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List of Acronyms

ACEC Area of Critical Environmental Concern

AIM Terrestrial Assess, Inventory, and Monitor

BLM Bureau of Land Management

DEM Digital Elevation Model

ES Ecological Site

ESD Ecological Site Description

ESG Ecological Site Group

MLRA Major Land Resource Area

NCA National Conservation Area

NOC National Operations Center

NGO Non-Governmental Organization

NPS National Park Service

NRCS National Resource Conservation Service

RMP Resource Management Plan

USFS United States Forest Service

USGS United States Geological Survey

WSA Wilderness Study Area

Abstract

The Uncompahgre Field office completed the first Terrestrial Assess, Inventory, and Monitor panel which it initiated in summer of 2022. This report summarizes the status and conditions of several key indicators from the AIM data set, in comparison to the reference conditions contained in Ecological Site Descriptions, or when required Ecological Site Groups. All comparisons are made using spatially explicit inferential statistics which allow for the interpretation of the percent of the field office falling into different categories which are meeting management conditions as specified in the Resource Management Plan (RMP).

Virtually all work was conducted in the statistical Programming language R, with occasional use of 'bash' and python. Most project 'styling' and report elements were designed using Latex in an Rmarkdown/Rstudio environment. All work was tracked using the version control software 'git' and is stored on a website known as Github, which contains logs of all major incremental changes in the sections. We hope that these steps make it easy for others to replicate our work, in short order build upon, and then readily surpass it.

Introduction

This document can be considered in two parts. The first four sections, i.e. the sections up to *'An Assessment of Drought Conditions'* are materials which do not *directly* relate to the analyses in the remaining portion of the document. Rather they reflect import processes in establishing the framework which we used to compare plots, and interpret results. The first section, *'Sample Design, Stratification, and Plots Weights'*, relates to an initial oversight regarding the sample design which made us very slightly re-assign the weights of plots associated with the three different areas of the field office which have different management objectives under the Resource Management Plan. The section *'Ecological Side Descriptions Completion Assessment'* was essential to evaluate the progress made, and remaining to be made, regarding the establishment of Ecological Sites, and their documentation via the formal Ecological Site Description process. Many of the plots had benchmarks which were very narrow, and did not reflect either uncertainty associated with sampling errors, or natural variation, we slightly adjusted the range of these benchmarks in *'Increase Range around Narrow Ecological Site Benchmarks'*. While we intended to use benchmarks directly from Ecological Site Descriptions, the Natural Resource Conservation Service still has many to complete in our area, largely focused in a Major Land Resource Area which contains most of our higher elevation lands. In order to utilize benchmarks across the entirety of the field office, we turned to Ecological Site Groups, a framework recently developed by the United State Geological Survey for the Colorado Plateau. We had to undertake multiple steps to ensure that the NRCS and USGS approaches would be congruent for the report which are detailed in *'Ecological Site Description and Ecological Site Group Comparison - Vegetation 'Benchmarks'*. The final section *'An Assessment of Drought Conditions'* relates to an aspect of the AIM sample design regarding time series data. Some other AIM analysts have made note of using time as a predictor of variation in responses within a single 5-year panel, which allows them to weight plots sampled in years with normal of above normal precipitation more than plots in drier years. However, we believed that the drought conditions over the sample period were too great to consider years in isolation, and pooled these data. These sections all served to inform how we carried out the analysis of the AIM indicators in the remaining work.

The second part, the next several sections, up to *'Plant Functional Diversity - Species'*, form the main body of work, and deal with a number of the Assess, Inventory, and Monitor, Indicators, for which we had enough data to investigate in a meaningful manner. These sections proceed in fashion of increasing ecological and biological complexity, but are written to *largely* be independent of each other; i.e. modular. Notably, the earlier sections refer to more detailed phenomena which are described in the introductions to the later sections. The first section *'Bareground'* documents how much soil across the field office is exposed to wind and precipitation. *'Soil Stability'* contextualizes how these soils, based on their aggregate stability, have different potential to soil erosion, and the effects of this process on the field office and adjacent areas. *'Noxious Species'* investigates the distribution of weeds across the field office, the taxonomic identities of them, and identifies areas with considerable presence of invasive species. We next turn to investigating the cover of each of the major plant functional types in comparison to reference conditions in *Plant Functional Diversity - Cover*, and believe this to be the most integral portion of the document. We then identify the number of species in the major, and some finer resolution, functional groups in '*Plant Functional Diversity - Species*'.

The remaining sections, *'Rare Species'* and *'Floristic Quality Index'*, show opportunistic applications of the AIM data set outside of the realm of Ecological Sites. In *'Rare Species'* we identify the species of conservation concern, across a variety of agencies and Non-Governmental Organizations, as well as species rare under two other non-conservation related metrics. Finally we compute the *'Floristic Quality Index'* and model it across the field office using Species Richness data, this metric is commonyl used in Midwestern and Eastern states, and we believed it's results were largely congruent to the efforts from the combined AIM indicators.

Executive Summary

The purpose of this monitoring project is to understand and quantify the current condition of upland vegetation resources within the Uncompahgre Field Office, Gunnison Gorge NCA, and UFO managed portion of the Dominguez-Escalante NCA and evaluate resource condition relative to the goals and objectives described in the respective resource management plans. Table 1 summarizes the specific vegetation monitoring goals and objectives from the resource management plans that monitoring data were evaluated against. Since the Gunnison Gorge NCA does not contain specific monitoring objectives we utilized the same goal of 80% of the landscape meeting desired conditions as the DENCA and special designations in the Uncompahgre plans. This monitoring project utilized BLM's Assessment, Inventory, and Monitoring (AIM) Strategy which provides a standardized monitoring strategy for assessing natural resource condition and trend on BLM public lands. The AIM Strategy provides quantitative data and tools to guide and justify policy actions, land uses, and adaptive management decisions. The AIM strategy uses core indicators for terrestrial monitoring that are ecologically relevant and clearly tied to the fundamentals of rangeland health. It is important to note that not only are the indicators standardized, but the methods used to collect the data are also standardized. This means that the same data are collected in the same way at each sampled site. The use of standardized methods helps ensure that AIM data are comparable.

The monitoring project leveraged a spatially balanced, stratified random sample design to make inference on approximately 871,000 BLM surface acres in the field office and NCAs. A total of 281 randomly selected plots were sampled over a five-year period from 2018 to 2022, 224 plots were sampled in the Uncompahgre FO, 38 plots in the Dominguez-Escalante NCA, and 19 plots in the Gunnison Gorge NCA (for the purpose of this report all analysis reflects the NCA boundary rather than the planning unit boundary). Every sample location was correlated to an Ecological Site Description or an Ecological Site Group to define whether any given site was within the range of natural variability or had the appropriate mix of plant functional groups or had healthy native plant communities as per the land use plans. Ecological Site Descriptions classify rangeland soils and vegetation into units with similar capabilities to respond to management and disturbance. ESD's provide an expected range of variability (benchmarks) for plant functional group cover/production, bare ground, and other ecologically relevant indicators about a distinctive Ecological Site. Ecological site groups are like ESDs however, they use quantitative approaches to generalize ecological site concepts based on unifying underlying soil, geomorphology, and climate patterns. ESGs also provide benchmarks expected within the range of natural variability of a particular ecological group for the indicators we collect in AIM.

This report focuses on four important ecological indicators to report upland vegetation conditions: bare ground, soil stability, invasive species, and plant functional group cover and composition (trees, shrubs, perennial grasses, and forbs). Additionally for these indicators, the ESDs and ESGs have established benchmarks for the expected range of natural variability. We compared the observed values of these indicators at each sample location to the expected values of the correlated ESD/ESG which then allowed us to use the sample design and a process known as weighted analysis to estimate the proportion of the landscape achieving desired conditions (benchmarks) for the indicators as well as an upper and lower confidence level for the proportional estimate (level of uncertainty).

Bare Ground

Within the Uncompahgre Field Office an estimated 66.3% (62.7 LCL, 69.9 UCL) of the public lands were within the range of natural variability for bare ground. The estimated proportion of lands in the Gunnison Gorge NCA within the range of natural variability for bare ground was 67.9% (55.5 LCL, 80.3 UCL), and for Dominguez-Escalante NCA 75.4% (66.4 LCL, 84.4). No management unit met resource management plan objectives for bare ground and however both NCA's had estimates of uncertainty which included the management objectives. Having too much bare ground may mean that the soil resource is more susceptible

Table 1: Resource Management Plan Goals and Objectives for Vegetation Resources

Uncompahgre Field Office
Objective: *Maximize native vegetation and natural processes by ensuring upland vegetation communities are within the range of natural variability, with an appropriate mix of plant functional groups, cover, and diversity, according to best available science on greater than 80 percent of vegetation communities in ACECs, WSAs, suitable WSR segments, and lands managed to minimize impacts on wilderness characteristics and on greater than 70 percent of vegetation communities on the remaining BLM-administered lands, over 10 years with 80 percent confidence.* *Manage for soil stability and productivity to maintain overall watershed health. Manage erosion to minimize downstream impacts from soil-related issues (e.g., sediment runoff, selenium, and salinity).*
Dominguez-Escalante National Conservation Area
PSV-DSS-OBJ-01: *Improve the plant composition of the D-E NCA's desert shrub/saltbush vegetation type to achieve public land health standards and move toward the following management targets:* • *80% (or more) of sampled acres contain adequate mixtures of warm and cold season grasses, shrubs and forbs* • *80% (or more) of sampled acres exhibit an acceptable composition of understory invasive plant species (<10% relative cover)* **PSV-PJW-OBJ-01:** *Manage for public land health standards in the D-E NCA's pinyon-juniper woodlands and move toward the following conditions in the D-E NCA's pinyon-juniper woodlands:* • *95% (or more) of sampled acres contain adequate mixtures of warm and cold season grasses, shrubs, forbs and trees* **PSV-SGS-OBJ-01:** *Improve the plant composition of the D-E NCA's sagebrush shrublands vegetation type to achieve public land health standards and move toward the following management targets:* • *80% (or more) of sampled acres contain adequate mixtures of warm and cold season grasses, shrubs and forbs* • *95% (or more) of sampled acres exhibit an acceptable composition of understory invasive plant species (<10% relative cover)* • *95% (or more) of sampled acres have acceptable levels (less than 50% relative understory cover) of crested wheatgrass* • *80% (or more) of sampled acres have moderate cover of sagebrush (10-30% cover)*
Gunnison Gorge National Conservation Area
VEG-C-1 *Public lands will be managed in accordance with Interpreting Indicators of Rangeland Health (Pellant et al. 2000).* ***VEG-C-2*** *Current vegetation studies will be continued, and new studies will be initiated.* ***VEG-C-3*** *Monitoring and studies will be carried out to help achieve sustainable populations of native plant species, with healthy native plant communities dominating the landscape.*

to wind and water erosion. Less bare ground than expected at a site may mean that there are other ecological issues including excessive litter generated from invasive species such as cheatgrass.

Soil Stability

Soil aggregate stability is another measure of soil erosion potential as well as an indicator of soil health, and presence of biological crusts. Soil aggregates are formed by natural processes including alternate wetting and drying and from the accumulation of organic substances derived from root exudates, roots, mycorrhizal fungi, soil microbes and their byproducts which act to cement soil particles into aggregates. Limited herbaceous understory plants, such as grasses and forbs, can contribute to lower soil stability values. Drought can also result in lower aggregate soil stability due to reduced litter cycling and suppressed soil biological activity. Within the Uncompahgre FO an estimated 39.9% (36.2 LCL, 43.7 UCL) of the landscape were found to be within the range of natural variability for soil aggregate stability and the proportion of the Gunnison Gorge NCA estimated to be achieving desired conditions was 54% (39 LCL, 69.1 UCL). There was an estimated 14.7% (7.6 LCL, 21.8 UCL) of the Dominguez-Escalante NCA found to be within the range of natural variability for soil stability. Relative to this indicator none of the management units met resource management plan goals. The entirety of the sampling did occur in a period of prolonged severe drought which likely has contributed to such low proportions of the landscape achieving desired conditions. However, low cover and composition of herbaceous plants that contribute to higher average soil stability are also a likely contributor to the low estimated observed.

Invasive Species

Of the 280 plots sampled during this monitoring project 81% of them detected non-native invasive plants present ranging from trace occurrences to dominant components of the vegetation community. This is suggestive that invasive non-native plants are now widespread in the field office. Many invasive species can disrupt ecological processes and competitively exclude more desirable vegetation. The most prevalent and widespread invasive species detected was cheatgrass occurring at 50% of the sample locations. The next most prevalent species detected was halogeten occurring at 23% of the sample locations. Halogeten was distinctly regionalized with most of the occurrences occurring in the Mancos Shale derived soils of the Uncompahgre and Gunnison River valleys. Ecological site descriptions and ecological site groups do not have invasive non-native species present in the reference state thus there are no descriptive references for the range of natural variability for invasive species. We utilized the RMPs goals and objectives to arrive at a threshold of 5% cover or less for the Uncompahgre FO and the Gunnison Gorge NCA to meet the intent of the overarching goals for vegetation relative to rangeland health.

We established <5% as our desired condition because sites with greater than 5% relative cover of invasive species have lower resistance to invasive species invasion and dominance and reduced resilience to disturbances. Utilizing the <5% benchmark for relative cover of invasive species an estimated 60.3% (LCL 56.7, UCL 63.9) of the lands managed by the Uncompahgre field office has less than 5% cover of invasive species. Again, relative to invasive species UFO managed lands do not achieve the RMP goal of 70% of the landscape meeting the objectives. In the Gunnison Gorge NCA an estimated 23.3% (LCL 10.6%, UCL 36%) of the landscape was found to have less than 5% relative cover of invasive species which is well below the desired condition of 80% of the landscape achieving desired conditions. Of the special management goals per stratum at the DENCA most objectives are being meet. Salt desert is meeting the estimate of 80% of land having less than 10% relative cover of invasives (LCL 69.9, UCL 90.1). Sagebrush was estimated as having 67% of land achieving goals (Estimate = 66.7%, LCL 31.3, UCL 100). If we assume similar goals for Pinon-Juniper on 80% of land than it is meeting objectives (estimate = 86.7%, LCL 76.6, UCL 96.7). The RMP also set a goal of 95% or greater of the DENCA landscape to be achieving this condition. Considering the overlap in the confidence intervals the DENCA is close to achieving this desired condition.

A caveat to the estimated presented here is that most of the invasive species detected are annual plants. Given the exceptional drought conditions experienced over the five-year sampling period, and the widespread nature of invasives, it is quite likely that notably less of the landscape in any of the three management units evaluated may be achieving desired conditions under more normal precipitation conditions.

Plant Functional Group Cover & Diversity

To answer the RMP questions regarding an "adequate mix of perennial grasses, forbs and shrubs" we utilized the plant functional group concept for analysis. Each correlated ESD and ESG describes an expected cover or production range of natural variability for each plant functional group i.e. trees, shrubs, grasses, and forbs. The appropriate mix of plant functional groups is perhaps the most fundamental key point for a site to be ecologically functioning, exhibiting good rangeland health, and achieving Colorado Public land health standards for healthy plant and animal communities. Lower than expected or the absence of key functional groups like perennial grasses and forbs can result in excessive bare ground and elevated erosional processes or decrease the plant community resilience to invasive species invasion and resistance to disturbances. Conversely, higher than expected values for woody species can result in similar issues and suggest that disturbance regimes such as fire may be missed resulting in a late successional condition or facilitating encroachment into less woody dominant ecological sites. A commonly observed issue in the

Uncompahgre Field Office is that sagebrush dominant ecological sites are often found to have low cover and diversity of perennial grasses and forbs and as a result the sagebrush has infilled resulting in shrub cover that far exceeds the range of natura variability for the ecological site.

For the Uncompahgre FO an estimated 34.4% (31 LCL, 37.8 UCL) of the landscape has the appropriate cover and composition of forbs, and an estimated 23% (19.9 LCL, 26.4) has the appropriate cover of perennial grasses. Neither functional group is close to achieving the goal of 70% of the landscape achieving desired conditions of being within the range of natural variability. The Uncompahgre FO has a concerning amount of the vegetation communities that lack the appropriate minimum cover of perennial grasses and forbs. No sites were deemed to be not achieving desired conditions for having too much cover of perennial grasses or forbs. An estimated 55.7% (52 LCL, 59.5 UCL) of the UFO landscape was found to have the appropriate cover of shrubs, and an estimated 53% of the landscape had the appropriate cover of trees. Most sites not achieving desired tree and shrub cover were in undisturbed sites where cover was commonly higher than expected likely attributed to late succession, while sites typically found to have lower than expected tree and shrub cover were in previously disturbed sites that have been systematically managed to remain treeless or previously burned and are in an early successional state. Again, neither functional group achieved land use plan goals of having 70% of the landscape within the range of natural variability.

The Gunnison Gorge NCA shows similar concerning issues with the herbaceous understory of its upland plant communities. None (0%) of the landscape was found to have the minimum cover of forbs relative to ecological potential, and an estimated 28.9% (15.9 LCL, 41.9 UCL) having the minimum cover of perennial grasses. The NCA also had an estimated 63% (48.8 LCL, 77.8 UCL) of the landscape with the appropriate cover of shrubs and estimated 71% (55.8 LCL, 86.4 UCL) with the appropriate cover of trees relative to ecological site potential. Tree and shrub cover was commonly lower than expected in the GGNCA due in part to drought related mortality in the low elevation saltdesert shrublands and dry site juniper woodlands that have been occurring over the last 20+ years. None of the plant functional groups in the GGNA met land use plan goals of 80% of the landscape achieving desired conditions. However, with the estimate of uncertainty the tree functional group does approach the RMP goals.

The DENCA also shows similar trends in that perennial grasses and forb cover are lower than ecological site minimums. An estimated 24.5% (14.9 LCL, 34 UCL) of the landscape had at least the minimum cover of forbs relative to site potential. For perennial grasses 35% (25.5 LCL, 44.5 UCL) of the landscape was found to be achieving desired conditions. The shrub functional group was found to be achieving desired conditions on an estimated 70.8% (62 LCL, 79.6 UCL) and the tree group was achieving desired conditions on an estimated 43.8% (31.6 LCL, 56.0 UCL). Similar drought and succession related impacts are driving tree and shrub estimates in the DENCA.

We utilized plant functional diversity as another line of evidence to help elucidate why such low estimates of the landscape are achieving plant functional group desired conditions. We found that the perennial grass functional group has the lowest diversity of the functional groups and commonly have far fewer species than what an ESD or ESG suggests could be expected. We found that highly palatable cool and warm season perennial bunchgrasses were absent or had fewer than expected species. The absence of deep-rooted perennial bunchgrasses does directly contribute to the low cover values observed above as these plants occupy a large aerial extent when present and have higher cover values than sod forming or shallow rooted bunchgrasses. The shrub functional group also exhibited low diversity, with many sites being dominated by or having near monocultures of broom snakeweed. Snakeweed was the most common plant observed in the project occurring at 72% of the sample locations and is regarded as a native invasive (noxious) species. Snakeweed exploits open niches caused by disturbances and does not compete well with other more desirable plants. The diversity of forbs was better than expected with the number and types of forbs commonly exceeding what ESDs suggested. Many forbs do not produce above ground biomass when conditions are poor and given the drought that has occurred over the sampling period forb cover may well be better the estimates made here under more normal precipitation.

Rare Species

The extensive plant species inventory conducted utilizing the AIM protocol also helped identify new occurrences of threatened species, BLM species, and rare species tracked by the Colorado Natural Heritage Program a partner agency in rare plant conservation. The project added five new occurrence records for the Colorado hookless cactus a species that was recently de-listed from the protections of the Endangered Species Act. Six new occurrences were documented for the BLM sensitive species Aromatic Indian Breadroot (*Pediomelum aromaticum*), San Rafael Milkvetch (*Astragalus rafaelensis*), Adobe desert-parsley (*Lomatium concinnum*), and Eastwood Evening Primrose (*Camissonia eastwoodiae*). The project also added the first record in the UFO for a newly proposed BLM sensitive species Paradox Valley blazingstar (*Mentzelia paradoxensis*) which will be added to the BLM sensitive species list in the coming months. These new records help inform decisions that can advance the conservation of these species and preclude their need for additional regulatory protections. Finally, the project identified 26 additional species that are tracked by the CNHP that will help to inform their records regarding rarity and distribution.

Conclusion

None of the vegetation community indicators evaluated here met land use plan goals and objectives for proportion of the landscape being within the range of natural variability. Across all three landscapes perennial grass and forb cover and diversity is low. Often well below minimum ecological site range of variability. As a result, undisturbed sites have experienced increases in shrubs including sagebrush and broom snakeweed and trees. Bare ground is often higher than expected in these types of sites with higher rates of erosion. We often observe substantial loss of the valuable topsoil resource in the interspaces of these sites which will make restoration exceptionally difficult. In previously disturbed sites we see similar conditions with fewer trees and shrubs. Invasive species are widespread in the three management units but appear to be less problematic in the Dominguez-Escalante NCA. The invasive species prevalence and dominance are a direct result of the plant functional group imbalance that the project has revealed. Invasive species exploit areas with low diversity and cover of more competitive vegetation. A diverse mix of perennial grasses and forbs within the range of variability for any given site offer the greatest competition to repel invasive species dominance. As a result, current vegetation conditions make large portions of the management units exhibit low resistance to invasive species invasion and dominance and have low resilience to disturbances such as fire and drought.

Past and current land uses combined with the cumulative impact of numerous droughts and more specifically the now 20+ yearlong mega drought have all contributed to current upland vegetation condition. The land use plan decisions for all three units need to be leveraged with other policies and regulations to effect meaningful change in upland vegetation condition and to avoid further degradation as well as maintain the valuable ecosystem services they provide. Greater emphasis should be placed on effectively reclaiming authorized disturbances, implementing, and appropriately managing restoration of degraded vegetation communities is also needed, and lastly resource allocations involving vegetation resources need to consider these changed conditions and make allocations that more closely reflect current conditions, and appropriate management decisions around persistent drought are needed to halt further resource degradation.

AIM Field Methods

The Bureau of Land Management uses the Terrestrial Assess, Inventory, and Monitor (AIM) program across it's surface managed lands to determine land health. The AIM program collects quantitative data of rangeland health using six core indicators. These indicators were selected by the BLM as they have been empirically shown to respond readily to management actions, and are straightforward to interpret. The four field methods utilized to collect these six indicators, are relatively quick to perform, can be learned quickly, and give repeatable measurements. Three of these methods have been widely employed in the natural sciences for over half a century, and the soil stability test was largely developed around the millennium by researchers whom went on to develop the AIM program. A comprehensive overview of these methods are provided in J. E. Herrick et al. (2021). Here we provide a brief recapitulation of the history, and common applications of the four main methods used by AIM.

All data collection performed for the AIM program is recorded by electronic data capture, that is data are entered directly into computer software. The software utilized for data entry are designed for survey work, and the National Operations Center (NOC) have developed AIM specific forms, populated with fields with limited entries to reduce errors. These data then undergo quality control by crew leads, and then substantial Quality Control by the NOC. Data which appear erroneous, are fixed when possible, or flagged as appropriate.

Species Inventory

The entirety of an AIM plot, a circle with a 30m radius and an area of ~2700 m^2, is used to record the presence of plant species (Figure 1). This area is systematically wandered, as the technician records the presence of each vascular plant species they encounter, and flag unknown species, for 15 minutes. In the event that the technician is still detecting un-recorded species on the plot at that time, they add an additional 2 minutes each time they find an unrecorded species, until they cannot detect another species in that interval. Species are then identified in field, or from collected specimens using Flora's.

Indicator	**Method**
Cover Bare Ground	Line-Point Intercept (LPI)
Vegetation Composition	Line-Point Intercept (LPI)
Vegetation Height	(LPI) + Height
Length of Bare-ground	Gap Intercept
Invasive Species	Species Richness; LPI
Soil Aggregate Stability	Soil Stability

Table 1: Core Indicator and Methods used to Collect Them

The entire area searched by this method is depicted in Figure 1, in beige. These data are used to detect plant species which may be invasive, of conservation concern, and to inventory the area.

Gap Intercept

The original formulation of this approach, **Line Intercept**, were developed in the 1940's in order to sample shrublands in an effective manner (Canfield (1941), Bauer (1943)). The commonly used formulation of Line Intercept is able to quickly determine the cover of physiognomically dominant shrubs, a metric which can be time intensive to record, such as Sagebrush. In the Gap Intercept utilization used by AIM, rather than measuring plants, the start and end positions of bare ground are measured; it can considered the inverse of the more common method.

AIM crews gather the total length of bare ground at three 25 m transects, in a tri-spoke design. Each line is orientated 120° from the others, and emanating out from a 'trample/sacrifice zone' an area at the center of a plot which is prone to damage and giving erroneous measures. The technician slowly walks the length of each transect, recording the 'Start' of bare soil, and the 'End' of bare soil, for each bare soil gap > 20 cm in length. The bare soil 'End', can be caused by live or dead vegetation, or embedded plant debris >~ 2 mm in diameter.

Two of three lines which the method is carried out on are depicted in Figure 1, as black lines.

Line-Point Intercept

The 'point' part of this method was developed in New Zealand in the early 1920's, the decision to align points along a line was developed and popularized in the 1950's to determine the abundance of plant species in areas (Goodall (1952)). Essentially, one has a transect tape - a line, and a predetermined number of locations they will record the presence of a species at, generally these locations will be equi-distant from each other.

AIM crews utilize the same three lines for this method which they use for Gap Intercept. There are 50 points located at 0.5 meter distances along these lines, where technicians drop a pin flag, and record each plant species which it passes through as it drops vertically. The types of debris, e.g. leafy or woody, and the object at the earths surface, e.g. bare soil, gravel, or plant base is recorded as well.

The 50 closely spaced diamongs along the Northern transect (vertically orientated) are depicted in Figure 1. An additional indicator, vegetation height, is collected at ten locations (cylinders) along these lines as well. Vegetation height records the tallest woody and herbaceous individual in a 15 cm radius located off the transect line.

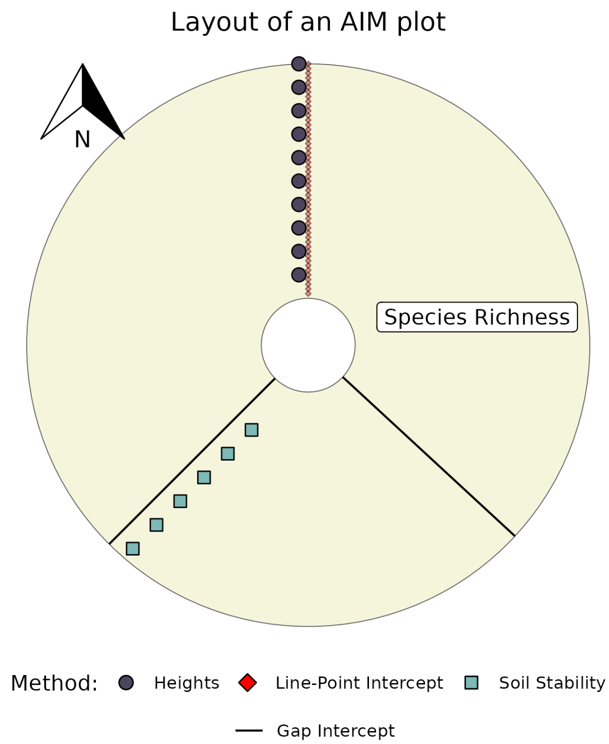

Figure 1: Example AIM plot, with locations of the four major methods illustrated. Note that while each line based method is performed three times, they are illustrated fewer times for clarity.

Soil Stability

This method was developed in the late 90's in order to approximate the susceptibility of soil to erosion (J. Herrick et al. (2001)). Directly measuring soil erosion potential is time consuming so this method utilizes the stability of **macro-aggregates** as a proxy for erodibility.

At six, generally equidistantly spaced, locations along each transect tape, technicians record the functional group of plant cover (if present), and collects a soil ped. After all 18 peds have been collected, they are then submerged in water and the dissolution of their particles as a function of time, and dipping cycles using a sieve, are recorded.

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Plot Weights & Locations

This AIM study was designed with three major principals in mind. The first, was maximizing the number of environments which it represents across the field office. The UFO field office varies by nearly 8400 feet, and displays a diversity of environments associated with this topographic gradient; and this sample design included this entire range of variation. The second, to ensure that these diverse areas were represented, ten major strata were identified (Table 1), in order to ensure that these areas had plots located in them in order to evaluate their ecological *integrity*. However, the management concerns, actions, and heterogeneity (internal dissimilarity) of these strata differ. To best understand the ecological context of these strata, and what management actions the BLM may take, the proportion of plots were tailored to each stratum. Areas of higher interest had more plots located in them. The third principal was that all plots be randomly placed within the stratum, which allows *inference* to be made from plots to the, unsampled, entirety of the stratum. This *weighted sample design* allows us to maximize statistical understanding while minimizing the field effort required to gain it. Finally, the entire field office was designated as the *sample frame*, i.e. the area of analysis (or a spatially explicit population), which statistical inference can be made to.

The initial sample design followed the general AIM implementation design to contain 255 plots to be sampled across five years. Sampling over a five year period is essential in order to ensure that plots are visited during periods wherein the vegetation is a state of *reproductive activity*, hence identifiable. A benefit of this prolonged time frame is that anomalous weather conditions are unlikely to affect the plots across the entire time period. For example, the condition of plots may be compared in a wet year, to a dry year, as referenced from a year with typical rainfall. The sample was spatially balanced into panels. The panel was balanced in order to avoid unexpected processes which cluster in space (are spatially autocorrelated), or in years of the design (temporal autocorrelation) and which may bias results over the sampling period. Each of these five panels is composed of a subset of randomly selected points within the sample design and avoids missing swathes of the *target frame* during the sample period due to an event such as a prolonged wildfire.

The location of the random plots for AIM are generated via computer and a number of them may not feasibly be sampled. In general potential plots are most often, anecdotally, rejected for: being located on steep slopes which are dangerous for field crews to sample, requiring access to cross private land to access BLM which landowners deny. The inability of these plots to be sampled reduces the spatial extent of the strata which we may use statistics to infer across, and increases the measures of uncertainty associated with these areas. In this section we review the original sample design, derive original sample weights for sub-units of the field office, and calculate the weights for each of these units after the sample design has been completed.

the sample design may be represented in simple mathematical terms Using an example from a stratum which is of high land management importance, sagebrush-steppe, we illustrate the site selection process. While the aerial extent of sagebrush-steppe in the target frame is roughly one quarter (0.246), the stratum makes up one third (0.33) of all plots

In order to convey the number of plots drawn in any stratum the following equation is representative.

$$n = \frac{N * \pi_i}{1/panels}$$

- N the total number of plots in the sample design, *i.e. 255*
- π_i the inclusion probability of a plot in each stratum being drawn, *e.g. 0.33 targets the placement of a third of all plots to be put into the Sage-steppe Stratum*
- $panels$ the number of design panels, *i.e. the number of years to stretch sampling across*

For sagebrush steppe we will then have:

$$16.83 \; = \; \frac{255 * 0.33}{1/5}$$

plots for each panel, which will round down to 16 in order to accommodate representation of some of our other strata.

The **weight** of a plot is the amount of acres it represents within a stratum, we utilize this metric to help derive *measures of uncertainty* while making inference from our sampled plots to the whole stratum.

$$W_i = \frac{Stratum_{area}}{n}$$

- W_i the weight acres, *i.e. the area which a plot represents within the target frame*
- $Stratum_{area}$ the total area of the stratum in the area of analysis *e.g. 214,023 acres of sage-steppe*
- n the total number of plots in a stratum *e.g. for a panel of sage steppe 16*

Each sagebrush-steppe plot will have a weight of

$$13,376 \; = \; \frac{214,023}{16}$$

acres per plot, this is the area which it represents.

Areas of Analysis

The original AIM design covers the entire target frame of the UFO field office. However, a few additional analytical and management sub-units exist within this extent. There are three planning areas each with their own management objectives. The two National Conservation Areas associated with the UFO - the Gunnison Gorge & Dominguez-Escalente - the latter of which is partially administered by the Grand Junction Field Office (GJFO), are associated with their own Planning Areas. Likewise both within those, and across the Field office, there are numerous Wilderness Study Areas (WSA's), and Area's of critical Environmental Concern (ACEC's). As discussed later, the UFO intends these areas to have higher proportions of certain vegetation metrics relative to the remainder of BLM land.

The Uncompahgre Resource Management Plan, completed in June 2019 covers 675,800 acres of the field office, and provides the definitive source of management objectives for the Field Office (summarized in VEG-OBJ-01 in Section II p. 23):

> "*Maximize native vegetation and natural processes by ensuring upland vegetation communities are within the range of natural variability, with an appropriate mix of plant functional groups, cover, and diversity, according to best available science on greater than 80 percent of vegetation communities in ACECs, WSAs, suitable WSR segments, and lands managed to minimize impacts on wilderness characteristics and on greater than 70 percent of vegetation communities on the remaining BLM- administered lands, over 10 years with 80 percent confidence.*" — RMP 2019

Table 1: Original Sample Design for the Entire Sample Frame

Stratum	Area (acres)	Prop. Area	Prop. Site	No. Plots	Plot Wt.
PJ	274,557	0.42	0.12	25	54,911
SS	170,022	0.26	0.33	80	10,626
SD	71,500	0.11	0.30	75	4,767
MMS	44,851	0.07	0.10	25	8,970
RI	34,453	0.05	0.05	15	11,484
GR	13,226	0.02	0.02	5	13,226
MC	12,042	0.02	0.05	15	4,014
PP	11,436	0.02	0.01	5	11,436
OT	10,943	0.02	0.01	5	10,943
AS	7,715	0.01	0.01	5	7,715

All land, less species status

"*greater than 70 percent of vegetation communities on (the remaining) BLM-administered lands*"

— RMP 2019

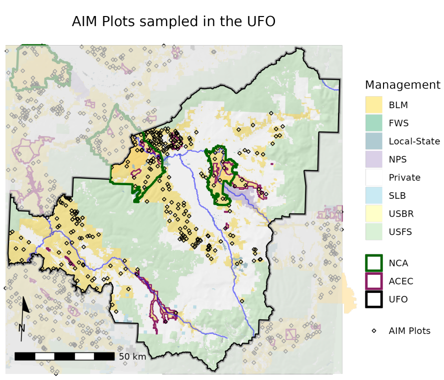

Figure 1: All plots sampled across the entirety of the field office

The majority of land administered by the field office does not fall under any special regulatory status, such as those which are present for the Areas of Critical Environmental Concern (ACEC's), Wilderness [Study] Areas, and National Conservation Areas. The initial AIM sample design covered the entirety of the Field Office, and did not separate out, or weigh, the special regulatory areas from the majority of BLM administered land. However, given that this area is managed differently than the following three other areas, and that it has different management objectives with more tolerance to certain kinds of uses, we split this area out retroactively. Here we assess the location of the AIM plots drawn in the original sample design, and using spatial techniques assign them to this area. We then determine the weights of plots under the scenario that all plots were sampled (Table 1), and under the realized condition that only certain plots were sampled (Table 2). Generally, we would only be performing the second action at the end of the sampling.

Table 2: Realized Weighted Sample Design for the Reamining Sample Frame

			No. Plots	
Stratum	Plot Wt.	Plot Wt. Ac.	Sampled	Rejected
PJ	1.087	54,911.337	23	7
SS	1.143	10,626.398	70	10
SD	1.471	4,766.685	51	6
MMS	1.316	8,970.282	19	10
RI	1.071	11,484.452	14	11
GR	0.833	11,021.501	6	9
MC	1.667	4,014.072	9	15
PP	0.714	8,168.550	7	7
OT	1.250	10,943.144	4	9
AS	1.667	7,715.088	3	7

The strata of Mixed Conifer and Aspen, both had fewer plots sampled than the sample design called for. Both of these areas are relatively rare across the field office, and accordingly had very few initial plots drawn for each of them. The inability to sample these is due to several of the drawn base points and over sample plots being located on parcels of land, which required permission from private land owners to access, but which were denied; or were on areas with very high slopes, which were unsafe to sample. While we are able to use these plot weights to infer across the field office on the whole, we note that a discrepancy existed between the actual distribution of these areas in the office and the plots classified as such (Section 2). While we regret not being able to sample these plots, most of the drawn locations were actually in Mixed Mountain Shrub, an area which was weighed to have extra plots and which was successfully sampled.

Aside from these two strata, all other strata had more plots sampled than expected. Because the area of inference which these plots speak to cannot change, the weight of each individual plot decreases in this scenario. In simpler terms, each plot represents a smaller unit of area. In the case of the Mixed Conifer and Aspend plots, each plot represents more area than intended.

Areas of Critical Environmental Concern (ACEC's) and Wilderness Study Areas (WSA)

The Uncompahgre Field office has 12 unique Areas of Critical Environmental Concern. The focus of these areas range from conserving Archaeological sites, sections of Rivers, and rare plant localities. Many of these areas are relatively small, and scattered across the entirety of the field office, the locations under this analytical unit will include those which cannot be combined the National Conservation Areas. Three of the ACEC's are nearly entirely contained within National Conservation Areas, and will not be included in the analyses for this section, but rather the NCA which they are surrounded by. Both of the field offices two wilderness areas, are also contained entirely within National Conservation Areas, and they also will be included in analysis and discussion within the NCA which contains them.

The office has four Wilderness Study Areas, one of which is largely encompassed by a National Conservation Areas, and will be included in the analyses there, rather than here.

Table 3: Original Sample Design for Areas of Critical Environmental Concern and Wilderness Study Areas

Stratum	Area (acres)	Prop. Area	Prop. Site	No. Plots	Plot Wt.
PJ	25,763	0.30	0.12	3	15,458
SS	15,034	0.17	0.33	10	7,517
SD	13,825	0.16	0.30	10	6,912
MMS	12,951	0.15	0.10	3	7,771
AS	5,163	0.06	0.01	1	1,033
RI	4,869	0.06	0.05	2	1,948
OT	3,926	0.04	0.01	1	785
PP	2,675	0.03	0.01	1	535
MC	2,516	0.03	0.05	2	1,006
GR	158	0.00	0.02	1	32

The reporting units of Areas of Critical Environmental Concern (ACEC's), and Wilderness Study Areas (WSA), and Wilderness Areas, have different management objectives relative to the remaining BLM administered surface area. Loosely, they are intended to have a higher percentage of the vegetation communities within the natural range of variation:

> "*greater than 80 percent of vegetation communities in ACEC's, WSA's...*" — RMP 2019

These areas were not intensified units within the original sample design, rather we split them out here using the original point draw for the field office. Here we calculate the initial sample weights for them using the same approach as for the remainder of BLM land, i.e. the acreage of each stratum is weighed against a targeted proportion of plots in the region. As our sample design was initiated and completed during a period of drought (Section 6), we dismiss the possibilities of making temporal comparisons across the sample panels. Accordingly, we have strata within these management units which: do not have a point per year panel (i.e. cannot be sampled each year). Subsequently, we do not have the initial ability to infer across the entire acreage of each stratum within them.

Strata with five or more plots, would allow for temporal analyses to be conducted on their data. Strata with less than five plots can only be treated as static entities within this time period.

Table 4: Number of Plots Drawn per ACEC

		No. Plots	
Name	Area (ac.)	Drawn	Sampled
Gunnison Sage-Grouse IBA	22,189	6	7
San Miguel River	21,535	9	3
Adobe Badlands	6,380	10	6
Paradox Rock Art	1,083	1	2
Fairview South (BLM Expansion)	608	1	1
Biological Soil Crust	385	0	0
Fairview North RNA	158	0	0
Needle Rock	83	0	0
Dolores River Riparian ACEC	1	0	0

As can be seen in table 4, not every Area of Critical Environmental Concern was able to have an AIM plot located in it. Roughly only half of them had an AIM plot located in them. This is due to them being relatively small in size, the number of AIM plots being relatively few. An alternative AIM sample design, for one of the frames, may require the placement of a plot into each of these individual areas in order to determine long term trends in them. This would allow for determining whether passive management actions in these areas were achieving goals.

Two of the ACEC's which had AIM plots located in them, Adobe Badlands and San Miguel River, did not have as many plots sampled within them as the sample design called for. In both instances this is most likely due to the plots being on hill slopes which were unsafe to sample. However, oversample plots were sampled in other ACEC units, leading to an appropriate number of plots being sampled across the entire area.

Table 5: Realized Weighted Sample Design for the ACEC-WSA

			No. Plots	
Stratum	Plot Wt.	Plot Wt. Ac.	Sampled	Rejected
PJ	1.00	15,457.54	3	0
SS	2.00	7,517.17	5	1
SD	1.25	6,912.41	8	4
MMS	0.50	3,885.26	6	2
AS	1.00	1,032.61	1	9
OT	1.00	785.16	1	4
PP	1.00	535.03	1	5

More AIM plots in Mixed-Mountain Shrub and Pinyon-Juniper were sampled than drawn in the sample design. Accordingly, these plots are weighted to speak to lower amounts of area than initially noted in the sample design. Fewer Sagebrush plots were sampled than intended, and the weight of the sample plots has been increased to compensate for the deficit.

Dominguez-Escalente National Conservation Area

Table 6: Original Sample Design for Dominguez-Escalante NCA

Stratum	Area (acres)	Prop. Area	Prop. Site	No. Plots	Plot Wt.
PJ	50,755	0.45	0.12	3	30,453
SD	21,640	0.19	0.30	10	10,820
SS	21,037	0.19	0.33	11	9,562
RI	6,825	0.06	0.05	2	2,730
GR	3,707	0.03	0.02	1	741
MMS	3,193	0.03	0.10	3	1,916
OT	2,626	0.02	0.01	1	525
PP	1,166	0.01	0.01	1	233
MC	695	0.01	0.05	2	278
AS	120	0.00	0.01	1	24

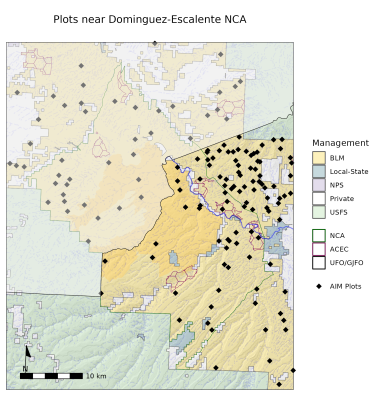

Figure 2: All plots sampled in the vicinity of the National Conservation Area, by both the UFO and Grand Junction Field Office. The GJFO is North (towards the top of the page) of the black line running across the map.

The Dominguez-Escalente National Conservation Area is joint administered with the Grand Junction Field Office, all values here refer only to the portions administered by the UFO field office. As previously mentioned, this area contains both a Wilderness Study Area (The Dominguez WSA), which follows the course of the Gunnison River. It also contains a Wilderness area (The Dominguez Canyon Wilderness), largely composed of difficult to access Pinyon-Juniper woodlands, which grade into Mixed Mountain Shrub in the higher elevations. This NCA also contains two Areas of Critical Environmental Concern (River Rim, and Escalente Canyon), both of which are essentially totally surrounded by the NCA. As expected, River Rim is four large parcels of land adjacent to the Gunnison River, three of which are in lands administered by the UFO field office. The Escalente Canyon ACEC is wholly located within the UFO administered portions of the NCA, and is comprised of Pinyon-Juniper Woodlands atop a mesa.

The total area of the Dominguez-Escalente NCA administered by the UFO field office is 111,930, of which 35,861 is comprised of the Dominguez Canyon Wilderness, and 6,539 by the two ACEC's. The lowest elevation bands in the area comprise riparian vegetation along the Gunnison River, and narrow riparian areas along a variety of the canyon bottoms feeding into the Gunnison. A large portion of the vegetation just outside of the riparian area is Salt Desert, and Pinon-Juniper composes the majority of the remaining areas, much of which is on top of Mesas. 35 total AIM plots were drawn throughout the area, this relatively high number is due to much of the area being categorized as being Sagebrush vegetation under the original sample stratification surface. Another contributing factor was tiny flecks of mixed woodlands at the upper boundary of the parcel, given that the office has small amounts of Aspen, Mixed Conifer, and Ponderosa Pine, these areas had points drawn in them. Realistically these areas are mostly mixed-mountain shrub.

The three plots drawn in the Aspen and Mixed Conifer strata were unable to be sampled, as mentioned due to the isolated and difficult topography of portions of the NCA. One plot fewer than desired in the drawn Sagebrush stratum was sampled, however more plots than expected were sampled in the Pinyon-Juniper and Salt Desert strata. Many of the over samples for 'Other', a stratum which was created by pooling many uncommon vegetation strata, were located in the Monument, but where not sampled. Many of these were located on escarpments.

Table 7: Realized Weighted Sample Design for Dominguez-Escalente

			No. Plots	
Stratum	Plot Wt.	Plot Wt. Ac.	Sampled	Rejected
PJ	0.60	18,271.62	5	1
SD	0.62	6,762.64	16	1
SS	1.22	9,562.11	9	1
RI	2.00	2,730.04	1	1
GR	0.50	370.69	2	3
MMS	1.00	1,915.90	3	0
OT	1.00	525.21	1	3
PP	1.00	233.20	1	1

Gunnison Gorge National Conservation Area

Table 8: Original Sample Design for Gunnison Gorge NCA

Stratum	Area (acres)	Prop. Area	Prop. Site	No. Plots	Plot Wt.
SD	20,195	0.32	0.30	6	16,829
PJ	18,330	0.29	0.12	2	7,332
SS	15,702	0.25	0.33	7	11,215
RI	2,829	0.04	0.05	1	566
OT	2,608	0.04	0.01	0	0
MMS	2,099	0.03	0.10	2	839
GR	941	0.01	0.02	0	0
MC	386	0.01	0.05	1	77
AS	109	0.00	0.01	0	0
PP	22	0.00	0.01	0	0

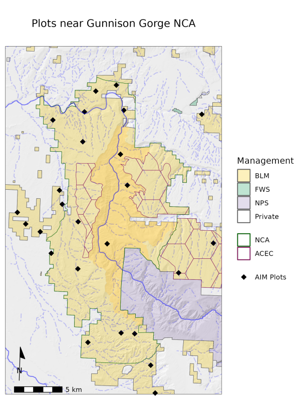

Figure 3: All plots sampled in the vicinity of the National Conservation Area

The Gunnison Gorge National Conservation Area encompasses 63,267 acres, and runs along the Gunnison River downstream of the Black-Canyon of the Gunnison National Park. Despite the enormity of the gorge, this area does not have significant topographic differences as are found on larger parcels of land, and as a result features only six of the original vegetation strata in the design. It is the only sub-unit of the AIM sample design which is comprised primarily of salt desert, which is featured prominently along the Western portions of it. It also contains significant portion of Pinon-Juniper, adjacent to the cliffs overlooking the gorge, and some of the field offices best Sagebrush-Steppe along the Eastern portions.

As previously mentioned, this NCA contains a wilderness area, and an ACEC. The Native Plant Community ONA which is located along the Western Rim of the gorge is roughly 3,788 acres, unfortunately no points were drawn in this area. The Wilderness in the center of the NCA is roughly 17,781 acres, and is a very popular rafting destination. Accordingly points which were located in this wilderness proved challenging to sample. A single portion of the 'Gunnison Sage-Grouse IBA', comprising roughly 5,666 acres overlays this NCA, and it is included in analyses of it, despite not having any plots drawn or sampled in it.

Table 9: Realized Weighted Sample Design for the Gunnison Gorge NCA

			No. Plots	
Stratum	Plot Wt.	Plot Wt. Ac.	Sampled	Rejected
SD	0.50	8,414.66	12	3
PJ	2.00	7,332.14	1	0
SS	1.17	11,215.39	6	1

Of the six strata which had plots drawn within the NCA, only three were successfully sampled. Twice as many Salt Desert plots were sampled as desired, and each of them accordingly has half the acres associated with them as intended. Only one of the two Pinyon-Juniper plots were sampled, hence it's weight is doubled to account for all acres in the original stratum. Fewer Sagebrush plots were sampled than intended, and each is weightde to cover slightly more acres than in the original design.

Summary of Plot Sampling Efforts

Considering all strata in each of the above units together, no clear biases existed in the success of plot sampling efforts with the exception of Mixed Conifer and Aspen plots. As previously indicated, these areas tend to be located at the Field Offices highest elevation locations. These localities are difficult to access due to 1) lack of permission from private land owners whose roads are required to access the area in an efficient manner, 2) steep slopes or treacherous terrain on plot making them unsafe to sample.

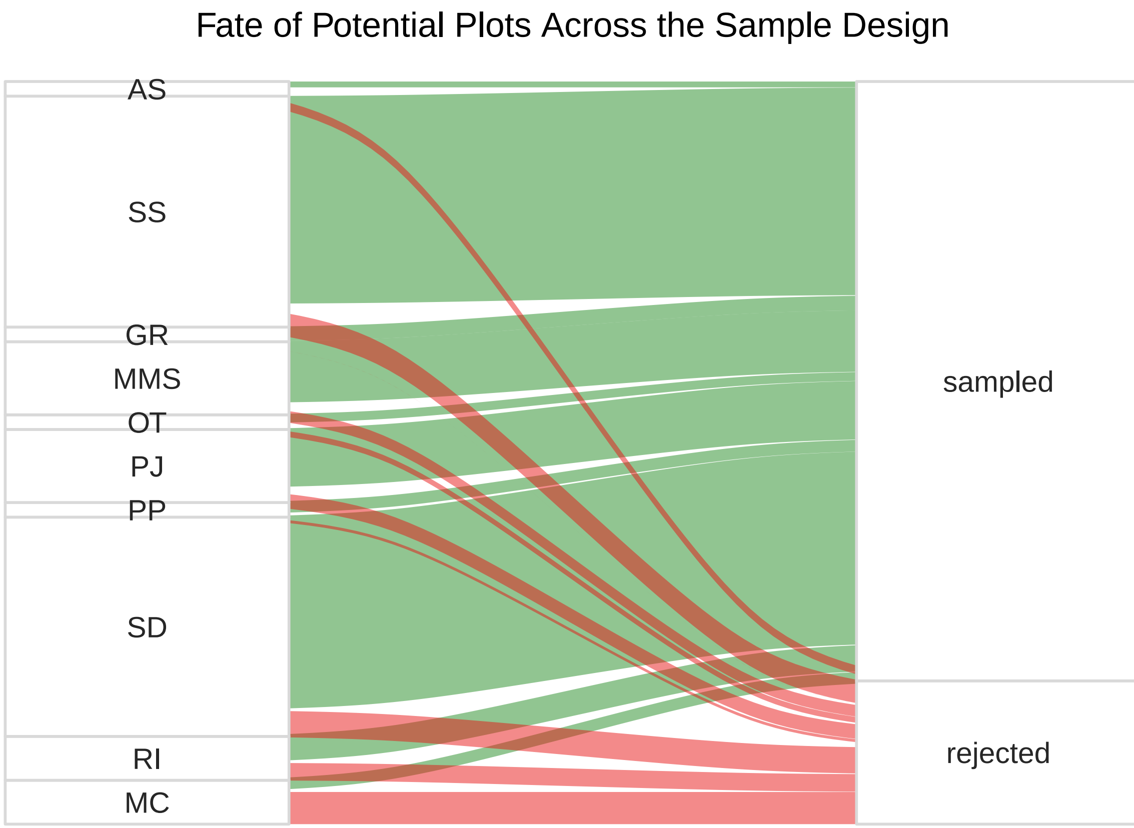

Figure 4: Fates of all potential AIM plots from the Sample Design

Vegetation Classification

The original AIM sample design for the field office utilized stratified random sampling within classified vegetation types which the plots would make statistical inference to. The vegetation types were composed of alliances, communities, and associations, from the GAP-Landfire National Terrestrial Ecosystems spatial data set (Program (2016)). These units were aggregated, by an expert at the field office, to form broader vegetation groups which could have a greater number of replicates per stratum for statistical purposes.

However, the GAP data set erroneously classifies many vegetation types at a non-negligible rate. Accordingly, many areas stratified as a certain vegetation type through the project did not, or seldom, featured the intended target vegetation. Several of the stratified sampling areas are in error, and while we cannot change the area associated with each plot, in certain instances the actual stratum which the plot was sampled in, and the actual cover of that stratum across the landscape will be considered as necessary.

For example, of the nine vegetation types which our AIM project was stratified across, a couple were seldom seen, such as 'Mixed Conifer' vegetation. The study area does contain this vegetation type, however it represents only a fraction of a percent of the field office, and the designated stratified area seldom coincided with it's actual presence. As the mapped areas did not correlate with the vegetation type, neither could the random plots drawn within it. Accordingly the acreage of these areas should, in several instances, be considered within actual vegetation types, which the plots were sampled in, in order that these data are interpreted in the appropriate context.

Additional problems were inherited with the vegetation types known as 'Other'. This is an aggregate developed from the *need* to classify the entirety of the field office. It functions as a catch-all designation for vegetation types which do not have adequate cover to form a broader stratification area and which have little semblance to other eight units. For example, a small patch of Blackbrush (*Coleogyne ramossisima*) on highway 90, gypsum terraces in the Paradox Valley, and escarpment vegetation with Stansbury's Cliffrose (*Purshia stansburyiana*) across the entire study area were placed into this designation. In hindsight as 'Other' is not a group wherein the members have any inherit similarities to themselves, we argue they should align with other groups which they share, even if only weak, affinities. Affinities between the gypsum terraces, to the salt desert, both soils which reduce the availability of free water for plant usage and result in barren to salt-tolerant vegetation are evident. Similarities between the escarpments of mesa's and the Pinyon-Juniper which occupies both the thin soil at the edges of the mesa, and the Pinyon-Juniper woodlands on the rocky soils at the toes of the escarpments, as well as those which are scattered throughout the Stansbury Cliffrose areas make this a tangible target for placement of these 'other' plots.

A final problem is associated with the need to classify bodies of water. Our field office the 'Uncompahgre', is named after a word of Ute origin which has various translations, but a central element of them is a reference to 'Red Rocks' and 'flowing water'. Our design stratum had 4% of the survey area designated as riparian, in part to hold surface rivers. However, given the allowance to shift plots 50m in the cardinal directions, the tri-spoke design of AIM plots requiring a 60m diameter, and the deployment of Lotic AIM during the sampling period, few to none plots remained in the riparian vegetation type. Given that this is a significant amount of cover it is addressed here.

In order to resolve these issues with the analysis of the AIM 2018-2022 sample design, we reclassify the field office into four major, and one very minor, vegetation groups which may accommodate a major swath of the lands in the field office. To accomplish this we use over 1600 random points across the entire extent of the field office, classified in Google Earth, with National Aerial Imagery Program (NAIP) aerial imagery and a couple simple spatial data products, as inputs to a simple random forest model which is projected onto the aerial extent of the field office.

Methods

NAIP Imagery from 2019 were downloaded from the official repository at 'Box' in Fall 2022. While decoding from MrSID (multiresolution seamless image database) to tif file formats, their resolution was reduced by a factor of four using the 'mrsiddecode' program (Vers. 9.5.1.4427) from LizardTech. This effectively reducing their resolution to 9.6 meters. Following decompression of MrSID data, all spatial data processing occurred using R version 4.2.1, and all computing performed on Linux Ubuntu (20 & 22) on multiple hardware. The NAIP raster tiles were united via mosaic, cropped to the extent of the Field Office's ownership, and masked to BLM administered surface areas. These data were then aligned with previously generated raster data sets derived from a 10 meter Digital Elevation Model (DEM); when required all re-sampling of these tiles were achieved using cubic spline interpolation.

A Gray Level Co-occurrence Matrix (GLCM) was created using the 'glcm' package to create a texture raster layer to aid in classification (Haralick et al. (1973), Zvoleff (2020)). Texture bands are, among other properties, capable of indicating the amounts of heterogeneity of habitat types across the landscape. Texture layers were produced using both NAIP natural color and infrared imagery. Texture statistics: mean, variance, homogeneity, contrast, dissimilarity, with windows of 5 in both direction, shifts in all directions (i.e. *Queen's case*), and the default value of 32 grey levels. NAIP data were processed to derive another preditive layer a Normalized Difference Vegetation Index (NDVI) band.

$$NDVI = (NearIR - Red)/(NearIR + Red)$$

NDVI is well suited for identifying sparsely vegetated areas, it was useful in distinguishing salt desert from all other strata, and help in distinguishing between Mixed-Mountain Shrub and Pinyon-Juniper.

To create data set for training a random forest model, all 469 sampled AIM and LMF points, from 2018-2022, as well as all drawn 2022 AIM points, were exported to Google Earth and 440 were classified. 400 random points were generated across the field office and 377 classified in Google Earth via the vegetation ecologist which lead the AIM sampling in 2022. An additional 885 regularly placed plots were drawn across the extent of the field office and 854 were classified. Unclassified computer generated points were generally those that fell upon a wide road, or were outside BLM Ownership. Unclassified AIM/LMF points were LMF points which were the second closely located plot, but under distinct record elements in the TerraDat database. In all instances points were buffered by a 30m radius, to create the dimensions of an AIM plot, and the single most dominant vegetation type was recorded.

To develop a random forest model, the data set of 1657 classified points were partitioned using a split of 0.7:0.3 for the training and testing sets (n = 1146, n = 488) using 'caret' (Kuhn (2022)). The data set was not balanced (number of plots per stratum: AS-7, MC-6, MMS-124, PJ-631, SD-235, SS-143) due to the natural varying presence of these vegetation types in the study area.

The number of mtry in the random forest model were tuned using the function 'tuneRF' with the number of try's set to 1000, a step factor of 1.5 and a relative minimum improvement in Out of Bag (OOB) error rate set at 0.01. The random forest model was then trained using 4 mtry and 1000 trees, all using the RandomForest package with parellel processing (Liaw & Wiener (2002), R Core Team (2022), Corporation & Weston (2022)).

Four simple diagnostic criteria were used to evaulate the predictions from out simple model.

$$Sensitivity = \frac{\text{true positives}}{\text{true positives + false negatives}}$$

Which can be thought of as the probability of the method giving a positive result when the test subject is positive.

$$Specificity = \frac{\text{true negatives}}{\text{true negatives + false positives}}$$

Which can be thought of as the probability of the method giving a negative result when the test subject is negative

$$Accuracy = \frac{\text{correct classifications}}{\text{all classifications}}$$

Which can be thought of as the probability of the method giving the correct result under all conditions. *Balanced accuracy* accounts for differences in the number of individuals in groups, because if the number of individuals in groups differ widely, accuracy may gave a false indication of the models performance. This happens when one group, with many records, is classified very well at the expense of other groups.

$$kappa = \frac{\text{observed agreement} - \text{chance agreement}}{1 - \text{chance agreement}}$$

For our model the overall accuracy was 0.773 (range 0.73 - 0.81 95 % confidence interval), due to an imbalance in the number of observations per group a more appropriate measure for evaluating the overall performance of the model, is the kappa, 0.628, which indicates substantial agreement.

Notably, the model has high rates of Specificity (median = 0.95, less Aspen and Mixed-Conifer), showing that when it predicts a vegetation type onto a pixel cell, the prediction is almost always correct; less so for prediction of Pinyon-Juniper (0.766). However, the model suffers from low Sensitivity (median = 0.71, less AS and MC), indicating that it is unable to detect all occurrences of a vegetation type. For example, the model is only able to appropriately classify half of the occurrences of Sage Steppe (0.475) and Mixed Mountain Shrub (0.596). Accordingly this model is susceptible to over-predicting the occurrence of Pinyon-Juniper, at the expense of Mixed Mountain Shrub and Sagebrush-steppe. This is to be expected given the sample imbalance, which contained many more plots PJ than other types. However numerous trials of reducing the number of Pinyon-Juniper plots did not significantly increase the quality of predictions.

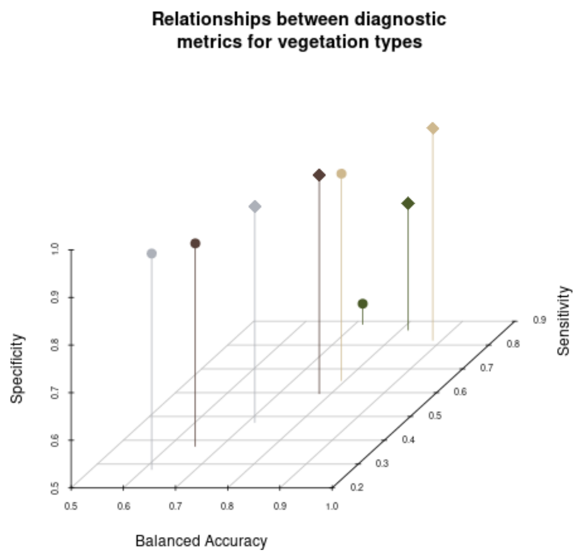

Figure 1: Three dimensional scatter-plot showing the relationship between three common metrics for evaluating the predictions of models. Circles represent the performance of the original stratification, and diamonds the new classification. Positions towards the upper right corner indicate more desirable qualities.

Further indicating, that the features used are not adequate for distinguishing between the transitional points where both Sagebrush-steppe phases into Pinyon-Juniper, and where shrubs increase in Pinyon-Juniper and it phases into Mixed-Mountain Shrub. In part we suspect this is an issue of the difficulty in splitting hairs required to generate the training data sets, and regrowth of historic vegetation in fire scars. While we believe this may be readily accomplished, given the objectives and goals of this process, we believe these are out of the current scope.

In order to determine the relative performance of our model to the original GAP classification a confusion matrix was also generated. A similar number of test points, 486, were used for both data sets. These points were only selected from the computer generated points in order to be independent of the data product, which the AIM plots were derived from. Several metrics indicate the original model is less accurate than the second model. The accuracy of the original model was 0.578 (range 0.53 - 0.63 95% confidence interval), a difference of roughly 0.19 to the new model. This test data set was also unbalanced and it's kappa, 0.368, serves as a better predictor or overall model performance in this case, a difference of roughly 0.26. Considering only the four major vegetation classes also present in the re-stratified model the original has similar rates of specificity (median = 0.93, less AS and MC), but similar to the re stratified model but has lower rates of sensitivity (median = 0.51, less AS and MC) (Figure 1).

On the whole the new model outperforms the older model in all comparisons except for that the sensitivity of the original PJ classification is higher than that of the secondary PJ classification (0.887 to 0.863) (Figure

1). In general their are multiple trade offs in the comparison of models, however the new model is both likely to correctly identify a the strata of a location, and to identify it correctly.

Comparison of Old and New Vegetation Models

The initial sample design only classified around 41% of the field office as Pinyon-Juniper. Based on our field experiences we believe this is a serious underestimate. Despite the need to incorporate several extra - or 'Other' vegetation types into Pinyon-Juniper to fit the four model vegetation classification, manual classification resulted in 60% of the 1600 points being categorized as Pinyon-Juniper. This result is much more in line with our observations, and expectations. That smaller amounts of land were categorized as Pinyon-Juniper by the initial strata actually had large affects on plot placements. Because these woodlands were intended to be sampled at a lower rate than other strata, this roughly 20% of area which is PJ, but missed in the original stratification was sampled more than intended. Thus resulting in lower amounts of information on other vegetation types, such as sagebrush, which we wanted to know more about. Our model was slightly greedy with it's classification of Pinon-Juniper, raising it to 62% of the total area. An important caveat with our use of the models here is that, these areas include many late successional areas, which have not had disturbances to maintain their phases in mixed grasslands.

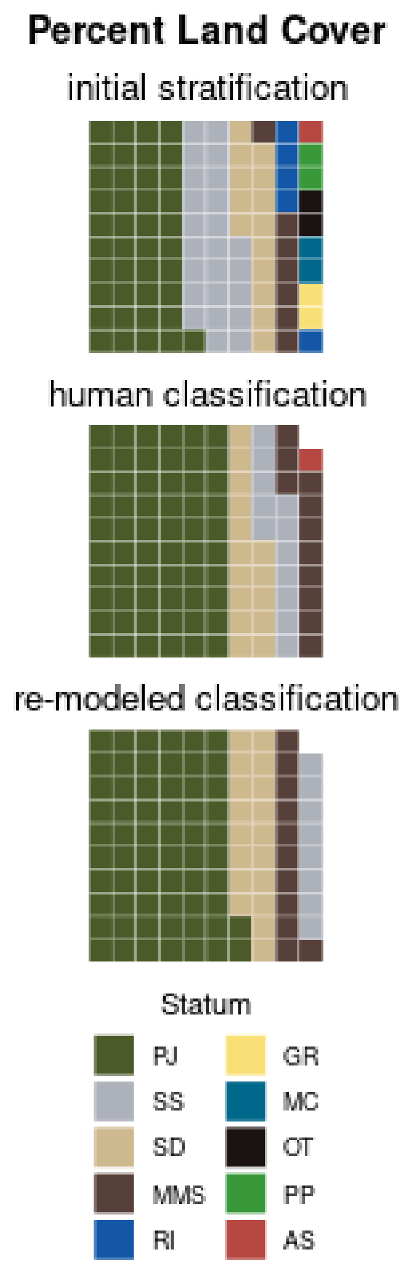

Figure 2: Changes in Predicted Land Cover

Estimates of the amount of Sagebrush habitat differed widely between stratification efforts. In our experience many areas which were classified as Sagebrush in the original classification were actually Pinyon-Juniper, and the age of the Pinyon-Juniper was great enough that this was unlikely to be a result of the different ages of aerial imagery utilized by both models. The original stratification had roughly 25% of the entire field office classified as Sagebrush, this value was too high. Our manual classification resulted in 12% of the field office being classified as Sagebrush, and our final classification from the model resulted in only 9% of land being classified as Sagebrush. We feel the true value is closer to 12% than the 9% predicted by the model. As mentioned, not as many plots were included in this strata as desired, due to the classification accuracy, rather many of the plots in this strata ended up being Pinyon-Juniper. Classifying sagebrush was difficult for us due to areas where it transitions into Mixed-Mountain Shrub and Pinyon-Juniper, i.e. what constitutes the transition point between these? Theoretically only small amounts of these two types are required to convert a land, and given Sage grouse preferences these make sense for our purposes.

The amount of Salt Desert is similar between all three methods, they agree the amount of salt desert in the field office is around 15% - 18%. In general, these predictions aligned well, and plots which were classified as salt-desert by the models actually were.

A notable incongruence is present with Mixed-Mountain Shrub between the systems. The original classification predicted very modest amounts of this stratum throughout the area, and predicted relatively high amounts of other forest types. Our manual classification, which retained Aspen and Mixed Conifer forest, detected nearly twice as much of this strata in the field office. The modelled cover was slightly less than this, and likely closer to a true value where 10% of the field office is Mixed-Mountain Shrub.

The relationships for the six remaining minor components is more complex, as is illustrated in Figure 3. This figure tracks the movement of each individual raster cell (or 'pixel') from the old to the new classification. As previously mentioned, the accuracy of the old system cannot be officially compared, but only discussed, because we reduced our classification process to the four major and two minor strata. For example, the original classification (lower half of circle), was correct in it's identification of Pinyon-Juniper, however it was unable to detect all of it. Hence the green line tracking from the bottom to the top, contains nearly every single pixel, except for a few which we were able to identify as Mixed-Mountain Shrub. Nearly all 4% of the Riparian stratum can be accommodated in Pinyon Juniper or the Salt desert stratum. This is sensible as nearly all ephemeral rocky canyon bottoms support Pinyon Juniper vegetation, and as the streams flow through the lower elevation portions of the field office, the riparian band is only a meter or two wide, and hence what an AIM crew would sample is Salt Desert. Most of the original Mixed Conifer was reassigned as Mixed-Mountain Shrub, with only a portion remaining in this stratum. Similarly, it appears roughly half of the Ponderosa Pine was assigned to Pinyon-Juniper and the other to Mixed-Mountain Shrub. Roughly half of the area classified as Aspen was maintained, and the other half was also reclassified as Mixed-Mountain Shrub. Most of the grassland stratum was transferred to Salt Desert

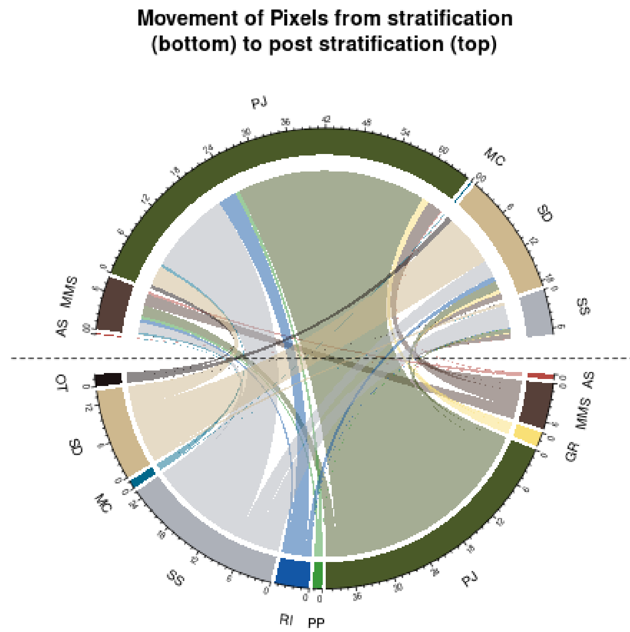

Figure 3: Depiction of how raster cells in the UFO are redistributed from the original sample design (lower half), to the the reclassified spatial product for the sample design (upper half)

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Ecological Site Desciption Completion

An *ecological site* is an area of land which has similar environmental conditions, e.g. climate & soils, and which will produce similar types of vegetation when both undisturbed and when subjected to the same type of disturbance, e.g. by wildfire (Butler et al. (2003)). The idea of Ecological Sites is a broad system of models, each of which seeks to capture these re-occurring motifs of climate and soils across the landscape. There goals are to guide land managers decisions at geographic levels at which they may effectively achieve desired outcomes, e.g. the level of grazing allotment. The aggregation of areas across the *landscape* reduce the amount of monitoring and research required to make informed land management decisions, and allow for the transfer of successful practices from one area to another. Each Ecological Site is a concept which inherently captures a range of variation across the landscape, and each site features varying degrees of dissimilarity. Given the resources available to land managers Ecological Sites form the most thorough and useful model for classifying natural lands.

Ecological Sites are developed by the Natural Resource Conservation System (NRCS). While the current conception of what an Ecological Site is has theoretical roots in the science of Ecology nearly 100 years ago, the formulation of specific Ecological Sites under the current paradigm began around 1997 (Brown (2010), Karl & Herrick (2010)). There popularity was a response to the failure of 'one size fits all' management decisions which were common throughout the 20$^{\text{th}}$ century (Bestelmeyer & Brown (2010)). The existence of Ecological Sites allow more fine tuned assessments of the possibilities which exist at each site, i.e. they can incorporate the observed number and identity of plant species and their annual growth and land managers can compare the current status of these attributes to the possibilities (Brown (2010)). Based on what multiple uses occur on this land, the land can then be managed to maximize it's benefits to all users.

The implementation of describing Ecological Sites by the NRCS occurs at the level of *Major Land Resource Areas (MLRA)*, and the *Land Resource Units (LRU)* within them. Different MLRA's, and there LRU's, are at varying states of completion. Anecdotally, nearly all ecological sites have been established for MLRA 34b and 36, while few have been developed for MLRA 46; in general more arid LRU's are more likely to have complete ecological sites relative to more mesic areas. MLRA 34b and 36 cover virtually all UFO land west of the foothills of the Cimarron, except for the highest elevation areas adjacent to Forest Service. Subsequent to the identification of an individual ecological site is recording and documenting a variety of it's ecological parameters in a written format known as an *'Ecological Site Description'* (Bestelmeyer & Brown (2010)). These documents follow a standardized format, but also vary in the degree of information which they contain. For example, most descriptions contain tables which include the *state and transitions models* which occur between vegetation, contain production values for vegetation in a *reference* state and phase, however some will contain production values for multiple states and phases (Bestelmeyer et al. (2010)). Given the amount of detail and expert knowledge required in the identification and development of an ES and then to write the description of the site, the creation of an ESD is a time intensive process (Bestelmeyer et al. (2010), Moseley et al. (2010)).

An important attribute of an Ecological Site Description (ESD) are *benchmarks*, ranges of attributes which reflect the variation observed in that ES while the area is in a certain state and phase. Benchmarks provide quantitative references to which land managers may compare lands to in order to contextualize there current status (Bestelmeyer et al. (2010)).

> "*In other words, the ecological site determines what is possible, the current state determines what is realistic, and the phase within a state conveys the current conditions and likelihood of future transitions.*"
>
> — Karl & Herrick 2010

We endeavor to make all management decisions based on the quantitative benchmarks in ESD's. Here we summarize the current levels of completion of ESD's across the UFO field office. As our management objectives rely on these benchmarks to determine whether our land is within the range of natural variability, they are an essential component of this.

Results and Discussion

Ideally, it would be possible to correlate every single AIM plot to an Ecological Site, and the description (ESD) for that site would be completed. This is unlikely due to two constraints, 1) the NRCS soil survey components can contain inclusions which are up to 15% of the area, 2) not all ESDs have been completed, and many are at varying stages of completion. While these provide limits on the information which can be gleaned from each plot for specialized management decisions, statistical procedures exist which allow for inference of overall land conditions regardless.

Of the 281 plots sampled by the AIM crews 237 were correlated to an ecological site, with a total of 41 unique Ecological Sites being sampled in the first AIM sample frame. As can be seen in Figure 1, most plots are located in just a few Ecological Sites, and most Ecological Sites had only a few plots sampled in them. As the plot locations were randomly stratified, this indicates that a few Ecological Sites make up much of the the Field Office. A few of the most common Ecological Sites are missing completed Descriptions. The lack of these few descriptions disproportionately affects the number of plots with existing quantitative benchmarks.

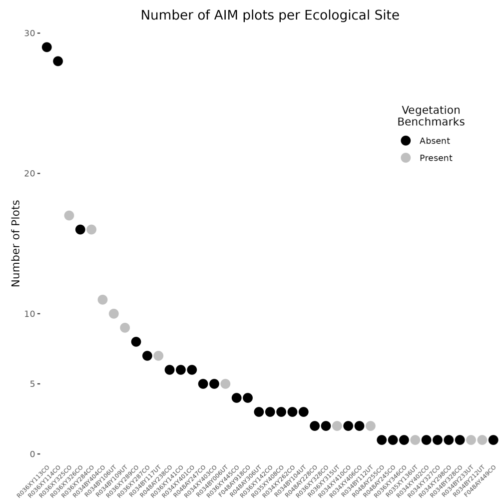

Figure 1: The number of AIM plots correlated to each Ecological Site, and whether each site has a written Description

Of all 281 plots sampled the total with the full set of quantitative benchmarks is 88. Roughly 16% of plots were not correlated to an Ecological Site, a proportion close to the 15% tolerance for inclusions across the map units. Of the plots which were correlated 47 were in Ecological Sites which do not yet have an Ecological Site Description which has made it to the publicly available draft stage. Of the 190 plots with ESD's, only 82 of these have quantitative benchmarks. Because of this, a variety of approaches will be used to estimate the quantitative benchmark values associated with many Ecological Sites. With the exception of the vegetation benchmarks, these will be discussed in each section.

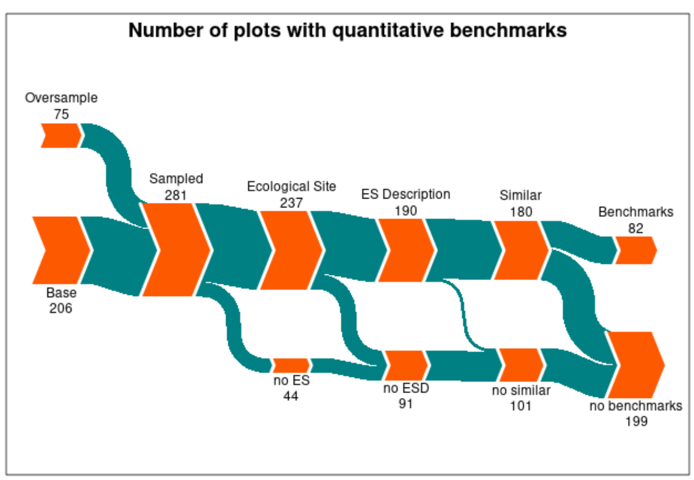

Figure 2: The fate of plots regarding Quantitative Benchmarks

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Increase range of Plant Functional Benchmarks

The quantitative benchmarks of Ecological Site Descriptions (ESD's) are meant to capture the ecological variation inherent in a state and phase under multiple conditions. These conditions range from years of consecutive drought to surpluses of moisture, and following multiple types of disturbance. The benchmarks are intended to capture the variation that would be found in this state and phase combination across the geographic and climatic extents of the Ecological Site in the MLRA for which it has been drafted or adapted. Some of the quantitative benchmarks, of the cover of plant functional groups, which we collected from ESD's were very narrow (Figure 1, top panel). In many of these instances the values reported in ESD's were more narrow than the uncertainty of the estimates of the true value of the population gleaned from a single AIM plot. It is apparent that several ESD developers did not emphasize the natural variability of the benchmarks while generating the vegetation benchmarks. This may be due to them only collecting quantitative vegetation data at a single point in time within the Ecological Site, accordingly it seems in multiple instances they may only have had a point of datum, and did not feel comfortable estimate the variation in the system.

Mean	**Range**	**Variation***
1 - 10	< 3	30.0%
11 - 20	< 4	26.7%
21 - 30	< 5	20.0%
31 - 50	< 6	15.0%
51 - 100	< 7.5	10.0%

Table 1: * calculated as $\frac{max(Range)}{\text{midpoint(Mean)}} * 100$

While the approach described above is prudent in the development of an ESD, it is not prudent for us to assume such narrow ranges of variation while evaluating land health. Benchmarks ranges which are very narrow may unduly penalize estimates of the amount of land under analysis which are meeting condition benchmarks. This can lead to a misleading assessment of land health conditions.

For example, if we seek to determine if plots within an ESD are meeting a standard - such as having from 20 - 30% grass cover, than given the time frame under which we sampled, plots with 20% grass cover have meet this condition. However, if this range of grass cover was reported as 24-26% than, a plot within the noted ranges of variation may not meet the standards. Here we seek to identify and broaden these estimates, we will use a simple method of *imputing* values in the context of *feature engineering.* A *linear model* will be fit to the benchmark values, which contain realistic ranges, and then the slope of this model will be used to fill in the missing values.

Ranges of estimated benchmark variation were estimated as being too low if they fell within the ranges in Table 1 & Figure 1 *top panel.* These 81 values were removed from the initial data set. The remaining 59 observations were used as **training** data for the linear model: lm(Range $\sim$ Mean + Functional Group). We believed that the variation associated with each measurement of range would decrease as the mean cover increases. In other words, as the mean of the cover estimate get's larger, the percent variation of the range of the benchmarks decrease.

This simple model served to explain a moderate amount ($r^2 = 0.28$) of the variation in the range of cover estimates. Based on these data it found moderate evidence that functional type has a effect on the slope of the trend line ($p = 0.0056$), and there was slight evidence ($p = 0.0106$) that increasing the mean of the values had a negative effect on the observed variation. However, this effect was much weaker than the additive co-variate of functional type; and the final results nearly exclude the term. Contratry to our expectations, for every one percent increase in the mean of the benchmark values the percent range in variation of the estimates increased by 0.001 percent.

Once the linear model was *fit*, the removed data points had both the Mean, or midpoint of their cover estimates, and functional types used together to predict what the Range of cover estimates would be for an ESD (Figure 1 middle panel). These estimates were then combined with the original data which meet the desired range of cover information to provide an altered quantitative benchmark data set (figure 1 bottom panel & figure 2.

For example, for the shrub cover benchmark which had a midpoint (mean) cover of 40% (Figure 1, top panel), and a range of ~2%, had a range of 38-42%. We widened this value to roughly 27-53%, maintaining the midpoint of 40%. After performing this process for all Ecological Site Descriptions which had narrow benchmarks (those beneath the dashed line in Figure 1, top panel), we recombined all functional group benchmarks as shown in Figure 1 panel 3, and figure 2.

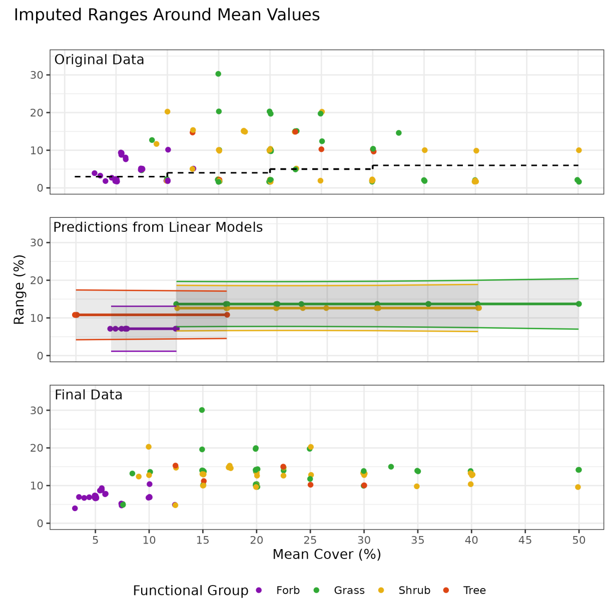

Figure 1: Original data (top), with values without sufficient variability beneath the black line. Predictions from linear models (middle). Results (bottom).

Ecological Site Description (ESD) and Ecological Site Group (ESG) Comparision of Vegetation Benchmarks

The development of Ecological Sites (ES) and their descriptions (ESD's) represent an enormous effort on behalf of the Natural Resources Conservation Service (NRCS) (Bestelmeyer (2015)). As discussed in Section 3, they do not yet form a continuous coast to coast, or across the field office data set (Twidwell et al. (2013)). In order to allay the significant amounts of effort required to develop these, and to make land management decisions in the interim, alternative classification systems have been proposed (Nauman et al. (2022)). One system which incorporates the NRCS ESDs, and which provides continuous coverage across our study area, are the Ecological Site Groups (Nauman et al. (2022)).

The Ecological Site Groups developed largely by United States Geological Survey (USGS) researchers at the Southwest Biological Science Center in Moab Utah, with assistance from Bureau of Land Management Colorado State Office Staff, are meant to both bridge the spatial gap in ESD development, and to provide a framework for land management decisions at a larger scale than that which generally occurs at a BLM Field Office. The Ecological Site Groups were developed by grouping together similar ESD's, and which will respond in a similar manner to management actions (Duniway et al. (2016)). To these initial groups of Ecological Sites, the field data which largely informed the ESD creation, most notably soil physical parameters, with the use of objective quantitative approaches, which make use of a simple *Machine Learning (ML)* method *random forest* were used to identify recurring environmental parameters in the data set into groups, and then project these onto a 'map' which covers the Upper Colorado River Basin.

While for most land management decisions at the Uncompahgre Field Office ESD's, when available, represent the best available scientific evidence to inform decisions, ESG's provide the best alternative, and in the future are likely to be influential for large scale decisions at the UFO. Herein we address questions regarding the similarity of estimates arising from ESG's and ESD's. In this report, we are using ESD's for the *reference state benchmarks* which they contain, the quantitative goals which we seek to compare land to, and seek to visualize the relationship between the benchmarks of ESD's to ESG's.

The goals of this section are to 1) determine which ESD's constitute the ESG's across the UFO? And 2) visualize the similarity in benchmarks within an ESG and the ESD's which it contains. A short investigation of how different methods of extracting the spatial data set to the ground truthed data, as well as a comparison of the accuracy of the predicted ESG's limited to the UFO area and ground truthed plots, is also included.

In the publication of Nauman et al. 2022 the accuracy of the final product was calculated as being 57.4% (Nauman et al. (2022)). However, In our experience certain predicted groups are often more difficult to reliably predict, and without sufficient testing sample sizes many spatial products will score higher, or lower, than these averages in areas with/without these features.

Methods

The mean fractional cover of vegetation was manually transcribed from Table 3 of Appendix A from Nauman et al. 2022. The spatial data product, a gridded surface of predicted ESG's, which was the outcome of the study was accessed from sciencebase.gov catalog on (Dec. 16, 2022, https://is.gd/c89lNz), three additional layers predicting soil geomorphological groups and another predicting climate zone were also downloaded. ESD quantitative benchmarks values, and AIM plot locations were cleaned in previous work (Section's 2 & 3). All analyses were performed in R version 4.2.2 using RStudio on Linux Ubuntu 22.04 LTS (R Core Team (2022), Sobell (2015), RStudio Team (2015)).

Test Raster Extraction Methods

To test whether we were extracting and processing the correct ESG from the gridded surface two comparisons of raster extraction methods were attempted. The first (hereafter: 'polygon'), which is typically performed at UFO, is to buffer the point to represent the actual area of an AIM plot and extract to that area, and choose the categorical class with the most cells per value (the statistical mode). The second (hereafter: 'point') option is to extract from the raster surface values from the gridded surface directly to the point geometry. The idea behind this is that the pixels from the gridded surface featured a 'splotchy' characteristic, a byproduct of not performing spatial operations to clean up the predictions (e.g. 'Focal Statistics'); oftentimes these central isolated cells are not artifacts of analysis, but rather true points which are swamped by the adjacent cells. The slightly higher performing method, 'polygon', was used for the duration of the analyses.

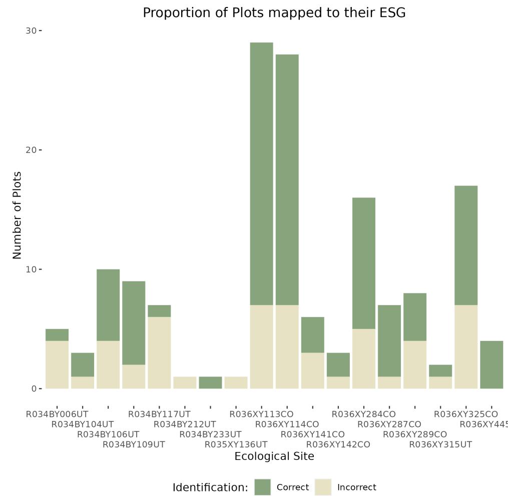

Figure 1: Proportion of Plots in ESDs which were included in the ESGS and should map over unequivocally to an ESG

Test Local Accuracy of Raster Dataset

To determine whether the raster dataset gives a higher or lower performance accuracy in the UFO portion of it's range the 157 AIM plots in 18 ESD's which are known to map directly to 10 ESG's were utilized. These plots were extracted from the ESG gridded surface and the proportion of each ESG which was correctly and incorrectly mapped were calculated (Figure 1).

Determine ESD and ESG Groupings

To inform a strategy for mapping individual ESDs directly to ESG's a bipartite network was created using all AIM plots with verified ESD's, all statistics (*see oneline supplement*) were calculated using the package 'bipartite' (Dormann et al. (2008)). This approach was deemed necessary given how many erroneous relationships were returned by the ESG extraction method (Figure 2).

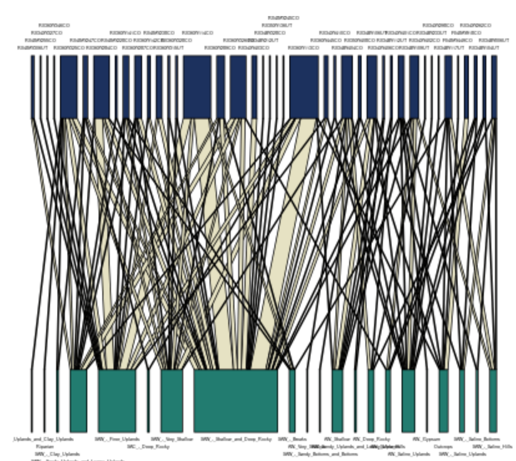

Figure 2: Initial Relationship Between Field Verified ESD and ESG extracted from the gridded surface

All observed ESD's were matched to ESG's in an incremental fashion. 1) ESD's which were used directly in the creation of ESG's were removed based upon a noted association in Appendix 6 of Naumen et al. 2022. 2) ESD's for which only a single AIM plot existed, had the ESG which values were extracted from it listed as it's ultimate mapped association. 3) ESD's with greater than 1 AIM plots associated with them, and which had all plots match an ESG were classified as such. 4) ESD's with over 65%, our local accuracy of the gridded surface, of their AIM plots mapping to a single ESG (Figure 3).

Following the removal of these ESDs from the initial dataset, the three Soil Geomorphologic Units (SGU) surfaces, and the Climate Zone surface were used to match the remaining ESDs to an ESG. Each of the remaining points had values extracted from each layer of the SGU surfaces, each representing the classification prediction from the three top performing classifier models. The most commonly occurring ESG (in others words the statistical mode), was then calculated considering each of the SGU layers per point, across all points in the ESD simultaneously. All climate zones were also extracted to the points, and the most commonly occurring climate zone per ESD was selected as the climate zone to classify these sites in the SGU framework, thus forming the ESG.

The results of these analyses led to the development of a draft ESD-ESG lookup table for the Uncompahgre Field Office for this project.

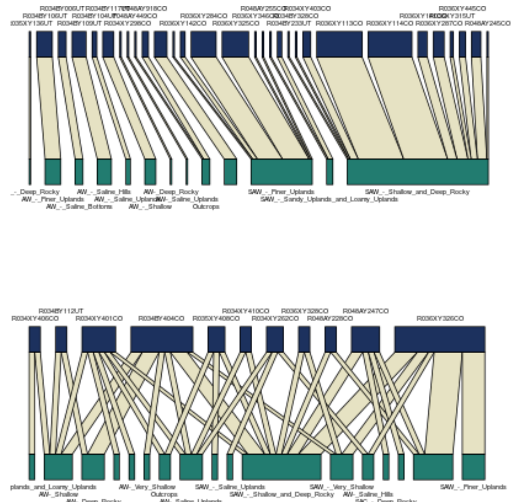

Figure 3: Relationships Between ESDs and ESGs midway through the cleaning process

Results & Discussion

157 AIM points were in the 18 ESD's which were directly involved in the creation of 10 ESG's and could be used for assessing the results from the alternative raster data extraction methods and for assessing the accuracy of the ESG gridded surface in the area of analysis. The method of extracting the ESG from the gridded surface to plot had little influence on the accuracy of the ESG. Regardless the method of buffering the point to the real size of the AIM plot was used as it slightly outperformed plot centers on the 157 known sites, 0.65 to 0.637. 157. The ESG gridded surface was able to correctly place 0.65 plots to the appropriate ESG, rather than the 57.4% reported for the entire study area (*Figure 1*). A caveat associated with this result is that our value is likely to be biased towards more common ESG classes. We accept this bias as the scale at which we make management decisions is so great.

The initial extraction of ESG to AIM points was problematic (Figure 2). Ideally, if ESD's were the entities which defined the ESG's, than each ESD (the upper boxes in the panel) would link to a single ESG (the lower boxes in the panel), an image of this relationship is displayed in Figure 4. As is apparent in Figure 1, this was not the case, however much of the error in these classifications lay with the classification of the gridded surfaces, rather than with the ESG concepts themselves.

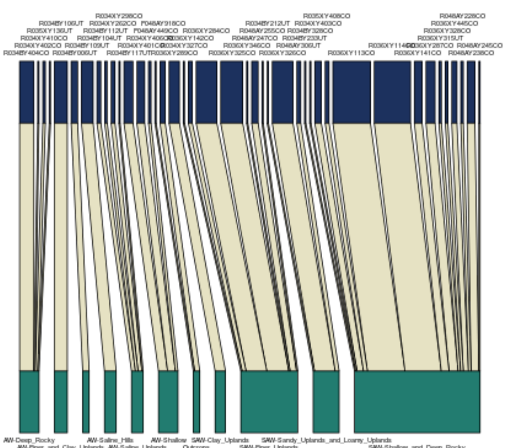

Figure 4: Relationship between ESDs and ESGs at end of process

In order to map these relationships we utilized a step-wise process. 1) 18 ESD's, associated with 157 plots, which were used directly in the creation of 10 ESG's were removed based upon a noted association in Appendix 6 of Naumen et al. 2022. 2) 8 ESD's for which only a single AIM plot existed, had the 5 ESG which values were extracted from them listed as there ultimate mapped associations. 3) There were 2 ESD's with greater than 1 AIM plot associated with them, which had all plots match the same 2 ESG's. These ESD's, F048AY918CO, R034XY403CO, were associated with 9 AIM plots, and mapped to: *Outcrops, SAW Shallow and Deep Rocky.* 4) Finally, 2 ESD's with over 65%, our local accuracy of the gridded surface, of there AIM plots mapping to a single ESG were recovered. These ESD's, R048AY238CO, R048AY306UT, were associated with 9 AIM plots, and mapped to: *SAW Shallow and Deep Rocky, SAW Sandy Uplands and Loamy Uplands.* At this point the classified ESD's and non-classified ESD's appeared as Figures 3 respectively.

The remaining 11 ESD's, associated with 54 AIM Plots, were all mapped to 6 ESGs via the methods of summarizing both all three SGU layers in conjunction with climate zones, from Nauman et al. 2022. The final lookup table of ESD to ESG mapping is in Figure 4. 11 of the 35 ESGs were present in the UFO, the number of plots per ESG ranged from 5 (AW Saline Bottoms and Bottoms) to 97 (SAW Shallow and Deep Rocky).

Of the three major climate zones defined in Nauman et al. 2022, the UFO does not contain the Semiarid-Cool zone.

The quantitative estimates for the cover of major vegetation types by group are presented in Figure 6. Note this figure does not contain 'Outcrops or Riparian ESG' both of which do not have ESD, nor are they target areas for management considerations. Riparian areas fall within the domain of Lotic AIM, and the ecology of outcrops is a management action over the geologic time scale.

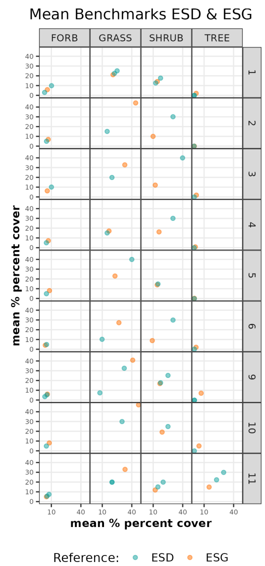

Figure 5: Fractional cover, note Outcrops and Riparian, have been removed

The covers of forbs, with both perennial -and to an extent annual- life cycles varies the least across lifeforms, their cover ranges from 4 to 8 (Fig. 5, Column 1). Given the variability inherent within the concept of an Ecological Site, and the years from which a large amount of the data for which the ESG cover values were calculated a marked majority of this cover should be constituted of perennial rather annual forbs. The cover of trees per Ecological Site Concept has the next least variability (Figure 5, Column 4).6 of 9 sites have < 5% tree cover, with the exception to this being the Finer Uplands, conceptually a very large ESG, which includes nearly all of the Pinyon-Juniper Woodlands, along with high notable amounts of Sagebrush sites. We presume that the inclusion of Sagebrush sites into this ESG is the reason that for this concept alone the tree cover in the ESG exceeds that noted in the ESD.

Greater divergences are observed between the estimates of cover for Grass and Shrub. The cover of grasses in two ESG's, are conceptually identical, however in the other seven ESG's the cover of grass notably exceeds that noted in the ESD's (Fig 5, Column 2). The relationships between the shrub cover values are more nuanced, 2 ESG's have conceptually identical cover values, three have very similar values, however for the remaining four ESG's the cover of Shrubs is less than noted in the ESD's (Fig 5, Column 3).

Conclusions

The relationship between the novel ESG's and the ESD's are less straightforward then may be offered by simple extraction of values from gridded surfaces. We find that Nauman et al. 2022 were largely hampered by the same issues as ourselves (supplemental online materials; 'cluster_ESD') in trying to map ESD's into meaningfully similar groups. We believe that we are the first persons to attempt to operationalize the ESG concepts which they utilize in their publication.

> "*We emphasize the need to corroborate mapped classes with field confirmation for site-specific management.*"
>
> — Nauman et al. 2022, P. 24

In regards to using ESG's we feel that the best option is to continue verifying the NRCS Ecological Site at all sites, and then using a spatial approach as above to determine which ESG an ESD is allied to. The additional use of the Dichotomous Key (Figure 3 of Nauman et al. 2022), with ocular estimates of Electrical Conductivity via 'sparkling' parameters of the soil may be used by crews in the future to develop a larger ground truthed data set for the UFO. Alternatively probes which may measure EC are readily acquired, easy to calibrate, and may operate with only small amounts of soil which has been collected in the field.

For vegetation, the cover of ESG's will be used as benchmarks.

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Drought

The AIM *sample design* contains five panels, one for each year, with the intention that temporal analyses can be conducted across the panels. Ideally, these panels will reflect consistent natural climatic variation. In others words, some years will be drier while other years will be wetter, but over the period the climate conditions will be average. Then the effects on sampling year can be considered when interpreting the results of analyses. However, we felt, and our constant monitoring of the University of Nebraska-Lincolns Drought Monitor over this period, supported the notion that we sampled during a period of immense drought. In fact, the sampling was concluded during a period of drought shown to be the most severe across Western North America since the year 800 A.D. (Williams et al. (2022)). Here we contextualize our sampling period within the known climatic variation in the field office, and discuss why we dismiss calculating AIM metrics separately for each year.

Contrary to the concept of Agricultural drought, the exploration and understanding of Ecological drought is nascent and not well defined (Crausbay et al. (2017), Slette et al. (2019)). In fact, numerous recent papers published in *peer-reviewed journals* have sought to both define this term, and formulate it in a way which it may be applied to the management of natural resources. A recent example of the term is:

> "***ecological drought*** *... an episodic deficit in water availability that drives ecosystems beyond thresholds of vulnerability, impacts ecosystem services, and triggers feedbacks in natural and/or human systems*"
>
> — Crausbay et al. 2017

Guidance regarding how to move from defining the term 'ecological drought' to using it this term is also recent, and limited. The most recent indication of making this step are:

> "*We suggest that future drought publications provide at least one of the following:* ***(a)*** *the climatic context of the drought period based on long-term records;* ***(b)*** *standardized climatic index values;* ***(c)*** *published metrics from drought-monitoring organizations;* ***(d)*** *a quantitative definition of what the authors consider to be drought conditions for their system*"
>
> — Slette et al. 2019

In this section of the report, we perform both operations **a, b**, and calculate our own drought metrics using the exact same equations, and software implementations used to calculate **c** as these organizations. We opt to calculate **c** ourselves due to the topographic complexity (rugged mountains) of the study area making the coarse scale models generated by other organizations of limited utility in understanding the Uncompahgre Field Office. Finally, for the purposes of this document we do define **d** for our field office, using the expertise of several Natural Resource Specialists in Western Colorado. Hence, in this section we meet not only the suggestion to meet one of the metrics above, but all of them.

Methods

Calculations of metrics which measure drought essentially reflect the relationship of 1) the amount of moisture entering an area, 2) and the amount of moisture leaving that area, and how 3) these values are balanced. When more moisture enters an area than leaves it, the moisture balance will be positive, when the moisture balance is negative for an extended period of time the area will in drought. Hence all equations for measuring drought may be conceptually simplified to

$$1_{MoistureIn} - 2_{MoistureOut} = 3_{Balance}$$

When $1_{MoistureIn} < 2_{MoistureOut}$ than $-3_{Balance}$ is negative and drier conditions prevail. In calculating drought the amount of moisture entering an area (1), is simply equivalent to the volume or precipitation, in any form (e.g. rain or snow). The amount of moisture leaving the area (2) via the process of evaporation is driven by many processes, such as the amount and intensity of sunshine, wind, temperature. The amounts of potential evaporation (the evaporation which would occur is sufficient amounts of water were present to allow) is commonly modelled using one of three models, which since their original developments have been slightly modified in numerous ways (Xiang et al. (2020)). In order of increasing complexity these are the Thornwaite, Hargreaves, and the Penman-Montieth equations (Thornthwaite (1948) Penman (1948), Hargreaves et al. (1985)). We chose to utilize the Penman-Montieth equation to estimate the Potential Evaporation in our study area, because both the Thornwaite and Hargreaves equations have been shown to under-estimate drought conditions in arid regions (Begueria et al. (2014)).

The most common and widely employed metric of measuring drought is the Palmer Drought Severity Index (PDSI). This metric is used by the USDA, and is highly effective for conveying information regarding drought in agricultural settings (Palmer (1965)). However, a metric preferred for calculating general drought outside of irrigated agricultural areas is the Standardized Precipitation Index (SPI) (McKee et al. (1993), Hayes et al. (2011)). One drawback of the SPI is that it does not account well for long linear trends, namely climate change. Using SPI would underestimate the values of drought in the study area, as drought is contingent upon air temperature and it's effect upon evaporation. To overcome this shortcoming the Standardized Precipitation Evapotranspiration Index (SPEI) was developed (Vicente-Serrano et al. (2010), Begueria et al. (2014)). We will employ the SPEI for visualizing drought.

Variable	**Source**
precipitation sum (mm)	GridMET (pr)
mean max temp (ºC)	GridMET (tmmx)
mean min temp (ºC)	GridMET (tmmn)
mean rel. humidity	$(RH_{max} + RH_{min})/2$
mean wind speed (km-hr-1)	GridMET (vs)
mean sun hours (hours)	r.sunhours ('sunhour')
mean solar radiation (MJ-m-d)	r.sun ('beam rad')
mean cloud cover (percent)	Wilson, EarthEnv
elevation in meters	EarthEnv 90m

Table 1: Variables used to Calculate Potential Evaporation

Metrics of sunshine were calculated using r.sun (total beam irradiance) and r.sunhours (daily sunhours) in Grass GIS (GRASS Development Team (2017)), on Linux Ubuntu 20.04.5 LTS. The function r.sunhours requires a year for which to calculate these values for, the year 2000 was selected as it represents a rounded midpoint of the temporal range of the climate variables (1979-2021), the linke value for r.sun was set at 2.35 (in lieu of the default value of 3.0), which is the mean of the annual linke value for mountains (2.75), and rural areas (1.9). Subsequent to these calculations, which were performed for each day of the year, the values were recalculated as monthly means. Subsequent to the calculations the total sun hours were subtracted from the mean monthly percent cloud cover dataset. All subsequent analyses were performed using R in the Rstudio Environment on Linux Ubuntu 22.04.1 (R Core Team (2022), RStudio Team (2015))

All climate variables, aside from cloud cover, were downloaded from gridMet using 'Get_Gridmet' (Abatzoglou (2013), Lovell & Benkendorf 2022). These variables were chosen in lieu of one of the two datasets from which they are derived, PRISM, due to that dataset lacking Wind Speed and Relative Humidity data (Abatzoglou (2013)). As gridMET data are distributed at a daily resolution, the values were aggregated to the mean of each month of each year from 1979 to 2021. The temperature values were converted to Celsius from Kelvin $x - 273.5$.

Cloud Cover data were downloaded from EarthENV, from the study of Wilson (2016), and were transformed back into percentages by multiplying $x * 0.01$, raster data are often transmitted with alterations of decimal points to reduce the file size.

The digital elevation model (DEM), which was utilized in the calculation of sunshine metrics and in the Penman equation, was downloaded from EarthENV, and re-sampled from it's 90m resolution to align with the grain of the other datasets - 4km (Robinson et al. (2014)).

SPEI was calculated for all 42 years between 1979-2021. Cells in rasters which contained missing (NA) values, these resulting from missing a value required for either the penman or SPEI calculations, were filled using the 'focal' function from the r package 'Terra'. This function calculated the mean of the 9 nearest pixels to the missing value. SPEI was calculated using moisture balances of the: 6, 12, 24 months preceding the current month of analysis. For example, the SPEI value for the Month of January 1981, under a scenario with a 6 month window, would go back as far as June 1980, while for a longer window such as 24 months, would go back to January 1979. Because our SPEI calculations included these windows, each data set could not-natively start at the same date. For example our dataset with the longest SPEI window, 24 months, exceeded the shortest windows by 18. Accordingly, we removed any months of values preceding the origin date for the start of the longest SPEI calculation intervals, and for ease of analysis and reporting began our background climate dataset in 1981.

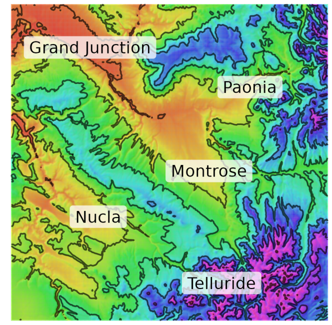

Figure 1: Stylized topographic map of the area of analysis

We subset the drought data so that the beginning period for all temporal extents has the same start date, January 1981. It is recommended to have 30 years of climate data to compare local conditions to. We will span the years from 1981-2015 as our 34 year background data. This will also allow a slight degree of buffering between our data sets, so that the starting times between our two datasets, the occurrence of drought during the AIM sampling, and the historic climate variables are not temporally replicated.

Results & Discussion

The area of analysis varies in elevation from 4500 feet (1370 m) in the red-rock deserts of the Colorado Plateau Western portion of the extent to over 14,000 feet (4265 + m) in the Alpine peaks in the SE of the extent (the Nucla and Telluride areas, respectively, *see figure 1*) . The mean annual amount of precipitation varies in these areas from a low of 7 inches (19 cm) to 51 inches (128 cm). In all areas of analysis precipitation arrives at two time periods. Large amounts of precipitation arrive in winter in the form of snow, which historically has melted slowly in spring making major contributions to soil moisture. In the afternoons of mid-summer days precipitation arrives in the form Monsoons.

The UFO has oscillated between drought status and times of excessive moisture (*see figure 2*). The historic values of this would define the natural climatic oscillations which the AIM panel design was intended to capture. For example the 1980's represent a relatively moist period in the history of the UFO, followed by a short drought around 1990. However, as can be inferred from the trend line in (*see Fig. 1 panel 4*), the average SPEI has decreased over time from the beginning of the calculated times to today. However, at no time prior to the initiation of AIM sampling has the drought reached as low as an SPI value as -1.3, which is most evident over a 24 month analysis time period (Fig. 1 panel 3).

Value	**Interpretation**
> -0.5	Near Normal
-0.5 to -0.7	Abnormally Dry
-0.8 to -1.2	Moderate Drought
-1.3 to -1.5	Severe Drought
-1.6 to -1.9	Extreme Drought
< -2.0	Exceptional Drought

Table 2: SPI Values Interpretation; B. Fuchs

The correlation between the values for SPEI which were calculated locally were compared to those calculated by the Consejo Superior de Investigaciones Cientficas group. SPEI calculated for the same three durations were downloaded from the spei website (https://spei.csic.es) on (Nov., 30th from 10-11 A.M. RMT). These data were cropped and resampled into the same resolution as the local data set using bilinear interpolation.

Using the SPEI values from the 12 month calculations evidence of a negative association between year and SPEI over the period of analysis (Jan 1981 - Dec 2021, p = 0.451) was found after accounting for the temporal correlation present in the errors of the linear regression model relating the two variables (model call to R: *nlme::gls, SPEI ~ time, m12, nlme::corAR1(form = ~1 | month), na.omit*). Each 1-unit increase in the values of time (in this case, month) was associated with a decrease in the expected value of SPEI of -4e-05 units (95% CI: -0.00013 to 6×10^{-5}). In other words, this model indicates that, on average, the area of analysis has become drier. This model has several limitations, which we will not address here, and advocate that it simply be used in a semi-quantitative fashion to indicate that a negative trend in SPEI values is evident.

While a year, 2019, where moisture exceeded evaporation occurred during the sampling for both 6 and 12 month windows, due to its 24 month period of moisture deficit we not consider it a year which may serve as an example of non-drought status in the time period.

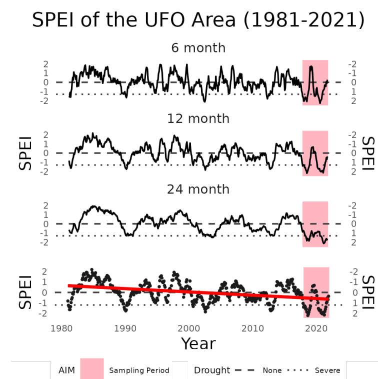

Figure 2: Mean Drought Status of the UFO area over time. Area is pink denotes the period of time which the AIM survey was conducted. Under each measurement window it generally has a negative value of SPEI.

The drought conditions were variable across not only time, as exemplified in Figure 2, but also to some extent in space, as shown in Figure 3. As would be expected the area experiences a SPEI value around 0 over longer periods of time, this being a result of a long term water balance. While many plants have physical adaptions for existence in areas with low amounts of precipitation, this balance is still required in order for the perpetuity of healthy vegetation communities, and even this vegetation may only survive under drought conditions for limited periods of time.

Drought is a multiscalar phenomena, and what it effects varies as a function of time. In a hypothetical example, shorter periods of drought, such as those of the six months leading up to a point in time, may only adversely impact the amount of reproductive output of a plant rather than it's growth. Whereas longer periods of drought, such as the last two years leading up to a point in time, may affect not only the reproductive output of a plant, but even it's own survival.

In the area of analysis, all duration of drought appear to be slightly more severe in lower elevation areas. However, all areas of the analysis were severely affected by the drought and the process can be considered nearly constant across the analyzed extent.

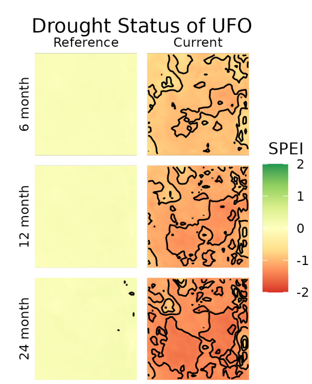

Figure 3: Maps comparing the historic mean SPEI values (left), and the current values (right)

A Land Health Assessment for a parcel of BLM land which was conducted in 2022, in the northwest portion of the UFO field office, states (Holsinger et al. (2022)):

> "*... All indicators are affected by drought conditions and continued drought is likely to have significant impacts on soil, hydrologic, and biotic integrity... The climate pattern of the last 21 years has likely inhibited the recovery of plant communities from the disturbances associated with insects, disease, and wild & domestic grazing (Pellant et al. (2020))...*"
>
> — BLM 2022

Summary

The area around, and which constitutes, the UFO has been in drought since the initiation of the AIM sample design. For these reasons we do not analyze each panel, which is intended to represent a year, separately. We also expect that the drought will have considerable effects on the following core indicator variables in particular:

- Percent cover of Annual Forbs/Grasses (decreased due to lessened plant growth)

- Percent cover of Bareground (increased due to lessened plant growth)
- Vegetative height of herbaceous cover (decreased due to lessened plant growth)
- Herbaceous litter (decreased due to lessened plant growth)
- Decedent (dead) plant cover (increased due to mortality of limbs)

In closing, in defining a SPEI value which captures the effects of drought in our study area, we feel that SPEI values < -1.3, for a duration of 12 months via the calculation capture a severe drought event. However, we feel that values of < -0.9 for 6 months periods are likewise a severe, rather than moderate, drought event, which may escalate to serious wildfire hazards.

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Introduction to the Analytical Section

The following seven sections (8-14) contain all of the analyses relating to whether the Uncompahgre Field Office is meeting benchmark objectives across it's domain. This portion of the report is of the most value to readers. All sections prior to this one, largely exist as documentation of steps and variables which we needed to utilize in the following sections.

Each of these sections attempt to address only a single type of benchmark (Sections 8-11), or a closely related component (Section 12-14). These sections all follow a similar format, somewhat reminiscent of the contents of papers in an academic journal. The first section (which is always unlabeled) resembles an introduction, and aims to jog the readers memory of critical concepts relating to the indicator. The second section 'Methods' is an abbreviated format of the approaches we took to analyze the data to address the benchmarks. This often has some contents which would be present in the 'Results' section of a journal, when we feel these help inform and illuminate the methodological processes. The final section generally comprises what would be considered the 'Results', 'Discussion' and 'Conclusions' in one area. We feel many readers will be able to skip to the final section 'Results & Discussion', without reviewing the first two.

As mentioned all of these documents were generated in Rstudio, and all sections are tracked using the git software, and available on Github. For detailed inspection of the methods please refer to Github, we tried to avoid the most technical aspects not only for brevity but also clarity.

https://github.com/sagesteppe/UFO_AIM_Panel1_Final_Report

Bare ground

Bare ground, constitutes the top layer of all soil which may be exposed to a falling raindrop, and in addition to vegetation does not include rock and gravel (Edwards et al. (2019)). A decrease in the total amount of vegetation, the litter which falls away plants, and soil crusts, contribute to an increase in bare ground (Edwards et al. (2019)). Increases in bare ground increase the susceptibility of soil to erosional forces from both wind and water. Erosion is a process which adversely impacts both natural and human modified areas (Section 9, Nouwakpo et al. (2016)). While invasive species tend to drastically alter the biotic context of ecological sites (Section 10), and different plant functional groups (Section 11) have differing effects on decreasing the potential of soil to erosion, the alteration of functional groups and shifting of an ecological site to cover of noxious and invasive species is not a zero sum game for soil retention. In other words non-native species can make contributions to protecting soil from erosion. Accordingly, here we determine whether an appropriate amount of vegetation, and biocrusts, remain on sites to prevent an increase in the risk of sites to erodibility.

Methods

TerrAdat summary data were downloaded from the ArcMap SDE (Spatial Database Engine) service layer on February 9$^{\text{th}}$ 2023, and imported into R. The references for bare ground cover come exclusively from the 'Reference Sheet' portion of Ecological Site Descriptions as these values were noted to differ from vegetation estimates in a few of the functional cover estimates. These values were available for 48 in the broader area around the Field Office.

The Ecological Site Groups do not contain values for bare ground cover, however they do contain the mean value of 'Total Foliar Cover' for each AIM plot within the concept. In order to generate a realistic estimate of bare ground for these plots, we assume that a relationship exists between the total foliar cover at a plot, and the proportion of bare ground. While the true relationship is expressed in the equation below:

$$100 - (\text{foliar cover} + \text{litter} + \text{rock} + \text{biocrusts})) = \text{bare ground}$$

We sought to simplify this relationship to:

$$100 - \text{foliar cover} = \text{bare ground}$$

To accomplish this a simple linear model was created, using the 188 AIM plots which had both verified Ecological Sites, and contained descriptions with cover reference. The linear model used the Total Foliar Cover as a predictor of Bare Ground. The values predicted from this model, for estimates of Total Foliar Cover from 0-100 were then rounded up to the nearest 5, e.g. an estimate of 1% bare ground would become 5%, to reflect variation in reference states.

Based on these data, there was very strong evidence that foliar cover affects bare ground, and serves as a moderately informative predictor of it (adj. $r^2 = 0.238$, $p < 000.1$), and that we can safely simplify this relationship.

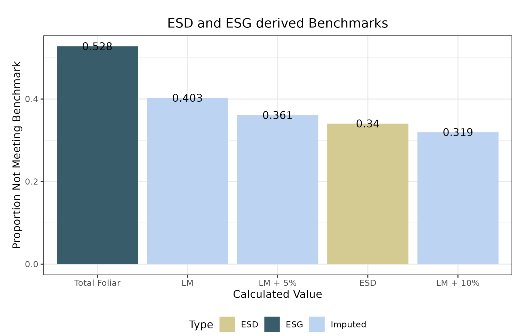

Figure 1: Relationship between Bare Soil and Foliar Cover

To determine whether our the simple use of Total Foliar Cover, or imputed values were capable of accurately estimate bare ground conditions, we compared four possible benchmark values inferred from the ESG covers using the 72 plots without ESD's, to the true ESD values calculated with the other 188 (Figure 1; with the latter group in beige). The first was the original value of Total Foliar Cover, using this metric 52.8% plots were classified as failing to meet benchmarks, a serious discrepancy, 18.7%, between those plots which had ESD benchmarks to compare themselves to which had only 34% plots failing to achieve benchmarks. This indicated that using the plots total foliar cover value would be an inappropriate proxy, and that imputed values may be more promising. The imputed values derived from linear models were a serious improvement over the last comparisons, results decreasing the discrepancy between the plots with known benchmarks which were failing and the linear model predictions of plots failing, 40.3%, to a difference of 6.2%.

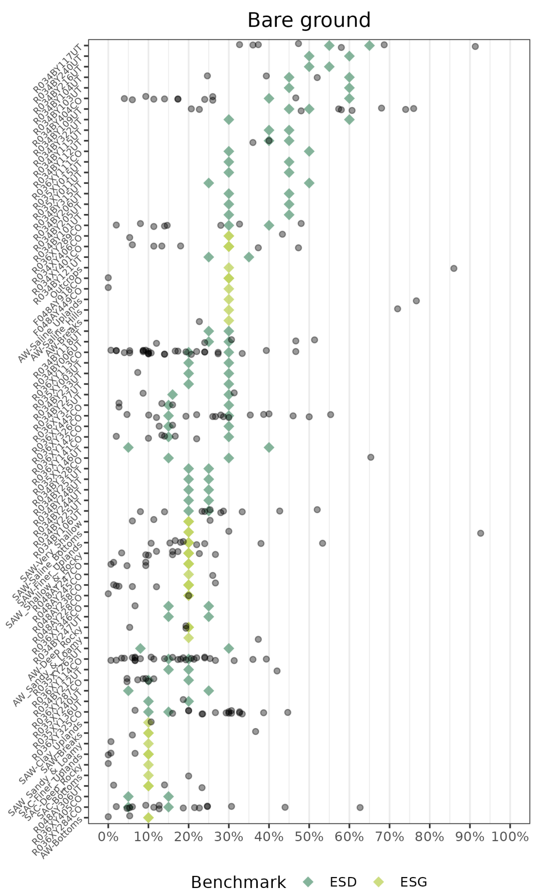

Figure 2: Benchmarks and Observed Values, all points to the right of the highest diamond are failing to achieve standards

To improve the estimates from the linear model we turned the predictions from the linear model into results more reflective of natural variation. We accomplished this by creating intervals within the range via rounding, more akin to bimodal concepts, to report estimates. Both values used for a range, 5% and 10%, produced results with a similar accuracy to the AIM plots with known benchmarks. Theoretically, there should be little biological reason for the groups of plots with and without ESD's in the same Major Land Resource Area's to differ extensively, and the known values should provide an accurate estimate of the unknown values. When using the buffer of 5% a difference of 2.1% was observed, and with 10% a difference of 2.1% was also observed. While several statistical frameworks would dictate the acceptance of the 5% buffer, we opted to use the 10% buffer. Because a sizable number of the plots in the groups under evaluation were in MLRA 48, which is generally well vegetated, and we expect it to have more land within bare ground reference conditions relative to the remainder of the field office. The 10% buffer has the same accuracy as the 5% buffer, but results in more plots achieving benchmark conditions.

For the final calculations of the proportion of land which was meeting or exceeding benchmarks. The 10% buffer was selected as the final benchmark standard for only the 72 plots which did not have Ecological Site Descriptions. The calculation of the proportion of lands meeting or failing to achieve benchmarks was carried out using 'cat_analysis' from the 'spsurvey' package with an 80% confidence level (Dumelle et al. (2022)).

Results & Discussion

Visual evidence suggests certain Ecological Sites were found to be outside of reference more often than others and may warrant concern (Figure 2). 'Semidesert Loam', 'Semidesert Sandy Loam' (R036XY325CO, R036XY326CO). Both of these sites are generally, coarse soiled, lower elevation Wyoming Sage Brush sites and tend to have wanting forb and graminoid components of their functional diversity. The site 'Loamy Foothills' (R036XY284), is similar to the above in all regards, except in having generally finer textured soils. Accordingly the loss of these functional components may be associated with this elevational trend On the other hand, the ecological sites 'Semidesert Stony Loam', 'Clayey Foothills', 'Semidesert Juniper Loam', and 'Mountain Pinyon', (respectively: R034BY404CO, R036XY289CO, R036XY113CO, R036XY114CO) tend to have less bare ground than would be expected under reference conditions. For the first three this may relate to soil loss and concomitant increases in the exposure of rock fragments, and invasive species, or indicate they are overgrown with woody species. For the last two this may indicate very dense cover of trees, perhaps due to lack of thinning of early succession sites. Further investigation of the relationships between ecological sites and the total cover benchmarks are warranted at the end of the second AIM sample design.

No administrative area had an estimated percent of land meeting the management objectives for bare ground (Figure 3). However, three areas the Dominguez-Escalente National Conservation Area, Gunnison Gorge National Conservation Area and BLM land in the UFO, had estimates of uncertainty around the estimate of land meeting benchmarks which included the management objectives. Dominguez-Escalente has a respectable sample size (n = 35), relative to the other special status areas, indicating that it's estimate (75.4% (LCL 66.4, UCL 84.4)) is unlikely to retract much with considerable sampling, and it is near meeting the bareground benchmarks. The confidence intervals for the Gunnison Gorge lands overlapped slightly with the objectives, 67.9% (LCL 55.5, UCL 80.3), however this may be in part due to relatively few plots (n = 16) which were sampled in the area, and with a narrowing confidence band these results may not be consistent. The field office at large, nearly overlaps (66.3% (LCL 62.7, UCL 69.9)) with the percent of land meeting objectives, and has a much larger sample size (n = 196), indicating these results are more stable. However, we should consider that we took a the slightly more lenient estimate on imputing our bare ground benchmark, i.e. the 10% estimate, and that the confidence interval for the 5% interval may not overlap. Whereas the estimate for the ACEC-WSA is broad, largely in part due to a small sample (n = 13), and the confidence intervals would be expected to contract significantly (43.1% (LCL 26, UCL 60.3)), towards the relatively low estimate.

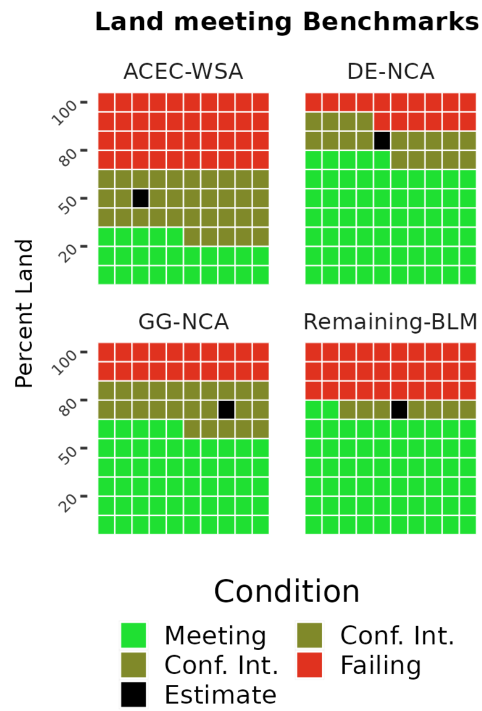

Figure 3: Percent land meeting reference benchmark conditions

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Soil Erosion Potential

AIM uses the stability of *(macro)aggregates* as an indicator of the potential for soil to erode (soil erodibility; Section VII) (J. E. Herrick et al. (2021), J. Herrick et al. (2001)). Soil aggregates are groups of soil particles which clump together to form individual strongly connected (micro)aggregates, which may then continually clump together with each other, and organic matter, to form larger less weakly connected (macro)aggregates (Totsche et al. (2018)). High numbers of macroaggregates at the soil surface have been shown to correlate strongly to reduce the effects of rain, and wind, on the loss of soil from areas (Barthes & Roose (2002), J. Herrick et al. (2001)). While the relationships between soil erosion, plant cover and functional type (Cerda (1998), Greene et al. (1994), Torre-Robles et al. (2023)), landform position (Swanson et al. (1988), Torre-Robles et al. (2023)), slope shape (e.g. concave, convex) (Canton et al. (2009), Torre-Robles et al. (2023)), and the cover of biocrusts are oftentimes complex (Leys & Eldridge (1998)), the quantitative observation that soils with low macroaggregate stability have greater rates of erodibility are always evident (J. Herrick et al. (2018)).

For this report, soil aggregates refer to relatively large coherent portions of soils (> 2mm diameter) (J. E. Herrick et al. (2021)). Soil microaggregates are continually being created by processes, such as the initial attraction of negatively charged clay and positively charged salts on silt particles, followed by cementation (Totsche et al. (2018)). Cementation is often achieved through organic matter and/or calcium carbonates and oxides, which then leads to biological processes. These involve the creation of numerous long organic (Carbon containing) molecules (generally polysaccharides), by organisms such as bacteria especially filamentous cyanobacteria, fungal hyphae, and plant roots, which act as 'glue' between these particles and will create macroaggregates (Six et al. (2004), Totsche et al. (2018), Moonilall (n.d.)). Soil macroaggregates in non-agricultural lands are continually quickly formed, and subsequently broken back into modified microaggregates by: certain wildland fire conditions (Urbanek (2013)), rapid drying and wetting, freeze-thaw cycles, some chemical interactions with water, and compaction (Le Bissonnais (1996)). When many more soil aggregates are being broken apart than are created areas become more susceptible to erosion from water, or wind (Leys & Eldridge (1998), Six et al. (2004)).

Soil erosion decreases water infiltration into the soil and less water is available to plants, reduces soil nutrients available to plants and microorganisms, removes soil carbon which foster soil microorganisms, and decrease root depth and space for plants; all leading to decreases in plant diversity, abundance and production (reviewed in Pimentel & Kounang (1998)). Accordingly, soil erosion may lead to the inability of an Ecological Site to support certain plant species essential to the maintenance of the site Bestelmeyer et al. (2015). In most instances this will tend to lead to a different or altered *state* and *phase*, generally with lower ecosystem diversity, to occur on the site (Bestelmeyer et al. (2015)). However, in severe instances soil erosion will lead to conversion of a site into a state from which land management agencies are unlikely to be capable of restoring basic ecosystem services absent extensive and costly inputs (Bestelmeyer et al. (2015)). Realistically in nearly all semi-arid lands utilized as rangelands, this equates to desertification.

Multiple other indicators collected by AIM interact to affect the implications of the Soil Stability findings, alterations in any of these metrics lead to increases in the potential for soil to erode. Increased lengths of bareground (interspaces) between individual perennial plants - whether alive or dead (hereafter: canopy gap), and increasing patchiness of perennial plants relative to each other (e.g. are plants only densely clumped in parts of a site?), interact with decreased heights of vegetation to protect soil from wind erosion (Bradley & Mulhearn (1983), Leenders et al. (2011), Mayaud & Webb (2017), Zobell et al. (2020), Webb et al. (2021)). The cover of biocrusts, especially lichens, mosses, and dark cyanobacteria, work to reduce both water and wind erosion (Leys & Eldridge (1998), Stovall et al. (2022)). As the shape and slope of the terrain which a plot is located on increases from concave through linear to convex soil is more prone to erode until settling downslope at the toe of a concavity (Canton et al. (2009), Torre-Robles et al. (2023)). Finally, increasing amounts of soil surface roughness achieved via rock and litter are able to reduce wind erosion (Raupach et al. (1993)). Surface soils with higher amounts of fine sands, are particularly more prone to erosion than soils with less sand or more coarse sands. **(NRCS Soil surveys San Miguel, Paonia & Ridgway)**. Work to combine all of these variables into predictors into a single model which is capable of predicting soil erodibility in Western North American Semi-Arid lands is still under way (Webb et al. (2021)). Current concerns regarding soil stability are to be compounded with climate change (Munson et al. (2011)), soil crusts, perhaps with the exception of 'light' cyanobacteria, are slow to regenerate. More episodic, and intense rainfalls are expected to increase soil erosion (Chen et al. (2018)).

Soil stability will be the only core-indicator in this report that is treated as a *categorical* variable. The way that crews collect soil stability means that it is an ordinal categorical variable, i.e. an object with discrete categories which are ordered. Soil stability measures are on a scale of 1-6, where '1' indicates little to no stability and '6' indicates very high stability. While it is tempting to treat these values as *continuous*, it is generally inappropriate to do so. For example:

> "Stability class 4: 10–25% of soil remains on sieve after five dipping cycles;
> Stability class 5: 25–75% of soil remains on sieve after five dipping cycles"
>
> — AIM 2021, V. 1 p. 51

As can readily be seen, from these two classes which are the most similar, breaks are of wildly different sizes (15% and 50%), and clearly violate this assumption. Another condition where ordinal values can be treated as continuous is when they represent a great range. Finally, we have few replicates per site. Soil stability is measured at only 18 locations per plot, roughly only half of the recommended observations for using parametric statistics. While non-parametric statistics are often applied to numeric data, they perform very well with small samples sizes. Accordingly, we end up in relatively the same place statistically by treating these values as ordinal categorical variables.

Methods

The first step in assessing whether the field office was achieving benchmark conditions regarding soil stability was to impute the measurements for these values at Ecological Sites which were lacking Descriptions, or which had incomplete descriptions. These values were imputed by *feature engineering*, however since they were ordinal categorical variables the *median* of the values were used.

A relatively high amount of Ecological Sites Descriptions (33 of 52, 63.5%) contained soil stability reference benchmarks under two conditions as well as a site 'average': 1) Interspaces (the distance between plant canopies), 2) Under Canopy (areas beneath plant cover), 3) Site 'average' (hereafter: median).

A hand-full of sites (n = 7) contained values for both under canopies and interspaces, but lacked a site wide median; a dozen sites had only one value (n = 9 'site', and n = 3 'interspaces'). To calculate these estimates for ESD's which were missing them, the median for each category was gathered using each observation (Figure 1). The missing values were then imputed in each ESD which was missing a value, as well as for each ES which was missing all values.

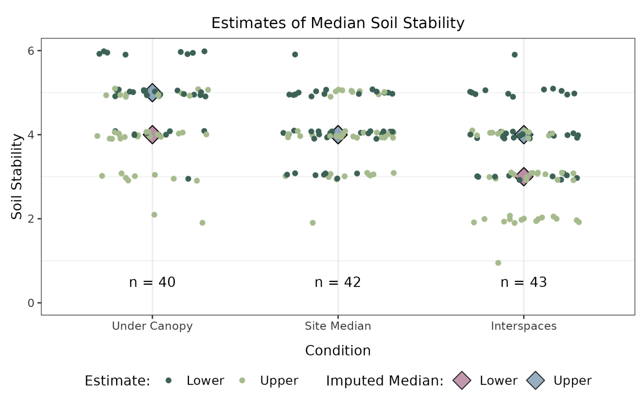

Figure 1: Specified benchmarks at all ESDs which included all three metrics and derived imputed value

The original AIM soil stability data were pulled from TerrAdat and imported to R. The median of Soil stability for each plot, under both of the conditions, and a site average, were calculated (Figure 1). These values then underwent categorical analysis using *cat_analysis*, in the 'spsurvey' package, with confidence interval of 0.8, (Dumelle et al. (2022)), and the 'local' (default) variance estimator.

```
## During execution of the program, a warning message was generated.  The warning
## message is stored in a data frame named 'warn_df'.  Enter the following command
## to view the warning message: warnprnt()
```


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## During execution of the program, a warning message was generated.  The warning
## message is stored in a data frame named 'warn_df'.  Enter the following command
## to view the warning message: warnprnt()
```


Results & Discussion

Of the 258 sites analyzed 95 were meeting the benchmark for median soil stability across the entire plot, a similar number of plots were meeting the lower benchmarks for canopy (85) and inter-spaces (98). Accordingly, hereafter for the sake of simplicity we only discuss the overall plot conditions, rather than all three benchmarks because it alone appears to capture this variation. Across the general UFO 39.9% (LCL 36.2, UCL 43.7) of land was meeting this benchmark condition, the estimates of uncertainty around the total area of the field office which were meeting these objectives were relatively narrow, and showed that roughly only half of the land in the field office was within reference condition for soil stability, and much of the area is more susceptible to erosion. Of the remaining three management areas none were meeting the objectives for the percent of land in acceptable condition. The Gunnison Gorge National Conservation Area had the highest estimate of stability, 54% (LCL 39, UCL 69.1), the breadth of the confidence intervals is due partially to the relatively small sample size (n plots = 18), and the true value likely falling much closer to the estimate, which more closely agrees with the estimates for BLM land overall. Relative to the other ares Dominguez Escalente National Conservation Area had much lower estimates of lands achieving soil stability benchmarks, 14.7% (LCL 7.6, UCL 21.8), with a sample size of 33 plots, hence these low values are not an artifact of sampling. While the Areas of Critical Environmental Concern-Wilderness Study Areas had a relatively small sample size (n plots = 12), their values, 20% (LCL 7.4, UCL 32.6), are congruent with the values for the remainder of BLM Land (Figure 2).

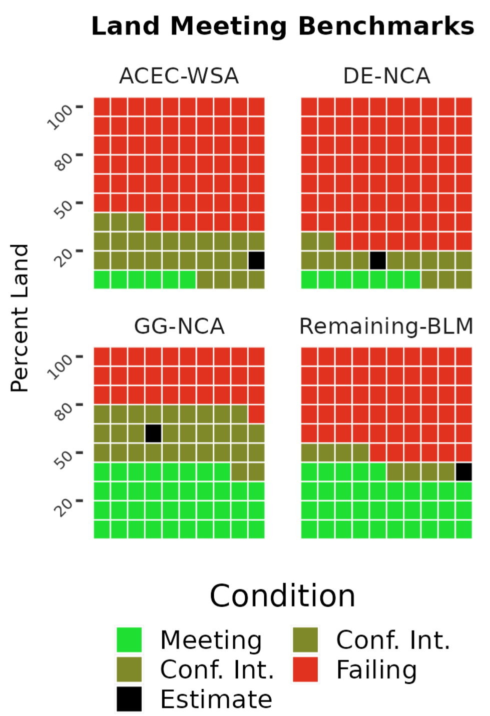

Figure 2: Percent land meeting reference benchmark conditions

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Invasive Plants

One of the first acts in the life of a plant, as a seed, is to move. Generally seeds travel only short distances, but less common long distance movements are known to occur naturally, and long distance movement occur regularly via humans. The long distance movement of seeds by humans, in recent history *i.e.* the last couple thousand years, may result in the seed germinating and the population (if any) which results from it surviving for only a few generations (*waif*) (Nesom (2000)). Seeds from introduced plant species that produce a population which is able to persist in the new ecosystem, and disperse across there new landscape, and which are incorporated into the existing vegetation with little alteration are considered *naturalized* (Nesom (2000), Pysek & Richardson (2010)). A subset of these introduced species which are able to persist may displace considerable amounts of plant species already in the landscape, and in so doing alter the composition of species at ecological sites (an *invasive* species) (Davies (2011), Evans et al. (2001), Pysek et al. (2012), Land Management (n.d.), *and reviewed in* Ehrenfeld (2010), Pysek & Richardson (2010)). Plants native to an area, but which are capable of displacing considerable numbers of other native species, and adversely affecting ecosystem function, are called *noxious* rather than *invasive.* That a few species can alter the properties of landscapes is at the core of the Ecological Sites state and transition concepts. While all invasive plants act noxious, not all noxious plants are invasive, as can be seen from a number of native species which act as aggressive weeds in disturbed settings and which prevent natural succession of vegetation.

Noxious and invasive species adversely affect nearly all ecosystem services offered by natural areas, increase fire frequency and intensity, and have enormous economic impacts (*reviwed in* Ehrenfeld (2010), D'Antonio & Vitousek (1992), Duncan et al. (2004), Fantle-Lepczyk et al. (2022), Crystal-Ornelas et al. (2021)). Invasive species at landscape scales have been shown to decrease plant species richness, taxonomic, functional and structural diversity leading to declines in habitat heterogeneity and adversely affect wildlife (Keeley & Brennan (2012), Ehrenfeld (2010), Klinger & Brooks (2017)). They pose serious threats to the well being of both wildlife, livestock, and humans, via interactions with historic land management alterations, to the fire cycle (D'Antonio & Vitousek (1992), Keeley & Brennan (2012)). The economic impacts of invasive species include enormous amounts of funds being channeled into their treatment to reduce fuel loads, treatments to curtail their spread into new areas, and losses in economic activity e.g. by displacement of grasses more suitable as forage for livestock.

In the Western cold deserts (The Colorado Plateau, Great Basin, and Columbia Plateau) invasive annual grasses pose the greatest challenge towards maintaining ecosystems and their multiple uses (Chambers et al. (2009)). A concern in the Uncompahgre field office is the increasing adaption of cheatgrass (*Bromus tectorum*), which is already present throughout the field office, towards higher elevation sites (J. T. Smith et al. (2022)). Based on the limited evidence currently available the encroachment of invasive species is of more adverse affect than is attributable to climate change or drought, although synergistic interactions between invasive species and drought still occur (Clarke et al. (2005), Lopez et al. (2022)).

Methods

Creation and maintenance of registries of invasive species often falls on the Department of Agriculture of the Federal and State governments (Quinn et al. (2013)). Given the focus of these agencies, these lists are generally focused on arable lands used for crop production, with less focus placed on natural settings (Quinn et al. (2013)).

To develop a list of invasive plant species for the study area, a semi-quantitative expert based assessment of introduced species 'IRanks', were extracted from the C-Values data prepared by the Colorado Natural Heritage Program (Section 13) (Morse et al. (2004), P. Smith et al. (2020)). This was combined with an AIM species attribute table extract from the vicinity of the study area, the latter data set contained a handful of synonyms which included codes not present in the former.

Once this list was developed and underwent review, we reprocessed our data to determine both the presence and absence of these species and recalculated percent cover. To determine what percent of the field office were meeting benchmark reference condition we developed three tiers of benchmarks, which were the same for all Ecological Sites. The first tier associated strongly with the Reference State Conditions, is if any individuals of any invasive species were detected on a plot during species richness, the site has failed. The second tier is if any individuals of any invasive species were detected on by Line-Point Intercept, the site has failed. The third benchmark, which we feel is the most ecologically informative, is if more than 5% of all plant cover is of invasives species, the site has failed. These plots then underwent categorical analysis using the function '*cat_analysis*' from the package spsurvey, with confidence intervals of 80% (Dumelle et al. (2022)).

```
## During execution of the program, a warning message was generated.  The warning
## message is stored in a data frame named 'warn_df'.  Enter the following command
## to view the warning message: warnprnt()

## During execution of the program, a warning message was generated.  The warning
## message is stored in a data frame named 'warn_df'.  Enter the following command
## to view the warning message: warnprnt()

## During execution of the program, 4 warning messages were generated.  The warning
## messages are stored in a data frame named 'warn_df'.  Enter the following
## command to view the warning messages: warnprnt()
## To view a subset of the warning messages (say, messages number 1, 3, and 5),
## enter the following command: warnprnt(m=c(1,3,5))
```

To detect whether Ecological Sites differed in their resistance to invasive weed invasion a Kruskall-Wallis test was used (Kruskal & Wallis (1952)). Kruskal-Wallis was used due to non-normal data (heavily right skewed), and a small number of replicates for each Ecological Site. The Kruskall-Wallis test offered evidence of a difference (p = 0.014), to detect which Ecological Sites differed from others a two-sided Dunn's test with Holms correction for multiple testing was used as a *post-hoc* test (Dinno (2017), Ogle et al. (2022), Holm (1979), Dunn (1964)). Once p-values were adjusted for multiple testing, as we had no *a-priori* hypothesis for which sites would be more resistant, there was no strong statistical evidence that any pairs were significantly different than any others. However, this is most certainly in part due to the very few number of plots per sites (Mdn = 5), and the large number of sites (total = 25), rather than an actual lack of resistance in Ecological Sites (Figure 1).

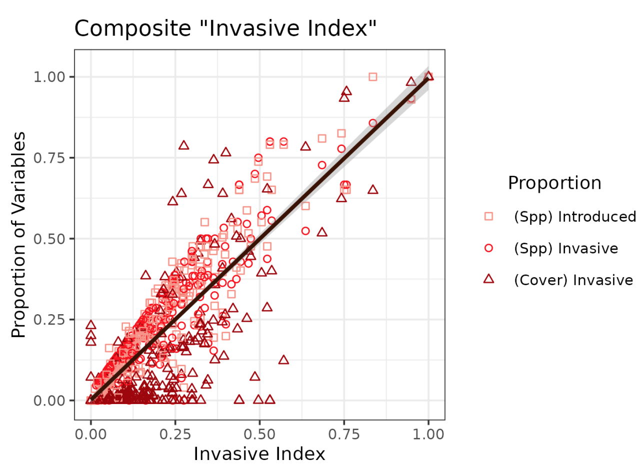

Figure 1: Invasiveness Index

To produce the estimates of the amount of land in each stratum in the Dominguez-Escalente NCA which was meeting objectives, each point was extracted to the derived vegetation cover product. The weight of each point was proportional to 'one' divided by the total number of points. This was to allow for a simplified format for weighing these plots.

Results & Discussion

The AIM sample design detected 74 naturalized species, and of these 63 invasive species were detected at the 275 plots. 227 plots had invasive species present, and 181 plots had more than one invasive species. As expected, the species which occurred the most often were *Bromus tectorum* (cheatgrass) n = 139, *Halogeton glomeratus* (saltlover) n = 64, *Chenopodium album* (lambs quarter) n = 61, *Lactuca serriola* (pirckly lettuce) n = 55, *Tragopogon dubius* (yellow salsify) n = 53 across all plots. Most of the invasive species were present across the entirety of the field office, except for roughly a dozen species which were isolated to localities between Delta and Paonia (Figure 3). Several of these species generally occur in wetter, higher elevation, habitats than most terrestrial UFO land and their spread to other UFO land is minimal; however a number of the populations are adjacent to USFS land (Figure 3).

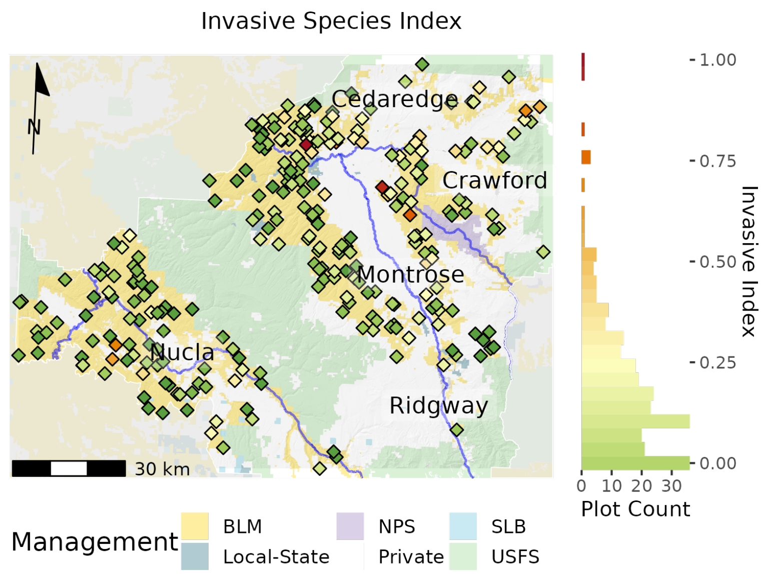

Figure 2: Invasive Index across the Field Office

A handful of the invasive species detected (Whitetops (*Cardaria chalepensis*, *Cardaria draba*, alternatively *Lepidum draba*), Canada thistle (*Cirsium arvense*), field bindweed (*Convolvulus arvensis*), Timothy (*Phleum pratense*)), especially in the Northern Portion of the field office are generally associated with slightly more mesic conditions than occur at most BLM land, especially adjacent to irrigated pastures. Unless these populations are entrenched near streams, their spread is likely curtailed by the general aridity of BLM land, but their successful extirpation, in the face of continual re-colonization from adjacent pasture lands, or if they have invaded wetlands, is unlikely. Baby's Breath (*Gypsophila elegans*) may have been introduced for roadside plantings, and is worth eradication efforts (Pringle (1993+)). Houndstongue *Cynoglossum officinale* is generally limited to higher elevation forested areas, and in the absence of forest fire it's spread may be slow on the portions of BLM land which it inhabits. A couple of these (Prairie Pepperweed (*Lepidium densiflorum*), prostrate knotweed (*Polygonum aviculare*)) are generally associated with heavily compacted soils, and tend to not spread aggressively outside of these areas.

A component of reference condition for all Ecological Sites is that invasive species are not present. By this metric the plots which would be meeting there benchmarks are quite low (227), however we feel another consideration is whether an invasive species was detected on the Line-Point Intercepts (Section VIII). As reference condition benchmarks still have considerable association with pre-Columbian times ecology, and much has changed in the interim, we believe each of these plots deserves another more liberal consideration. Of the plots which had invasive species, they were detected via line-point intercept at 167 of them. A more apt final comparison of the relative abundance of invasive species at plots is performed, plots with over 5% relative cover of invasive species are considered to not be meeting benchmarks at these.

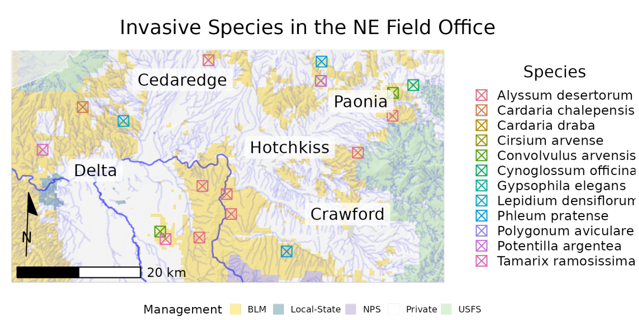

Figure 3: Uncommon Invasives in the Northern Field Office

There were 44 species which were detected on LPI lines at 167 plots. The proportion of all vegetation cover which was invasive at plots varied from 0% to 100% (Mdn = 2.69, mean = 12.53). In general, the few plots with exceptionally high cover of invasive species were typically adjacent to roads, and private lands, in low elevation areas near Delta (Figure 2).

A caveat with detecting invasive species is that, in the study area, many to most of them are annuals. Given the exceptional drought conditions (Section 6) under which field work was conducted, it is quite likely that the estimated cover of them by plot are notably lower than can be expected during more normal conditions (Mack & Pyke (1984), Bowers (1987)). While the presence of invasive species at plots is unlikely to change, their abundance are likely underestimated in more normal conditions. The proportion of species at a plot which are introduced, or invasive, and the proportion of invasive species cover at a plot were highly correlated; the mean of these three indicators were taken at a each plot and combined into a single metric, 'Invasibility Index' (Figure 1). We suspect that this index is more indicative of the status of a plot in a year with more normal precipitation, as can be seen in Figure 2, the data from the sampling period generally has a large cluster of invasive cover in the lower left of the plot from (0 - 0.25 on the index axis, beneath the line), which is more representative of the potential for invasive cover to expand rapidly under normal conditions.

A wide range of invasive species were present throughout the field office in a variety of habitats, and at varying abundances (Figure 2). When either the presence of an invasive species on plot (Species Richness data; Figure 4 Panel 1), or the presence of an invasive species on 'line' (Line-Point Intercept; Figure 4 Panel 2), were used as a benchmarks all four areas of analysis within the field office failed to have adequate areas meeting benchmarks. These results were not unexpected, and the more modest benchmark of plots where invasive species compose less than 5% of the cover of all vascular plant cover (Figure 4 Panel 3), had one area - the Dominguez-Escalente National Conservation Area, which was meeting benchmarks (Estimate = 78.2%, LCB 69.2%, UCB 87.3%). The Area's of Critical Environmental Concern (ACEC's) - Wilderness Study Area's (WSA) (estimate = 36.9%, LCB 19.5%, UCB 54.3%), and the Gunnison Gorge National Conservation Area (estimate = 23.3%, LCB 10.6%, UCB 36%) failed to meet management objectives for being in reference condition, and despite relatively small sample sizes had estimates of areas meeting objectives lower than the remaining BLM Lands (estimate = 60.3%, LCB 56.7%, UCB 63.9%) with minimal amounts of overlap between their confidence intervals. This indicates that these two management areas may be worth focusing resources on invasive species treatments, and that Dominguez-Escalente warrants attention before the cover of invasive species, already present throughout the area (Figure 4 Panel 1) increase.

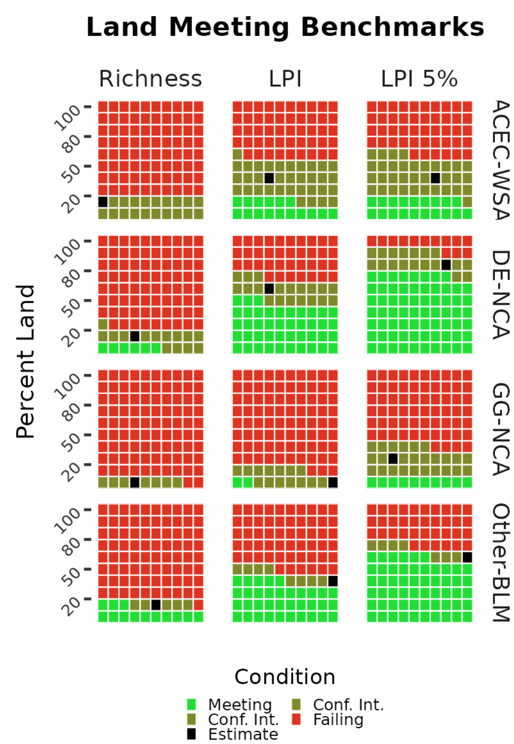

Figure 4: Percent of Each Area meeting Benchmarks

Of the special management goals per stratum at the Dominguez Escalente National Conservation Area most objectives are being meet. Salt desert is meeting the estimate of 80% of land having less than 10% relative cover of invasives (Estimate = 80%, LCB 69.9%, UCB 90.1%). Sagebrush while including the confidence intervals had a very small sample size and was not estimated to be meeting it's goals (Estimate = 66.7%, LCB 31.3%, UCB 100%). If we assume that a goal for Pinon-Juniper is less than 80% of land having relative cover of invasive species below 10% than it is meeting objectives (Estimate = 86.7%, LCB 76.6%, UCB 96.7%).

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Plant Functional Diversity - Cover

While each plant functions differently in an ecosystem, the degrees of dissimilarity which exist between all species are unequal, allowing them to form natural groups. This observation has given rise to the notion of *Plant Functional Types*, shared attributes which unite similar species, and which bind how they affect ecosystems. Plant functional types are often the easiest form of vegetation data to measure, hence great amounts of work have been conducted on how they affect ecosystem function.

In Western Colorado, four plant functional types are often used to evaluate rangeland conditions. These types are: Trees, Shrubs, Grasses, and Forbs (or herbs), and each has been linked to affecting rangelands in multiple ways. Theoretically their distributions and abundances are driven by variations in soil moisture throughout horizons (O. Sala et al. (1997)). Accordingly, in nearly all instances a mix of each of these groups, less trees, is best to maintain ecosystem services on BLM Land. In the UFO which features extensive Pinon-Juniper Woodlands, trees when present, are included in this mix on ecological sites where they represent the climax vegetation community.

Semi-arid lands which are utilized as rangelands across the world are experiencing several common issues relating to shifts in the composition of their plant functional types (Archer & Predick (2014), Eldridge et al. (2016), Maestre et al. (2016), Diaz et al. (2007), Dalgleish et al. (2010)). Namely, decreases in grasses are occurring at the same time as increases in woody species, and a decrease in the cover and number of species of perennial forbs while annual forbs increase (Diaz et al. (2007), West & Yorks (2006)). In certain areas, the increases - or encroachment of - woody species may be split into encroachment of trees, and the transition to a shrub state in ecological sites which do not support trees.

The current increases in shrub cover relative to the cover of the herbaceous components of vegetation are problematic for a variety of reasons. The increase in trees within mixed grass-shrubland sites may decrease water available to grasses, forbs, and shrubs which then favors non-native annual grasses (McIver et al. (2022)), as domestic livestock and wildlife depend on palatable grasses, forbs, and shrubs this decreases the ability of our lands to support either. Increases in shrubs at the expense of perennial grasses and forbs may increase the severity of site level drought (Wilson et al. (2018)), further shrubs and trees may foster higher severity fires. Increases in shrubs decrease soil stability, allowing increased erosion, increasing 'dust on snow', and poor air quality (Munson et al. (2011)). Decreases in perennial grass may reduce competition with non-native annuals from overtaking sites Sheley & James (2010), Corbin & D'Antonio (2004), and a diversity of grass species may be the most effective prevention (Belnap & Sherrod (2008)). A decrease in forbs adversely affect wildlife both directly and indirectly, by decreasing the quality of habitats for species such as the Gunnison Sage-Grouse (Pennington et al. (2016)).

Methods

Numerous inconsistencies exist between what the USDA Plants Database classifies a shrub, sub-shrub, or forb relative to how ecological sites classify them into functional groups. These inconsistencies are permeated into the Colorado state species list utilized by AIM to calculate the plant functional group cover summaries. To appropriately compare conditions on the ground to ecological sites or ecological site groups we had to reassign certain plant species to the appropriate functional group.

Very good agreement existed between our species reclassification summaries and TerrAdat plant functional group summaries (Figure 1), however outliers existed within the shrub and forb functional groups. Species outliers from Figure 1 were manually investigated for their functional classification in Ecological Sites and were reassigned. A total of 1760 site functional group pairs were utilized for this process. By the end of the process 1485 of these pairs had identical values when rounded to 1 decimal point (a tenth of a percent), of the remaining 275 records, 120 had less than a one percent difference in cover, and 166 were less than a 1.5%

difference. By the end of the process the Pearson correlation coefficient for trees (n = 269, $r = 0.99999$) and grass (n = 400, $r = 0.99971$) indicated the values were essentially identical, and most likely diverged merely according to rounding during internal computations. We further added the groups of Sedges (Cyperaceae), and Rushes (Juncaceae), into our calculations of grass cover. This is important as they are included in Interpreting Indicators of Rangeland Health (IIRH), and hence the ESD cover estimates, and are likely to be included in the ESG estimates. They are likely to make a notable difference in higher elevations sites, Mixed-Mountain Shrub & Aspen, where Elk Sedge (*Carex geyeri*) may be abundant.

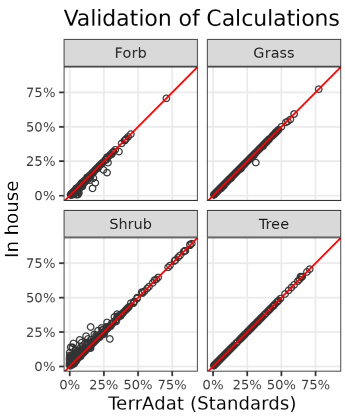

Figure 1: Comparision of calculations

The difference between the remaining functional groups was greater due to whether a species was considered in one group or the other. For Forbs and Shrubs there was a consistent discrepancy regarding a single abundant species, *Gutierrezia sarothrae* (Broom Snakeweed). Based on a review of ESD's, most authors considered this a sub-dominant shrub, a group which is combined into their estimates for Shrub Cover. Accordingly, we utilized this assessment of *Gutierrezia sarothrae* and included it as a shrub in our recalculations. As a result, we have a slightly larger discrepancy between our estimates and the TerrAdat estimate of shrub cover (n = 436, $r = 0.99189$). Since many ecological sites also consider succulents, specifically the genus *Opuntia*, a shrub in their shrub cover estimates we also included it in our calculations of Shrub cover. This also leads to a discrepancy associated with estimates of forb cover (n = 380, $r = 0.99542$). However, a greater number of values diverge between the TerrAdat summary of cover and our reclassified summaries, of the 275 records which diverge by $> 0.1\%$ cover, 188 of them are associated with Shrubs, and the correlation here is much lower at $r = 0.97911$. We were unable to match up the functional groups from these two sources beyond this point.

Results & Discussion

Forb Cover

Benchmark forb cover is generally low across all Ecological Sites in the study area, with a maximum expected cover at any site of 15% and the median of all mean values across all sites 5.5%. In general these estimates focus on perennial species, as after they germinate nearly all species will produce above ground biomass every year of their lives. Compared to annual forbs, which may not be apparent in drier years - and remain as seeds, and are known to have considerable variation in there year to year abundances as a response to precipitation.

Across all BLM land which is not managed as an NCA, WSA, or ACEC, the estimate of the total amount of land which is achieving benchmarks is 34.4% (LCL 31%, UCL 37.8%), similar to the estimates for all ACEC-WSA areas 20.4%, and the Dominguez-Escalente NCA 24.5%. Results for the Gunnison Gorge are much lower, none of the 19 sampled sites were meeting benchmarks, indicating none of the land is. That roughly only 1/3 of all BLM administered land is meeting these objectives was unexpected.

While nearly all plants produce less above-ground biomass during drought, the largest reduction in forb cover may be due to natural and climate induced mortality of individuals and the lack of recruitment of perennial forbs from the soil seed bank which replenishes local populations (Eziz et al. (2017), Casper (1996) Munson et al. (2022)). In other words, many individual plants are dying, and new individuals are not being recruited from seed during the prolonged drought. The establishment of both long and short lived forbs seems hampered during drought periods, and it may take several years after the cessation of a drought for the cover of perennial forbs to return to pre-drought conditions (Anderson & Inouye (2001)).

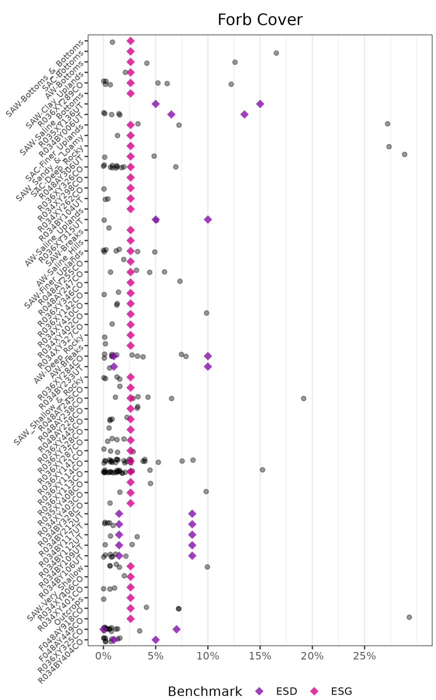

Figure 2: Benchmarks and Observed Values

It is probable that short lived perennial forbs, which generally only live a total of 2-4 years past their germination (Dalgleish et al. (2010)), have widely decreased in areas during the drought (Torang et al. (2010), Anderson & Inouye (2001)). Many long-lived perennial forbs seem to generally persist for one to two decades (but up to four are noted), and once established (i.e. they reach reproductive maturity) are able to survive disturbances, such as drought (Treshow & Harper (1974), Lauenroth & Adler (2008), Morris et al. (2008)). However, given the duration of the current drought, and the merely episodic periods of normal moisture (Section 6) it is possible many of the long lived perennials have suffered non-drought induced mortality. Because of these conditions forbs have not recruited individuals from the seed bank. Recovery of the above ground cover of both forms of perennial forbs may require periods of from 2-5 years, or more, in mesic habitats (Anderson & Inouye (2001)) and longer in xeric habitats (Figure 2).

Areas in the field office which may be the most affected by forb declines may be those ecological sites with inherently lower water holding capacity; such as those with skeletal soils, high clay content, and shallow depths to bedrock, e.g. Salt Desert and considerable portions of Pinyon-Juniper Woodland. Soil depth can be a highly influential factor on survival of perennial forbs during times of drought (Davison et al. (2010), Nicole et al. (2011)). Accordingly, ecological sites supporting Wyoming Big-Sage, e.g. Gunnison Sage-Grouse habitat, are less affected due to greater soil depth and higher water holding capacity. When a lack of forbs is observed at these sites, is likely to be stemming from issues other than drought.

Aspects regarding the nature of perennial forbs are discussed further in section 12, but given the timing of sampling relative to the drought (Section 6), these results are not surprising. A more worrisome metric would relate to the species composition of forbs which are present at plots, rather than collective cover of them during periods of drought. This is because forbs have been shown to recover after droughts, granted they are initially present in the soil seed bank, partially because they have seeds which tend to have strong longevity, and partially because most species can produce some seed even under adverse conditions (LaForgia et al. (2018), Loydi & Collins (2021)). If these forbs are even present patchily throughout an area, than it appears secondary dispersal (e.g. movement of seeds by an animal) of their seeds will allow a number of them to reach suitable micro-sites for possible reestablishment of individuals and then a population under climatically favorable conditions (Olano et al. (2012)).

Woody Plant Encroachment

The cover of woody plants, both shrubs and trees, is observed to be at the upper end or beyond the reference benchmark values for cover at nearly all Ecological Sites (Figure 3 & 4). Greater cover of woody plants relative to reference condition is a common occurrence in nearly all arid and semi-arid rangelands globally (Bestelmeyer et al. (2018), Archer et al. (2017)). While some disagreement over the exact mechanistic causes of increases in woody plant cover exist in the literature, commonly attributed causal factors include: 1) An altered fire-cycle, 2) increasing atmospheric CO_2, 3) improper grazing by livestock (Bestelmeyer et al. (2018)). These initial drivers may lead to feedback loops enforced by changes to soil fertility which cause the shrub and or tree-encroached status of these sites to perpetuate (Bestelmeyer et al. (2018)).

Drought and insect induced mortality in Sagebrush and Pinyon-Juniper woodland ecological sites have been locally apparent in the field office, and across the Southwest, over the last 20+ years (Gaylord et al. (2013), Floyd et al. (2009)). Where mortality has occurred it threatens to create conditions which allow for high severity wildfire or diminished ecological function (Baker & Shinneman (2004)). While the species of shrubs and trees which grow in semi-arid lands are considered less responsive to droughts than forbs and grasses, due to depths which many of their roots are able to reach to draw soil moisture, drought predisposes them to insect induced mortality and die back (Gaylord et al. (2013), Winkler et al. (2019)). Shrubs and trees are considerably longer lived than either forbs and grasses. Hence recruitment of these species are limited during dry periods, the effects of background mortality on plant cover is likely to have only marginal effects at the time scales over which the current drought is occurring, and current reductions are likely to be the direct results of drought (Shinneman & Baker (2009)).

Areas with dense stands of similarly aged trees are likely to experience high mortality due to a lack of self thinning processes and the competition between densely colonizing trees for limited water (Baker & Shinneman (2004)). These dense stands are often times the result of historic vegetation treatments, or other severe disturbances which led to a very large cohort of shrub and trees species germinating and attempting to develop simultaneously. Given the high density of these individuals and the long time which it takes for them to decompose, if a source of ignition occurs these areas are likely to allow fires to spread rapidly and burn 'hot'. Conditions which lead to wildfire's which are difficult to manage.

We suspect that ecological sites which are being encroached upon by Pinon-Junipers, will have more mortality than those sites, which have them as the climax vegetation. It has been observed that their encroachment more commonly occurs at the lower elevation sagebrush sites, than mixed-mountain sites. Since the sage-brush habitat is generally closer to human habitations, and important for sage-grouse, these areas represent those which funding and management efforts can be focused towards. Further, as few of the sage-brush species resprout, fires can have longer lasting impacts there than in several other areas.

Shrub Cover

Across all BLM land which is not managed as an NCA, WSA, or ACEC, the estimate of the total amount of land which is achieving benchmarks is 55.7% (LCL 52%, UCL 59.5%), similar to the estimate for the Gunnison Gorge NCA 63.3%. Both the ACEC-WSA areas 51%, and the Dominguez-Escalente NCA 70.8%, had higher estimates of land meeting benchmarks, and the confidence intervals slightly overlapped the goal of having 80% of these lands being within the realm of natural variability, for this particular indicator. Accordingly overall, between one half and two thirds of all BLM lands were meeting this benchmark.

We observed that despite the drought, and except for Salt Desert areas, nearly all plots across all Ecological Sites will have shrub covers exceeding the reference benchmarks (Figure 3). Further the extent of Ecological Sites which have elevated shrub cover are expected to be greater than for trees for multiple reasons. Chief among them are that, in the study area, shrubs have faster growth rates than trees, and a greater number of shrub species than trees species allow them to grow in more numerous habitat types. Further we expect that a great number of re-sprouting shrubs compose considerable amounts of this cover, in lieu of non-resprouting shrubs such as most of our species of sagebrush.

Reductions in shrubby plant cover, while maintaining and enhancing other functional groups, is difficult to implement at a landscape scale, and success varies considerably by ecological site (Ding & Eldridge (2022)). Accordingly, we expect that many areas of the field office which had been treated before the advent of the current Ecological Sites have already had shrub cover return. In shrub sites, like sagebrush, where non-woody shrubs species are now dominant restoration should emphasize reestablishing the balance in functional groups cover and composition while minimizing soil impacts due to erosion that has happened or may with soil disturbing approaches.

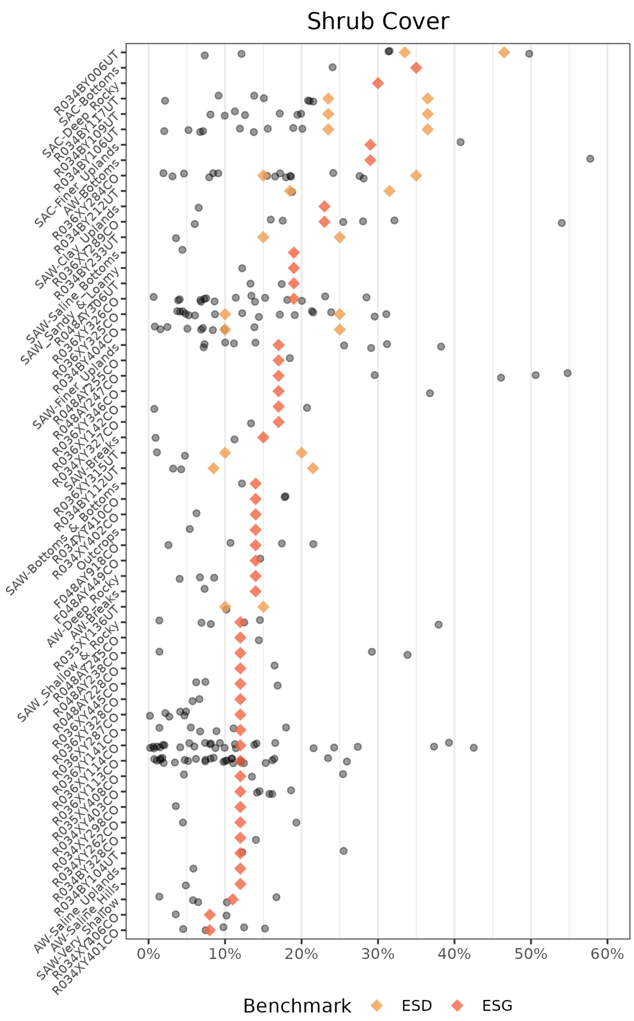

Figure 3: Benchmarks and Observed Values

This is the sole functional diversity metric which is computed in two manners. For sites which are in the Salt Desert stratum, per the spatial product created in (Section 2), we consider plots to be failing if they do not have cover of shrubs meeting the lower benchmark boundaries, and do not penalize sites which may have cover's exceeding the upper benchmark value. For sites outside of the salt desert stratum, they are considered to be out of reference only if there observed covers exceed the upper benchmarks.

Ecological sites where shrubs are greatly reduced i.e. Salt Desert ecological sites, especially areas which have historically been composed primarily of shadscale (saltbush) (*Atriplex* spp.), require restoration by re-introducing more shrubs. Cover of most palatable species of shrubs, especially winterfat (*Krascheninnikovia*) & sages (*Artemisia*), in these areas was greatly reduced by improper livestock utilization upwards of a century ago (Blaisdell & Holmgren (1984)). While passive efforts have been made to facilitate the establishment of shrubs at these sites, the very slow re-generative process, combined with invasive annuals, climate and seasonal effects on usage, have not always shown the desired results (Jonas et al. (2018)).

The recruitment of many species of shrubs in arid and semi-arid environments has been observed to be rare and episodic, requiring periods of above average precipitation for several consecutive years (Ackerman (1979), Meyer & Pendleton (2005)). Further, reductions in shrubs, and minor endeavors of assisted migration are being explored. The central idea behind this is that the velocity required by a species to track climatically optimal habitat is greater than there propagules can move, especially given fragmented habitats which reduce the connectivity of natural lands (Loarie et al. (2009), Meyer & Pendleton (2005), Corlett & Westcott (2013)).

Tree Cover

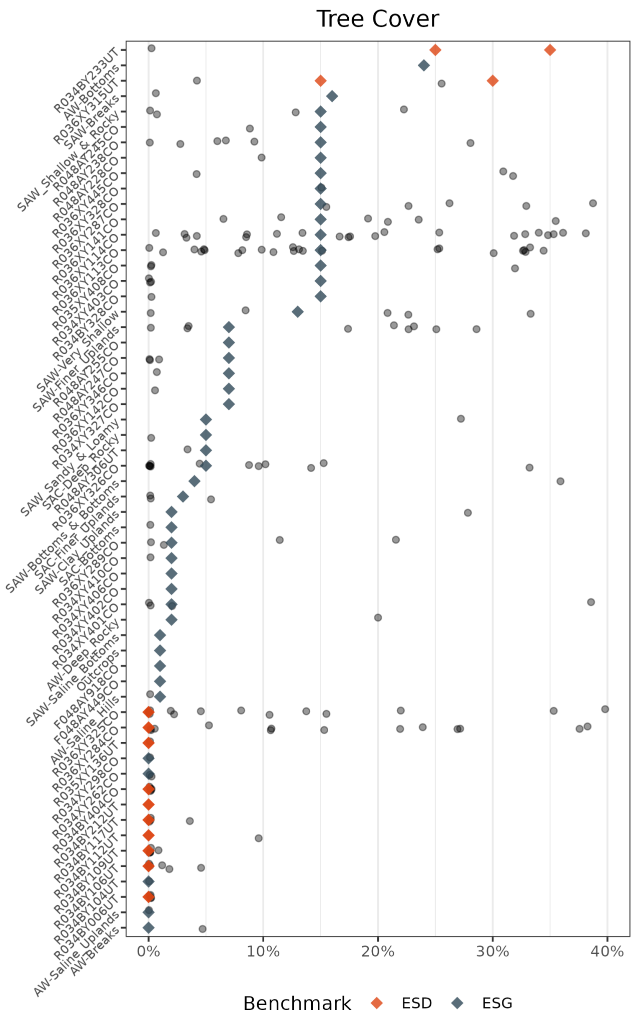

Figure 4: Benchmarks and Observed Values

Across all BLM land which is not managed as an NCA, WSA, or ACEC, the estimate of the total amount of land which is achieving benchmarks is 53.4% (LCL 50.2%, UCL 56.6%), similar to the estimate for the Dominguez-Escalente NCA 59.4%. Both the ACEC-WSA areas 83.7%, and the Dominguez-Escalente NCA 71.1%, had higher estimates of land meeting benchmarks, and for the Gunnison Gorge the confidence intervals slightly overlapped the goal of having 80% of these lands being within the realm of natural variability, for this particular indicator.

The same general observations, trends, expectations, and reasoning behind an increase in tree cover is shared as discussed in the shrub section. Mortality of portions of trees is expected less on trees than shrubs.

Many historical Pinyon-Juniper vegetation treatments, were conducted throughout the study area (Pilliod et al. (2017)). However, akin to a great proportion of other such treatments globally, where the goal was to increase big game and livestock habitat, most of these were marginally effective (Ding & Eldridge (2022)). This is likely due to the potential of those ecological sites not including a mixed grass-shrubland more desirable for livestock use.

Currently three trajectories for Pinyon-Juniper cover exists in the study area. The **first** is the regrowth of Pinyon-Juniper on sites which were historically treated, and cannot support mixed grass-shrublands, and which are being allowed to naturally re-vegetate. The **second** is that a lack of certain disturbances, e.g. fire, at certain Ecological Sites is resulting in increases in Pinyon-Juniper as natural process of *succession*, i.e. these are parts of the landscape where the climax vegetation community is PJ, but which via management may be kept in a grass-shrub condition. The **third** trajectory is that Pinon-Juniper are encroaching into ecological sites where trees are not expected due to the absence of naturally occurring processes, and are starting to promote their own expansions via feedback loops. In areas such as this multiple functionalities of the land are reduced, and are sites where vegetation treatments are highly desirable from a variety of ecosystem services perspectives (Yahdjian et al. (2015)), and which are capable of regaining ground for wildlife and livestock usage (Anadon et al. (2014), Archer et al. (2017), Morford et al. (2022)).

The higher cover of trees than expected throughout the study area identifies ecological sites which vegetation treatments which involve tree removal can be implemented. Due to the effects of woody encroachment on the production of species which are used as forage by livestock, removals of low percentages of woody species at select ecological sites might have strong effects with minimal effort. Prioritization of sites where treatments will offer the most ecosystem services, such as the most productive sites in terms of forage production, and areas with species of wildlife which are susceptible to higher predation via tree encroachment.

Perennial Grass Cover

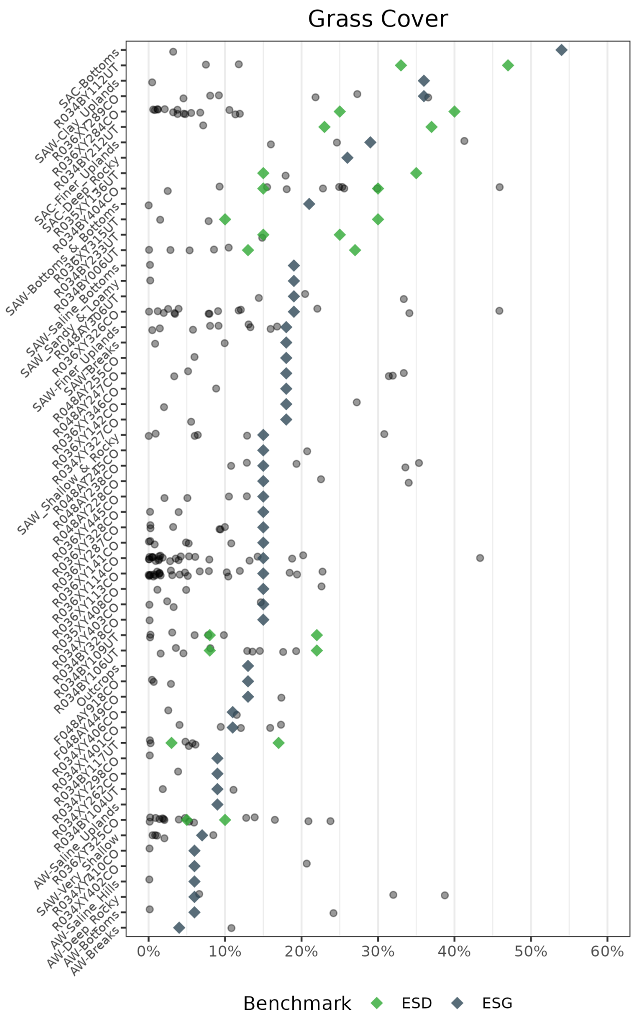

Figure 5: Benchmarks and Observed Values

Across all BLM land which is not managed as an NCA, WSA, or ACEC, the estimate of the total amount of land which is achieving benchmarks is 23.1% (LCL 19.9%, UCL 26.4%). Results for the other areas, were comparable, but quite as low, with Dominguez-Escalente NCA 35%. The ACEC-WSA areas 16.3%, and the Dominguez-Escalente NCA 28.9%. No confidence estimates for the amount of areas meeting benchmarks came close to meeting management objectives.

It is difficult to determine the extent to which grasses of the Colorado Plateau will reduce their above ground growth in response to drought. Various studies have found that grass production decreases during drought, and during periods of highly variable precipitation, however the extent of reductions are variable (Gherardi & Sala (2015), Staver et al. (2019), Munson et al. (2022)). While other studies show that the amount of biomass produced by grasses is quite resilient to drought (Byrne et al. (2017)), and that moisture limitation reductions in grass growth are largely buffered by legacy effects (in this case, a single normal year of precipitation, e.g. 2018, can offset the next few years of dryness and *vice versa*) (O. E. Sala et al. (2012), Reichmann et al. (2013)). More recent studies on the Colorado Plateau have shown reduced growth of C4 grasses, partially due to variability in Monsoons, and C3 grasses via reduced cool season precipitation (Munson et al. (2022), Hoover et al. (2021)). However, given the distinctive growth forms of grasses (i.e. generally columnar), it is unlikely that their cover would be found to be much lower via the methods employed by AIM, unless high levels of mortality occurred.

Compared to perennial forb duration perennial grasses in general appear to live for considerably longer, many from 4-7 decades (Lauenroth & Adler (2008), Treshow & Harper (1974)) but many of the grasses in the sampling areas may only persist 1-1.5 decades. Many clonal (in particular rhizomatous & stoloniferous) grass species, such as some species of *Bouteloua* appear much less likely to undergo mortality of the whole plant (De Witte & Stocklin (2010)) relative to bunch-grasses (but see Winkler et al. (2019)). Mortality due to drought is expected to reduce cover measurements of grasses more than reductions in above ground biomass. While grasses tend to have deeper roots than forbs, soil textures and depths (Chamrad & Box (1965), Griffin & Hoffmann (2012)) still mediate drought effects. A manipulative experiment which sought to determine the effect of drought on five grass species in the Colorado Plateau observed mortality of roughly 25% of all individuals under the ambient treatment (similar conditions to what the UFO experienced), largely attributed to Indian Rice Grass (*Achnatherum hymenoides*), tracked in ambient conditions over the time period 2011-2018 (Winkler et al. (2019)).

Similar to forbs, we expect little to none recruitment of new grass individuals from the soil seed bank. However suspect it is unlikely that has considerably high a proportion of the members of this functional group would have died off over this period, independent of drought induced mortality (Morris et al. (2008), Winkler et al. (2019)) as perennial forbs, and given their average rooting depths relative to forbs should be more drought tolerant (O. Sala et al. (1997)). Accordingly we expect estimates of grass cover to be at the lowest end of the benchmarks.

Metrics Combined

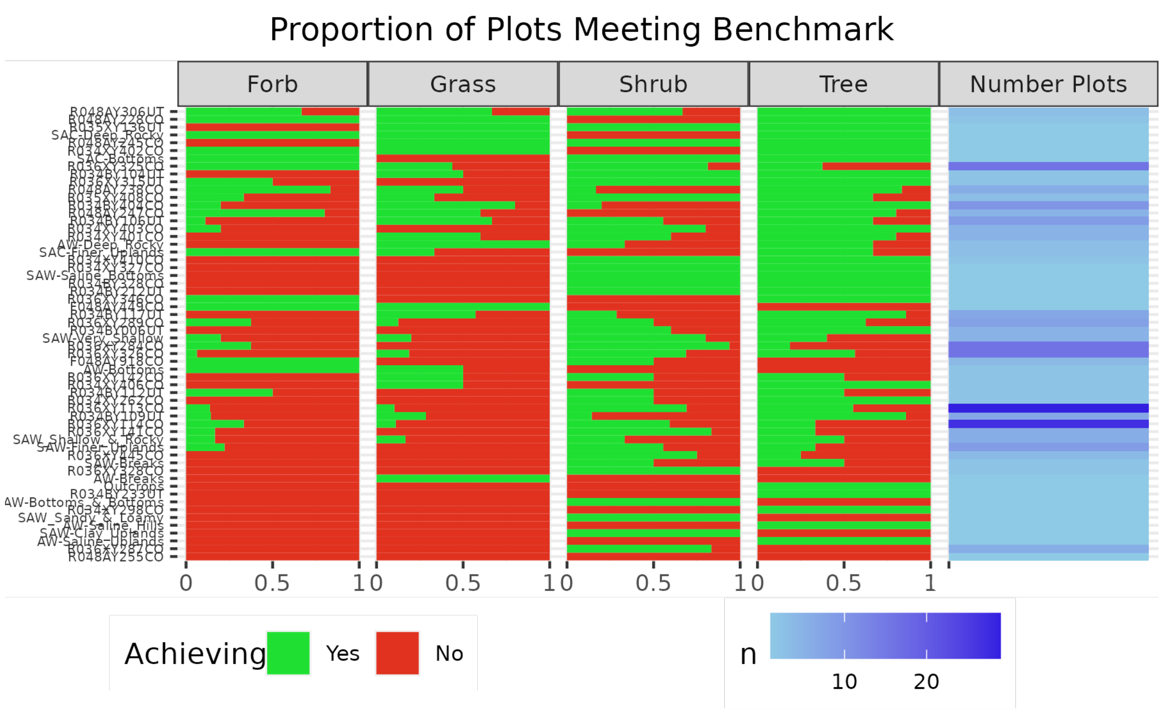

Figure 6: The Proportion of Plots in Each ESD which are meeting Reference Benchmarks

Few clear trends emerge regarding which Ecological Sites are failing to meet certain benchmarks, and the relationships between the functional diversity benchmarks which they are no t meeting. It is evident that few plots in few Ecological Sites are meeting either forb of grass cover benchmarks, and the only Ecological Sites which appear notably different here are those with only a single plot which was sampled in them. In general most ecological sites were achieving benchmark goals for shrub and tree cover, with the few exceptions being ecological sites which lacked replicates. This illustrating a notion that woody encroachment is not a large issue in the area of analysis, but rather that the loss of non-woody species within the remaining inter-spatial areas is concerning.

The feature engineered cover of a lower forb benchmarks are used across all plots in lieu of the mean values from ESG's, which we felt was too high of a value relative to the other forb benchmark values included in the Ecological Sites Descriptions. The use of the median value means that more plots are passing benchmark conditions, than were under the pure ESG schema (original results not pictured here).

Each individual Ecological Site, or Ecological Group, varies in the proportion of all plots located in them which are within reference condition. Once a greater number of replicate plots are sampled per Ecological Site (see Figure 6, panel 5 *(right most)*), and combined with digitized management records, these data may form an approach towards understanding the resilience of different Ecological Sites in the UFO to management actions. The Ecological Sites are arranged via descending order of the total proportion of vegetation types and plots which are achieving benchmarks. These results currently largely reflect the

We can combine the number of plots, and their weight acres, within each stratum which had all four major functional groups within reference to develop a sense of how well the RMP objectives are being meet (Figure 6).

Results by management areas in general do not differ greatly. From a broad perspective, roughly a quarter of lands in the field office is meeting standards for forb or grass cover, except for the Gunnison Gorge National Conservation Area which has no land meeting objectives for forb cover (Table 1). No areas are meeting the benchmarks for Tree cover either, although most management areas have roughly one half of their land achieving. This indicates the need to explore tree thinning, or removal, operations in certain areas, as funding permits and needs require, to benefit wildlife via modification of habitat or to decrease the threats of wildfires to adjacent human population base (Baker & Shinneman (2004), Shinneman & Baker (2009)). Given the historic reductions in fire cycle this is an issue which requires a great many decades before coming back into resolution, but given the current awareness of the problem, management actions are now underway which will do so. Roughly half to two-thirds of land are achieving shrub cover objectives, and two areas have confidence intervals which do (51% (LCL 30.7, UCL 71.3)), of very nearly (70.8% (LCL 62, UCL 79.6)), include the land cover targets. On the whole, the results taken together indicate that the study area is failing to meet metrics for plant functional diversity, with only areas 2 having confidence intervals which even overlap the management objectives. However, the perennial grass functional group presents the most serious concern. The median estimate of land within any area meeting objectives for cover of perennial grasses is 26%, across the entirety of the field office the estimate of lands meeting objectives for grasses is a low 23.1%, the ^{2nd} lowest proportion of land out of all benchmarks and areas.

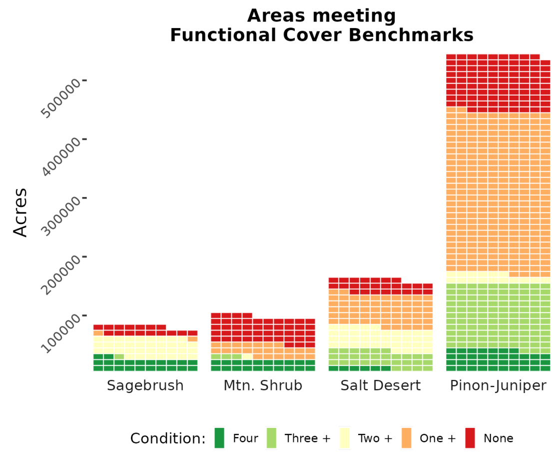

Figure 7: Total area of each stratum and the overall status of benchmarks

Table 1: Land Meeting Benchmarks by Administrative Unit

Goal	Management	Plots	Group	% Land Meeting
80%	ACEC-WSA	12	Forb	(10-) 20.4% (-30.8)
			Grass	(6.7-) 16.3% (-25.9)
			Shrub	(30.7-) 51% (-71.3)
			Tree	(71.7-) 83.7% (-95.6)
	DE-NCA	35	Forb	(14.9-) 24.5% (-34)
			Grass	(25.5-) 35% (-44.5)
			Shrub	(62-) 70.8% (-79.6)
			Tree	(50.7-) 59.4% (-68.1)
	GG-NCA	19	Forb	(0-) 0% (-0)
			Grass	(15.9-) 28.9% (-41.9)
			Shrub	(48.8-) 63.3% (-77.8)
			Tree	(55.8-) 71.1% (-86.4)
70%	SampleFrame	200	Forb	(31-) 34.4% (-37.8)
			Grass	(19.9-) 23.1% (-26.4)
			Shrub	(52-) 55.7% (-59.5)
			Tree	(50.2-) 53.4% (-56.6)

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Plant Functional Diversity - Species

In this section we discuss the three major forms of diversity. *Alpha-diversity* - or species richness, is simply the number of species in a space at a point in time. The scale at which we discuss $\alpha - diversity$ is generally from the size of the footprint of a car to a football field. This metric refers to areas which we are intimately familiar with and may traverse readily on foot. *Gamma-Diversity*, represents the richness of species in a larger area, generally a landscape. For example, we may readily discuss the $\gamma - diversity$ of the Dominguez-Escalente National Monument. In all instances the $\alpha - diversity$ of sites are nested within the gamma diversity of an area. $\gamma - diversities$ -in this case the number of species- exceed those of $\alpha - diversities$, both due to the relative uncommonness of many species, which are often not present across the entirety of the landscape, and due to the large changes in the type of species supported by differing habitat types. These turnovers across the alpha diversity of sites, the difference in species present at sites, comprise *beta-diversity*. High rates of $\beta - diversity$, or dissimilarity of sites, foster high rates of gamma-diversity (Whittaker (1972)).

Evolution, the process largely mediating the maintenance of diversity, is survival of the fittest. However, the conditions of the test which may constitute the 'fittest' are nearly endless. Many species which exist in the same location in space, have distinct characteristics which allow for species persistence; dry years favor some species, while wet years favor others, some require more sun, while others thrive with less, and there are myriad permutations and combinations of these settings. The cover of these species ebb and flow with the usual weather and disturbances within the climate zone of the site (Hoover et al. (2014)). These trends are especially important for the production of forage and browse, over the life of most large animals, they will have to feed on what they have available. No single species is the fittest at a site on a time scale which the BLM manages land for, and having multiple species within the same functional groups in space is the only stable strategy for management.

While Ecological Site Descriptions do not provide true measures of $\alpha - diversity$ many of them do contain lists of taxa which may be considered *dominant* or *subordinate* at a site (Avolio et al. (2019), Grime (1998)). In general, it seems that the Shrubs, Trees, and Grasses at an Ecological Site would be considered *dominant*, and the forbs *subordinate*; keep in mind these terms refer to immediate ecosystem cycling effects, and the services offered by forbs to insects and then larger animals remain substantial (Avolio et al. (2019)). These species which have high amounts of biomass, and ground cover, may be thought of as a core groups of species which are essential for the functioning of an ecosystem (Grime (1998)), and each of the dominant species have been theorized to have conceptual effects as large as their cover. When dominant and subordinate species are lost from an area, it also has effects on the remaining species - most of which are relatively uncommon across the landscape (Grime (1998), Whittaker (1965)). Considerable research has shown that having a diverse suite of plant species allows areas to: 1) Produce more forage in both a single year, and across different weather scenarios (Vogel et al. (2012), Hoover et al. (2014)), 2) recover from disturbances such as fire, or compaction (Tilman & Downing (1994)) 3) and resist degradation such as from the encroachment of noxious weeds (Weisser et al. (2017), Avolio et al. (2019), Allan et al. (2011), Gaitan et al. (2014), Sheley & James (2010), Isbell et al. (2011), Oakley & Knox (2013), *and reviewed in* Maestre et al. (2016), Oliver et al. (2015)).

Information on production at Ecological Sites implies some superficial, yet essential, components of plant diversity. Here we determine what proportion of species identified in ESD production tables are present at each AIM plot, how many are missing, and whether any species are uniformly missing. We also combine plots by Ecological Site to determine $\gamma - diversity$ by site, and compare the relative turn over in species composition within each Ecological Site which has replicate plots.

Methods

In order to determine which, and how many, species are noted to be dominant members of the vegetation at an Ecological Site, all ESD's for which an AIM plot was verified to were checked. All conditions (e.g. all

State/Phase combinations which were present) from the reference tables were copied from these manually into spreadsheets, and data were cleaned using R. The USDA symbols utilized in these tables, were verified to match to the look up tables which were created for our project, to ensure that the same species between the two data sources could not be 'missed' due to using different abbreviation codes. Given that only two species (*Pinus edulis*, *Junipers osteosperma*) were ever considered to be true 'Trees' under both systems, this lifeform was dropped from all analyses to focus on the more variable groups.

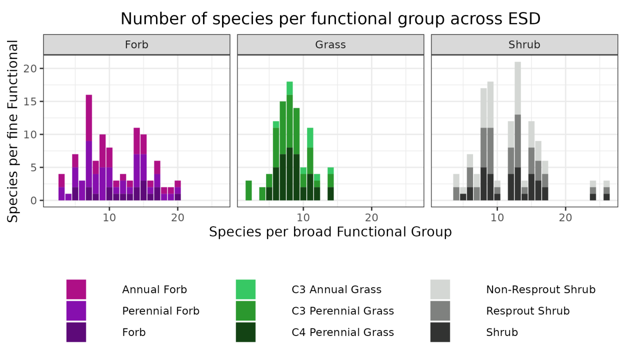

Figure 1: Number of Species Per Functional Groups

In order to be able to make comparisons across functional groups, for each species present in both the ESD's and our plot based data, we ensured that they drew from the same attribute table. This was especially important for situations where subshrubs are alternatively classified as forbs or shrubs depending on context. After performing this, we recovered both the 'coarse' functional groups used by the AIM team, which correspond largely to the botanical notion of 'lifeforms'. Subsequently, using the developed attribute tables finer functional groups, developed locally, which reflect the propensity of major groups of these lifeforms to respond to various disturbances, such as drought (e.g. C3 and C4 grasses), and fire (non-resprout and re-sprouting shrubs). In instances where more fine scale functional data were not available, such as was the case for sprouting potential of many shrub species, then species were left at this level rather than attempting to determine an appropriate group for them.

The calculation of all, previously mentioned, forms of diversity were not carried out for each individual plot. All plots had $\alpha - diversity$, the number of unique species per plot calculated. However, plots which lacked Ecological Sites, were unable to have $\gamma - diversity$, and subsequently $\beta - diversity$ calculated. Ecological Sites which lacked reference tables were able to have these calculations performed, as they are independent of the Descriptions. Calculations of $\beta - diversity$, wherein only plots belonging to the same Ecological Sites were compared, were performed using the 'vegdist' function from the package 'vegan', with standard defaults and as Sorenson-Dice dissimilarity (Oksanen et al. (2022), Sorensen (1948), Dice (1945)).

$$\text{Sorenson-Dice Index} = \frac{(2 \times \text{No. Species Site } 1 \cap \text{Site } 2)}{(\text{No. Species Site } 1 + \text{No. Species Site } 2)}$$

Wherein the number of species which are shared ("∩") between two sites are multiplied by 2, and divided by the total number of species at both sites.

The data collected by two distinct methods, Line-Point Intercept, and Species Richness, were both compared to the ESD reference data to determine which more closely reflected the benchmark values. After visual exploration of these data displayed in several plots (including Figure 2), it was determined that the Species Richness data more adequately reflected the Benchmarks values and it was used for all subsequent analyses.

In order to determine whether enough plots have been sampled to reliably estimate the number of species which are present per Ecological Site ($\gamma - diversity$), rarefaction and extrapolation curves from the iNEXT package were used with confidence intervals of 0.8, and 50 bootstrap replicates (Hsieh et al. (2022), Chao et al. (2014)).

Results & Discussion

Species richness data contained a total of 7525 records, which after removal of unidentified material left 6227 species records at 276 plots which were verified to Ecological Sites. 30 Ecological Sites had more than two AIM plots located in them, and had $\beta - diversity$ calculated for all plots 224 located in them.

While the species within any one reference table in an Ecological Site Descriptions are noted to represent the dominant species across a gradient which stratifies the Ecological Site, such as elevation, many of our plots contained an adequate number of these species per functional group. In fact, when considering the richness data, which we suspect is a more apt data set for comparing to the ESD taxa, many of our plots had the number of species expected in the reference table, for both the Forb and Shrub functional groups (Figure 2). For Shrubs it appears that the Ecological Sites which are meant to have the greatest number of shrub species do not reach these goals; and that perhaps only for these sites is the species representing the gradients of the Ecological Site utilized. Below we will make several comparisons of the number of species per plot to 3/4 of the number of species in each functional group in the production table. It appears unlikely that the production tables represent variables collected in a consistent manner; however they offer clear, and useful, insights.

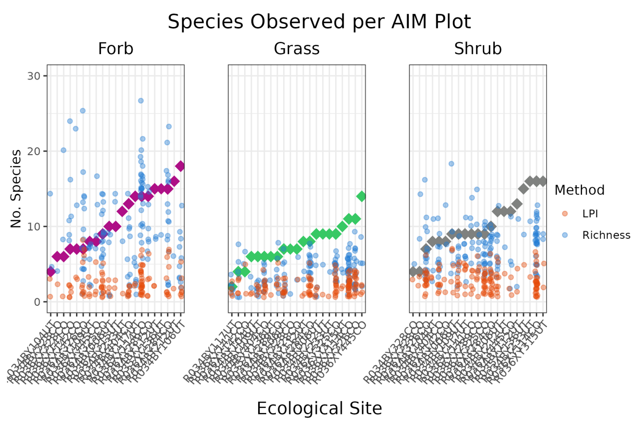

Figure 2: Number of Species Per Coarse Functional Groups

In regards to $\alpha - diversity$, the number of identified plant species varied widely by plot, from 2 - 54 (Med = 23). The 28 plots with the most species in them were in 10 Ecological Sites, this subset of plots had a minimum of 35 species per plot. Of the top 28 plot subset the Ecological Sites with the highest observed $\alpha - diversity$ were: R036XY114CO (plots = 7), R048AY238CO (plots = 3), R036XY326CO (plots = 3), R036XY113CO (plots = 2), R036XY141CO (plots = 2), R036XY284CO (plots = 2). The percent of all plots which had less than three-quarters the number of species noted, as were in the production table 43.5% , 86.5% , 62.9% for forbs, grasses, and shrubs respectively (Figure 2). The percent of all plots which had fewer than three-quarters the number of Annual and Perennial Forb species than in the production table were 38.6%, 42% (Figure 1). The percent of all plots which had fewer than three-quarters the number of Annual C3 Grasses, Perennial C3 Grasses, C4 Perennial Grasses species in the production table were 66.7%, 85.6%, 92.6%. The percent of plots with fewer than three-quarters Non-Resprouting Shrubs, Resprouting, and unclassified shrubs species as in the production table were 61.9%, 55.6%, 69.1%. As previously mentioned no benchmarks can be formed from the attribute tables, however the lack of grasses is concerning, and indicates that not only is there a lack of cover of this functional group (Section 11), but that this may be related to a lack of functional diversity brought on by a lack of species diversity.

Concerns regarding the species composition of shrubs and grasses may be related to the prevalence of *Guiterrhiza sarothae* (Section 13 Figure 1). Empirical evidence supports a theory that *Guiterrhiza sarothae* recruits when surplus resources become available, and that this resource availability is spurred by a lack of resource utilization by species which exploit similar niches, such as Grasses and Shrubs (Davis et al. (2000), Ralphs & McDaniel (2011)). The decline in these other species are likely due to disturbances (Ralphs & McDaniel (2011)). However, once established at moderate densities *Guiterrhiza sarothae* has a competitive advantage and inhibits recruitment of other grasses and shrubs (Ralphs & McDaniel (2011)).

The mean $\beta - diversity$, that is the dissimilarity between plots varied from 0.54 - 0.88, with lower values indicating greater differences between plots; and where the Ecological site with the most dissimilar plots, on average, contained just over half (0.54) of the same species as each other. The range of Sorenson-Dice values for the 25 most dissimilar plots were from 0.45 - 0.6. The 25 most dissimilar plots were in 9 Ecological Sites, with the following having the most plots in them: R036XY113CO (1), R036XY141CO (1), R036XY284CO (1), R036XY315UT (2), F048AY918CO (3), R034XY401CO (3). The median of beta-diversity of the major functional groups varied from forb (0.82), grass (0.55), shrub (0.62), indicating that turnover in forbs is much greater than for the other two groups. This is to be expected given that there are a greater number of forb species than grasses and shrubs. Given the low amounts of $\alpha - diversity$ observed for grasses, these results may indicate that the same few grass species are being observed across plots. The turnover in species composition for the minor functional groups was not analyzed as more replicates appear to be required.

The $\gamma - diversity$ of Ecological Sites varied widely in the number. However, the accuracy of the current estimates of $\gamma - diversity$ are limited by the number of replicates per Ecological Site. The estimated Sample coverage per Ecological Site ranged from 23.44 to 91.62 with a median of 65.82, 80% confidence interval; in general it appears only 3-4 plots are required to gain Sample coverage estimates about 50%. In other words, we do not yet have enough replicates per Ecological Site to make definitive statements regarding the species richness of each site, but clear trends have emerged. The Ecological Sites with the greatest number of species are R036XY114CO (spp. = 201), R036XY113CO (spp. = 187), R036XY284CO (spp. = 143), R036XY326CO (spp. = 131), R048AY238CO (spp. = 131). The predicted estimates of the number of species which are to be expected, in the UFO, at these sites with further sampling are: R036XY114CO (spp pred. = 267, LCL = 227, UCL = 306), R036XY113CO (spp pred. = 280, LCL = 229, UCL = 331), R036XY284CO (spp pred. = 249, LCL = 190, UCL = 308), R036XY326CO (spp pred. = 228, LCL = 179, UCL = 277), R048AY238CO (spp pred. = 240, LCL = 186, UCL = 295) (80% confidence intervals). It appears that the high rates of $\gamma - diversity$ observed at two Ecological sites: R036XY114CO, R036XY326CO, are likely to relate to the large geographic expanses of them, rather than high rates of $\beta - diversity$. While for the other sites, as expected $\beta - diversity$, is the primary factor fostering high rates of $\gamma - diversity$.

The Ecological Sites above, which have the greatest net number of species have forb (74-126 med. = 90), grass (15-20 med. = 19), shrub (26-50 med. = 39) species per functional group. The number of forbs at these ecological sites range from annual-forb (12-27 med. = 24), perennial-forb (50-100 med. = 74). The number of grasses at these sites range from c3-annual-grass (1-1 med. = 1), c3-perennial-grass (9-15 med. = 14), c4-perennial-grass (3-5 med. = 5). The number of shrubs vary from non-resprout-shrub (7-15 med. = 13), resprout-shrub (8-16 med. = 10), shrub (9-21 med. = 16), (Figure 1).

In summary, it appears that grass functional and taxonomic diversity are much lower than expected. It also appears that shrub diversity is wanting. Monitoring of forb diversity should be followed up several years post drought to determine if it rebounds. Returning to grasses, the number of species, in only a handful of plots, meets or exceeds the numbers of grass species noted present in reference conditions in all ESD's. Observations suggest that the grass species which appear to be missing from expanses of the field office include two species of *Sporobolus* (*S. airoides*, and *S. cryptandrus*), *Koeleria macrantha*, with noted declines in *Hesperostipa comata* and *Aristidia purpurea* in areas as well.

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Rare Species

"Rarity is one of those concepts that suffuses our culture: it defies precise definition and when used by the scientist it is often given a spurious accuracy to satisfy our need for precision."

— V.H. Heywood 1988

A connotation where rare species are synonymous with legal protections exists in popular culture (Kruckeberg & Rabinowitz (1985), Gaston (1994)). However, rarity is the normal condition under which an enormous amount, if not the majority, of species in all kingdoms of life exist, and only a subset of these species are at risk of extinction (Enquist et al. (2019), Flather & Sieg (2007)). By definition rare species are organisms which are inherently difficult to detect in nature relative to 'common' species (Rabinowitz (1981)), but see Kondratyeva et al. (2019) for elaborations on rarity. One of the most consistently supported observations in ecology, both empirically and theoretically, is that the majority of species in any one location are represented by only a few individuals (Preston (1948), Stohlgren et al. (2005), Manzitto-Tripp et al. (2022)).

Rare species encode enormous amounts of functional diversity to an area and have been shown in multiple cases to imbue an ability for areas to respond to disturbance (Isbell et al. (2011), Leitao et al. (2016), Mouillot et al. (2013), Oliver et al. (2015)). While we focus on large functional groups in Sections 11 & 12, each of these groups has enormous variation within them, and due to the sheer number of rare species, they comprise most of the variation within these groups (Kondratyeva et al. (2019), Mouillot et al. (2013)). In addition to allowing areas to respond to disturbance, they are also capable of reducing the possibility and severity of biological invasions (Lyons & Schwartz (2001), Oakley & Knox (2013)).

A popular conceptual framework to discuss rare species may be considered which contains three dimensions, 1) the geographic expanse of the species, 2) their relative restriction to particular habitats, 3) and the number of individuals per population - 'size' (Rabinowitz (1981)). Collectively the interaction between these traits can result in a matrix with eight cells along three axis (Table 1). Seven of these cells represent traits associated with rare species, six of which frequently occur, and one which is seldom -if ever- observed (Rabinowitz (1981)). The rare species which receive most of the attention, are those which are restricted to particular habitats across narrow geographical extents, *'narrow (local) endemics'* (Table 1 & 2) (Kruckeberg & Rabinowitz (1985)). In general narrow endemics tend to be the species which require special designations to ensure their habitats undergo minimal alterations (Harnik et al. (2012)). However, the remaining types of rarity still call for documentation by land management agencies.

Table 1: Conceptual forms of Rarity

Seven Forms of Rarity
From Rabinowitz 1981

Geo. Range:	Large		Small	
Habitat:	Wide	Narrow	Wide	Narrow
Population Size				
Large, dominant somewhere	Locally abundant over large range in several habitats	Locally abundant over large range in a specific habitat	Locally abundant in several habitats but restricted geographically	Locally abundant in a specific habitat but restricted geographically
Small, non-dominant	Constantly sparse over large range in several habitats	Consistently sparse in a specific habitat but over a large range	Constantly sparse and geographically restricted in several habitats	Constantly sparse and geographically restricted in a specific habitat

A typology of rare species based on three characteristics: geographic range, habitat specificity, and local population size.

> *"Many species are abundant in portions of their range, but uncommon in others Brown et al. (1995), Ter Steege et al. (2016)"* — Enquist et al. 2021

Considering the conceptual model in Table 1 we see that a majority of species in the Uncompahgre Field Office which would be considered rare are likely to have 'Large' Geographic Ranges (left two columns, note the upper left most entry represents common species). That most rare species have large geographic ranges seems to the typical condition in temperate regions.

These three cells of rare species are less likely to be have special designations (ESA, SSS) because as a species they are fundamentally at lower risk of extinction due to the decreased likelihood of all populations going extinct via the same causes (Table 2) (Flather & Sieg (2007)). However, at edges of there geographic ranges these species are likely to have state protections, because the species may lose the few populations (local extinction) which exist in those administrative units. Biologically, these widespread rare taxa may have interesting properties, relating to their relative positions in the range of the species distribution.

In particular, these common rare species, especially when at the edges of their distributions, either geographically or climatically - often have populations which have notably different genetic constitutions than populations near the centers (Hampe & Petit (2005), Oldfather et al. (2020), reviewed in Pecl et al. (2017)). Populations which are expanding into new geographic ranges, largely following shifts in climates are termed *leading edges*, and those populations persisting at the edge of the extent geographic ranges are noted as *trailing edges*. Conserving trailing edge populations at the local level is important as they may contain many forms of genes which are pre-disposed to adapting to climate change (Hampe & Petit (2005)). Further these populations may end up being essential for adaption on up-slope Forest Service Lands, where our border with them faces alterations associated with severe fires and which may require immediate seed sourcing to recover a stable state (Parks et al. (2019)). Theoretically the lowlands of the UFO are capable of receiving migrants to them from a *leading edge*, however the up slope travel required to enter the basins (e.g. the Paradox Valley), slows steady-state dispersal, reducing the chances of immigrants to relatively infrequent events, such as seeds being stuck to muddy bird feet (Nathan et al. (2008), Jordano (2017)). While these events have largely shaped the global distribution of biodiversity, their infrequent occurrence generally means they occur on timescales outside of land management considerations (Nathan et al. (2008)). However, that many species which are in the field office via long-distance dispersal, but which are currently uncommon, but may greatly expand under climate change is probable.

Table 2: Local Examples of Rarity

Seven Forms of Rarity
From Rabinowitz 1981 - Species modified to Field Office

	Large		Small	
Geo. Range: Habitat: Population Size	Wide	Narrow	Wide	Narrow
Large, dominant somewhere	Common	*Camissonia eastwoodiae*	*Sclerocactus glaucus*	*Pediomelum aromaticum*
Small, non-dominant	*Draba oligosperma*	*Cypripedium calceolus* ssp. *parviflorum*		*Eriogonum pelinophilum*

A typology of rare species based on three characteristics: geographic range, habitat specificity, and local population size.

The other half of the Table 1, the two right columns with 'Small' Geographic Ranges represents species which are very well tracked by entities and are generally of conservation concern. Species with 'Small' Geographic Ranges and 'Wide' Habitat Specificity (column 3) would be expected to be encountered at numerous AIM plots. These taxa are almost always generally noted as rare by the State, and BLM, and may be considered threatened or endangered by the United States Fish and Wildlife Service (USFWS), the agency which administers the *Endangered Species Act* (Table 2); but tend to be quite abundant across the landscape within which they reside. Finally the column at right represents the species at the fundamental core of our notions of 'rarity'. These taxa are generally warranted legal protections as human modification of their habitats has the possibility to result in catastrophic declines of populations and subsequently the species. We will utilize pre-compiled tracking lists, from the Endandered Species Act, BLM Sensitive Species, and NatureServe, to address the species which aggregate in this end of the table.

In this section we seek to identify all rare species of conservation concern. We identify all singleton species, those encountered only once, as well as all records under the 25th quantile (Gaston (1994)) of observations, to identify plants which are locally rare within the Field Offices administrative areas. We use a variety of Governmental and Non-Governmental datasets to identify the rare species of conservation concern which were encountered throughout the field office.

Methods

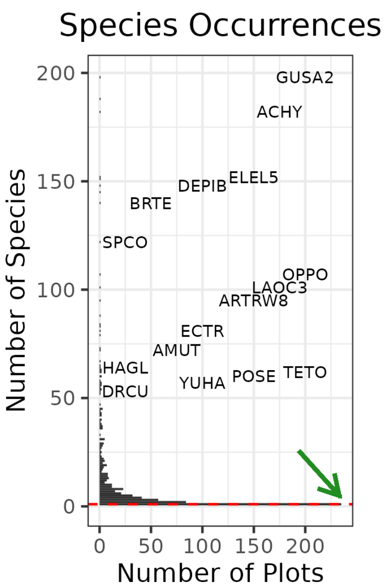

Figure 1: The Commonness of Rarity; This plot shows the number of times which a species was encountered at an AIM Plot. Both the 'singletons' and Gastons Quintile are on the red line. A randomly selected subset of more abundant species are selected to contextualize the few abundant species

Rare Species of Known Conservation Concern

To identify species which are rare, and are species of conservation concern, the Colorado Natural Heritage Program (CNHP) registry of rare plants was downloaded directly from the website in Winter of 2023. Using the list of C-values, also from the CNHP website, which contained an entry for nearly every species in the state (less *Mentzelia paradoxensis*), we determined the appropriate mapping between the plant species symbol used in the AIM database, and the official USDA codes used in all CNHP work. The CNHP registry of rare plants was then subset and used as the definitive source for each organizations tracked species.

Rare but untracked Species

The number of plots which a species was recorded at were counted. This list was filtered for species which only occurred at a single plot, our ecological version of the botanical 'singletons' collection. This list also had the 25th quantile calculated on it using the 'quantile' function from base R.

Results & Discussions

The original AIM Species Richness data set contained 7525 plot based observations from 276 plots. After removing 113 morphotypes which were not identified (1.5% of all records), dropping the 17 records which were not identified beyond genus (0.23% of all records), and removing 37 synonymous taxa at the same plot (0.5%), 7390 records were left. These records represented 680 distinct terminal taxa, i.e. final taxonomic units - species were not double-counted with their sole infraspecific (varieties or subspecies) taxa. These records are more or less in accord with the states most current Floristic treatment (Ackerfield (2015)), and the USDA Plants database.

The CNHP rankings include many species which are considered rare by agencies with different foci and intentions than the BLM. While their initial list is comprised of 540 taxa, different organizations and agencies have different criteria for interpreting and classifying susceptibility of a species to extinction and loss of populations. The most rigorous and selective conditions, the Endangered Species Act, are enforced the United States Fish and Wildlife Service (USFWS), whom maintain the only official registries and implement the evaluation procedures for 'Threatened' and 'Endangered' species. These categories represent species with high probabilities of extinction in the wild, in the near future, due to anthropogenic changes; Endangered Species represent the more severe category of the two. Colorado contains 15 species tracked by the USFWS, eight of these species are threatened with extinction, and six are immediately endangered. Of the species tracked under the Endangered Species Act one species, *Sclerocactus glaucus*, was found at five plots.

The Bureau of Land Management officially tracks plants which are of conservation concern on their lands, and which may be petitioned to be elevated to the more stringent categories implemented by the Fish and Wildlife Service, but which are mostly undergoing further assessment. As the BLM is apart of the federal government, these tracked taxa do not contain any redundancies with the Endangered Species Act list. BLM Colorado has 65 sensitive species. Of the four BLM sensitive species, *Pediomelum aromaticum*, *Astragalus rafaelensis*, *Lomatium concinnum*, *Camissonia eastwoodiae*, were found at a total of six plots.

Several Non-Governmental Organizations (NGO's) also maintain their own information on species of concern, utilizing different methods and assessments than Government Agencies. A large portion of there goals are to form networks which are global in scale, rather than restricted to state actors, allowing for more comprehensive views of biological ranges and processes. Accordingly, they have a more integrated global perspective on species, and then make assessments of susceptibility to extinction at administrative units to assist local planners. One such agency is NatureServe. NatureServe uses a tiered ranking system, from 5-1, with lower values indicating susceptibility of a taxon to extinction at a either a Global or State level. Values '3' and below are taxa that warrant conservation considerations. The number of low value species, are greater at lower administrative levels, oftentimes due to species ranges crossing multiple administrative units. The number of S3 ('S' short for 'State') or lower (S2, S1) species in Colorado is 519, and the subset of these which are globally tracked species with G3 ('G' short for 'Global') or lower ranks is 219. These two lists are not independent of the government data, for example the State list contains 15 of 15 FWS species, while the global list contains 14 of them. In addition, the state list contains 64 of the 65 BLM Sensitive Species, while the global list contains 53. We subtract these species from these two lists and end up with a total of 440 species on the state and 152 species on the global lists to avoid confusion in reporting. We further remove the species present in the state list from the global list reducing the state list to 291, which maintains the global list at 152. Of the state species 21, were found at a total of 47 plots, for a total of 50 records. Of the state species 13, were found at a total of 37 plots, for a total of 40 records.

A fascinating rare plant recorded on the AIM plots was *Mentzelia paradoxensis* J. J. Schenk & L. Hufford, a taxon described as new to science in 2010. This was found at a single plot, in the Paradox Valley, the locality from which it was collected in by prolific Intermountain West botanists Noel and Patricia Holmgren. This species hypothesis is so new, relatively little testing of it has been carried out, and to date the Colorado Natural Heritage Program has not evaluated the conservation status of it. Botanical collectors have determined it to be present in both the Paradox and the nearby Big Gypsum Valley, in both instances growing on stream terraces with elevated gypsum content.

In regards to species which are rare within the UFO, but not of conservation concern, two methods 'singletons' and 'Gastons Quantile', returned identical results. Both methods suggest that there are 235 rare species. The convergence of values implicates two methodological limitations. In regards to singletons, these records generally come from floristic inventories, wherein a well trained botanist is unconstrained by the dimensions of plots, and is able to roam a large area using their knowledge of an area and intuition. In the case of a field office wide inventory, they would be able to allocate considerable less time to certain Ecological Sites with very few species, in lieu of spending more time in areas with many species. We do feel after several more AIM sample frames this will start to deliver quite effective results. Regarding Gastons Quintile, or the lower 25% of records, this may indicate an inconsistency in goals of survey work. Many ecologists in the era in which this metric was derived were oftentimes explicitly surveying for species diversity and abundance metrics, whereas AIM serves to characterize landscape units, as delimited by geomorphology. Accordingly, the ecologist of yesteryear strove to maximize variation between plots, while the rangeland ecologist of today seeks to maximize statistical inference across landscapes. That such a significant number of species only occurred at a single plot, may in part reflect that a single Sample Frame of AIM data is not enough to start to gather information on the biotic composition of this field office. However, we see no reasons that future results for Gaston's Index would be significantly different than the species composition presented here, and believe that many of the singleton's identified here, would be singleton's again after combing these data with those in the next sample frame.

Of the 235 species identified by Gaston's and the Singleton method 18 overlap with the 26 species of conservation concern which were found on plots. It is expected that these approaches would give different results, given that rare species tend to be abundant in portions of their ranges.

As evidenced here, the field office has a variety of rare plant taxa which contribute enormously to the biological diversity of the field office.

Table 3: Species found and plots found at.

Scientific Name	Authority	Plots
Sclerocactus glaucus	FWS	SS-230, SS-234, SD-379, RI-202, RI-204
Astragalus rafaelensis	BLM	SD-362
Camissonia eastwoodiae		SD-405
Lomatium concinnum		SD-412, RI-210, OT-126
Pediomelum aromaticum		SD-401
Cleomella palmeriana	NatureServe - Global	SD-348
Draba rectifructa		MMS-093
Erigeron nematophyllus		SS-320
Eriogonum leptophyllum		SS-292
Lepidium crenatum		SS-270 , MMS-094
Mentzelia cronquistii		SD-401
Oreocarya longiflora		PJ-156, RI-208, SD-357, SS-267, SS-264, RI-199, OT-113, SD-355, SD-363, RI-198, SS-255, SS-259, SD-368, SD-359, SS-271, GR-025, SS-316, PJ-165, GR-026
Penstemon breviculus		PJ-156, PP-172, PJ-146, SD-362, SS-292, SS-284, SS-288
Penstemon teucrioides		MMS-101
Polypodium saximontanum		SS-248
Stanleya albescens		SD-367
Townsendia glabella		MC-056
Astragalus monumentalis var. cottamii	NatureServe - State	MMS-088
Calochortus flexuosus		MMS-096
Commelina dianthifolia		SS-302
Erigeron lanatus		PP-180
Eriogonum leptophyllum		SD-353
Mirabilis alipes		MMS-076, SD-342 , PJ-138
Oxytropis parryi		SS-260
Pediomelum megalanthum		RI-196

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Floristic Quality Index

Floristic Quality Assessments (FQA) utilize the vascular plant species at a site as an indicator of habitat quality. The fundamental assumption guiding the use of plants as indicators of habitat quality is that different species respond differently to the types and frequencies of disturbance. At one end of this disturbance spectrum are species which are able to persist, or may be introduced to an area, after certain types of anthropogenic disturbance - e.g. compaction of soil via heavy vehicles. On the other end of the spectrum are taxa which may only grow in areas which receive episodic natural disturbances characteristic of their ecosystem - e.g. a 100 year flood in a wetland. The subjectively estimated likelihood of a species either persisting, or being removed from a site due to disturbance is expressed as a Conservatism Value (C-Value). C-Values range from 1 to 10 for plants native to North America, and 0 for plants introduced to the continent since colonization by European Settlers, with plants which are not tolerant of disturbance being at the upper end of the spectrum.

The use of FQA are uncommon in the Bureau of Land Management, perhaps in part due to the FQA originating in the Midwest, and the assignment of C-values being a task which requires considerable amounts of resources (Spyreas (2019)). A further requirement which hampers the utilization of these metrics are that each individual plant species, often times including subspecies, is assigned a separate C-value, the number of land management professionals which are capable of distinguishing taxa at these resolutions, and have time assign C-values, are limited (Kramer & Havens (2015), Ahrends et al. (2011), Morrison (2016)). Other possible limiting factors are that the FQI indices have been traditionally associated with the portions of Natural Resources focused on designation of parcels for conservation and preservation, rather than management of these parcels.

While C-values exist for virtually all states East of the Continental Divide, Colorado is one of only two states with significant surface lands administered by the BLM which has existing C-values for every documented member of it's flora (Spyreas (2019)). In fact BLM Colorado staff, including the lead state botanist, assisted in developing the states C-values (Smith et al. (2020)). Here we utilize FQA, to supplement our formal AIM analysis, and to develop a map of the habitat quality for lands managed by the Uncompahgre Field Office.

The Floristic Quality (FQ) Assessments are comprised of two core indices Mean Coefficient of Conservatism (Mean CoC) and Floristic Quality Index (FQI) (see Appendix A for equations). While many novel permutations of these calculations exist, they seem to offer little additional insight to the main pair of indices and appear only useful for niche applications (Spyreas (2019)). As the general goal of an FQA is the assessment of parcels of land, the location of study areas across different habitat types is an accepted use, and as we used a weighted stratified sample design our points meet the assumptions implicit in the sample design (Spyreas (2019)). FQ Assessments have yet to converge on a standardized size for conducting the species inventory, and while Mean-CoC is affected by plot size FQI is relatively robust against small plot size. Regardless in similar systems plots of similar size as AIM plots have been shown to be adequate for noting enough species to calculate both main indices (Spyreas (2016)). In several applications C-values have been shown to be stable across sampling time, in part perhaps due to a propensity for many species at a site to share the same C-values (Spyreas (2016), Matthews et al. (2015), Bried et al. (2013)). C-values have also been shown capable of distinguishing habitat variability more effectively than two traditional diversity metrics (Simpsons and Shannons) (Taft et al. (2006)). Practitioners of varying degrees of skill are likely to have minimal effects on the estimates of the FQA indices due to the species encoding some degrees of redundant information (Bried et al. (2018), Spyreas (2019)). Further, the calculations of Mean C and FQI, are relatively independent of the species richness, or diversity at any one plot, and are capable of reflecting the C-values of the species present, rathern than other diversity metrics *per se*.

Utility and Limitations of FQA FQA scores are tied to the regional list of C-values, accordingly they cannot be compared across regions with independently developed C-values, in other words the FQA scores of sites in the mixed grass prairies of Kansas and Colorado are incomparable. Scores have the potential to be misleading if compared across major habitats, (e.g. comparing Sagebrush to Salt desert) but generally appear robust against this (Spyreas (2019)). The indices are relatively boundless, e.g. we could visit plots which we designate high quality and use them as benchmark for FQI values, but we cannot incorporate metrics e.g. land > 4 is 'good', until we perform these activities to determine locally relevant measurements.

> *'... tolerance of anthropogenic disturbance and exclusivity to remnant habitats are the only validated criteria for defining FQA.'*
>
> *"...FQA conveys two things about high conservative species: (1) All else being equal, they have greater conservation value, and (2) they reflect a site's history of minimal disturbance and degradation."*
>
> — Spyreas 2019

Accordingly, in our scenario we are interested in using FQA to identify sites which have histories of evident degradation.

Methods

Cleaned AIM species richness data were imported from TerrAdat and joined to the CNHP C-values using the lookup keys developed for the Functional Diversity in Section 11. All species from the plot based species richness which were not unambiguously identified to terminal taxon were removed from analysis. The Mean Coefficient of Conservatism and Floristic Quality Index were calculated via the formulation in Appendix 1. To determine whether their were bias in the Mean-CoC values between the strata, a linear model with a single set categorical predictors and a single continuous response (an 'ANOVA', or Analysis of Variance) was used, with a Kruskall-Wallis Post-hoc test with 95% Confidence Intervals. The strata used for detecting differences were those developed in Section 2.

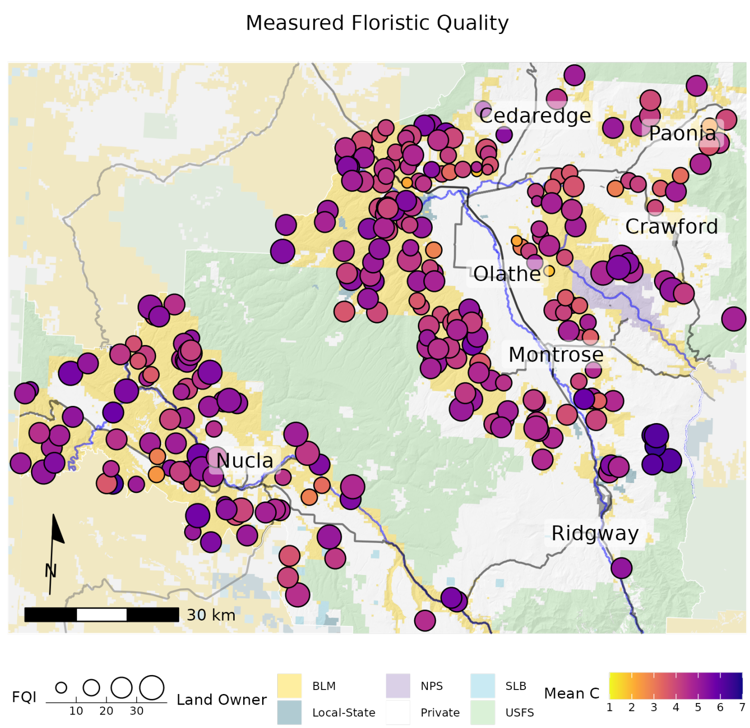

Figure 1: FQI Values observed at AIM Plots

The floristic quality assessment data are theoretically independent of ecological sites. Accordingly, we can create a statistical model using the plots which were sampled, and using this model predict the values of floristic quality across the field office onto a map. Based on Figure 1, and our field experience collecting these data, we suspected that the variables which most strongly predict Mean-CoC scores, and which we could readily acquire or create were: 1) distance to nearest road, 2) patch size of federal public lands, 3) human population density within ~ 10 km (~6 miles), 4) elevation. These analyses occurred across the extent of the mapped area in Figure 1), at a resolution of 90 meters. All of these variables were used as predictors of Mean-CoC scores in our statistical model.

Predictor variables were created using the following processes. To calculate the distance of each pixel of BLM land from the nearest road, the U.S. Census Bureau's 'roads' data set was acquired through 'tigris' and simplified using 'st_simplify'; the distance function of 'terra' was used to calculate the distance from each pixel to the nearest road (Walker (2022), Hijmans (2022), Pebesma (2018)). The U.S. Census Bureau roads data set contains nearly all of 'major' dirt and gravel roads across the field office, included many used for historic Uranium mining activities. To calculate the patch size of federal lands, i.e. all area across BLM, Forest Service, and National Park Service management areas etc., federal lands were queried from the PAD-US database and then areas dissected by roads were erased; the area of each parcel were then calculated using 'st_area' from sf, and converted from meters squared to hectares (Gergely & McKerrow (2022)). The patches were then converted into a raster using 'rasterize' from terra, and the size was saved as an attribute. To create estimates of population density within a distance of ~ 10km of each pixel of BLM Land, population density data at 30m resolution were downloaded from HDX, and summed to 90m resolution; these values were then summed using rolling windows in terra (Tiecke et al. (2017)). 90 meter resolution elevation data were acquired from EarthEnv (Tuanmu & Jetz (2014)). All data sets were cropped and re-sampled to a template to ensure optimal alignment of raster cells.

To determine whether any predictor variables would violate assumptions of independence, by being collinear, variance inflation factors were calculated using the 'function' vifstep in the package usdm with a theta cut off 10 (Naimi et al. (2014)). This function was used on a subset of 5000 random pixels regularly dispersed across the extent of the sample area, which had their value of all predictors extracted from them. No theta scores were high enough that they suggested any combination of variables should be removed from the analysis, but notable collinearity existed between 'elevation' and 'X coordinates'; this is likely due to the increase in elevation from the Colorado Plateau into the Rocky Mountains, which follows a longitudinal (West to East) trend.

In order to create a model which could capture the variation in our data and generalize them to predict Mean-CoC, a maximal model of the term *glm(mcoc_r ~ road_dist * pop_density * patch_size * elevation * xcoord * ycoord)* was created. This model was then fed into the 'dredge' function from the package MuMIn, which creates all smaller models, and evaluates them in an information theoretic framework (Bartoń (2022)). All tops models, those with ΔAIC < 2.0, were each selected for further analyses. Each model was checked for the effect of spatial autocorrelation in the residuals by identifying neighboring points using 'graph2nb' and converting them into three neighbor lists using 'nb2listw' (both functions from 'spdep'), the Moran's Index for each model was very low, despite the Moran's Index for the predictor variables being very high and showing significant clustering (Bivand (2022)). Subsequent to the interpretation of the maps produced by the stacked model outputs, considering that the model including the coordinates did not suffer from autocorrelation, and that the X coordinate was often selected as a predictor in top models, we believe that minor collinearity between the X coordinate and the strongest predictor Elevation indicated that the term was redundant.

A new full model, without the coordinates, was created in the form *glm(mcoc_r ~ road_dist * pop_density * patch_size * elevation)* and passed to the 'dredge' function in MuMIn. The results of the top models, those with ΔAIC $<= 2.13$ were checked for spatial autocorrelation, using the same methods as the models above. The general rule of thumb for only utilizing models with ΔAIC < 2, was bent in order to accommodate two more models which incorporated terms we have observed to be ecologically relevant, and are well supported in the landscape ecology primary literature (Symonds & Moussalli (2011)). After evaluation of the stacked prediction map from these top models, it was used for the final analysis, in part because removal of the x-coordinates made it more easy to interpret, and it was able to utilize the other predictors which are known to affect species compositions, both universally and strongly.

All six of the optimal models, which were in the ensemble model, were individually predicted into space and the weighted means, based on each models AIC weight, were used to generate an ensembled prediction layer (Symonds & Moussalli (2011), Bartoń (2022), Dormann et al. (2018)). Spatial predictions of the statistical model were performed using terra Hijmans (2022). The ensembled prediction was then clipped to the extent of UFO BLM administered lands.

Discussion

The Mean-CoC values varied by stratum (Analysis of Variance (ANOVA)), follow-up tests (Kruskall-Wallis) indicated that Salt Desert and Sagebrush sites did not differ from each other, nor Mixed Mountain Shrub and Pinon-Juniper from each other (Figure 1). Results indicate that all members of these two groups differed across them, e.g. Sagebrush differed from both Mixed Mountain Shrub and Pinon-Juniper and *vice versa*. The two lower elevation strata, Salt Desert and Sagebrush, are generally more accessible and have higher recreational land uses, and had the lowest Mean-CoC. While the values did vary, the extent of the variation was minor, with the range of variation from the median values 4.18 - 4.86 and a mean range of 4.21 - 4.76, given that these values occur over a range of 0-10, we considered this to be a negligible difference. Which was not indicative of major biases in the C-values ascribed to plants themselves, but rather indicative of actual land conditions, and that all main vegetation types in the UFO could be analysed together.

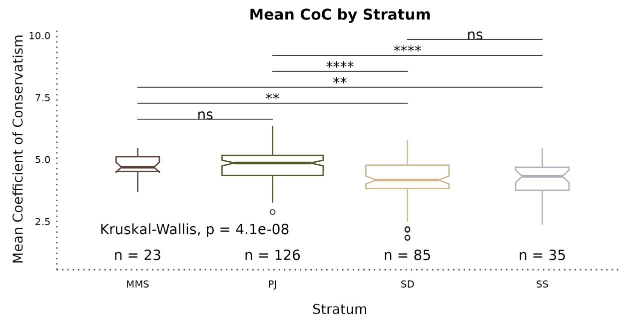

Figure 2: Comparision of median values by Stratum

The generalized linear models were used to predict the floristic quality of unsampled areas on BLM Land. Based on AIC model selection a total of 5 top models were retained, and these were then combined into a single model, of which the weights used to make a stacked ensembled prediction of the Mean-CoC. The results from these top models indicate that all predictors, elevation, human population density, Federal land patch size, and distance to nearest road, affect the Mean-CoC, but that by far the strongest predictor is elevation (Figure 4, Appendix). Elevation was such a useful predictor, that under certain statistical paradigms (e.g. forward or backward selection via null hypothesis) our models may be simplified to include it as the only predictor. However, given that C-values reflect the tolerance of a species to anthropogenic disturbances this approach would be questionable. Rather we suspect that elevation is strongly associated with disparities in land use intensity, but more importantly also the inability of lower elevation sites to recover from previous disturbances relative to higher elevation areas. The other predictors had

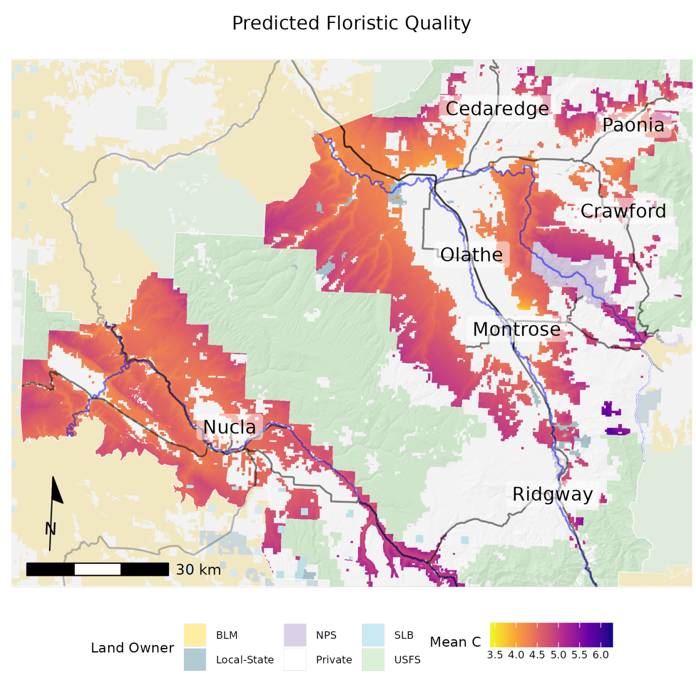

Figure 3: Consensus predictions of FQI Values across the UFO Field Office (excluding lands East of Paonia along the North Fork of the Gunnison

the expected effect on Mean-CoC as ex-
pected, Mean-CoC slowly increases as nearby human population density decreases, and FQI slowly increases as the size of the patch of natural lands and the distance from the nearest road increase.

From the predicted Mean-CoC values we can infer that habitat quality follows a consistent trend across the field office (Figure 2). In general, the lowest elevation sites display the lowest values of FQI and Mean-CoC. This is especially apparent between Montrose and Grand Junction on Highway 50. These and several other areas of the Salt Desert along Highway 50, and near Crawford are areas of concern. However, not all Salt Desert is inherently in this condition as can be seen from the values for a moderately sized parcel of BLM managed land immediately Southeast of Montrose; indicating that both anthropogenic disturbances and elevation in these areas are causal factors for these low values. Areas with low FQI values represent highlight potential restoration needs and the Mean-CoC prediction model can be a tool to help prioritize such actions. The highest elevation portions of BLM managed land are in the best habitat condition, and very good conditions range into Pinon-Juniper, including most of the disturbed areas around historic mining in the West End. By these metrics considerable portions of the sagebrush habitat occupied by Gunnison sage-grouse near Crawford are also in good condition.

Subjective interpretation suggests these results seem very similar to those generated by the much more time intensive comparison of Ecological Sites to Quantitative Benchmarks. This relationships considers serious consideration in the use of FQA as a proxy of site condition, and may warrant applications for rapid surveys prior to certain land use decisions, in time frames when AIM cannot be implemented. It also provides a second independent set of data which may help to contextualize sites which were unable to have their Ecological Sites correlated, are lacking Ecological Site Descriptions, or are difficult to interpret.

Appendix A - Indices

Mean Coefficient of Conservatism

$$\overline{C} = \frac{\sum C_i}{S}$$

Where:

$\overline{C}$ is the Mean Coefficient of Conservatism, or for short Mean C
S is the number of species included in the calculation
C_i in particular C is the Conservatism Value (C-Value), for each $_i$ of the S at the site
$\sum$ is an operator, meaning that we will calculate the sum of all C-Values, C

Floristic Quality Index

$$FQI = \overline{C} * \sqrt{S}$$

Where:

$\overline{C}$ is the Mean Coefficient of Conservatism, or for short Mean C
$\sqrt{S}$ is the square root of the number of species included in the calculation

Equations from Swink et al. (1994), and modified for simpler formulations.

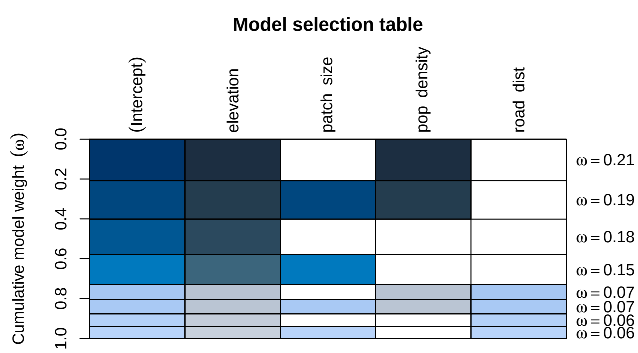

Figure 4: Contributions of each term to each model, and each model sized by contributions to ensemble prediction. Note the left-most column is not a predictor, and only the top 6 models were used.

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Glossary

Assess, Inventory, and Monitor: "The objective of the Assessment, Inventory, and Monitoring (AIM) Strategy is to provide a standardized monitoring strategy for assessing natural resource condition and trend on BLM public lands. The AIM Strategy provides quantitative data and tools to guide and justify policy actions, land uses, and adaptive management decisions."

annual/biennial/perennial: Plants differ in the lengths of their lives, and at which ages they are able to undergo reproductive activity. **Annual** plant species both reproduce and die within their first year of life. **Biennial** plant species tend to grow for one year, and flower and die in their second. **Perennial** plant species grow for two or more years, and oftentimes reproduce multiple times along this period.

benchmark: In this context, a benchmark is an informed objective or goal which to strive towards. These tend to be quantitative, or semi-quantitative, and reflect best scientific opinion and management knowledge regarding the capacity of land.

categorical: Relating to statistics, objects which are able to be defined as discrete groupings. For example Counties, are discrete categories in the state of Colorado. The attributes of objects which themselves are **continuous** may be recorded in a fashion which makes them **categorical** data. Categorical data often require different statistical approaches than continuous data.

continuous: Relating to statistics, information which follows a natural gradient, and is collected in a matter reflecting this. For example, the height of a human in feet is categorical, but height in large fractions of an inch are continuous. Continuous data often require different statistical approaches than categorical data.

C3/C4 grass: Two major photosynthetic pathways exist in grasses. C3 photosynthesis is the main form of photosynthesis in the plant kingdom, and aligns with the Cool-Season Grasses. C4 photosynthesis, present in the warm season grasses, is postulated to be an evolutionary adaption which makes grasses more competitive in moisture limited areas.

confidence intervals: A range of estimates around a parameter, such as the mean, of a population calculated from the sample. A confidence level of 0.8 should generate results that include the true parameter of the population in 80% of all instances.

domain: An area in space which delimits the spatial extent of an analysis. Usually the domain we used in this study was a square box, buffer 5km from the most extremes edges of the Field Office.

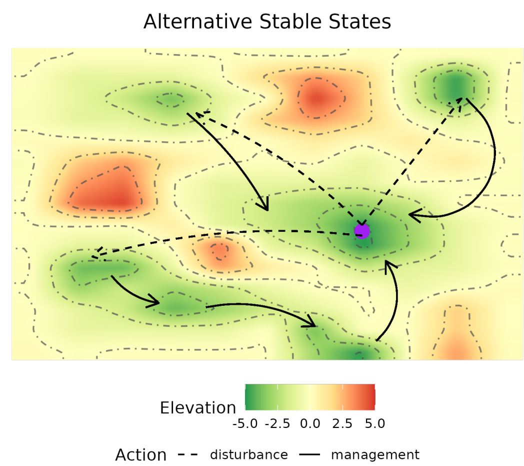

Figure 1: Theoretical Illustration of States (green) and Phases (three lower left green areas). Each dashed line indicates a type of disturbance which pushed an area into an alternate stable state from the **reference** state (center). Management actions to return the area to another state often require more work than it took to move it away (hysteresis, more curved lines have higher amounts). Each Green basin has a varying depth, those with are isolated by there own depths, or other features (red), such as the upper right green, has a high **resistance** to changing from alternate state to another. The width of the basins relate to the **resilience** of them, many minor disturbances will move the ball up from the deepest portion of the basin (not pictured), but it can readily return towards the center of it.

Ecological Site (Description): An ecological site **(ES)** is an area of land which is subject to roughly the same environmental factors, e.g. climate & soils, and which produce similar types of vegetation when both undisturbed and when subjected to the same type of disturbance, e.g. by wildfire. The **Description (ESD)** contains information regarding this site, and in particular the type of vegetation which it supports.

ensemble model: A single statistical model which takes multiple other models as inputs, and creates an average of them.

feature engineer: The development of new features from existing ones. We use the term here, but we are mostly imputing data into features which are missing them.

grain: The resolution of a dataset in both time and space. For example, if you had a rain gauge and checked it every hour of every day in March, you would have precipitation data at an hourly **grain**, and could also transform this to different grains like: **hourly**, **daily**, or a **monthly**. Spatial grains are analogous, and often form the most 'notable' component of a *raster*, i.e. the cell sizes.

impute a statistical process to replace missing data with substituted values

indicator: A set of features which are known to correlate to with a more challenging to measure metric, and serve as a proxy of it. The **AIM** dataset collects data on many features, which indicate overall site health.

inference: The utilization of applying statistical approaches and modelling to a sample to make judgement about a **population**.

invasive (species): A species which until recent history (e.g. within the holocene), has been restricted to certain major geographic areas, and was introduced to different areas by human activities, and which is **noxious** in this area.

landscape: A similar geographic area with several re-occurring features.

Line-Point Intercept: A quantitative method for measuring the cover of an object relative to a plane, e.g. the soil surface. A single line, such as a measuring tape, is unrolled and at pre-determined intervals along its length a pin, which emulates a point, is dropped. The presence of features of interest, such as plants and rock fragments, are recorded at each location the pin is dropped. This method is used for many of the AIM indicators.

noxious (species): A species which has features making it undesirable in certain contexts. These features generally pertaining to it's competitive exclusion of other species.

panel: In statistics this refers to datasets, where the same sample is revisited over time. For example, all of these plots were in the first of several AIM panels, each of which will be 5 years.

peer-reviewed journals: These provide the major forum for sharing academic research and ideas. The peer-review process entails sending an article, which an author would like to publish in a journal, to independent experts on the topic who verify that the work meets the standards which the journal requires for publication of work.

population: A group of individuals to which statistical *inference* can be made. The population is defined before it is sampled from.

raster: A format for transferring geographic data. A raster is, generally, a square grid where each cell depicts the value of an attribute, and is tied to a geographic location. The attributes which are usually stored in a raster are **continuous** such as *elevation.*

reference state the initial stable state which an area was in without human disturbance

resilience "the capacity of ecosystems to reorganize and regain their fundamental structure, processes, and functioning (i.e., to recover) when altered by stresses like drought and disturbances like inappropriate livestock grazing and altered fire regimes" (Chambers et al. 2017)

resistance "the capacity of ecosystems to retain their fundamental structure, processes, and functioning when exposed to stresses, disturbances, or invasive species" (Chambers et al. 2017)

reproductively active: The period during which a plant is reproducing, i.e. has flower and fruits. Many diagnostic features which are required to identify plant species are found only during this period.

sample frame: In statistics, the portion of the population of interest which can be sampled. In the Natural Sciences the sample frame may have to preclude areas which are dangerous to sample (high slopes), or very inaccessible (isolated Wilderness Areas).

target frame: In statistics, the total population of interest. This population can not always be ascertained, but in our study aligns with the sample frame.

TerrAdat: A centralized repository for AIM data which has undergone quality control and assurance steps. Also contains statistical summaries of a wide range of **indicators**.

weighted sample design: An approach which applies different probabilities of inclusion to members of the population, hence results in a random sample with more members of certain portions of the population than others. This is often used to provide adequate samples of groups with little representation.

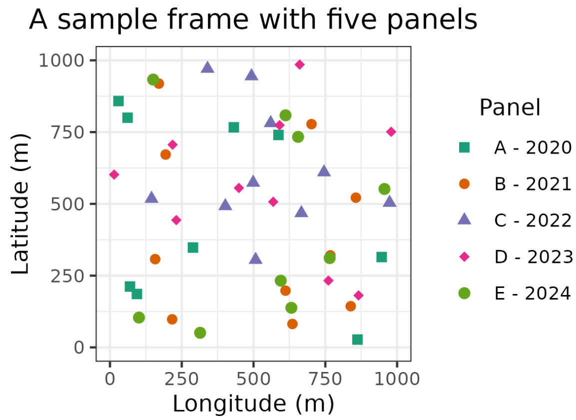

Figure 2: Graphic Illustrating a Target Frame, Sample Frame (the outer dimensions of the box), and which will be sampled as a single Panel.

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