpine community dissimilarity across parks did not change between first and last sampling times. Boxes represents the interquartile range and the bold line represents the median.

Table 3. Species richness model results showing parameter estimates and standard error in parentheses.

	Dependent variable:			
	Species Richness			
	GSD	RMN	YNP	ALL
Year	0.009 (0.005)	−0.003 (0.009)	0.022** (0.007)	−0.007 (0.009)
Visitation				−0.00000 (0.00000)
Precipitation	0.001** (0.0004)	0.001* (0.001)		0.001* (0.0004)
Temperature		−0.196* (0.099)		0.061 (0.035)
Pseudo R2	0.224	0.144	0.313	0.158
Observations	96	117	64	277
Log Likelihood	−320.642		−222.821	
Akaike Inf. Crit.	653.285		455.642	
Note:				$^{*}p<0.05$; $^{**}p<0.01$; $^{***}p<0.001$

Table 4. Non-native species model results showing parameter estimates and standard error in parentheses.

	Dependent variable:		
	Proportion Non-native		
	GSD	RMN	YNP
Year	0.001 (0.160)	−0.004 (0.172)	0.008 (0.283)
Precipitation	0.005 (0.013)	0.002 (0.012)	
Temperature		−0.703 (2.005)	
Pseudo R2	0.422	0.339	0.088
Observations	163	216	128
Log Likelihood	−11.887	−14.480	−7.360
Akaike Inf. Crit.	35.774	42.961	24.720
Note:			$^{*}p<0.05$; $^{**}p<0.01$; $^{***}p<0.001$

Table 5. Diversity model results showing parameter estimates and standard error in parentheses.

	Dependent variable:			
	Shannon's Diversity Index			
	GSD	RMN	YNP	ALL
Year	0.004* (0.001)	−0.006* (0.003)	0.008*** (0.001)	0.002 (0.005)
Precipitation	0.0003* (0.0001)	0.001*** (0.0002)		0.001 (0.0003)
Temperature				0.015 (0.026)
Pseudo R2	0.770	0.622	0.962	0.696
Observations	12	15	8	35
Log Likelihood	36.623	12.619	14.193	53.574
Akaike Inf. Crit.	−61.247	−15.237	−20.386	−77.149
Bayesian Inf. Crit.		−11.697	−20.068	
Note:				$^{*}p<0.05$; $^{**}p<0.01$; $^{***}p<0.001$

Discussion

In this study I used long-term alpine vegetation monitoring data gathered using GLORIA protocol to explore how vegetative communities in the Rocky Mountains are changing both locally and regionally and to what degree invasive species are contributing to changes. I assessed species richness, diversity, community change and proportion of nonnative species within three U.S. national parks and assessed richness, diversity, and community similarity across parks over time. At GSD and YNP there were positive effects on species richness over time, however, at RMN there were negative effects on species richness, though time was only a significant predictor for species richness at YNP. Precipitation was a driving factor for species richness at both RMN and GSD. At no parks were the proportions of nonnative species significantly changing over time, though there were slight positive effects at GSD and YNP and negative at RMN. Diversity is increasing over time at GSD and YNP, however at RMN it is decreasing. Mean annual precipitation was an additional significant predictor for change in diversity at both RMN and GSD. Regionally, no significant changes were detected in species richness or diversity, though there were negative effects on species richness and positive on diversity over time. When mean dissimilarities across parks were grouped into first and last sampling years, the vegetative communities showed no evidence of becoming similar over time, however, NMDS ordination revealed shifts of vegetative community composition at both RMN and GSD.

While there was little statistical evidence for increasing nonnative species in alpine environments of any parks, the appearance of *Hieracium aurantiacum* in the 2019 sampling of the lowest elevational ecotone in RMN supports the hypothesis that nonnative species are expanding their altitudinal ranges and encroaching on the alpine vegetation communities in RMN. *Hieracium aurantiacum* (orange hawkweed) is a rhizomatous perennial herb native to

Europe, studies show that it can suppress pollination, germination, or seedling growth of surrounding species (Murphy, 2001) and produces an antibiotic that can be detrimental to other plants (Curtis, 1959). Thus, *Hieracium aurantiacum* could have consequential impacts on native species in alpine environments.

This study was considerably shorter than similar alpine studies (11 years vs. 20-100 years) (Britton et al., 2009; Jurasinski & Kreyling, 2007; Ross et al., 2012), and it's possible that invasive plant prevalence will increase over longer timescales. Altitudinal range of nonnative species is positively correlated with residence time (Pyšek et al., 2011). Considering the explosive rise in visitation to national parks over these last eleven year (Fig. 2a) and the association of human activity with alien species introduction (Dainese et al., 2017; Pauchard et al., 2009), extended monitoring time may reveal that nonnative species are invading the alpine study areas. However, my findings support the theory that alpine ecosystems are resistant to invasion (Inouye, 2020), at least on small timescales. Pauchard et al. (2009) attributes the pattern of invasion resistance at high elevations to low invasive propagule pressure, limited human disturbance and harsh abiotic conditions like lower temperatures, decreased length of growing season, increased duration of snow cover, frost frequency and UV-B exposure. These factors will continue the change and facilitate invasion as park visitation increases, abiotic factors become more inviting and propagule pressure increases with warming climate.

Despite lack of significant change in the plot-level proportion of nonnative species, this study revealed significant local-scale vegetative community changes, indicating surprising dynamism in alpine environments. Species richness increased significantly over time at YNP, dominant species changed over the course of the study and diversity significantly changed at all three parks, signaling pronounced community turnover in alpine ecosystems. Britton et al. (2009)

found similar dynamism over 20-40 years in Scottish highland vegetation with increases in species richness and decreases in diversity. In agreement with Elmendorf et al. (2012) who found that alpine response differed based on soil moisture and Ross et al. (2012) finding that alpine community change was associated with precipitation, vegetation community response at the parks was generally better predicted by mean annual precipitation than by year of sampling. In additional to temperature warming, climate change can increase the stochasticity of seasonal weather patterns resulting in increasingly dynamic shifts in alpine vegetative communities that favor alpine generalists at the expense of alpine specialists (Britton et al., 2009). NMDS ordination further supports precipitation and temperature driving community change: parks showed similar compositional directional movement over time on axes associated with temperature and precipitation. The association of community change with elevation could also indicate community movement to higher elevation. With increased study length that incorporates more stochastic precipitation events and changes in annual temperature, the broader patterns of increases in species richness and decreases in beta-diversity in the parks may be more apparent.

On the regional scale, no significant changes were detected in species richness, diversity, or community similarity, revealing that regional alpine communities are more stable than expected over short time spans. This finding contrasts similar studies which revealed regional signs of alpine vegetative community homogenization over times ranging from 20-100 years in both the Swiss Alps and the Scottish Highlands (Britton et al., 2009; Jurasinski & Kreyling, 2007; Ross et al., 2012). This result could be due to the short length of this study and the fact that this study compared sites over a greater distance (~760mi) than others (6mi, 111mi, 450mi); detection of homogenization would be expected to take longer due to increased dispersal barriers.

Shifts in dominant species over the course of the study could indicate community shifts from forb to graminoid-dominant alpine communities. In 2009 the dominant species found at RMN was the forb *Minuartia obtusiloba,* by 2019 the dominant species was the graminoid *Carex rupestris.* Similarly, dominant species at YNP shifted from *Carex phaeocephala* and *Polygonum bistortoides*, a graminoid and a forb, to *Carex elynoides* and *Poa glauca*, two graminoid species over the course of the study. Climate change is likely to cause shifts in plant growth form among alpine communities (Chapin et al., 1996; Chapin & Shaver, 1996; Yang et al., 2011), which could explain the shift in dominant species growth form in the national parks. In an alpine grassland subject to drought and rewetting periods, the community responded by shifting from legumes to more water-use efficient grasses (Tello-García et al., 2020). The shift to dominant graminoids could be explained by changes in annual precipitation, which was a better predictor overall for community changes in the parks.

Besides future studies in the Rocky Mountain alpine extending monitoring time, future studies should explore community change based on vegetative functional traits. A key difference in this study compared to other studies exploring alpine community change over time is the lack of specific analysis of alpine plant growth forms. I instead focused on species richness, diversity measures, and community composition while many studies in alpine systems also analyze changes in plant growth form cover and abundance. Focusing on broad groups like functional traits may make changes in communities more detectable as precipitation stochasticity influences dominant species and woody species encroach on alpine ecosystems (Elmendorf et al., 2012; Scharnagl et al., 2019; Zorio et al., 2016). Lastly, in this study, all models exploring changes within individual parks removed the effect of elevational ecotone, it’s possible that looking specifically at individual ecotones will reveal changes happening at the smaller scale and make changes more detectable on a shorter timescale, preliminary data investigation revealed differing dynamic changes in diversity and species richness within elevational ecotones.

In conclusion, the alpine vegetative communities in GSD, RMN and YNP show relative resistance to invasion by nonnative species in the last ten years, however, sighting of a new invasive species at RMN's lowest sampled alpine elevation indicates that resistance may be waning. Inherent resistance should give managers a window of time to implement measures of protection for fragile alpine ecosystems before nonnative species expand into alpine environments and outcompete endemic species for resources. Alpine vegetative communities in the parks are dynamically changing in response precipitation which can be expected to become more stochastic as climate changes, causing shifts in dominant alpine vegetation to generalist species. In addition to the need for longer-term data, further exploration into fine-scale changes within elevations ecotones of each park could reveal encroaching woody growth forms which could be incorporated into alpine management plans focused on protecting the diversity and high rates of endemism in alpine environments.

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CHAPTER 4

From conflict to collaboration: Preventing future wild horse gathers at Sand Wash Basin

Introduction

Across the western United States, approximately 86,000 wild horses wreak havoc on natural ecosystems (Daly, 2021). Wild horses are considered an introduced species that competes with native wildlife for important resources like forage and water (Daly, 2021). With few natural predators, wild horses require intensive human intervention to manage their population size (Philipps, 2018). However, the wild horse is a cultural icon that represents the untamable spirit of the American West with which many citizens identify (The Wild and Free-Roaming Horses and Burros Act, 1971). Being culturally revered, horses are often kept for recreation and as pets and play important roles in American traditions. In addition, the modern horses' ancestors have a deep evolutionary history in North America, and some argue that American feral horses are a native species (Kirkpatrick & Fazio, 2010) comprising a distinct subpopulation, and as such should be listed as an endangered species (Friends of Animals & The Cloud Foundation, 2014). The conflict between horses' cultural relevance, question of belonging, and the ecological damage caused by overpopulation presents a unique challenge for managers that immediately makes any traditional wildlife management techniques, like lethal control, distasteful (Daly, 2021). Wild horse herd management at Sand Wash Basin, just west of Craig, CO, exemplifies this conflict.

Sand Wash Basin is one of 177 U.S. Herd Management Areas (HMA) managed by the U.S. Bureau of Land Management (BLM) specifically for wild horses (Bureau of Land

Management, n.d.-a). The horses of Sand Wash Basin are beloved by many. Photographers travel from across the United States to photograph them, their lineage is carefully documented by enthusiasts, and the stories of many of the individual horses are told through online blogs, forums, and photographers' websites (Brown, 2021; Webber, 2021). The high-profile nature of the horses resulting from this attention likely contributed to the public backlash from the most recent wild horse removal effort, termed a "gather."

An estimated ~900 wild horses roamed the HMA before the September, 2021 removal effort (Draplin, 2021). In this highly controversial gather, two privately contracted helicopters herded 683 horses up to 15 miles into a canyon where trap corrals were set (Webber, 2021). The large-scale gather occurred in summer heat, and required horses to navigate steep terrain covered in sharp rocks and shale (Webber, 2021). Helicopter gathers such as the one that occurred at Sand Wash Basin often disconnect herds, separate foals from mares, and injure horses (Webber, 2021). The Sand Wash Advocate Team (SWAT), a local advocacy group, and the American Wild Horse Campaign (AWHC), a national advocacy group, along with an outpouring of public concern for the welfare

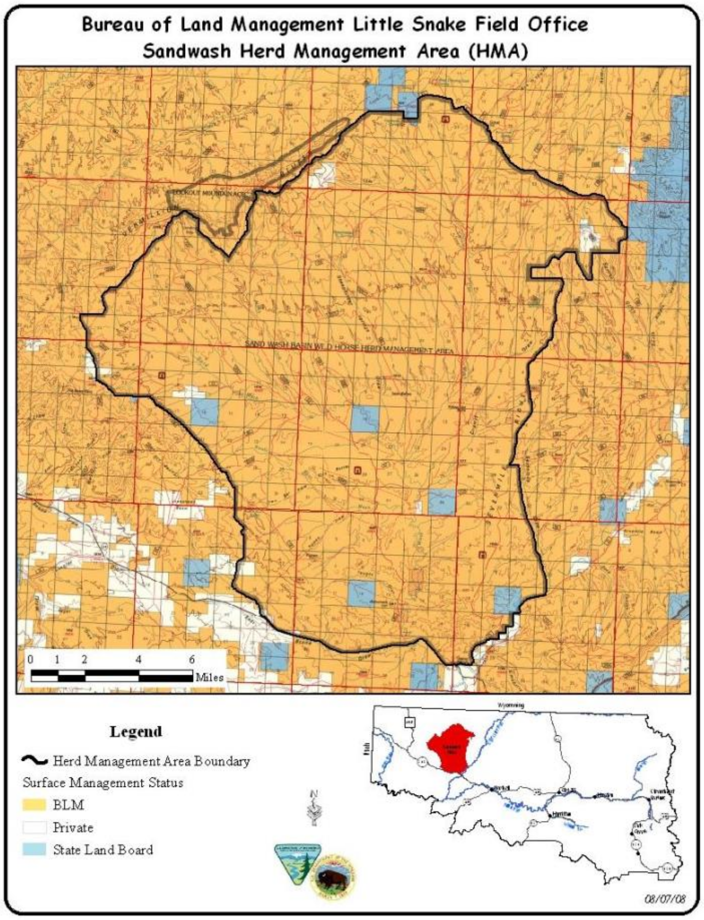

Figure 1. Map of Sand Wash Basin Herd Management Area (BLM, 2016).

of the Sand Wash Basin horses brought the 2021 gather to the attention of Colorado Governor Jared Polis (American Wild Horse Campaign, 2021). Gov. Polis and first gentleman Marlon Reis proceeded to send a letter to the Secretary of the Department of the Interior and the BLM Deputy Director of Policy and Programs urging the gather to cease and requesting a six month moratorium on gathers to explore more humane solutions that would involve the public and advocacy groups as stakeholders (Polis, 2021). The BLM stopped the gather early and released 49 horses back onto the HMA at the request of SWAT (Kafer, 2021).

Following the gather, the horses were transported to a holding facility on East Cañon Correctional Complex grounds in Cañon City (Brown, 2021). Just a few years ago these pens held ~200 horses, but today they hold nearly 3,000 (Brown, 2021). In the past, several horses held at the facility died due to starvation, injuries and colic, or were euthanized for various reasons, causing SWAT and other advocacy groups' concerns regarding the safety and well-being of the Sand Wash Basin horses (Brown, 2021).

While held at the facility, horses are prepared for public adoption events, or if determined unfit, are shipped to farmlands or pastures across the country where they are put to long-term pasture (Brown, 2021). Despite a $1,000 monetary adoption incentive, adoption rates have not kept up with capture rates and holding costs are borne by American taxpayers (Draplin, 2019). While horses are in the care of the BLM, U.S. taxpayers pay an additional estimated $50 million annually for their care (Draplin, 2021); this cost is projected to rise if horse populations are not kept under control (Philipps, 2018).

East Cañon Correctional Complex inmates provide basic care to horses held at the facility as part of a collaboration effort between the BLM and the Colorado Department of Corrections: the Wild Horse Inmate Program (WHIP); in the past inmates were involved in training the horses

(Brown, 2021; Colorado Correctional Industries, n.d.), a program that received national attention, inspired similar programs in other states and resulted in lower rates of recidivism than the national average (Colorado College History Department, 2022). Horse care is a highly coveted job for inmates and equips them with hands-on horse handling, blacksmithing, and farrier skills (Brown, 2021), and participants have gone on to teach horse riding programs, become ranch hands, and start ranches after release (Colorado College History Department, 2022).

The September 2021 gather effectively reduced the Sand Wash Basin wild horse population; however, wild horse populations can double in size every five years (Kafer, 2021). If left to grow unchecked, Sand Wash Basin will soon be overrun with horses, making the positive effects gained by the latest gather temporary. A long-term management solution must be reached to keep the horse population in check that also ensures their humane treatment, allows the ecosystem and native wildlife to thrive, secures a sustainable inmate rehabilitation program, and keeps taxpayer costs down.

Management Challenges

Until the 1970s, North American wild horses were treated poorly and were often hunted for commercial purposes, eventually leading to the unanimous passing of the Wild Free-Roaming Horses and Burros Act in 1971 (Bureau of Land Management, n.d.-b). Under this act, the Secretary of the Interior is responsible for the protection and management of wild horses on public lands while maintaining a healthy ecological balance and may designate specific areas as sanctuaries for wild horse preservation (The Wild and Free-Roaming Horses and Burros Act, 1971). The Secretary is required to consult experts to set population limits for HMAs and excess horses are to be humanely captured and offered for adoption; any horses that exceed adoption demand are to be destroyed in a humane and cost-efficient way (The Wild and Free-Roaming Horses and Burros Act, 1971).

The Sand Wash Basin HMA was established soon after the passing of the The Wild and Free-Roaming Horses and Burros Act and remains one of 201 HMAs, reduced from the original 303 (Moretti, 2020). The horses at Sand Wash Basin share their designated habitat with various wildlife including greater-sage grouse, elk, deer, pronghorn, coyotes, mountain lions, badgers, golden eagles, and bald eagles (Wild Horse Warriors for Sand Wash Basin, 2022). The area is also open to recreation activities such as hiking, primitive camping, dirt biking, and OHV/ATV use (Craig Chamber of Commerce, 2022). Sheep are allowed to graze in the HMA as they cross through from summer to winter feeding areas; however, their consumption is only approximately 4% of available forage and water and thus are not of particular concern at Sand Wash Basin, unlike other HMAs which allow more livestock grazing (Kafer, 2021). Due to overpopulation of wild horses, the rangeland suffered years of overgrazing and recent drought put both the horses and native wildlife at risk of starvation (Kafer, 2021).

AWHC expressed concern about appropriate management levels (AMLs) set by the BLM under The Wild and Free-Roaming Horses and Burros Act, claiming that AMLs set in the 1970s were not based on scientific research or assessment of rangeland but instead were administrative decsions (American Wild Horse Campaign, 2020a). In 2013, the National Academy of Sciences reviewed the wild horse program and noted that AMLs should be adapatable and that proper transparency in how AMLs are set is lacking (American Wild Horse Campaign, 2020a). Most controversy around AMLs is based around shared range with livestock, where livestock take priority over the wild horses (American Wild Horse Campaign, 2020a). Sand Wash Basin allows a negligable amount of livestock grazing which likely doesn't affect AML as greatly as other HMAs (Kafer, 2021); regardless, transparency and adaptability are still lacking.

In addition to controlling the horse population at Sand Wash Basin with gathers, BLM and SWAT collaborate to administer Porcine Zona Pellucida vaccines (PZP), a fertility treatment delivered via darting (Sand Wash Advocate Team, 2021). PZP works through an immunological response that prevents fertilization and does not affect hormones, thus wild horse behavior is minimally affected (American Wild Horse Campaign, 2020b). PZP is effective for approximately two years (The Humane Society of the United States, 2022), is approved by the National Academy of Sciences, and successfully controls wild horse populations in three HMAs (American Wild Horse Campaign, 2020b). Unfortunately, due to the dispersed nature of the horses across Sand Wash Basin's expansive area and the lack of sufficiently trained administrators, past application efforts were only minimally effective in herd suppression (Bureau of Land Management, 2016).

Additionally, the BLM works with Wild Horse Warriors for Sand Wash Basin (WHW), a local advocacy group, to install water sources, improve holding ponds and habitat, and repair fences (Wild Horse Warriors for Sand Wash Basin, 2022). Despite these efforts over the past 13 years, the horse population expanded and decimated the available forage to the point that BLM deemed the latest gather necessary.

Stakeholders

Secretary of the Interior & the Bureau of Land Management

The Secretary of the Interior in partnership with the BLM is charged with managing the wild horse population in Sand Wash Basin for the good of the horses, the native wildlife, and the greater ecosystem (The Wild and Free-Roaming Horses and Burros Act, 1971). Both show openness to alternative herd control methods by engaging with SWAT and WHW to administer PZP and improve the habitat at Sand Wash Basin in order to sustain the herd through droughts (Sand Wash Advocate Team, 2021; Wild Horse Warriors for Sand Wash Basin, 2022). The BLM provides transparency by publishing gather schedules in advance and allows the general public to view corrals and gathers (Webber, 2021). While charged with maintaining ecological balance that sometimes necessitates large gathers, the BLM demonstrates through their willingness to collaborate with advocacy groups that they intend no harm to the horses but are bound by law to maintain ecological balance within the HMA where the horses have almost no predators.

Advocacy Groups: SWAT, WHW, AWHC

The local and national advocacy groups SWAT, WHW, and AWHC act as the voice for the wild horses at Sand Wash Basin. These groups wish to see healthy wild horses run free and live wild. Through their various collaborations with the BLM to administer PZP (Sand Wash Advocate Team, 2021) and provide water during droughts (Wild Horse Warriors for Sand Wash Basin, 2022), these advocacy groups demonstrate that they too are open to finding ways to work within the confines of The Wild and Free-Roaming Horses and Burros Act's limitations. Most importantly, they are motivated to find alternative ways to control the population and improve the overall habitat so the land can support the horses alongside other wildlife.

Prison Inmates

Prison inmates are integral to gathered horses' care before adoption events (Colorado Correctional Industries, n.d.). Though they constitute an underrepresented stakeholder, their voices and opinions should be heard on the issue of Sand Wash Basin. In a hypothetical future where gathers are no longer necessary, inmates will suffer a substantial loss of opportunity to better their immediate and long-term situations. Continued access to programs that provide desirable skills and connection with the horses is important for their development and chances of avoiding recidivism (Colorado College History Department, 2022).

Taxpayers

U.S. Taxpayers represent an interested party that is often unaware of the costs associated with gather events and gathered horses' care. While taxpayers, like inmates, have little voice in the matters of Sand Wash Basin, their presence is a reminder of the fiscal responsibility needed in any solution for controlling the wild horse population at Sand Wash Basin.

Proposed Solution

A solution for herd management at Sand Wash Basin must incorporate and engage all stakeholders. While the BLM is bound to The Wild and Free-Roaming Horses and Burros Act for population management at Sand Wash Basin, there is still room for greater public transparency, collaboration with advocacy groups and exploration of alternative management methods that prevent the need for stressful and expensive gathers. The following suggestions for herd management can alleviate conflict among stakeholders and unite them under a common goal: care for the wild horses and the ecosystem the horses and wildlife depend upon.

To address the ongoing threat of overpopulation in Sand Wash Basin, BLM and SWAT should develop a more robust PZP treatment program that will ensure the herd size will remain stable for years to come. To overcome the obstacles that rendered PZP treatment efforts ineffective in the past, the new treatment program should enlist and train more PZP administrators and explore efficient ways to track horse herds across Sand Wash Basin. PZP treatment costs approximately $30 per mare, which is considerably less than the housing and the adoption incentive cost for a single horse (Draplin, 2019), bringing the taxpayer burden down over time. PZP should be provided by the BLM as a herd management cost which will foster a better relationship between BLM and SWAT.

Every one or two years, a census should be conducted on the wild horses at Sand Wash Basin to detect excess horses early which will reduce the number of horses that need to be removed in a single event. In doing this, more humane capture methods can be used such as the bait and capture method. The bait and capture method uses a lure such as water, food or saltlicks to draw small numbers of wild horses into a corral where they are then shut in; this method reduces risk of injury and physical stress and it allows family groups to stay together (McDonnell & Torcivia, 2020). An alternative method that shows promise uses drones as lures which drew entire herds of up to 123 animals into corrals within 38 minutes in preliminary tests (McDonnell & Torcivia, 2020). Advocacy groups should be encouraged to participate in censusing the population and reporting interim census numbers to the BLM so that timely action can be taken to maintain the wild horse population size within AML limits, currently set to 163-363 (Bureau of Land Management, n.d.-a), thereby avoiding need for gathers.

Partnership with advocacy groups to improve the rangeland habitat through water supplements, holding pond construction, and native plant reseeding should continue. Habitat improvement benefits not only the wild horses but also the native wildlife. Because the range is currently in a degraded state, this will be necessary if the area is to support wildlife into the future. With drought events projected to increase in frequency over time as climate warms and weather patterns change (Dai, 2011; Pokhrel et al., 2021), constructing water holding ponds will be integral to the success of all species in Sand Wash Basin and thus is a worthwhile pursuit.

To address the captured horses that are currently in captivity at East Cañon Correctional Complex, BLM should encourage advocacy groups to participate in adoption events as well as advertisement for events and recruitment of adopters. Advocacy groups can assist with locating private reserves to accept horses and can provide important family group and lineage information to ensure that family groups can stay together whenever possible, thus reducing horse stress.

Inmates at East Cañon Correctional Complex can be incorporated into the wild horse population management through assistance with tracking horses as well as range improvement and maintenance. Providing new ways for inmates to engage with the program ensures their future participation and allows them to continue bettering their immediate and future situations as the number of captured horses held at the complex decreases.

Lastly, BLM should conduct a full, transparent AML review for Sand Wash Basin that uses an unbiased, third-party service. Regular range condition assessment and appropriate AML should be conducted to account for fluctuating conditions as climate change affects forage and water availability. Assessment results should be made readily available to the public and to advocacy groups to allow time for collaboration on herd reduction techniques if reduction is deemed necessary.

Conclusion

Events leading up to the recent overpopulation of wild horses at Sand Wash Basin provides an opportunity for stakeholders to reflect on how management can be improved. Collaboration among stakeholders can provide a path to solutions that satisfy all parties' needs, uniting them under the desire for wild horse well-being and a balanced ecosystem on which horses depend. Through the implementation of a robust PZP darting program, habitat improvement, censusing, and transparent range assessments in collaboration among the BLM, advocacy groups, and inmates, the future of the wild horses at Sand Wash Basin will be improved, taxpayer cost will be reduced and a bright future for inmates will continue.

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