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Grazing and Climate Effects on High Elevation Meadow Resources

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by

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ABSTRACT

Semi-arid rangelands cover roughly 41% of the Earth's land surface, and house more than 38% of the human global population. The Greater sage-grouse (*Centrocercus urophasianus*) has commonly been used as an umbrella species for restoration of sagebrush ecosystems in rangelands, due to its status as an indicator of overall rangeland health. Scarce mesic resources may lead to an energetic bottleneck for juvenile sage grouse, limiting fitness and survival rates. Mesic and ground-water dependent ecosystems found in the Great Basin of North America are heavily utilized by livestock and wildlife throughout the year. It is important for land managers to understand how intensity and timing of grazing affect the temporal availability of mesic commodities utilized by species like sage-grouse. This dissertation quantifies changes in the timing of availability of mesic sage-grouse resources across grazing and climatic gradients in high-elevation meadows. The methods include both on the ground and remotely sensed tools, and the correlations between them are assessed. The results suggest that field determined phenology, phenocam Green Chromatic Coordinate (GCC), Phenocam Normalized Difference Vegetation Index (NDVI), and Landsat NDVI are all highly correlated, with slight de-coupling occurring at the end of the growing season. Timing of growth varied in these ecosystems depending on yearly precipitation and vegetative type. Arthropod taxa abundance responded differently to grazing management and environmental variables in these meadow communities. Coleoptera abundance during peak sage-grouse usage periods had an increase of roughly 40% in some meadows with increased grazing intensity, while Formicidae abundance saw a 22% decrease. Near-surface cameras had varied success with predicting peak insect abundance levels. Sage grouse usage of the meadows was highly linked to growth seasons of vegetation, with slight decoupling occurring with growth seasons derived from phenocam GCC in drier years. Little correlation was seen between peak sage grouse usage of the meadows and peak capture rates of arthropods, this was true for all insect groups (Coleoptera, Formicidae, and Lepidoptera).

DEDICATION

To my wife Sahara, thanks babe.

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PREFACE

The objectives of this dissertation have both directly and indirectly been addressed throughout the three chapters below. Each chapter consists of a manuscript formatted for publication in a peer review journal. Manuscripts and references are formatted to meet the requirements of each respective journal. Since each chapter is created to be its own manuscript, overlap in methods does occur across the chapters.

Chapter 1: Changes in meadow phenology in response to grazing management at multiple scales of measurement. Published in Remote Sensing in 2021.

Chapter 2: Shifts in sage-grouse arthropod food sources across grazing and environmental gradients in upland meadow communities. In review at Arthropod-Plant Interactions.

Chapter 3: Comparison of methods used for determining the impact of grazing on synchrony between sage-grouse usage and food source availability.

INTRODUCTION

Over 70% of the forage consumed by livestock worldwide is grown on rangelands (Lund 2007). Livestock management in rangelands can have long-lasting effects on wildlife diversity and habitat (Krausman et al., 2009; Alkemade et al., 2013). Mesic and ground-water dependent areas within rangelands are of major concern to both wildlife managers and producers due to the large amounts and variety of quality forage available in those habitats during seasonal drought. Understanding how management approaches across multiple years of implementation can alter the availability of mesic resources is crucial for developing land use strategies which harmonize human interests and the preservation of threatened species. Various management techniques have been developed throughout the years to promote mesic habitat conservation and maintenance (Swanson et al. 2015). Managing grazing to account for plant recovery through the use of timing, duration, and intensity has proven to be an effective form of mesic habitat restoration (Goss and Roper 2018). The implementation of remote sensing tools such as high-resolution cameras and satellite imagery can allow for the continuous monitoring of plant usage, and may prompt changes in grazing regimes to occur which will lead to more adaptive management approaches.

This dissertation seeks to 1) quantify changes in the timing of availability of mesic resources across grazing and climatic gradients in high-elevation mesic systems, 2) develop and test remote sensing monitoring techniques that can accurately capture those shifts, and 3) determine if those shifts create dissonance between the forage timing of a wildlife species (the greater sage-grouse, *Centrocercus urophasianus*) which uses those resources and the timing of when those resources are most available. By focusing on these objectives, this dissertation can deepen the understanding of how human use of these mesic systems potentially impacts wildlife species that heavily rely on said systems. Additionally, this dissertation can further clarify which monitoring tools can most accurately and easily track resource abundance and availability.

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CHAPTER 1. Changes in Meadow Phenology in Response to Grazing Management at Multiple Scales of Measurement

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Abstract

Riparian and ground-water dependent ecosystems found in the Great Basin of North America are heavily utilized by livestock and wildlife throughout the year. Due to this constant pressure, grazing can be a major influence on many groundwater dependent resources. It is important for land managers to understand how intensity and timing of grazing affect the temporal availability of these commodities (i.e., biodiversity, water filtration, forage, habitat). Shifts in forage or water availability could potentially be harmful for fauna that rely on them at specific times of the year. Seven meadow communities, each consisting of three distinct vegetative communities, were grazed at three intensities to determine the relationship between grazing management and phenological timing of vegetation. The agreement of on-the-ground measurements, near-surface digital cameras (phenocams), and satellite-based indices of greenness was examined for a two-year period (2019–2020) over these grazing and vegetative community gradients. Field determined phenology, phenocam Green Chromatic Coordinate (*GCC*), and Landsat Normalized Difference Vegetation Index (*NDVI*) were all highly correlated and the relationship did not change across the treatments. Timing of growth varied in these ecosystems depending on yearly precipitation and vegetative type. Communities dominated by mesic sedges had growing seasons which stopped earlier in the year. Heavier grazing regimes, however, did not equate to significant changes in growing season. Ultimately, shifts in phenology occurred and were successfully monitored at various spatial and temporal scales.

Keywords: GCC, grazing management, groundwater dependent ecosystems, Landsat, NDVI, phenocam, phenology

Introduction

Understanding how groundwater dependent areas located within rangelands are affected by various management techniques is crucial to conservation. Their importance in arid and semi-arid ecosystems is generally disproportionate to their overall size [1,2]. Riparian areas boost landscape biodiversity [3], filter water [4], provide forage and water for livestock [5], and serve as suitable seasonal and year-round habitat for a wide variety of key taxa [6,7].

Livestock tend to congregate in these systems, due to increased water and forage availability [8]. Improper grazing management has been shown to reduce vegetation, leading to destabilized streambanks and erosion [9,10]. Over-grazing can also decrease leaf area, nutrient stocks, photosynthetic capability, reproductive success, and vegetative growth [11–13]. It should be noted that varying plant communities can lead to different responses in grazing. Skaer et al. [14] found that grazing at Palo Corona Regional Park (Monterey County, California) led to a 15% increase in cover of exotic annual forbs but had little to no effect on the cover of native perennial forbs and grasses.

One of the main obstacles that land managers face is understanding how disturbance affects the high temporal variability that accompanies plant establishment and growth [15]. Phenology, or the seasonal recurrent activity of plants and animals, is the term used to describe this trait [16]. In future years, climate change may result in widespread shifts in phenology across a large range of taxa [17–19]. Managers may have to alter land use practices to compensate for these shifts. Assessing how phenology changes over gradients of temperature, moisture availability, and land use will help to clarify how lifecycle timing may be altered in the coming years.

Grazing can affect phenology in a number of ways. Both grazing and grazing exclusion can indirectly modify phenology by influencing environmental factors like soil nutrients, soil moisture and temperature, and litter build-up [20–23]. Li et al. [24] showed that grazing exclusion led to a thicker litter layer, which suppressed water loss by soil evaporation, and resulted in a later vegetation green-up period. Zhu et al. [25] tested this hypothesis in alpine meadow grasslands in northern Tibet. They found that after 2-4 years, grazing exclusion significantly advanced the green-up and flowering dates of several low-growing, shallow rooted species.

Groups of researchers, such as the USA National Phenology Network, have developed general phenological monitoring techniques that allow for a greater consistency between studies [26]. Within the Great Basin, several unique challenges hinder the monitoring of plant phenology. Given the vast and rugged landscapes, many areas are difficult to access and sample. Additionally, many of these ecosystems consist of a wide variety of herbaceous and woody vegetation, each plant type having different phenological patterns. This spectrum obstructs the application of a singular comprehensive monitoring technique [27].

Understanding the relationships between various spatial and temporal resolutions of phenology, could potentially allow for extrapolation of patterns seen from in situ field observations to larger landscapes seen from satellite imagery [28–30]. One way in which the gap can be bridged between field observations and satellite imagery is through the use of repeat photography from ground- or tower based digital cameras (hereafter referred to as phenocams) [31,32]. Average daily greenness index values, such as the Green Chromatic Coordinate (GCC), calculated from phenocams can be used to fit curves which estimate growing season metrics [32,33]. GCC has been shown to be highly sensitive to slight changes in leaf colors in summer and more pronounced color changes in the autumn [34]. This may prove to be useful in arid environments where green-up and plant growth occurs fairly quickly following rainfall pulse events in late spring and early summer [35–37]. Using phenocams aids in increasing temporal resolution, broadening the spatial footprint of field observations, and reducing the costs of field-based observations [38]. Browning et al. [27] showed the utility of phenocams in arid environments by comparing phenological metrics of black gramma (*Boutella eriopoda*) and mesquite shrub (*Prosopis glandulosa*) across field observations, phenocams, and satellite time series in a Chihuahuan desert grassland. They found that field observations of canopy greenness and start-of-season date were highly correlated with both satellite and phenocams imagery. Systems with more heterogeneous vegetation would require satellite-derived indices with finer spatial resolution. Two of the most common satellite tools used for calculating broadscale phenology indices are Moderate Resolution Imaging Spectroradiometer (MODIS) near-infrared (*NIR*) and red reflectance data, and Landsat Normalized Difference Vegetation Index (NDVI) derived data [31,39–41]. In regard to groundwater dependent riparian meadows, the spatial resolution of MODIS *NIR* data (250 X 250 m) is too coarse to accurately monitor the area without including neighboring communities [42,43]. Landsat NDVI derived data has been shown to have high enough spatial resolution (30 X 30 m) to track the phenology of many groundwater dependent meadows [34,44].

This study aims to (1) establish baseline phenological metrics, and therefore better understand seasonal productivity and forage availability, in several heavily grazed upland meadows within the Great Basin, (2) quantify the effects of differing grazing management strategies on phenological metrics in the context of year-to-year climate variability, and (3) determine the relationship between on-the-ground phenology measurements, phenocam GCC, and Landsat NDVI data within these heterogeneous systems and across a spectrum of different grazing regimes.

Methods and Materials

Site Description and Treatments

Several high elevation meadows, referred to as the Haypress meadow complex, are located in the Desatoya Mountains of the Central Great Basin (39°27′ N, 117°36′ W; 39°19′ N, 117°42′ W). Over the past 10 years (2010–2020), annual precipitation in this complex ranged between 175 and 419 mm, with an average value of 315 mm. Annual mean temperature ranged from 4.6 to 7.6 °C, with an average of 5.8 °C. These values were obtained via the Google Earth Climate Engine (http://climateengine.org (accessed on 1 October 2021), [44]). Four meadows were selected for this study; they were given names based on their relative locations or the grazing intensity associated with them: (1) Lower at 2322 m, (2) Middle at 2365 m, (3) Upper at 2438 m, and (4) Uncontrolled at 2320 m. For the Lower, Middle, and Uncontrolled meadows the soil type for the first 14 inches of soil was a gravelly loam; lower than 14 inches the soil type was a gravelly clay loam. In the Upper meadow, the soil type for the first 12 inches was a gravelly clay loam; beneath that it became an extremely cobbly loam [45].

The vegetation of the meadows, while diverse, was delineated into three distinct classes. These classes were (1) the dry meadow community, which consisted of an overstory of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) and an understory of primarily Douglas' sedge (*Carex douglasii*), (2) the mesic meadow community, which consisted of Douglas' sedge, arctic rush (*Juncus arcticus*), and a mixture of meadow forbs (ex. *Chorispora tenella, Symphyotrichum ascendens*), and (3) the wet meadow community, which consisted of Nebraska sedge (*Carex nebrascensis*), arctic rush, and a mixture of facultative wet forbs (ex. *Montia chamissoi, Veronica americana*). Vegetation surveys were conducted in 2020, using standard line-point intercept methods [46], to verify the differentiation of these groups (Figure 1). On average, forbs had the highest amount of cover in the wet communities (29%), graminoids in the mesic communities (71%), and shrubs in the dry communities (56%).

Historically, grazing in these meadows consisted of yearlong grazing by feral horses and managed cattle grazing. Haypress livestock grazing occurred in the following manner: for two consecutive years, grazing occurred in May and continued through mid-June; the following two years, use occurred mid-August through September. Livestock use was generally limited to one month, with around 700 cow–calf pairs having access to the meadows but could last up to six weeks depending on the year. In 2019, cattle were brought in on the first half of the grazing rotation (mid-June).

In the winter of 2019, pipe-rail fencing was installed in the complex. The perimeters of the Lower, Middle, and Upper meadows were fenced completely, while the Uncontrolled meadow was not fenced. For the three fenced meadows, cross fencing was constructed which cut each meadow in half. The higher elevation portion of each meadow was designated as part A, and the lower part B. This allowed for a total of seven separate units to be treated and measured (Table 1). In the summer of 2020, cattle were brought into the complex in mid-June. Section A of each meadow was grazed such that the major forage species of the riparian areas had a mean stubble height of roughly 10 cm [47] (approximately 3 days of grazing). Section B of each meadow was left ungrazed, and the Uncontrolled meadow was left unmanaged (allowed for constant grazing by both cattle and horses to occur in the meadow throughout the summer).

Field Methodology

The Line-Point Intercept (LPI) method [46] was used to calculate the percent cover of the wet, mesic, and dry subregions of each of the seven meadow sections in June of 2018. The number of transects, length of transects, and number of points along transects was tailored to each area of vegetation (Figure 2, lengths of transects ranged from 3 to 20 m, number of transects per community ranged from 3 to 9). Monitoring was set up in such a way that at least 100 data points could be generated from each LPI measurement (points ranged from 100 to 120).

In 2019 and 2020, these same transects were used to monitor the phenology and height of the meadows bimonthly, throughout the growing season (May to October). At 25%, 50%, and 75% of the total length of each transect, a 1 m hoop was placed and average height (collected using a fiberglass folding ruler) and phenology of each species in the hoop was recorded (Figure 2). Average phenology was determined using simplified methods outlined in Moore et al. [48], where each species was assigned one of 8 different phenological stages: Leaf development (L), Initial inflorescence (I), Mature inflorescence (Im), Inflorescence– pollen released (Ip), Initial seed development (S), Mature seed development (Sm), Dropped seed (Sd), and Desiccated (D). If a species was grazed to the point where phenology could not be ascertained, a rating of G (Grazed) was given. A total of roughly 9–15 hoops were placed per community type for each of the meadow regions.

Phenocam and Landsat Methodology

In fall of 2017, four phenocams (StarDot NetCam SC 5MP IR-enabled cameras using complementary metal oxide semiconductor (CMOS) image sensors, StarDot Technologies, Buena Park, CA, USA) were installed in four of the meadow sections previously discussed, in fall of 2018 the other three were installed. Methods of installation and use for the phenocams were sourced from the PhenoCam Network (https://phenocam.sr.unh.edu/webcam/tools/ (accessed on 1 October 2021)) (Table 1). Additionally, time-domain reflectometry soil moisture sensors (CS-616, Campbell Scientific, Logan, UT, USA) were installed at depths of 10, 20, 35, and 50 cm at each of the phenocam sites. Eight RGB images at a resolution of 1296 X 960 pixels were taken daily between 12:00 pm and 3:30 pm. All data and images were downloaded monthly during the growing season, and once every three months during the winter.

Phenopix, the R statistical program package [49], was used for the calculation of all phenological metrics derived from the phenocam images. The package allowed Regions of Interest (ROIs) to be manually defined that were representative of the wet, mesic, and dry communities within each meadow section. The average digital numbers, which ranged from 0 to 255 and were extracted for red (RDN), green (GDN), and blue (BDN), of the pixels within each ROI were used for the analysis. These numbers were converted into GCC values, a relative percent index, using the following equation:

$$\mathrm{GCC} = \mathrm{GDN}/((\mathrm{RDN}+\mathrm{GDN}+\mathrm{BDN})) \quad (1)$$

GCC values were filtered using the night, spline, and max filter options in the phenopix package. An additional novel filter using the blue chromatic coordinate, was added to remove values influenced by snow (see Snyder et al. [43] for more details). A method proposed by Gu et al. [50] was used to apply a double logistic fit to the filtered sub daily GCC values. One thousand replications were used for an uncertainty analysis to determine how well predicted values fit observed. The Gu et al. [50] threshold method calculates four phenophase dates for each fitted set of data: an upturn date (UD) when GCC of vegetation begins to increase consistently, a stabilization date (SD) when vegetation approaches maximum GCC, a downturn date (DD) when GCC starts to consistently diminish, and a recession date (RD) when vegetation reaches a seasonal low. The double logistic fit determined by Gu et al. [46] was also applied to the collected soil moisture data and the RD phenophase DOY was determined for each individual meadow across the two years.

Landsat NDVI data were extracted from Landsat 8 Optical Imager (OLI) satellite images using the Climate Engine (http://climateengine.org (accessed on 1 October 2021)) [44]. Polygons were created for each meadow section and NDVI was determined for 2019 and 2020. NDVI values were calculated as Equation (2):

$$\mathrm{NDVI} = ((\mathrm{NIR}-\mathrm{Red}))/((\mathrm{NIR}+\mathrm{Red})) \quad (2)$$

where *NIR* is the near infrared at-surface reflectance and *Red* is the red at-surface reflectance. The polygons created were large enough to encompass dry, mesic, and wet vegetation communities, and were at least one pixel in size (30 m X 30 m). Climate Engine applied a cloud mask to the Landsat top of atmosphere (TOA) reflectance data. The cloud masking removed medium and high confidence snow, shadow, and cirrus clouds using the Quality Assessment Band provided in the Landsat Google Earth Engine (GEE) collection.

Statistical Methodology

A random forest model was used to determine which variables had the greatest importance in predicting the on-the-ground phenological state of vegetation. Random forest models are used in the ecological community when dealing with issues such as auto-correlation [51,52]. This is due to these models being non-parametric, using bootstrap aggregation, and implementing feature randomness. Table 2 shows the variables used in the analysis. The field data collected was input to the *randomForest* package [53] in the R statistical program [54] to create and test the model, with four simplified phenological stages used as the output: Leaf (L), Inflorescence (I), Seed (S), and Desiccated (D). The analysis was completed using 1000 trees and was set to test three variables at each split. Out-of-box estimate of error was used to measure the strength of the model and the mean decrease in Gini coefficient was used to measure the importance of each variable.

To further test the strength of correlation between field- and camera-based observations, the relationship between the four field-based, simplified phenological stages and the four camera-based phenophase dates was analyzed. The date where the highest percentage of each simplified phenological stage was seen in each community type of each meadow region was calculated. These values were then plotted against their corresponding camera- based phenophase dates (UD-L, SD-I, DD-S, and RD-D) and repeated measures correlation coefficients were calculated, with meadows used as the subject for repeated measure. Factorial ANOVAs and Tukey tests were utilized to determine if there were differences in UD, SD, DD, and RD dates across vegetation communities, years, and grazing interactions. Meadow location was used as a randomized block variable. Since the grazing treatment began in 2020, the analysis was treated as a before–after design where a significant grazing treatment effect would appear in the grazing–year interaction. Similar ANOVAs and Tukey tests were created for field phenology dates (max observance of L, I, S, and D). Data met the assumptions of homoscedasticity, independence, and normality. Root mean square error (RMSE) and Pearson's *r* were used to determine the level of correlation between Landsat derived NDVI and camera-derived GCC. Comparisons between Landsat NDVI and camera-derived GCC were completed using data from the dates of the Landsat imagery.

Results

Phenology Stage Modeling

Grazing treatments were successfully implemented and monitored in 2019 and 2020. In 2020, grazed meadows were brought down to a 10 cm stubble height in the wet communities, with cattle tending to focus their attention on the wetter areas and leaving the other sections of the meadows largely undisturbed for the period they were allowed in the exclosures. Phenological data of the vegetation across all meadows were collected, both in person and remotely, and a random forest model was successfully created to predict phenological stages in the Haypress complex (L-Leaf; I-Inflorescence; S-Seed; D-Desiccated). The out-of-box estimate of error (OOBE) obtained by the model was

33.56%. Across the phenological classes, there were significant differences in error. Phenological stages D and L were correctly predicted most frequently by the model, with class errors of the stages being 0.2772 and 0.1985, respectively. The model had a more difficult time with predicting stages I and S, with class errors of 0.5962 and 0.8703. DOY, *GCC*, life form, and soil moisture were the variables with the highest level of importance based on their mean decreases in Gini. Camera, year, grazing, and meadow were the variables with the lowest (Figure 3).

Phenocam and Field-Based Metrics and Correlations

Different vegetation communities were effectively delineated from the phenocam images and phenophase dates were extracted (UD—upturn date; SD—stabilization date; DD—downturn date; RD—recession date) (Figure 4). A statistically significant difference in DOY for SD, DD, and RD phenophases was found across both vegetation community type and year, however not for meadow or grazing–year interaction. For the UD phenophase, significant differences were only seen across years (Table 3). The mean difference between 2019 and 2020 was highest for the DD phenophase (-42.13) and lowest for the UD phenophase (-14.13). A Tukey post-hoc test revealed significant pairwise differences only for the SD mesic-dry comparison in 2019. In 2020, community differences were seen for mesic-dry and mesic-wet comparisons across all phenophases except UD (Figure 5). No differences between wet-dry communities were statistically significant in either 2019 or 2020.

The date when each phenological stage was most frequently observed in the field was unique for both years. Consistently, across both meadows and years, the phenology stages that had the closest maximum observed DOYs were I and S (Figure 6).

Significance of variables for field-observed data was similar to that previously discussed for phenocam data. Meadow and grazing–year interaction were not significant for any phenology stage, while year was significant for all stages (Table 4). Vegetative community was only significant for S, with significant interannual differences solely occurring between the mesic and other two communities in 2020. Correlation between camera-based phenophase dates and field-based phenological stages was very high (Figure 7). Year-to-year changes led to some variability in the repeated measures correlation coefficients of the data, with a range of 0.725–0.942 across the two years and all community types. All correlation values were above 0.90, except for the mesic in 2020 (0.725).

Grazing intensity, as shown in the ANOVA model, had little effect on phenophase date (Figure 8). For the UD, SD, and DD phases, the treatment that had the latest average date varied across community type. For example, for the UD phase in 2020, the grazed meadows had the latest average date in the dry communities (102), ungrazed had the latest average date in the mesic communities (109), and the uncontrolled meadow had the latest date in the wet communities when compared to the averages of the other treatment groups (110). Additionally, when looking at the same community type, the treatment with the latest average date was never the same across all phenophases. The RD phase was slightly different in that, in 2019, the meadow that would be uncontrolled had the latest date in the dry communities, and the meadows which would be ungrazed had the latest average date for the wet and mesic communities. In 2020, however, the uncontrolled meadow had the latest date compared to the averages of the other treatments across all communities. It should be noted that these shifts did not equate to significance in the ANOVA model.

Landsat and Phenocam Relationship

Due to the spatial resolution (30 X 30 m) of the Landsat data, metrics were calculated and used at the whole meadow level, instead of at individual community levels. Across all meadows, treatments, and years, Pearson's *r* values demonstrated a strong correlation between Landsat NDVI and phenocam GCC, and RMSE values demonstrated strong similarities in the prediction errors of the comparisons (Table 5). RMSE values never rose above 0.108, and Pearson's *r* values never dropped below 0.91. For both 2019 and 2020, the meadow with the highest Pearson's *r* value was Lower B (0.983 and 0.984). The meadow with the lowest Pearson's *r* value for both years was Upper A (0.911 and 0.912).

Soil Moisture

Determining the RD dates for the soil moisture of the meadows demonstrated the effect year-to-year differences had on meadow soil moisture retention (Figure 9). The average RD DOY for 2019 was 190, while the average DOY for 2020 was 144. The treatment with the latest DOY in 2019 was the grazed (192 average), and the treatment with the latest DOY in 2020 was the uncontrolled (162).

Discussion

Influences of Climatic and Community Variables

Prism data showed that the study encompassed both the wettest year over the past decade in the meadow complex (2019, 419 mm annual precipitation) and the driest (2020, 175 mm annual precipitation). Interestingly, the effect this change in precipitation regime had on plant phenology was more distinct in the strictly camera-based data than in the random forest (RF) model, which combined both field- and camera-based observations. From the RF model, year to year variability was designated as the second least important variable (as determined by its mean decrease in Gini). In contrast, soil moisture differences within the season of growth had one of the highest levels of importance. When analyzing year to year differences picked up by camera-derived phenophases, the changes were more perceptible, with the dry year of 2020 having consistently earlier phenophases across the meadows, which is expected due to earlier snow melt. It should also be noted that within the RF model, temperature variability within the growing season was much less important in determining phenological stages than DOY, life form, and soil moisture.

The type of vegetation community also played a less important role in determining phenological stage than expected within both the RF model and the ANOVA model for the field data. The ANOVA model only showed community significance for the S phenology stage, and the RF model had community as the sixth most important variable (out of 11). While delineation of these community types was determined to be fairly accurate, overlap may have occurred leading to error. Encroachment of species from one community type into another was common, and is clearly demonstrated in Figure 4 [55,56]. Conversely, when comparing camera-derived dates, the ANOVA showed significant differences between communities for all phenophases except UD. The major differences in phenophases were seen between mesic vegetation and the other two community types. For both years, very little variation was seen in the UD dates (probably due to high soil moisture in all meadows early in the year). In 2020, however, as the growing season progressed, the phenophase dates of the mesic communities came earlier.

This could have been due to the mesic community soils drying out quicker than the wet community soils, as well as the woody shrubs in the drier communities having deeper roots and naturally later seasons of growth than the mesic sedge communities. Regardless, for the earlier portion of the year, there was not much difference between vegetation communities. It should also be noted, that while phenophase UD dates started at roughly the same time for both years across all vegetation community types, the greatest disparity between years was at the downturn and end of growing season (i.e., DD and RD) in the mesic communities (Figure 7). This demonstrates that the mesic communities were not only affected the most by interannual environmental changes, but also by year-to-year environmental changes. The clarity provided by the phenocams shows the importance that multiple temporal resolutions of observation can have on understanding phenology in these complex ecosystems.

The RF model as a whole, had issues with differentiating between phenological stages within the meadow complex. While the model was fairly accurate in predicting when vegetation reached stages L and D, the error linked with stages I and S was significantly higher. This may largely be explained by the difficulty associated with properly identifying the two stages in the field. Certain species required greater observer experience in recognizing phenological shifts and human error may be present in the data. Dense sagebrush cover may have also obscured vision and made it difficult to ascertain field metrics of understory forbs and graminoids. Additionally, variability in genotypes and small-scale variability caused by competition or edaphic factors could lead to differences of phenology of individual species [57,58]. This could have allowed for a high level of irregularity in individual species’ flowering and seeding time, irregularity which the model may not have detected.

It should be noted, as well, that when the two stages were differentiated, they usually occurred fairly close together in the context of the growing season as a whole. The dates where the highest percentage of observed species were in the I and S stages were very close across all meadows, while the dates for L and D were much further from the next closest stage. This would make it very difficult for the model to separate the I and S stages from one another. In future iterations of the model, the I and S stages could possibly be combined into a single group. While more basic, this would still help managers and researchers in understanding when plants would be susceptible to grazing and could influence management decisions.

Grazing Impact on Phenology

One of the major limiting factors of this study was the deficient number of meadows that had an uncontrolled grazing treatment. This was due, in large part, to time and monetary restrictions. When analyzing the effects grazing had on phenology, the RF model seemed to lack the ability to pick up variation across treatments. This was demonstrated by grazing having the third lowest level of importance among the variables used. Camera- derived phenophases and field-observed phenology dates also showed little variation between grazing treatments. Some variation was seen within the RD dates, but this was not deemed significant. With more years of grazing management, and more meadows included in the uncontrolled treatment, differences could potentially be seen.

We theorize that any differences seen between the grazing treatments in the future may be partially explained by soil moisture being retained in the soil later in the season. We fitted double logistic curves comparable to the ones fitted for GCC to calculate the "phenophase dates" for the soil moisture values in each of the meadows (Figure 9). The uncontrolled meadow had a similar RD DOY for soil moisture in 2019 to the other meadows, but in 2020 the RD DOY shifted much later into the year when compared to the others. Zhang et al. [59] hypothesized and observed that increased grazing in alpine meadows initially led to lower levels of transpiration from vegetation, allowing for moisture to stay later in the soil. Eventually, however, long term grazing led to an increase of bare ground and evaporation, limiting soil moisture in comparison to less grazed areas. Similar trends were shown in other studies across rangeland ecosystems [60–62]. If similar grazing patterns were to continue in Haypress, a shift in soil moisture to earlier RD dates could potentially occur within the uncontrolled meadow. Alternatively, sustained grazing has the potential to delay plant development (i.e., reduce plant size), and in so doing limit water uptake until later in the season. In a study conducted by Virgona et al. [63], they demonstrated that prolonged grazing in wheat fields could potentially elongate the growing season and delay development. They stated that within their experiment, for every 4-5 days grazed, antithesis and maturity was delayed by 1 day. This and other studies demonstrate how grazing can affect the maturation of vegetation [64–66]. This delayed necessity for water uptake could be another explanation for why soil moisture in the uncontrolled meadow was higher when compared to other meadows and may continue to be in future years.

Relationships between Spatial Resolutions

Phenocam derived GCC demonstrated a strong correlation with field observations by being the variable with the second highest degree of importance in the

RF model. Additionally, Landsat *NDVI* maintained a high level of correlation with phenocam GCC across all years, treatments, vegetation communities, and sites observed (see supplementary Figure S1 for more detail). These strong relationships make a clear argument for the ability to develop statistical models which could extrapolate phenology metrics to other Great Basin Meadow systems using Landsat. With a wider sample size of meadows, years, and grazing treatments, and more effort to define and model the relationship between phenocam and satellite imagery, Landsat data could also potentially be used to extrapolate grazing pressure to other meadow systems.

Conclusions

Overall, this study shows how remote sensing and on-the-ground science can work in coordination to create a deeper understanding of phenological changes across gradients. Within the Haypress complex, more years of observation need to be incorporated in order to examine vegetational shifts in years that do not embody opposite ends of its climatic spectrum. Furthermore, additional meadows could be included to analyze how heavily grazed systems change in relation to properly managed environments and see if these systems emulate similar patterns to previous studies. Understanding the influence grazing has on these areas, in the context of phenology, will allow for the implementation of management strategies that do not create dissonance between the timing of meadow resources and the needs of those that rely on them.

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Tables

Site	Camera Viewshed (ha)	Meadow Area (ha)	Lat/Long (Decimal Degrees)	Camera Orientation (Cardinal Direction)
Upper A	0.09	0.64	39.317479/-117.697128	NE
Upper B	0.11	0.29	39.315936/-117.698013	NNW
Middle A	0.10	0.73	39.322538/-117.703238	NE
Middle B	0.04	0.11	39.320682/-117.703979	NNE
Lower A	0.18	0.56	39.331126/-117.701456	NNE
Lower B	0.15	0.66	39.330403/-117.701739	NNW
Uncontrolled	0.12	0.33	39.325874/-117.688641	E

Table 1. Meadow and phenocam information, including the area of the phenocam field of view (i.e., camera viewshed), the areal extent of the meadows, latitude and longitude of the meadows, and the orientation of each phenocam.

Variable	Definition
DOY	Day of year
GCC	Green Chromatic Coordinate value derived from phenocams
Life.Form	Life-form (i.e., graminoid, forb, graminoid-like, shrub)
Moisture	Soil moisture
Temperature	Temperature (degrees Celsius)
Community	Plant community type (i.e., dry, mesic, wet)
Grazing	Grazing intensity (i.e., uncontrolled grazing, managed grazing, ungrazed)
Year	Year of data collection
Meadow	Meadow of data collection (Upper, Middle, etc.)
Camera	Portion of meadow (i.e., high elevation, low elevation)

Table 2. List of variables used in the creation of a random forest model for the prediction of phenological stages in the meadows.

Variable	Analysis	UD	SD	DD	RD
Year	ANOVA Prob > F	0.0006	<0.0001	<0.0001	0.0001
Community	ANOVA Prob > F	0.06	0.0005	0.0012	<0.0001
Grazing/Year Interaction	ANOVA Prob > F	0.992	0.929	0.289	0.164
Meadow	ANOVA Prob > F	0.06	0.332	0.359	0.067

Table 3. *p*-values for vegetation community types, years, meadows and grazing interactions derived from factorial ANOVAS, across all phenophases. Phenophase stages are upturn date (UD), stabilization date (SD), downturn date (DD), and recession date (RD).

Variable	**Analysis**	**L**	**I**	**S**	**D**
Year	ANOVA Prob > F	<0.0001	<0.0001	0.0016	0.003
Community	ANOVA Prob > F	0.522	0.0525	0.0248	0.244
Grazing/Year Interaction	ANOVA Prob > F	0.782	0.588	0.945	0.889
Meadow	ANOVA Prob > F	0.067	0.377	0.947	0.092

Table 4. *p*-values for vegetation community types, years, meadows and grazing interactions derived from factorial ANOVAS, across all maximum percentages of observance for each phenology stage. Phenology stages are leaf growth (L), inflorescence (I), seed (S), and desiccated (D).

Year	**Meadow**	**Pearson's *r***	**RMSE**	**Prob > F**
2019	Lower A	0.963365	0.107	> 0.0001
	Lower B	0.983981	0.104	> 0.0001
	Middle A	0.978051	0.087	> 0.0001
	Middle B	0.958376	0.075	> 0.0001
	Upper A	0.911001	0.061	> 0.0001
	Upper B	0.991294	0.137	> 0.0001
	Uncontrolled	0.970543	0.075	> 0.0001
2020	Lower A	0.946754	0.065	> 0.0001
	Lower B	0.982728	0.061	> 0.0001
	Middle A	0.950454	0.056	> 0.0001
	Middle B	0.913333	0.039	> 0.0001
	Upper A	0.912522	0.08	> 0.0001
	Upper B	0.966456	0.087	> 0.0001
	Uncontrolled	0.945573	0.059	> 0.0001

Table 5. Root mean square error (RMSE) and Pearson's correlation coefficient (*r*) between phenocam GCC and Landsat *NDVI* for all meadow areas.

Figures

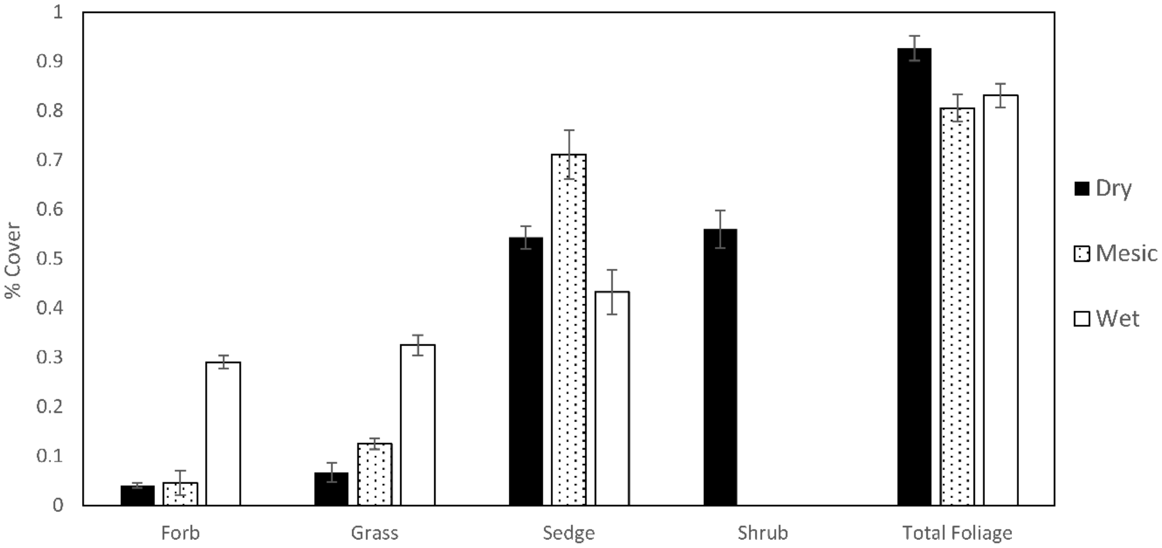

Figure 1. Line point intercept data grouped by percent foliar cover of functional type at four meadows in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

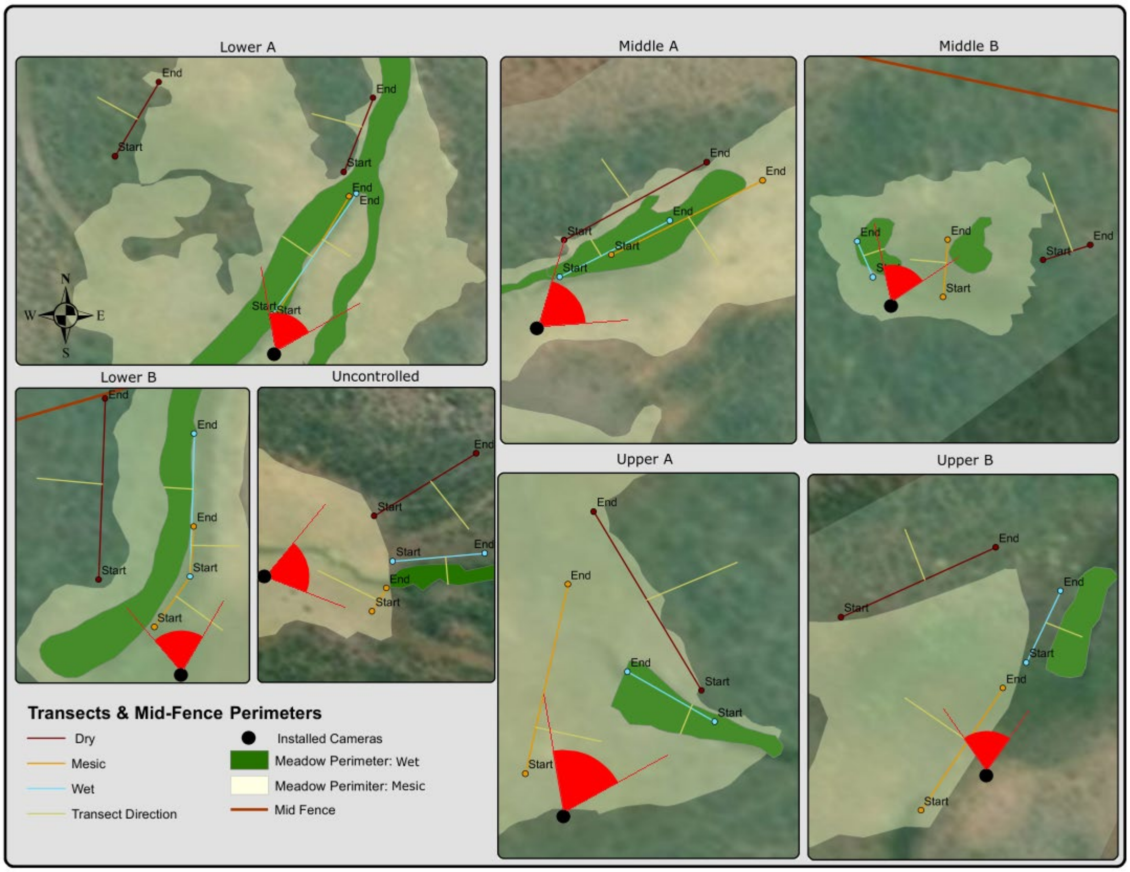

Figure 2. Layout of seven meadow sections in regard to transect position and length, phenocam position and direction of view, and vegetation community position in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

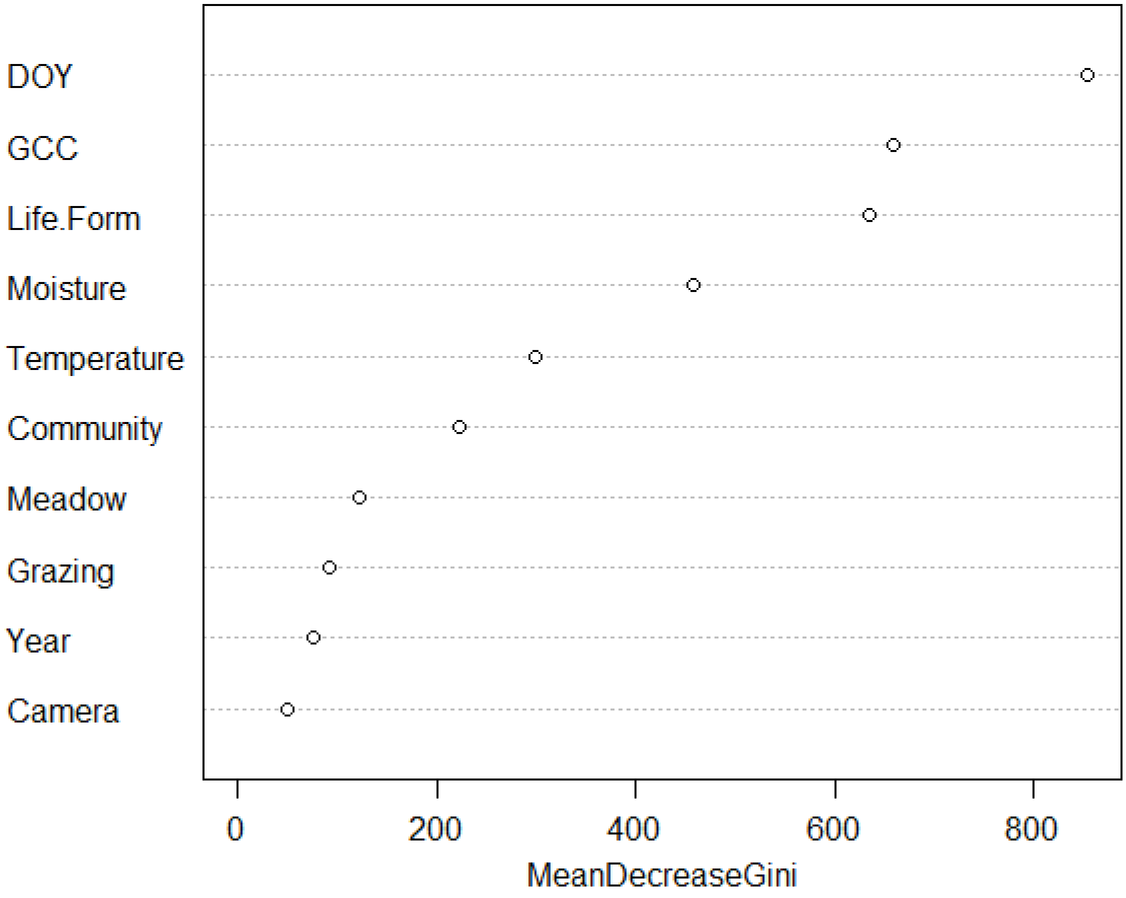

Figure 3. Importance of variables used to create a random forest model for the prediction of phenological stages in the meadows. Importance was calculated based on mean decrease of Gini.

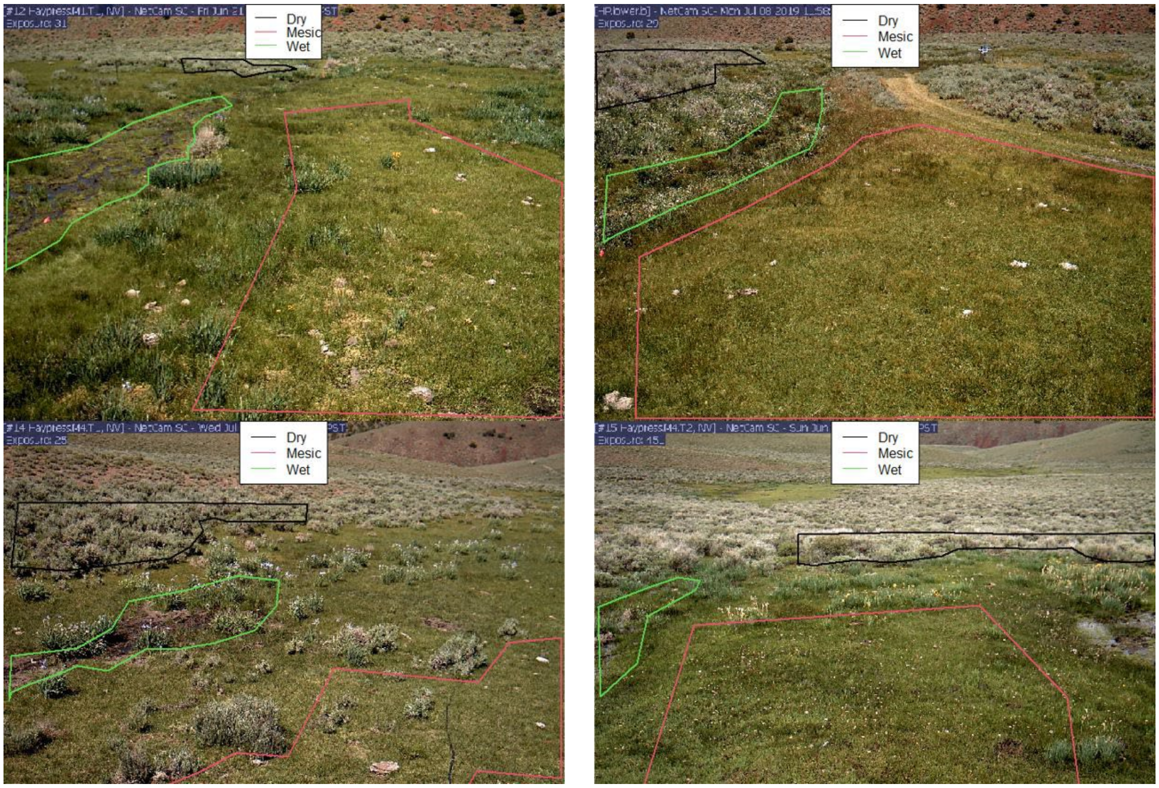

Figure 4. Examples of phenocam regions of interest (ROIs) for dry, mesic, and wet vegetation communities in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

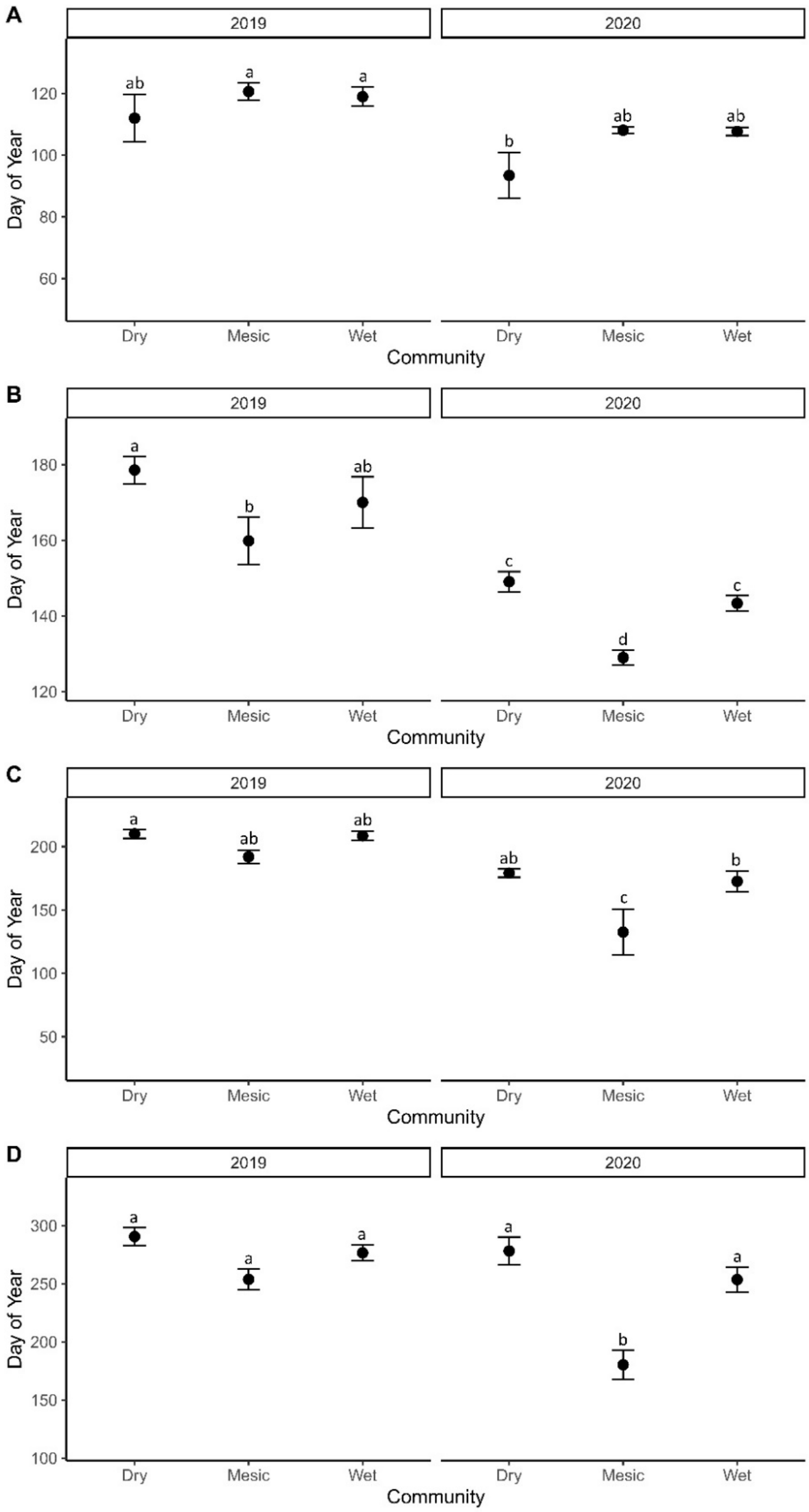

Figure 5. Mean (± SE) phenophase values across vegetation communities and years. Phenophase stages are (**A**) upturn date (UD), (**B**) stabilization date (SD), (**C**) downturn date (DD), and (**D**) recession date (RD). Different letters indicate significant differences.

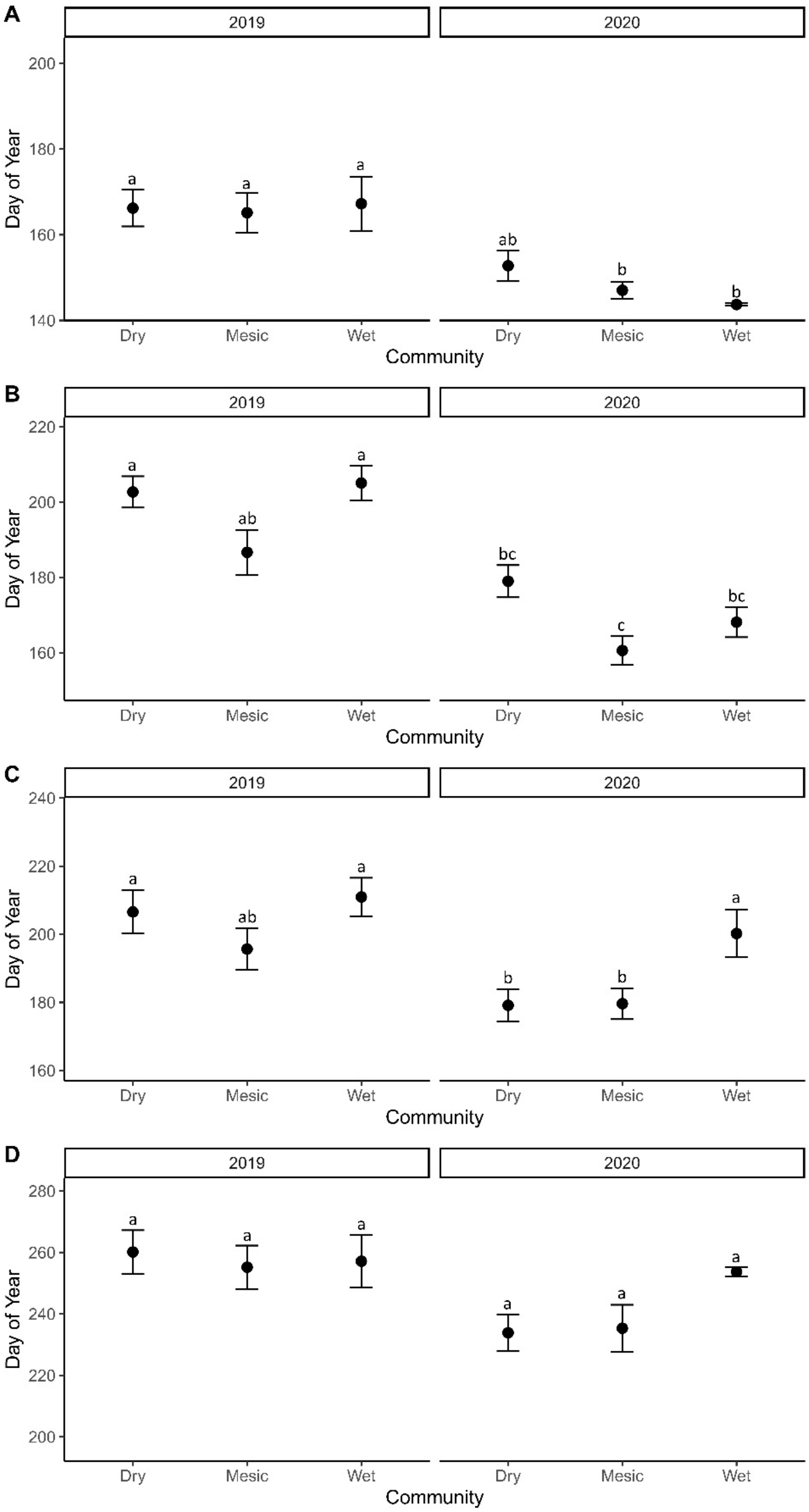

Figure 6. Mean (± SE) phenology values across vegetation communities and years. Phenology stages are (**A**) leaf growth (L), (**B**) inflorescence (I), (**C**) seed (S), and (**D**) desiccated (D). Different letters indicate significant differences.

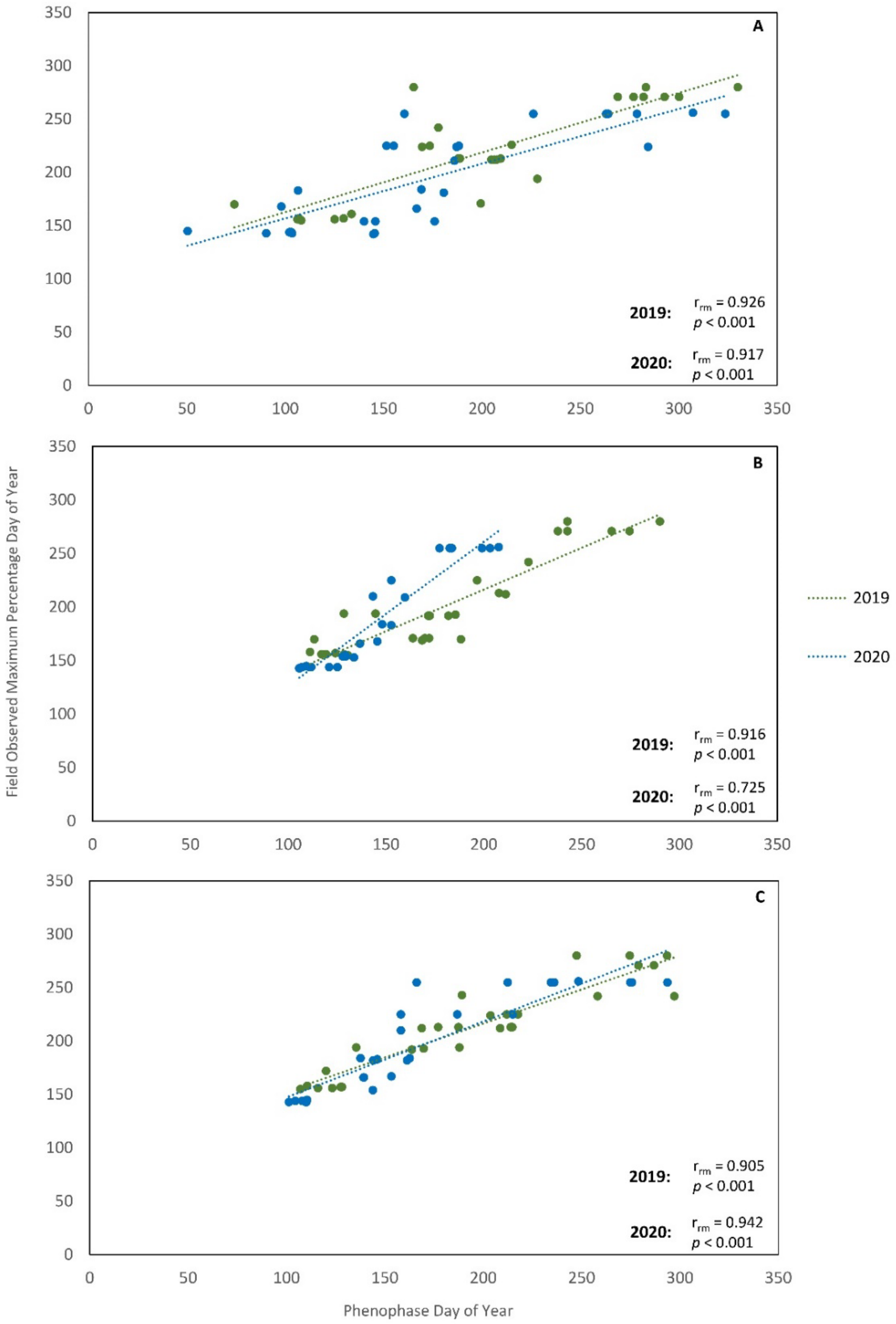

Figure 7. Plotted average day of year (DOY) for maximum percentage of observance for each phenology stage and camera-derived phenophase for (**A**) dry vegetative communities, (**B**) mesic vegetative communities, and (**C**) wet vegetative communities. Repeated measures correlation coefficients (r_{rm}) and p-values were calculated for each year and community type.

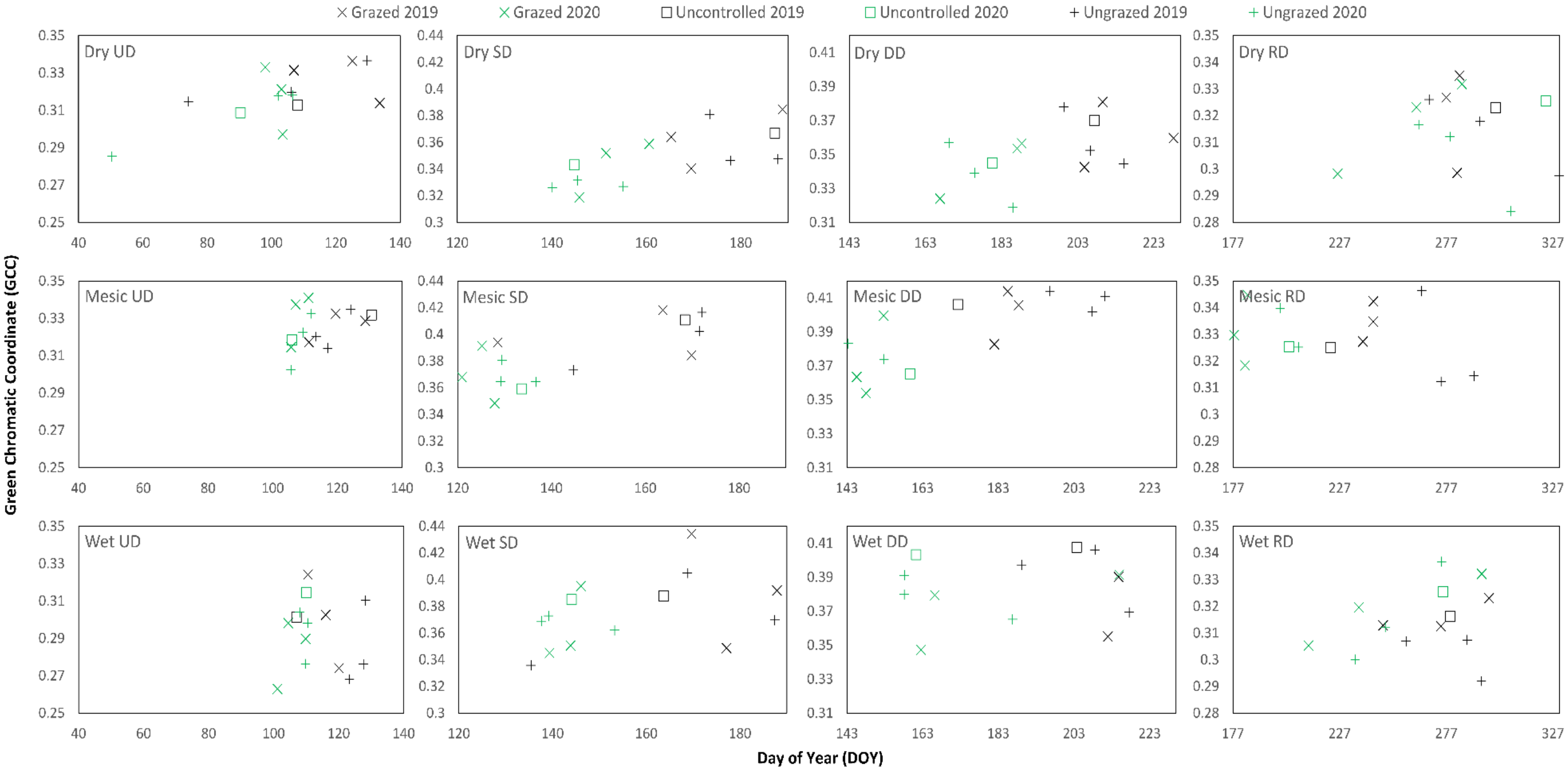

Figure 8. Data for average phenophase day of year values (DOY) in relation to their respective Green Chromatic Coordinate values (GCC) across all meadow treatments. It should be noted that in 2019, all meadows were treated as uncontrolled. In 2020, grazing treatment actually began. Designating meadows as grazed, ungrazed, and uncontrolled in 2019 was done solely to show differences in the groups of meadows across the years. Phenophase stages are upturn date (UD), stabilization date (SD), downturn date (DD), and recession date (RD).

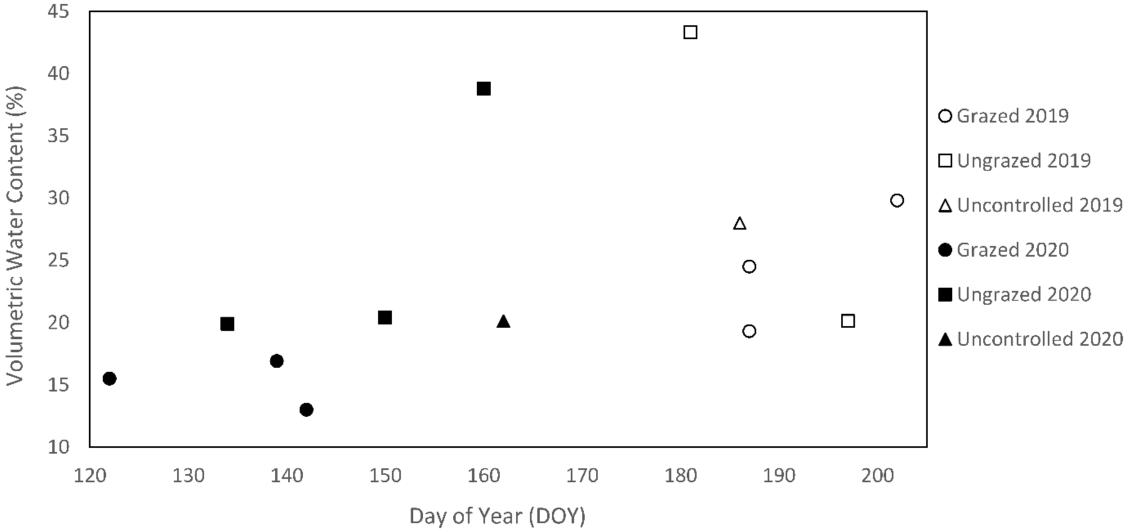

Figure 9. Average recession dates (RD) in relation to their soil volumetric water content (%) values for grazed, ungrazed, and uncontrolled meadows. It should be noted that in 2019, all meadows were treated as uncontrolled. In 2020, grazing treatment actually began. Designating meadows as grazed, ungrazed, and uncontrolled in 2019 was done solely to show differences in the groups of meadows across the years.

CHAPTER 2. Shifts in Sage-Grouse Arthropod Food Sources Across Grazing and Environmental Gradients in Upland Meadow Communities

In review at Arthropod-Plant Interactions

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Abstract

Groundwater dependent systems are extremely important habitats for a wide variety of taxa in the Great Basin of North America. The impacts of grazing on these habitats cause shifts in resources and subsequent change in species composition. The Greater sage-grouse, a keystone species of Great Basin ecosystems, rear brood in these areas during spring and summer months, utilizing forbs and arthropods. To examine the impact of grazing on arthropod abundance in these ecosystems, seven meadows, each made up of three unique vegetative communities, were grazed at three intensities across two years (2019-2020) and monitored for environmental variables and abundance of arthropods during peak sage-grouse utilization periods. Additionally, the relationship of field measurements and near-surface digital cameras (phenocams) was examined to better understand how remote sensing technologies can be used to monitor these insect abundance shifts on larger scales. Arthropod taxa abundance responded differently to grazing management and environmental variables. Coleoptera abundance during peak sage-grouse usage periods had an increase of roughly 40% in some meadows with increased grazing intensity, while Formicidae abundance saw a 22% decrease. For year-to-year environmental variability in precipitation, Lepidoptera abundance was 115% higher in the drier year, while Coleoptera was 64% lower. Near-surface cameras had varied success with predicting peak insect abundance levels. Lepidoptera and Coleoptera capture rates had strong correlations with phenological indices derived from phenocams, while Formicidae had much weaker relationships.

Key Words: grazing management, groundwater dependent ecosystems, phenocam GCC, phenology, plant insect interactions, insect abundance

Introduction

When compared to their actual size, groundwater dependent ecosystems play a very important role in arid and semi-arid ecosystems. (Naiman and Decamps 1997; Fesenmeyer et al. 2018). An example of the importance of these habitats can be found in the Greater sage-grouse (*Centrocercus urophasianus*), a keystone species for many sagebrush ecosystems in the Great Basin of North America. Sage-grouse frequent groundwater dependent meadows during their brood rearing period in the spring and summer. During this crucial time, mesic habitats allow them to utilize the high-quality forage found there, such as forbs, succulent vegetation, and arthropods (Wallestad 1971; Crawford et al. 2004). These ecosystems can be extremely sensitive to disturbance and inter-annual precipitation variability which may lead to declines in ecosystem services (e.g. soil loss, limited forage capacity and water storage) (Reynolds et al. 2007; Zhao et al. 2017). Understanding how these disturbances and climatic variability affect various ecological processes is important for deciding what conservation and restoration practices should be implemented (Hardegree et al. 2012; Hardegree et al. 2016).

In a comprehensive review, Briske et al. (2011) identified the spatial and temporal distribution of cattle and livestock grazing pressure as two of the most impactful factors in conservation outcomes. As the human population increases, the demand for livestock products continues to grow leading to increased forage demand and use of natural resources (Thornton 2010). Many studies have demonstrated that livestock can be managed in such a way to promote and maintain ecosystem services within grazed areas (Matejkova et al. 2003; Marty 2005; Xu et al. 2018). Conversely, clear evidence demonstrates how, when managed improperly, grazing can lead to a degradation of rangeland (Fleischner 1994; Belsky et al. 1999; Pringle et al. 2004). Ultimately, the impact that grazing can have on community productivity is highly dependent on the type of management strategy used by producers, as well as the various climatic and ecological factors that influence resilience (elevation, precipitation, soil, etc.) (Milchunas and Lauenroth 1993; Wardle et al. 2004).

Grazing and other disturbances have the potential to affect the high temporal variability (i.e. phenology) that accompanies resource availability (Browning et al. 2019). Phenology is widely recognized as an extremely sensitive biological response (Parmesan and Yohe 2003; Cleland et al. 2007). Within the Great Basin of North America, the phenology of plant taxa can significantly affect the habitat quality of many species. For example, in the spring and summer both adult and juvenile Greater sage-grouse consume forbs generally found in ground-water dependent zones (Klebenow and Gray 1968; Klebenow 1969; Martin 1970; Wallestad and Eng 1975; Drut et al. 1994). Changes in forb phenology due to grazing may alter the amount of forage available to Greater sage-grouse during this time.

Additionally, a large portion of the Greater sage-grouse chick's diet consists of arthropods, most of which are insects (Crawford 2004; Gregg and Crawford 2009). Arthropods contain nutrients such as crude protein, calcium, and phosphorus crucial for the development of Greater sage-grouse chicks (Barnett and Crawford 1994; Greg et al. 2008). Many studies have reported that Formicidae and Coleoptera are the most common insects in the sage-grouse diet during spring and summer months (Klebenow and Gray 1968; Drut et al. 1994). More recently, Lepidoptera larvae have been shown to be another key food source (Gregg 2006; Ersch 2009). Cattle grazing has the potential to limit insect diversity and shift arthropod community composition to species that are better equipped to handle the disturbance induced by herbivores. In the Mongolian steppe, Clements et al. (2018) demonstrated that intensive grazing regimes can lead to a 30% decrease in the frequency of certain species of Diptera*,* while moderate-to-light grazing can potentially increase the diversity of Diptera species overall. Changes in arthropod community composition could potentially alter fitness and survival of higher trophic levels, such as sage-grouse, that depend on arthropods as food sources.

Other biotic and abiotic factors besides grazing pressure can lead to shifts in insect population size and availability. Habitat accessibility has a significant influence on insect population fitness, and has been proposed as an important drivers of species distribution and richness (Townsend 1989; Morris 2003). In southern India, microhabitat (tree trunks, dead logs, ground, canopy foliage, etc.) altered the distribution of 13 Orthoptera species (Jain and Balakrishnan 2011). Of the 13-cricket species examined, 10 showed 100% selection fidelity for a specific microhabitat. This demonstrates how habitat, on smaller ecological scales, can impact when and where certain types of insects may be found. In groundwater dependent meadows, where multiple plant community types can occur within close proximity, more research is needed to understand how insect abundance changes due to disturbance and environmental gradients, and the potential trophic implications for species that rely on those populations as food sources (Arcoverde et al. 2018).

Studying plant phenology helps researchers understand when resources are available for bird and insect communities. There are many ways to monitor phenology, resulting in a spectrum of both the temporal and spatial inferences that can be made from these studies (Denny et al 2014). These methods range from using satellite remote sensing to individual field observations. The utilization of repeat photography from digital cameras (hereafter referred to as phenocams), is reliable (Klosterman et al. 2014; Melaas et al. 2016), increases time-based resolution, widens the spatial extent of observations, and reduces the costs of phenological monitoring (Sonnentag et al. 2012). While indices derived from phenocams have mostly been applied to plant communities (Piao et al. 2019; Snyder et al. 2019), understanding how these indices correlate to insect abundance levels could help broaden our understanding of insect population shifts on a landscape scale.

This study aimed to: (1) quantify the influence of environmental and grazing variables on the timing of arthropod abundance in high-elevation meadows within the Great Basin, (2) assess the impacts of varying grazing management strategies on total abundance of insect food sources for sage-grouse, in the framework of yearly climatic variability, and (3) ascertain the connection between on-the-ground insect abundance measurements and phenocam data, that provide metrics of plant community growth and greenness, across these meadow communities.

Methods and Materials

Site Description and Treatments

Details of treatment design for this project can be found in Richardson et. al (2021). Briefly, several high elevation meadows, referred to as the Haypress meadow complex, are located in the Desatoya Mountains of the Central Great Basin (39°27’ N, 117°36’ W; 39°19’ N, 117°42’ W). Four meadows were selected for this study and were named based on their relative locations or the grazing intensity associated with them: (1)

"Lower" at 2322 m, (2) "Middle" at 2365 m, (3) "Upper" at 2438 m, and (4) "Uncontrolled" at 2320 m.

The vegetation of the meadows was delineated into three distinct classes: 1) the dry meadow community, which consisted of an overstory of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) and an understory of primarily Douglas' sedge (*Carex douglasii*), 2) the mesic meadow community, which consisted of Douglas' sedge, arctic rush (*Juncus arcticus*), Sandberg bluegrass (*Poa secunda*), and a mixture of meadow forbs (ex. *Chorispora tenella, Symphyotrichum ascendens*), and 3) the wet meadow community, which consisted of Nebraska sedge (*Carex nebrascensis*), arctic rush, and a mixture of facultative wet forbs and grasses (ex. *Montia chamissoi, Veronica Americana, Poa pratensis, Glyceria striata*). Vegetation surveys were conducted in 2018 to verify the differentiation of these groups (Richardson et al. 2021).

Grazing in these meadows consisted of yearlong grazing by feral horses and managed cattle grazing. Haypress livestock grazing occurred in the following manner: for two consecutive years, grazing occurred in May and continued through mid-June; the following two years, use occurred mid-August through September. Livestock use was generally limited to one month, with around 700 cow–calf pairs having access to the meadows but could last up to six weeks depending on the year. In 2019, cattle were brought in on the first half of the grazing rotation (mid-June).

In the winter of 2019, pipe-rail fencing was installed in the complex. The perimeters of the Lower, Middle, and Upper meadows were fenced completely, while the Uncontrolled meadow was not fenced. For the three fenced meadows, cross fencing was constructed which cut each meadow in half. This allowed for a total of seven separate units to be treated and measured (Table 1). In the summer of 2020, cattle were brought into the complex in mid-June. One section of each fenced meadow was grazed such that the major forage species of the ground-water dependent areas had a mean stubble height of roughly 10 cm (Hall and Bryant 1995, approximately 3 days of grazing). The other section of each fenced meadow was left ungrazed, and the Uncontrolled meadow was left unmanaged (allowed for constant grazing by both cattle and horses to occur in the meadow throughout the summer).

Seven phenocams (StarDot NetCam SC 5MP IR-enabled cameras using complementary metal oxide semiconductor (CMOS) image sensors, StarDot Technologies, Buena Park, CA, USA) were installed, one in each of the meadow units. Methods of installation and use for the phenocams were sourced from the PhenoCam Network (https://phenocam.sr.unh.edu/webcam/tools/) (Table 1). Additionally, time-domain reflectometry soil moisture sensors (CS-616, Campbell Scientific, Logan, UT, USA) were installed at depths of 10, 20, 35, and 50 cm at each of the phenocam sites. Daily green chromatic coordinate (GCC) values were derived from the repeat imagery taken from the phenocams, and peak of season (POP) GCC was calculated for 2019-2020 for each community type within each meadow section. The methods used for GCC and POP calculations can be found in Richardson et al. (2021)

Field Methodology

In May of 2019, pitfall traps were randomly installed within the meadow complex. Three traps were installed in each vegetative community type of each meadow section. Traps were constructed using methods from Brown and Mathews (2016). Twelve oz red plastic cups were buried up to their lip at each location and filled to three-quarters full with a mixture of 50% propylene glycol antifreeze and 50% water. Three, 30 cm long, pieces of metallic flashing were installed into the ground running triangularly (roughly at 90-degree angles from one another) from the cup. A cover of plywood was nailed into the ground over the trap (to protect from rain as well as to keep unwanted debris from falling in) using 8” galvanized nails, leaving a 5 cm gap above the trap (Fig. 1).

Every two weeks between the months of May and August, each trap was drained and the contents were stored in 70% ethanol. This process was repeated in 2020, with installation of traps within each meadow subset (Fig. 2). After each visit, samples were sorted into one of eight groups determined by class, order, or family and the number in each group was recorded. The groups used were Formicidae (due to high abundance of ants in each sample, this family was counted separately from its order), Hymenoptera (wasps, bees, etc.), Coleoptera (beetles), Diptera (flies), Lepidoptera (adult and larval moths, butterflies, and skippers) Orthoptera (crickets, grasshoppers, and katydids), Arachnida (spiders, ticks, mites, etc), and Other (this group encompasses any arthropod that did not fall into one of the previously mentioned groups).

The Line-Point Intercept (LPI) method (Herrick et al. 2005) was used to calculate the percent cover of the wet, mesic, and dry subregions of each of the seven meadow sections in June of 2018 (before the fencing was built) and June 2020 (after the fencing was built). The number of transects, length of transects, and number of points along transects was tailored to each area of vegetation (lengths of transects ranged from 3 to 20 m, number of transects per community ranged from 3 to 9). Monitoring was set up in such a way that at least 100 data points could be generated from each LPI measurement (points ranged from 100 to 120). Additionally, in 2019 and 2020, these same transects were used to monitor vegetation height of the meadows bimonthly. At 25%, 50%, and 75% of the total length of each transect, a 1 m hoop was placed and average height (collected using a fiberglass folding ruler) of each species was recorded. To determine the amount of grazing that occurred in each meadow throughout each year, all phenocam pictures were manually analyzed and the number of cows seen within each meadow area was recorded. Horses were not included in this count due to there being zero horses seen in all meadows except for the uncontrolled, and the number of horses seen via phenocams in that meadow was negligible in comparison to the number of cows seen.

Statistical Analysis

Canonical correspondence analysis (CCA) was used to interpret the relationship between the daily capture rates of the arthropod groups and the various explanatory variables monitored throughout this study (Table 2). Daily capture rates were calculated by taking the amount of each group found in a trap and dividing it by the days since the trap was last emptied. The *vegan* package (Oksanen et al. 2020) in the R statistical program (R Core Team 2013) was utilized for this analysis. The abundance count data was log transformed, and all other variables were scaled. Scaling was performed by dividing each variable’s centered data by their standard deviations. This was done to standardize all variables, giving them a mean of 0 and a variance of 1, and therefore making it easier for the model to compare data. The significance of the variance explained by all explanatory variables was determined using a Monte Carlo permutation test (*anova.cca* function in the *vegan* package). Additionally, significance of the variation explained by each individual explanatory variable was determined by removing variation of all other variables in the model. To correct for multiple testing, a Holm's correction was used.

For key insect food sources of sage-grouse (Coleoptera, Formicidae, and Lepidoptera) (Klebenow and Gray 1968; Drut et al. 1994; Gregg 2006; Ersch 2009), total average trap abundance was calculated, and linear mixed effects regression models were created (*glmmTMB* package in R, Magnusson et al. 2017) to determine the relationship between insect abundances and grazing intensity within the context of vegetative community and year-to-year climatic variability. In these models, meadow section was considered a random effect, and year, grazing intensity (number of cows spotted in the phenocams), and community type were considered fixed effects. Once again, the abundance data were log transformed, and grazing intensity was scaled. The models fit all assumptions of linearity, independence, homoscedasticity, and normality. Coefficients were interpreted using the method from Cornel Statistical Consulting Unit 2012. Tukey pairwise comparison tests were used to understand differences among individual community types across the two years.

To better understand the relationship between insect food source abundance and phenocam metrics, the peak of season day of year (POP DOY) for daily capture rate of Coleoptera, Formicidae, and Lepidoptera across all vegetative communities, meadow sections, and years was plotted against POP DOY derived from the phenocams. Repeated measures correlation coefficients, where meadow section was used as the subject for repeat measure, were determined for each insect group.

Results

Meadow Arthropod Abundance

The study included the wettest year over the past 10 years in the meadow complex (2019, 419 mm annual precipitation) and the driest (2020, 175 mm annual precipitation) as determined from PRISM data (http://climateengine.org (accessed on 10 February 2022), Huntington et al. 2017). Across the two years, 139,660 individual arthropods were collected (Table 3). From 2019 to 2020, Formicidae, Hymenoptera, and Lepidoptera collection numbers increased (20.74, 358.76, and 341.63% respectively), while Arachnida, Coleoptera, Diptera, Orthoptera, and Other collection numbers decreased (6.28, 29.49, 15.02, 3.78, and 1.24% respectively). The total variance explained by the constrained axes in the CCA was 40.8% (Table 4). While less than the total variance explained by the unconstrained axes (59.2%), the effect of the constrained axes was still significant as shown by the Monte Carlo test ($p < 0.01$). When focusing on the first two constrained axes, 29% of the total variance was elucidated (15% and 14% respectively). The variance explained by each individual variable was significant, except for percent bare soil and percent total foliar cover (Table 5).

Relationships between group bi-weekly abundance and environmental/management factors are shown in Fig. 3. Abundance of Coleoptera and Arachnida was positively correlated to grazing intensity, soil moisture, and forb cover, while negatively corelated to year collected, sedge cover, and stubble height. Abundance of Diptera, Orthoptera, and Other groups were positively correlated with day of year and soil temperature. Lepidoptera and Hymenoptera were positively correlated to sedge cover, while negatively correlated to soil moisture. Formicidae was positively correlated to shrub cover.

Insect Food Sources: Grazing Impact and Relationship with Phenocams

In regards to total growing season abundance for insect food sources of sage-grouse, the linear mixed effects models demonstrated that individual groups react differently to year-to-year climatic shifts, grazing intensity, and vegetative community type (Table 6).

In the model, average trap abundance of Coleoptera and Lepidoptera for all meadows saw significant changes from 2019 to 2020. Coleoptera abundance had a 64% decrease, while Lepidoptera abundance had a 115% increase (Table 6). For scaled grazing intensity (SGI), only Formicidae and Coleoptera showed significant changes. Formicidae abundance had a 22% decrease for every unit increase in SGI, while Coleoptera saw a 40% increase. All three groups changed significantly among vegetative community types (Fig. 4). With Formicidae, both mesic and wet communities were lower in overall abundance when compared to dry communities, however, only the change in the mesic community was significant (39% decrease, Table 6). Conversely, both mesic and wet communities had significant increases in total abundance for Coleoptera when compared to dry communities (183% and 63%). Mesic communities had the only significant change from dry communities for Lepidoptera, with an 88% increase. Abundance shifts between community types were not similar between both years studied (Fig. 4). For Coleoptera, the shifts were much greater in 2019, and for Lepidoptera, the shifts were much greater in 2020.

The strength of correlation between insect POP and phenocam POP varied greatly across groups and community types (Fig. 5). All relationships were positive (i.e. an increase in phenocam POP typically was followed by an increase in insect POP), except for Formicidae in the mesic community type. Absolute values of repeat measure correlations (r_{rm}) ranged from 0.128 to 0.844. The four strongest relationships occurred for Coleoptera in the mesic and wet communities (r_{rm} values of 0.819 and 0.844), and Lepidoptera in the dry and wet communities (r_{rm} values of 0.749 and 0.699). Formicidae had no significant relationships.

Discussion

In ground-water dependent ecosystems, arthropods deliver a myriad of supporting processes. Pollination, waste removal, decomposition, and being food sources for species such as sage-grouse are all vital services made available by arthropod communities (Ersch 2009; Mata et al. 2014). The CCA demonstrates how different the reactions of arthropod groups can be to their environment (Fig. 3). A number of studies have established that site-insect interactions vary across species. Threlfall et al. (2015) showed that while vegetation diversity positively impacted bee communities, it had little effect on ground-nesting beetles, which in turn were more affected by impervious soil cover. These fluctuations in arthropod groups illustrate how communities can become imbalanced through ecological events and lead to limited diversity and environmental services. It should be noted that in the CCA performed, the variation explained by percent total foliar cover and percent bare ground was not significantly correlated to arthropod abundance, which may be due to these measurements being too aggregated. While total foliar cover was relatively constant across community types (ranged between 66% and 76%), the type of foliar cover present varied greatly (forb, shrub, sedge, and graminoid). Variation in habitat type has been shown to increase insect diversity and abundance (Thomas and Marshall 1999; Philpott et al. 2014), and this study further reinforced that relationship.

Both Coleoptera and Lepidoptera had higher levels of abundance outside of drier, sagebrush dominated communities. Interestingly, while Coleoptera abundance was higher in both mesic and wet plant communities according to the linear model, Lepidoptera generally was most abundant in only mesic, sedge dominated communities. Research has shown that Lepidoptera tend to pupate in saturated soil that is neither excessively dry or has large amounts of standing water (Shi et al. 2021), and in areas with high levels of herbaceous vegetation cover for larvae to reside and feed on (Kobori and Amano 2003). The mesic communities in these meadows (with the highest vegetation cover, most of which is made up of sedges, and limited standing water) created a unique environment that was well suited for the life cycle of Lepidoptera species.

Both the CCA and the linear mixed effects models demonstrated a high correlation between Formicidae abundance and areas where shrub cover was high (the dry community was dominated by *Artemisia tridentata*). Formicidae abundance in arid rangeland communities has been positively correlated with increased prevalence of shrub communities (Majer 1983; Nash et al. 2000), and negatively correlated with areas associated with seasonal waterlogging (Hoffman 2000; Andersen et al. 2015) due to many ant species nesting underground and being susceptible to flooding. Formicidae are generalists found across a wide range of community types and structures (Lach et al. 2010). In our study, while more ants were found in the dry community type, a larger number of ants were still collected in both the wet and mesic communities than both of the other insect food groups (Fig. 4). Formicidae could still be a viable source of food for sage-grouse in the communities where they may not be as prevalent. More research should be conducted to understand the dietary requirements of sage-grouse, and how insect food sources meet those requirements. While ants were captured 3-4 times more than beetles across the different communities, it is unclear if those populations have a greater dietary impact for sage-grouse due to how much larger many beetles are than ants. In future studies, questions like how does the size of individual species of insect prey correlate to sage grouse utilization and brood rearing success should be asked.

In Haypress, Arthropod abundance shifts were defined in very broad terms. Due to the sheer number of arthropods counted, determination of individual species was not feasible. While this study may be able to define patterns of arthropod groups, it is unclear whether those patterns are applicable to individual species within said groups. Take Formicidae as an example. In a study conducted by Arcoverde et al. (2017), they found that three common ant species, *I. sanquineus*, *Pheidole* sp. 1, and *Monomorium* sp. D., all reacted to grazing differently based on their preference for more open habitat. If all three species were to reside in the same meadow community, grazing or some other disturbance may lead to an overall increase of abundance in Formicidae. However, that increase would not be applicable to all species, but rather it would be indicative of more disturbance resilient species outcompeting less resilient species. Classification to the species level of arthropods collected in Haypress would help to clarify how shifts in arthropod abundance affect sage-grouse health and survival. Certain species that sage-grouse prefer (due to size or other nutritional factors) may remain constant under more intensive grazing regimes and changes in climate, while the overall abundance of the arthropod group could increase or decrease.

Interestingly, all three food source groups were impacted differently by yearly climatic changes. Formicidae trap abundance levels across the collection years remained high, despite a 20% decrease in total abundance from 2019 to 2020. Coleoptera abundance decreased significantly from 2019 to 2020, and Lepidoptera abundance increased. This study included both the wettest year over the past decade in the meadow complex and the driest. Coleoptera (specifically carabids) tend to react positively to wetter winters and springs. A number of studies have shown that carabid abundance increases in areas where abnormally high levels of precipitation and flooding have occurred (Eyre et al. 2006; Lessel et al. 2011). Other research has demonstrated the negative impact frequent rain events can have on Lepidoptera larvae. Increased rainfall has the potential to increase the falling rate of larvae from vegetation, which in turn can lead to higher mortality (Kobori and Amano 2003). Additionally, lower temperatures and higher moisture in the winter (which was seen in 2019 when compared to 2020) can inhibit overwintering pupae survival (Huang and Li 2014).

In regards to grazing impact on insect food source abundance, only Formicidae and Coleoptera saw significant shifts across grazing intensities. Between the two groups, grazing had a much larger effect on Coleoptera. Coleoptera tended to increase with increased grazing pressure. Carabid beetles were one of the most common families seen in the studied meadows. These beetles are thought to be potential indicators for disturbance (Lovei and Sunderland 1996; Niemela et al. 2000). It should be noted that a number of studies have shown that the use of pitfall traps can lead to a mobility bias, where more mobile species (like Coleoptera) are more likely to encounter the traps and be caught (Hancock and Legg 2012; Brown and Matthews 2016). To combat this bias a sufficient number of traps were used, and a sufficient time between collections occurred for an accurate representation of arthropods to be caught. For why Coleoptera increase with increased grazing pressure, it has been theorized that reduced grazing of meadows could lead to increased structural complexity of vegetation, which in turn could impede ground beetle forage ability and ultimately decrease abundance (Smith et al. 2014; Pozsgai et al. 2022). The CCA of this study showed vegetation height having a negative relationship with Coleoptera abundances, giving further credence to this hypothesis. Multiple other studies have demonstrated similar relationships (Nijssen et al. 2001; Sanderson et al. 2020). Suominen et al. (2003) suggested that there may be an optimal level of grazing where Coleoptera abundance would be at its highest. Formicidae, on the other hand, tend to be more reliant on vegetation, and have been shown (in this study and others) to be negatively impacted by more intense grazing regimes (Nash et al. 2000; Ajerrar et al. 2020). Over time, with continued grazing treatments, these changes in abundance may become more prominent.

Shifts in capture rates during peak sage-grouse-usage were accurately mirrored by phenocams across many of the vegetation types and insect groups studied. Phenocam GCC has been shown to be highly correlated to plant production and growth (Knox et al. 2016; Richardson et al. 2021). Groups like Lepidoptera larvae are highly dependent on vegetation for protection and food (Zalucki et al. 2002). Coleoptera and Lepidoptera had strong levels of correlation between their POP and the POP of phenocams in two of the three community types, while Formicidae’s POP correlation was weak across all community types. Formicidae's low level of correlation may have been affected by its capture rate data exhibiting weak or nonexistent peaks. Only a percent of ant colonies is dedicated to foraging. This subset does not represent the total number of individuals in the colony (Lach et al. 2010). If some foraging individuals are killed off due to harsh conditions, there may be many more to replace them without having to wait for a reproductive cycle, potentially leading to their collection rate curves being more uniform than parabolic. Also, it can take several years for some ant colonies to build up enough workers and resources to begin producing reproductives, so their life cycle (as a colony) is much longer that beetles and moths that reproduce once (or more) a year (Lovei and Sutherland 1996; Zalucki et al. 2002). For communities that are more highly correlated, several studies have demonstrated that phenocam derived growth indices and satellite derived growth indices are highly paralleled (Snyder et al. 2019; Richarsdon et al. 2021). Statistical models could be created that extrapolate certain insect phenological metrics to other Great Basin meadow systems using satellite imagery, allowing researchers and producers to have a better understanding of how management choices affect arthropod communities.

Overall, this study shows how we can balance human interests and the preservation of threatened species. Grazing intensity was not ultimately detrimental to insect abundance, and even permitted some insect taxa to thrive. More research can be done to define arthropod-plant relationships in these systems to better understand the mechanisms for community shifts made by grazing. In addition, defining dietary and nutritional contributions from arthropod groups to sage-grouse survivability and health could help researchers better understand what management tools could be used to limit sage-grouse mortality.

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Tables

Site	Camera Viewshed (ha)	Meadow Area (ha)	Lat/Long (Decimal Degrees)	Camera Orientation (Cardinal Direction)
Upper Grazed	0.09	0.64	39.317479/-117.697128	NE
Upper Ungrazed	0.11	0.29	39.315936/-117.698013	NNW
Middle Grazed	0.10	0.73	39.322538/-117.703238	NE
Middle Ungrazed	0.04	0.11	39.320682/-117.703979	NNE
Lower Grazed	0.18	0.56	39.331126/-117.701456	NNE
Lower Ungrazed	0.15	0.66	39.330403/-117.701739	NNW
Uncontrolled	0.12	0.33	39.325874/-117.688641	E

Table 1. Meadow and phenocam information, including the area of the phenocam field of view (i.e., camera viewshed), the areal extent of the meadows, latitude and longitude of the meadows, and the orientation of each phenocam.

Variable	Definition
DOY	Day of year
GCC	Green Chromatic Coordinate value derived from phenocams
Moisture	Soil moisture (Volumetric Water Content %)
Temperature	Air temperature (degrees Celsius)
GrInt	Grazing intensity (Number of cattle spotted in the phenocam photos)
Year	Year of data collection
Height	Height of primary forage species
TotalFoliar	Percent total vegetative cover
BareSoil	Percent total bare soil
Graminoid	Percent total graminoid cover
Forb	Percent total forb cover
Shrub	Percent total shrub cover
Sedge	Percent total sedge cover

Table 2. List of explanatory variables used in the creation of the canonical correspondence analysis (CCA) describing the relationship between arthropod abundance and environmental variables in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

Taxon	2019	2020	% Change
Arachnida	11,292	10,582	- 6.3
Coleoptera	9,419	6,641	- 29.5
Diptera	9,309	7,911	- 15.0
Formicidae	26,891	32,469	20.7
Hymenoptera	565	2,592	358.8
Lepidoptera	233	1,029	341.6
Orthoptera	450	433	- 3.8
Other	9,984	9,860	- 1.2
Total	68143	71517	5.0

Table 3. Total arthropods collected in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. between 2019 and 2020.

Axes	Inertia	Proportion	Rank
Total	0.447	1	
Constrained	0.182	0.408	7
Unconstrained	0.265	0.592	7

Table 4. Inertia and proportion of variance explained for all axes in the canonical correspondence analysis (CCA) describing the relationship between arthropod abundance and environmental variables in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

Variable	Df	Chi-square	F value	Pr (>F)
Year	1	0.008	13.796	<0.05
DOY	1	0.012	22.938	<0.05
Height	1	0.007	9.477	<0.05
GCC	1	0.003	4.877	<0.05
Moisture	1	0.004	7.747	<0.05
Temperature	1	0.004	7.781	<0.05
GrInt	1	0.006	11.042	<0.05
BareSoil	1	0.001	0.838	0.209
Forb	1	0.002	4.029	<0.05
Graminoid	1	0.004	7.170	<0.05
Sedge	1	0.002	4.293	<0.05
Shrub	1	0.009	15.575	<0.05
TotalFoliar	1	0.001	1.931	0.130
Residual	618	0.337		

Table 5. Significance of variables in the canonical correspondence analysis (CCA) describing the relationship between arthropod abundance and environmental variables in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

Insect	Variables	Estimate	Std. Error	z value	Pr (>|z|)
Coleoptera	Intercept	4.878	0.456	10.694	<0.01
	Year 2	-0.497	0.259	-1.923	<0.05
	Grazing Intensity	0.378	0.125	3.022	<0.01
	Mesic Community	1.042	0.241	4.321	<0.01
	Wet Community	0.603	0.241	2.5	<0.05
Formicidae	Intercept	7.141	0.357	19.995	<0.01
	Year 2	-0.326	0.202	-1.609	0.108
	Grazing Intensity	-0.205	0.098	-2.099	<0.05
	Mesic Community	-0.402	0.189	-2.133	<0.05
	Wet Community	-0.285	0.189	-1.509	0.131
Lepidoptera	Intercept	0.628	0.124	1.143	<0.05
	Year 2	0.765	0.312	2.455	<0.01
	Grazing Intensity	-0.034	0.151	-0.229	0.818
	Mesic Community	0.634	0.291	2.183	<0.05
	Wet Community	-0.042	0.291	-0.146	0.884

Table 6. Coefficient estimates, standard errors, z values, and *p* values of variables in linear mixed effects models that show the relationship between sage-grouse insect food source abundance and year-to-year climatic variability, grazing intensity, and vegetative community type in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

Figures

Fig. 1 Example of a pitfall trap located in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

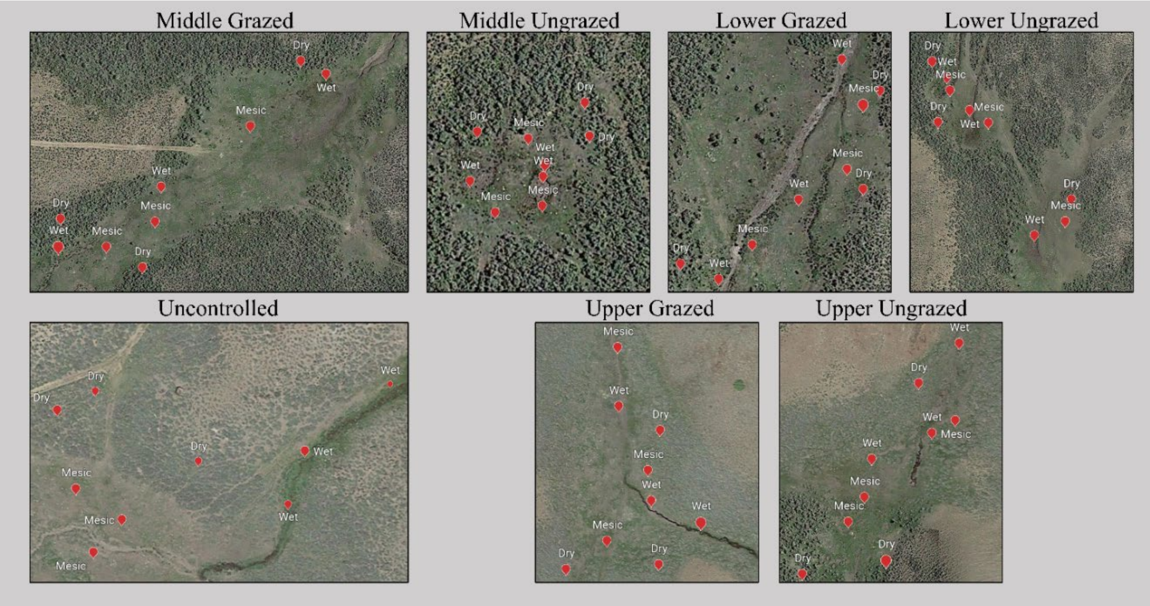

Fig 2. Pitfall trap locations for the seven meadow sections in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. Names of traps correlate to the community type in which they were installed.

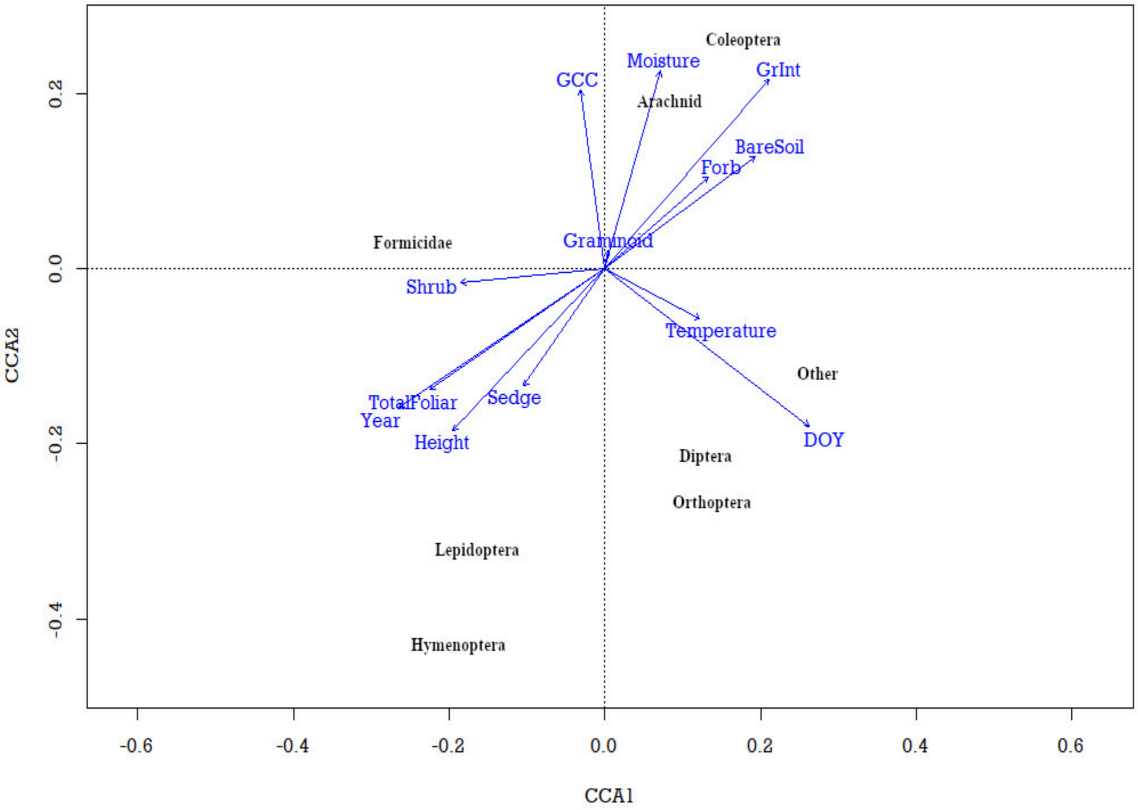

Fig. 3 Canonical correspondence analysis (CCA) describing the relationship between arthropod abundance and variables in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. Influence of variables on arthropod group abundance is represented by blue arrows.

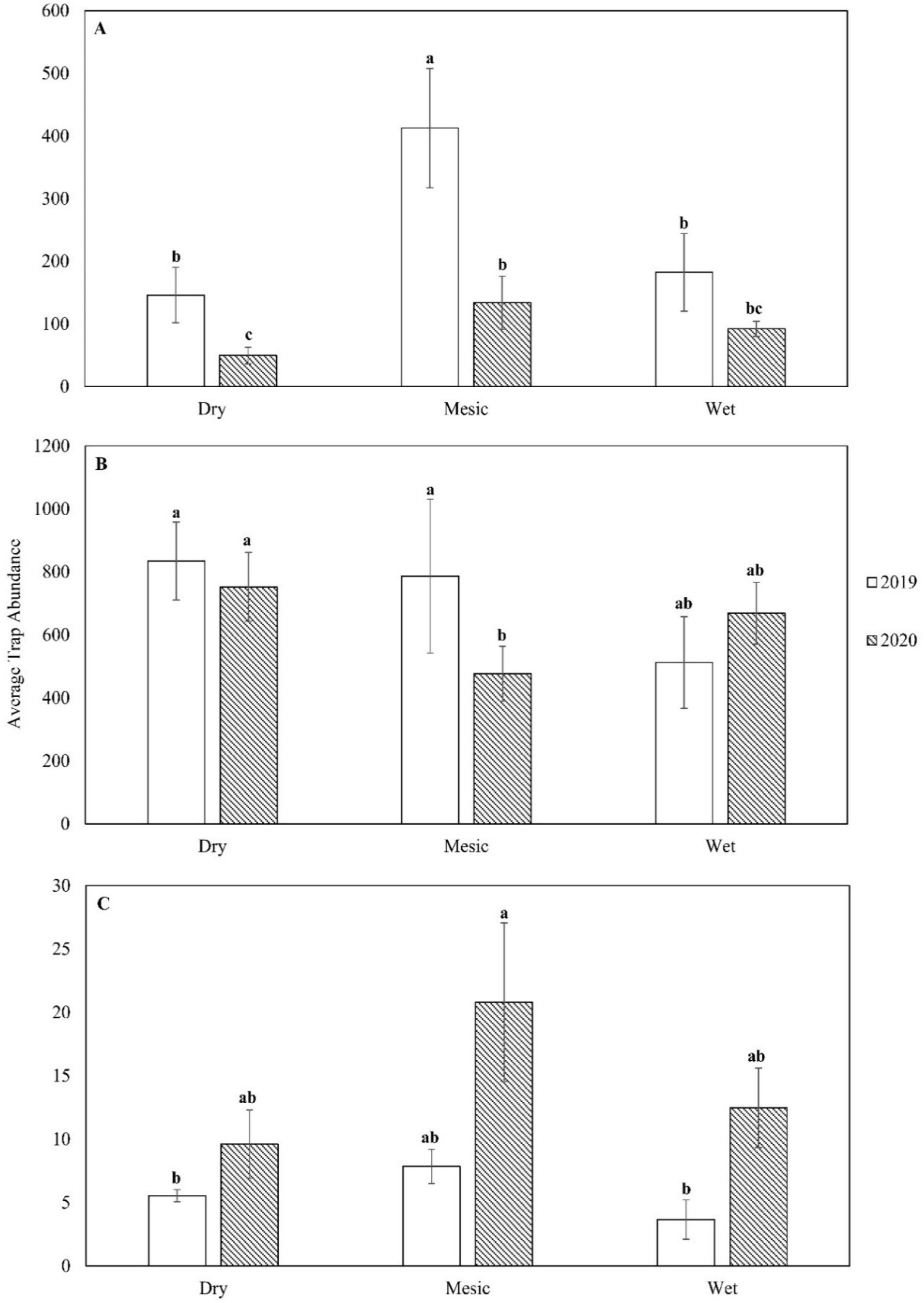

Fig. 4 Average insect trap abundance across all meadows and vegetative community types in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. for A) Coleoptera, B) Formicidae, and C) Lepidoptera. Letters indicate differences among abundances ($p < 0.05$).

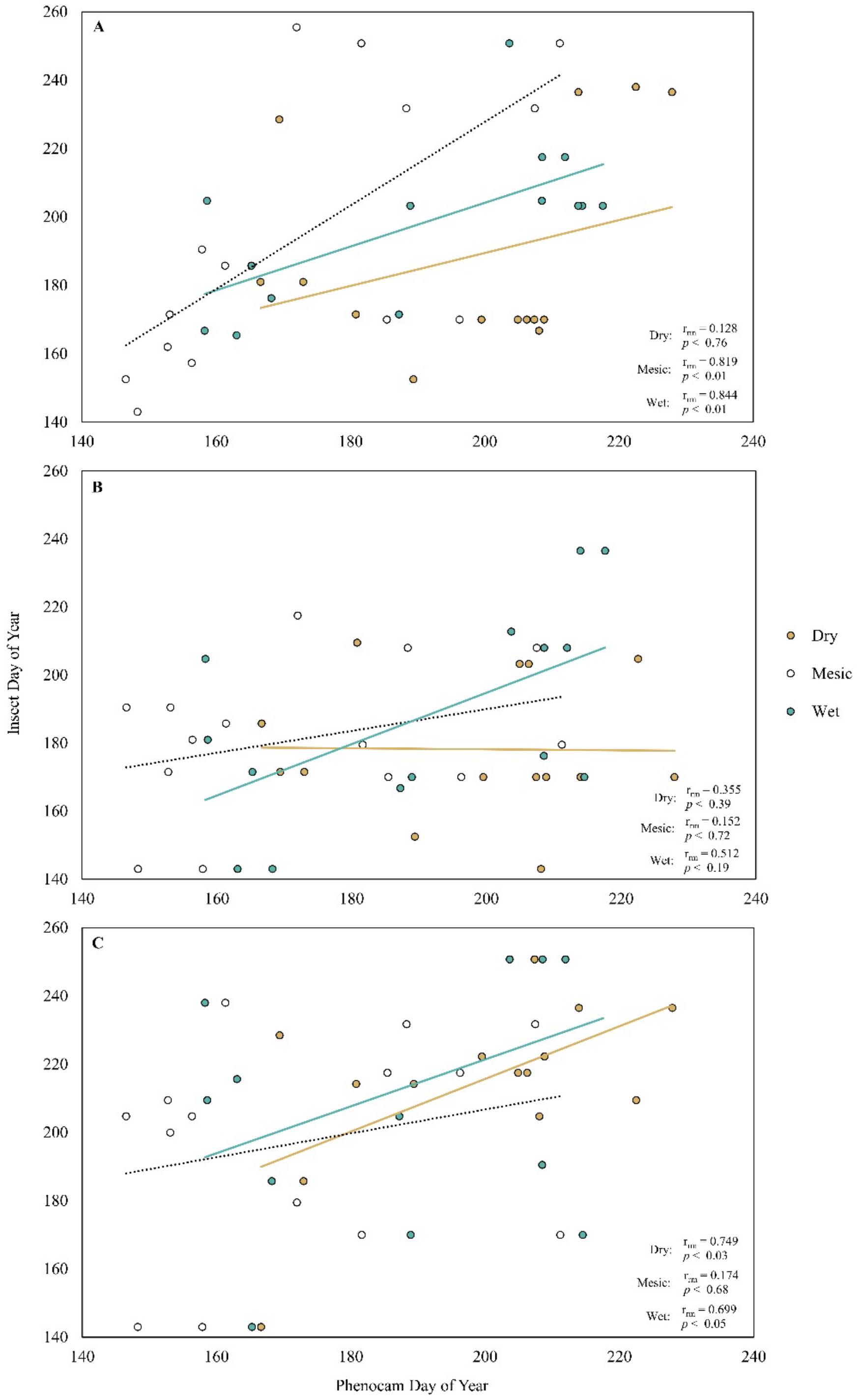

Fig. 5 Insect peak of season day of year (POP DOY) and green chromatic coordinate (GCC) POP DOY derived from phenocams across three unique insect groups, A) Coleoptera, B) Formicidae, and C) Lepidoptera, and three vegetative community types in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. from 2019-2020. Repeated measure correlation coefficients (r_{rm}) and *p*-values were calculated for each individual insect group and community with meadow as the repeated measure.

CHAPTER 3. Comparison of Methods Used for Determining the Impact of Grazing on Synchrony Between Sage-Grouse Usage and Food Source Availability

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Abstract

Over the past 60 years, the Greater sage-grouse has declined in extent and abundance across the West. It has been theorized that nest timing evolved in relation to food accessibility. Scarce mesic resources may lead to an energetic bottleneck for juvenile sage-grouse, limiting fitness and survival rates. Grazing is one such mechanism that may influence the timing and abundance of resources available for sage-grouse. Seven meadow communities were grazed at three intensities across three years (2019-2021) to determine the relationship between grazing management and synchrony of sage-grouse usage and food source availability. Near-surface digital cameras (phenocams) and satellite imagery were used as remote sensing tools for monitoring vegetational grouse food sources. Phenocam Green Chromatic Coordinate (GCC), Phenocam Normalized Difference Vegetation Index (NDVI), and Landsat NDVI were all highly correlated, with slight de-coupling occurring at the end of the growing season. Phenocam GCC was able to detect impacts of grazing, which led to delays in end-of-season recession dates. Sage-grouse usage of the meadows was highly linked to growth seasons of vegetation, with slight decoupling occurring with growth seasons derived from phenocam GCC in drier years. Little correlation was seen between peak sage-grouse usage of the meadows and peak capture rates of arthropods, this was true for all insect groups (Coleoptera, Formicidae, and Lepidoptera).

Keywords: GCC, NDVI, Landsat, sage-grouse, phenology, food source abundance

Introduction

The Greater sage-grouse (*Centrocercus urophasianus*) has commonly been used as an umbrella species for restoration of sagebrush ecosystems, due to its status as an indicator of overall rangeland health (Storch 2007; Hagen 2011). Over the past 60 years, sage-grouse have declined in extent and abundance across the West (Knick et al. 2003). While the extent of this decline has varied across landscapes, the overall decrease has been largely due to sagebrush habitat reduction caused by anthropogenic factors (Gregg et al. 1994; Shroeder and Baydack 2001; Holloran et al. 2005; Hagen 2011). Nielsen et al. (2015) showed that from 1965 to 2015, the decline of male bird attendance at leks was estimated at about 66 percent. Due to this decrease, the U.S. Fish and Wildlife Service (USFWS) has reviewed the species to determine if it required protection under the Endangered Species Act (USFWS 2010) several times. Maintaining and adapting the conservation efforts surrounding sage-grouse will require continuous monitoring and documentation of the effects of habitat restoration and informed management practices on population change and survival (Dahlgren, Guttery, et al. 2016, Stoner et al 2020).

For sage-grouse, brood-rearing periods are generally the most energetically and resource demanding phases of their lives (Martin 1987; Blomberg et al 2017). Sage-grouse chicks hatch in late spring and need access to forbs and arthropods, which are high in crude protein, phosphorous, and other nutrients important for chick development (Barnett and Crawford 1994; Shroeder 1997; Rosentreter 2004; Gregg et al. 2008). In the summer months, both adult and juvenile sage-grouse consume forbs and arthropods (Patterson 1952; Klebenow and Gray 1968; Klebenow 1969, Martin 1970, Peterson 1970, Wallestad and Eng 1975, Pyle 1993, Drut et al. 1994). Studies have shown that sage-grouse hens with dependent broods show strong selection for mesic habitats (Wallestad 1971; Fischer et al. 1996; Atamian et al. 2010; Gibson et al. 2017; Street 2020).

Mesic and groundwater dependent habitats offer high quality forage and succulent vegetation during periods of time when vegetational production in other ecological communities across rangelands may be low due to seasonal drought (Fischer et al. 1996; Donnely et al. 2018). Stoner et al. (2020) demonstrated that sage-grouse have synchronized their most energetically taxing life stages (nesting and brood rearing) with the natural progression of growth (phenology) seen in utilized vegetational communities. Within the confines of their study, sage-grouse nest initiation followed the start of season for plant growth on average by 22 days, and hatch dates fell on or just prior to the peak of the growing season. Declines in ecosystem services (e.g. soil loss, limited forage capacity and water storage) may be seen in mesic habitats due to their sensitivity to disturbance and inter-annual precipitation variability (Reynolds et al. 2007; Zhao et al. 2017). Scarce mesic resources may lead to an energetic bottleneck for juvenile sage-grouse, limiting fitness and survival rates (Blomberg et al. 2012; Donnelly et al. 2016).

Grazing is one such mechanism that may influence the timing and abundance of resources available for sage-grouse. In their 2015 decision, the USFWS did not identify livestock grazing as a conservation threat for sage-grouse (USFWS 2015), however they did identify improper grazing as a potential local threat for the species. Improper grazing is of course very hard to define (Dettenmaier 2018). The impact that grazing can have on community productivity is highly dependent on the type of management strategy used by land owners, as well as the various climatic and ecological factors that influence plant resilience and growth (elevation, precipitation, soil, etc.) (Milchunas and Lauenroth 1993; Wardle and Bardgett 2004).

A number of studies have demonstrated how grazing affects the timing of environmental resources. Han et al. (2015) showed that grazing shifted senescence timing of several observed species, and shrank the overall growing period for the communities in their study. Conversely, Lebon et al. (2014) hypothesized that plant compensation after grazing may occur due to a release of bud dormancy, modification of competition between plants for resources (Crawley 1990; Center et al. 2005), an increase in plants primary metabolism (Dyer et al. 1993; Poveda et al. 2010), increased photosynthesis (Thomson et al. 2003), or soil fertilization by herbivore dejections (DeAngelis 1992). Studies have tried to validate this hypothesis with mixed levels of success (Augustine and McNaughton 2003; Loeser et al. 2004; Austrheim et al. 2014). Ultimately, grazing effects are variable and can change via management choices such as animal type, timing of grazing, stocking rate, duration of grazing, and frequency of use (Heitschmidt & Walker 1996; Briske et al. 2008; Teague et al. 2008; Veblen, Nehring, McGlone, & Ritchie 2015).

There is a growing roster of tools available to researchers for monitoring shifts in plant growth across ecosystems (Denny et al. 2014, Browning et al. 2021). Methods for phenology data collection include using satellite remote sensing and individual plant field observations. An increasingly common tool for remotely sensed growth data is near-surface digital cameras mounted on towers (phenocams) (Zeng et al. 2013; Zhuo et al. 2017; Luo et al. 2018). Phenocams can take multiple images daily, and can fill gaps in time and space missed by satellite and in-situ measurements (Browning, Snyder, &

Herrick 2019). A wide variety of growth metrics have been derived from phenocams, two of the most common being the Normalized Vegetation Difference Index (calculated from phenocams equipped with near-infrared filters) and the Green Chromatic Coordinate (calculated from extracted digital numbers from the red, green, and blue bands) (Filippa et al. 2016; Diao & Li 2022). Generally, NDVI is associated with changes in leaf structure (i.e. leaf area), whereas VIs associated with RGB band lengths, like GCC, are more associated with changes in leaf color (Filippa et al. 2018). Understanding the relationships between these phenocam metrics and metrics obtained from satellite data would help to clarify the limitations and strengths of each remote sensing product as a monitoring tool. (Browning et al. 2015; Brown et al. 2016; Morisette et al. 2021). Few studies have compared phenocam NDVI or GCC to satellite NDVI in heterogenous mesic communities, especially within the context of grazing pressure gradients (Snyder et al. 2019; Richardson et al. 2021; Snyder et al [IN REVIEW]).

This study aims to (1) establish the relationship between phenocam GCC/NDVI and satellite NDVI phenology metrics across several heavily grazed upland meadows within the Great Basin, (2) verify seasonal movement and phenology of greater sage-grouse, and correlate their need of certain food sources with the timing of when those food sources are most available, and (3) discover how changes in phenology due to climatic and disturbance related variables may create dissonance between forage timing and need.

Methods and Materials

Site Description and Treatments

Details of treatment design for this project can be found in Richardson et. al (2021). Briefly, several high elevation meadows, referred to as the Haypress meadow complex, are located in the Desatoya Mountains of the Central Great Basin (39°27’ N, 117°36’ W; 39°19’ N, 117°42’ W) (Fig. 1). Four meadows were selected for this study and were named based on their relative locations or the grazing intensity associated with them: (1) "Lower" at 2322 m, (2) "Middle" at 2365 m, (3) "Upper" at 2438 m, and (4) "Uncontrolled" at 2320 m. Vegetation communities within the meadows ranged from dry communities dominated by mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) to mesic/wet communities dominated by a mixture of graminoids and forbs (*Carex douglasii, Carex nebrascensi, Veronica Americana, Poa pratensis)* (Richardson et al. 2021).

Grazing in these meadows consisted of yearlong grazing by feral horses and managed cattle grazing. Haypress livestock grazing occurred in the following manner: for two consecutive years, grazing occurred in May and continued through mid-June; the following two years, use occurred mid-August through September. Livestock use was generally limited to one month, with around 700 cow–calf pairs having access to the meadows but could last up to six weeks depending on the year. In 2019, cattle were brought in on the first half of the grazing rotation (mid-June).

In the winter of 2019, pipe-rail fencing was installed in the complex. The perimeters of the Lower, Middle, and Upper meadows were fenced completely, while the Uncontrolled meadow was not fenced. For the three fenced meadows, cross fencing was constructed which cut each meadow in half. This allowed for a total of seven separate units to be treated and measured (Table 1). In the summers of 2020 - 2021, cattle were brought into the complex in mid-June. One section of each fenced meadow was grazed such that the major forage species of the ground-water dependent areas had a mean stubble height of roughly 10 cm (Hall and Bryant 1995, approximately 3 days of grazing). The other section of each fenced meadow was left ungrazed, and the Uncontrolled meadow was left unmanaged (allowed for constant grazing by both cattle and horses to occur in the meadow throughout the summer). It should be noted that sage-grouse were able to get past the fencing and access all meadows.

Phenocam and Landsat Methodology

Seven phenocams (StarDot NetCam SC 5MP IR-enabled cameras using complementary metal oxide semiconductor (CMOS) image sensors, StarDot Technologies, Buena Park, CA, USA) were installed, one in each of the meadow units. Methods of installation and use for the phenocams were sourced from the PhenoCam Network (https://phenocam.sr.unh.edu/webcam/tools/). Eight RGB images and eight RGB + Near Infrared (NIR) images at a resolution of 1296 X 960 pixels were taken daily between 12:00 pm and 3:30 pm. These sets of images were taken 30 s apart to allow for the movement of the camera's mechanical filter. All data and images were downloaded monthly during the growing season, and once every three months during the winter.

Phenopix, the R statistical program package (Filippa et al. 2016), was used for the calculation of all phenological metrics derived from the phenocam images from 2019-2021. The package allowed Regions of Interest (ROIs) to be manually defined that were representative of each mesic/wet area of the meadows (Fig. 2). These areas are where the majority of forbs and herbaceous material are located, which as discussed previously, are of extreme importance to both juvenile and adult sage-grouse in the spring/summer months. The average digital numbers (DN), which ranged from 0 to 255 and were extracted for red (RDN), green (GDN), and blue (BDN), of the pixels within each ROI, were converted into green chromatic coordinate (GCC) values, a relative percent index, using the following equation:

$$GCC = GDN/((RDN+GDN+BDN)) \quad (1)$$

GCC values were filtered using the night, spline, and max filter options in the *phenopix* package. An additional novel filter using the blue chromatic coordinate was added to remove values influenced by snow (see Snyder et al. (2019) for more details). A method proposed by Gu et al. (2009) was used to apply a double logistic fit to the filtered sub daily GCC values. One thousand replications were used for an uncertainty analysis to determine how well predicted values fit observed. The Gu et al. (2009) threshold method calculates four phenophase dates for each fitted set of data: an upturn date (UD) when GCC of vegetation begins to increase consistently, a stabilization date (SD) when vegetation approaches maximum GCC, a downturn date (DD) when GCC starts to consistently diminish, and a recession date (RD) when vegetation reaches a seasonal low.

The same ROIs were used to calculate NDVI from the phenocams. NDVI was determined from Equation (2) (Petach et al. 2014):

$$NDVI = ((NIR-Red))/((NIR+Red)) \quad (2)$$

NIR is the near infrared value and Red is the RDN value. Exposure values were extracted from each image and the DNs from the images were adjusted by their exposure with Equation (3):

$$DN_A = DN/\sqrt{E} \qquad (3)$$

DN_A is the new DN value and E is the exposure time. Petach et al. (2014) goes into much greater detail on the exposure adjustment for the StarDot phenocams. Furthermore, if an RGB image had a lower exposure value than their corresponding RGB + NIR image, those images were deleted from the dataset (Snyder et al. 2019). The adjusted NIR DN value of each image was extracted using Equation (4) (Petach et al. 2014):

$$DN_{NIR\_A} = DN_{(RGB + NIR)\_A} - (0.30 \times DN_{R\_A} + 0.59 \times DN_{G\_A} + 0.11 \times DN_{B\_A}) \qquad (4)$$

DN_{NIR_A} is the adjusted NIR DN value, $DN_{(RGB + NIR)\_A}$ is the adjusted RGB + NIR DN, and DN_{R_A}, DN_{G_A}, and DN_{B_A} are the adjusted red, green, and blue DN values (Snyder et al. 2019). The majority of the phenocam NDVI values calculated were negative, due to phenocams providing DN, not actual reflectance values (Liu et al. 2017). Similar to phenocam GCC, methods from Gu et al. (2009) were used to calculate NDVI UD, SD, DD, and RD from each camera and for each year.

Landsat NDVI data were extracted from Landsat 8 Optical Imager (OLI) satellite images using the Climate Engine (http://climateengine.org) (Huntington et al. 2016). Polygons were created for each meadow section and NDVI was determined for 2019-2021. The polygons created were large enough to encompass all the meadow, and were at least one pixel in size (30m X 30m). Climate Engine applied a cloud mask to the Landsat top of atmosphere (TOA) reflectance data. The cloud masking removed medium and high confidence snow, shadow, and cirrus clouds using the Quality Assessment Band provided in the Landsat Google Earth Engine (GEE) collection.

To determine the amount of grazing that occurred in each meadow throughout each year, all phenocam pictures were manually analyzed and the number of cows seen within each meadow area was recorded. Horses were not included in this count due to there being zero horses seen in all meadows except for the uncontrolled, and the number of horses seen via phenocams in that meadow was negligible in comparison to the number of cows seen.

Field Insect Methodology

In May of 2019, pitfall traps were randomly installed within the meadow complex. Three traps were installed in each vegetative community type of each meadow section. Traps were constructed using methods from Brown and Mathews (2016). Twelve oz red plastic cups were buried up to their lip at each location and filled to three-quarters full with a mixture of 50% propylene glycol antifreeze and 50% water. Three, 30 cm long, pieces of metallic flashing were installed into the ground running triangularly (roughly at 90-degree angles from one another) from the cup. A cover of plywood was nailed into the ground over the trap (to protect from rain as well as to keep unwanted debris from falling in) using 8” galvanized nails, leaving a 5 cm gap above the trap.

Every two weeks between the months of May and August, each trap was drained and the contents were stored in 70% ethanol. This process was repeated in 2020 and 2021, with installation of traps within each meadow subset. After each visit, the number of Lepidoptera, Coleoptera, and Formicidae were counted from each sample (these three groups were chosen based on their common usage by sage-grouse as a food source, Gregg 2006; Ersch 2009). Daily capture rates were calculated by taking the amount of each group found in a trap and dividing it by the days since the trap was last emptied. Date of highest capture for each group was calculated for 2019-2021.

Greater Sage-Grouse Monitoring

Intensive on-the-ground monitoring of sage-grouse movement and survivorship was conducted throughout the years of 2019-2021. All telemetry procedures were conducted according to the USGS sage-grouse telemetry protocol (USGS, 2015). Field crews located Greater Sage-grouse and attached GPS 'backpacks'. GPS locations updated automatically 3 to 5 times per day and did not require field crews to rediscover them to collect the locations. In 2021, due to complications with aging batteries, data was recorded sparsely and was not updated past July of that year.

The GPS data of sage-grouse locations was plotted alongside the meadows (ArcGIS Online platform, ESRI, Redlands, CA, USA), and 150 m buffers were created around each meadow subsection. All GPS data that fell within the buffers was considered points of time when sage-grouse could potentially move in and use meadow resources.

Day of year when peak number of GPS data points were within the 150 m buffer was calculated for both 2019 and 2020, 2021 was not included.

Statistical Methodology

To create a comprehensive picture of the effects of grazing pressure on in-situ vegetation metrics, repeated measures correlation coefficients, where meadow section across years was used as the subject for repeat measure, were determined for the interaction between phenophase dates (UD, SD, DD, and RD) and grazing intensity (number of cows spotted in the phenocams) across both phenocam GCC and phenocam NDVI.

To understand the relationships between phenocam GCC/NDVI and Landsat NDVI, root mean square error (RMSE) and Pearson's *r* were used to determine levels of correlation between Landsat NDVI and both phenocam GCC and phenocam NDVI for 2019 - 2021. A linear mixed effects regression model was created (*glmmTMB* package in R, Magnusson et al. 2017) to determine the relationship between Pearson's *r* and year-to-year climatic variability. In this model, meadow section was considered a random effect, and year, phenocam metric type (whether Landsat NDVI was compared to phenocam GCC or NDVI), and the interaction between phenocam metric type and year were fixed effects. Additionally, phenocam (both GCC and NDVI) phenophase dates (UD, SD, DD, and RD) were plotted against Landsat phenophase dates. Repeated measures correlation coefficients, where meadow section across years was used as the subject for repeat measure, were calculated for each phenophase of each interaction.

The period of time between the UD and RD was considered the length of growing season (LOS) for each respective meadow. The ratio of GPS data points that occurred within the growing season of each meadow and the total number of GPS data points in the vicinity of each meadow was calculated for 2019-2020. This metric showed what percentage of the usage of the meadows occurred during the growing season of the mesic resources. Additionally, the day of year when peak number of GPS data points were within the buffers was subtracted from peak insect capture rates for all insect groups for 2019-2020 to create a metric which showed the variation between peak grouse usage and peak insect availability. The reason for why a similar metric to what was done with vegetation (calculating UD and RDs and determining number of GPS points within season) was not repeated for the insects collected was because UD and RD dates were not able to be accurately created for insect capture rates. This was due in large part to only having 8 dates of capture for each sub-region, as well as individual insect groups not following perfectly parabolic rates of capture (see Chapter 2).

Two mixed effects regression models were created to determine relationships between differences in time of sage-grouse usage and food sources, and grazing intensity within the framework of climatic variability. The first model had a binomial distribution and used the log-odds (log-transformed odds ratio) of sage-grouse residing within the buffers during the vegetational growing season as the response variable. Meadow section was used as a random effect, and year, grazing intensity, remote sensed metric type (whether Landsat NDVI, phenocam GCC, or phenocam NDVI was used to calculate UDs and RDs), and the interaction between remote sensed metric type and year as fixed effects. The second model was a linear mixed effects regression model that followed a very similar structure. The only alterations were that the difference between peak grouse usage and peak insect capture was used as the response variable, and remote sensed metric type was replaced with insect group in the fixed effects. All linear models fit assumptions of linearity, independence, homoscedasticity, and normality.

Results

Grazing Effects

Precipitation varied greatly between the years of the study (http://climateengine.org (accessed on 1 September 2022), Huntington et al. 2017). Annual precipitation in 2019 was 419 mm, in 2020 it was 175 mm, and in 2021 it was 356 mm. Phenophase dates (upturn date – UD, stabilization date – SD, downturn date – DD, and recession date – RD) for both phenocam GCC and phenocam NDVI followed the trend of being later in the season when wetter years occurred. For UD, SD, and DD, the average difference between phenocam GCC and phenocam NDVI across all years never exceeded 16 days. However, the average RD for phenocam NDVI across all years was 40 days higher than the RD for phenocam GCC. The difference between RD dates for phenocam GCC and phenocam NDVI was greater in drier years.

The strength of correlation between phenocam day of year (DOY) and grazing intensity was generally very low across all phenophases and phenocam metrics (Figure 3). The only correlation to be significant was for RD derived from phenocam GCC (0.603). All relationships were positive or uniform, and besides RD for phenocam GCC, they ranged from 0.113-0.312. The relationship between phenocam GCC and grazing intensity was stronger than the relationship between phenocam NDVI and grazing intensity for all phenophases, except SD.

Remote Sensing Tool Relationships

All Pearson's *r* correlations between Landsat NDVI and phenocam GCC/NDVI remained above 0.74, with both the highest (2019 – 0.99) and lowest (2021 – 0.74) values being between Landsat NDVI and phenocam NDVI (Figure 4). The linear mixed effects model performed for the correlations demonstrated that yearly differences did occur both overall and when interacting with metric type (Table 1). On average, the correlations in 2020 were 0.05 less than correlations in 2019, and the correlations in 2021 were 0.02 less. Phenocam NDVI in 2021 had the lowest average correlation value (0.87), but also had one of the largest ranges (0.74 – 0.94).

When looking at the phenophase dates derived from the remote sensing tools, the relationships between Landsat NDVI and phenocam GCC/NDVI varied significantly (Figure 5). All repeat measure correlations stayed above 0.6, except for the correlation between Landsat NDVI RD and both phenocam GCC and NDVI RD (0.47 and 0.29 respectively). Across all phenophases, the correlation between Landsat NDVI dates and phenocam GCC dates were higher than the correlation between Landsat NDVI dates and phenocam NDVI dates. The highest level of correlation for both phenocam GCC and phenocam NDVI occurred for the SD phenophase (0.847 and 0.809 respectively).

Sage-Grouse Usage and Food Source Availability

Throughout the three years of study, 6 individual birds were equipped with GPS units, with three frequenting the Haypress meadow complex. The range and frequency of when sage-grouse frequented each meadow varied greatly (Figure 6). In both years, the most GPS data points occurred around the middle meadows. Over 100 data points were recorded within the buffer of the middle meadows for each year. All other meadows never had more than 40 GPS points for any given year. It should also be noted that in 2020, the drier of the two years, birds tended to frequent the meadows earlier than in 2019. In 2019, the first GPS data point to be picked up in the vicinity of the meadows occurred in May, while in 2020 the first GPS data point occurred in March (Figure 6). Grouse visited every meadow for both 2019 and 2020.

In the mixed effects model that looked at the log-odds of sage-grouse visiting meadows during the growth season of vegetation, the interaction between remote sensed metric type and year had the largest effect size (Table 2). Neither grazing intensity nor year by itself had significant effects on the response variable. When the ratios of GPS data points that fell within growing seasons and the total GPS data points in the vicinity of the meadows was plotted, ratios calculated from phenocam GCC growing seasons in 2020 had the lowest average across all remote sensed metrics and years (Figure 7). The average ratio across all metric types and years was 0.61, meaning that more often than not, grouse visited the meadows during the vegetational growing season. The range of ratios was from 0 to 1, meaning that there were meadows that never had grouse visit them during the growing season. The majority of GPS data points that were outside of the growing seasons occurred after the RDs, not before the UDs.

On average, insect peak of season occurred earlier than sage-grouse peak usage for Coleoptera, Formicidae, and Lepidoptera (Figure 8). Differences ranged from -150 (Formicidae, 2019) to 116 (Lepidoptera, 2020). Values changed very little between years (Figure 7). These observations were mimicked in the linear mixed effects model, with none of the variables used having significant or large effects (year, insect type, grazing intensity, or the interaction between year and insect type) (Table 3).

Discussion

It has been theorized that nest timing evolved in relation to food resource availability and abundance (Dunn 2004). Stoner et al. (2020) demonstrated that female sage-grouse matched the energy needs of chicks with periods of time when sources of protein were readily available (they used the window between start and peak of season from Moderate-Resolution Imaging Spectroradiometer [MODIS] sensor NDVI data). Models have shown a correlation between vegetation green-up and sufficient accumulation of precipitation followed by warm temperatures (Post et al. 2022). Due to climate change and human impacts, the timing of green-up may become more variable in the on-coming years. This current study, aimed to further clarify the level of synchronization that occurs between sage-grouse usage and food resources in mesic habitats across climatic and grazing gradients. While understanding how climatic and disturbance pressures affect timing of food resources for sage-grouse in critical stages (nesting and brood-rearing) of their development would be better for the future of monitoring grouse population fitness and survival, the current study still sheds light on how those resources may change under different management strategies and how those changes may lead to desynchronization with overall sage-grouse usage.

Grazing Impacts on Phenology

When looking at grazing impacts on vegetation phenology in this study, both phenocam GCC and NDVI saw very little change in early phenophase dates (UD – upturn dates, SD – stabilization dates, and DD – downturn dates) within the context of grazing pressure. Only phenocam GCC saw a strong positive correlation between grazing intensity and recession dates (RDs). Grazing effects on plant phenology have been shown to be highly variable and dependent on intensity of grazing, timing of grazing, and the ecological condition of the initial site. Richardson et al (2021) hypothesized that short term delays in RDs due to grazing in the Haypress meadow complex may partially be explained by soil moisture being retained in the soil later in the season. Zhang et al. (2019) observed that short term grazing led to reduced transpiration in vegetation in high alpine meadows. As grazing continued at unsustainable rates, bare-ground increased leading to higher evaporation and limited soil moisture later in the season. Other studies have linked chronic grazing to shorter growing seasons, as well as limiting the number of flowering plants in a system (Han et al. 2014; Balata et al. 2022). However, proper grazing management has been associated with the maintenance of light and space availability (limiting litter build up and controlling dominant species), leading to increased productivity and length of growing season (Helman et al. 2015; Charles et al. 2017). Snyder et al. (*IN PRESS*) demonstrated that when compared to less grazed meadows in the Desatoya mountain range, the Haypress meadows tended to have lower integrated GCC and GPP. In the Haypress meadow system, initial delays in RD from sustained grazing pressure, may not continue if the same levels of grazing intensity remain.

For why grazing pressure shifts were seen by phenocam GCC and not phenocam NDVI, it should be noted that RDs from phenocam NDVI were much later than RDs from phenocam GCC. Several studies have shown that phenocam NDVI lagging behind phenocam GCC is due to GCC being more sensitive to shifts in the ratio of dry/green biomass, while NDVI responds more to variations in leaf area index (Fillipa et al. 2018; Luo et al. 2018). If the hypothesis of Richardson et al. (2021) is true, and delays in RD within the Haypress complex are because of vegetation having access to soil moisture for longer (delaying plant browning), then it would make sense that vegetation indices more sensitive to dry biomass would pick up differences created by grazing.

Relationships Between Metrics and How They Synchronize with Grouse Usage

Landsat imagery is a readily available and cheap tool that can be used for monitoring shifts in phenology. This study showed high levels of correlation between Landsat NDVI and both phenocam NDVI/GCC (Figure 5), demonstrating its effectiveness of mirroring what is being observed on the ground. It should be noted of course, that the correlation between Landsat phenophase dates and phenocam phenophase dates was lowest when looking at RDs. Multiple studies have shown that Landsat end of season metrics tend to decouple from on the ground end of season metrics (Snyder et al 2012; Snyder et al. 2019). It has been hypothesized that the stronger agreement between satellite and phenocam indices earlier on in the growing season is due to spring leaf out occurring more rapidly and homogenously across the landscape, while fall senescence and changes in leaf color occur more gradually, therefore the onset of spring is more easily (and consistently) detected than the onset of autumn (Klosterman et al. 2014; Snyder et al. 2019; Zhang et al 2018). Overall, these relationships between Landsat NDVI and phenocam GCC/NDVI show how using more than one remote sensing tool is key to understanding all shifts that may occur in a growing season across multiple years of study.

In this study, the correlation between phenocam GCC and Landsat NDVI was higher than the correlation between phenocam NDVI and Landsat NDVI across all phenophase dates (Figure 4). One reason why this might be the case is due to how phenocam GCC went through a more rigorous filtering process than phenocam NDVI. Snow images were not removed from phenocam NDVI datasets, as well as pictures with exposure rates that indicated significant degradation of raw image quality (above 1600, Snyder et al. 2019). These processes could have led to higher correlation of phenophase dates between Landsat NDVI and phenocam NDVI. However, it should be noted that correlations between actual Landsat NDVI and phenocam NDVI values were not significantly different from Pearson correlations of actual Landsat NDVI and phenocam GCC values (which was the initial reasoning for the level of filtering that did occur for phenocam NDVI).

UD and RD were used as metrics for the beginning and ending of the growing season because of how they signify the ending and beginning of stagnation in both NDVI and GCC (see supplementary Figure S1 for more detail). The binomial mixed effects model demonstrated no real impact of grazing intensity on synchrony between grouse usage and mesic meadow community growing seasons, indicating that more than two years of data will be needed to see if any effects can be observed. Overall, it is not clear if the birds preferred fenced areas over non-fenced areas. While all GPS location data points did not occur during the growing seasons of the meadows, the majority did. In the drier year, grouse tended to be in the vicinity of the meadows earlier, demonstrating that they were able to mirror shifts in peak food source intervals created from climatic variability demonstrated in Chapter 1 (drier years led to earlier seasons of growth). It should be noted that the ROIs from the phenocams only covered the mesic areas of the meadows (not sagebrush dominated areas). Sage-grouse that frequented the meadows later in the year would still have access to sagebrush as a food source, however those resources unique to mesic communities would not be available. Interestingly, the major difference seen in synchrony between grouse usage and mesic meadow community growing seasons was between phenocam GCC and the other remote sensing tools in 2020. It has already been established that GCC is more responsive to variation in dry/green biomass. For the drier year (2020), the meadows ‘browned’ out very quickly, leading to NDVI RD lagging more severely behind GCC RD. What this means for sage-grouse is that for drier years, birds that frequent meadows later in the season may still have access to mesic resources (higher NDVI) but the quality of said resources may be much less than what would be found in wetter years (lower GCC).

When looking at the differences between peak arthropod capture dates and peak sage-grouse usage dates, significant variation between year, grazing intensity, and insect type did not occur. Peak arthropod capture dates on their own were highly inconsistent. Across all years and meadows, peak capture dates never fully lined up for all three insect groups (see Chapter 2). Differences in peak capture dates in a single meadow could be as high as 80 days across insect groups. This lends credence to the idea that not all insect populations follow the same life history patterns, and throughout the growing season grouse may focus on different species based on what is most available or abundant at the time.

Overall, this study demonstrates the complex interactions that occur within high elevation meadows across different trophic levels. Admittedly, two years of observational grouse data is not enough to fully understand how grazing and climatic impact affect food source availability in the Haypress meadow complex, let alone across larger landscapes. However, it does give researchers insight into the direction these systems can move toward. Monitoring the phenological stages of herbaceous vegetation in mesic communities across the Great Basin of the North America may be one way in which producers can model estimates of nest and brood initiation with sage-grouse. This study strengthens that argument by showing how sage-grouse do mirror shifts in mesic vegetation availability.

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Tables

Variable	Estimate	Std. Error	z value	Pr (>|z|)
Intercept	0.945	0.021	43.25	<0.01
Year 2020	-0.053	0.025	-2.11	0.034
Year 2021	-0.026	0.025	-1.05	0.292
Phenocam NDVI	0.02	0.025	0.8	0.421
Year 2020: Phenocam NDVI	-0.005	0.035	-0.14	0.888
Year 2021: Phenocam NDVI	-0.066	0.035	-1.85	0.064

Table 1. Coefficient estimates, standard errors, z values, and *p* values of variables in a linear mixed effects model that shows the relationship between the Pearson's *r* correlation between Landsat NDVI values and phenocam GCC/NDVI values, and year-to-year climatic variability, phenocam metric type, and the interaction between phenocam metric type and year, in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A from 2019-2021.

Variables	Estimate	Std. Error	z value	Pr(>|z|)
Intercept	0.028	0.817	0.034	0.972
Phenocam NDVI	0.64	0.239	2.675	<0.01
Landsat NDVI	1.435	0.282	5.088	<0.001
Year 2020	-0.153	0.251	-0.609	0.542
Grazing Intensity	-0.0002	0.0006	-0.464	0.642
Phenocam NDVI:Year 2020	0.753	0.352	2.138	0.032
Landsat NDVI:Year 2020	1.082	0.36	-3.003	0.003

Table 2. Coefficient estimates, standard errors, z values, and *p* values of variables in a binomial mixed effects model that shows the relationship between the log odds of sage-grouse being in the vicinity of mesic/wet meadows during the vegetational growing season, and year-to-year climatic variability, grazing intensity, remote sensing tool, and the interaction between remote sensing tool and year, in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A from 2019-2020.

Variables	Estimate	Std. Error	z value	Pr(>|z|)
Intercept	-29.15	35.846	-0.813	0.416
Formicidae	-6.107	26.989	-0.226	0.821
Lepidoptera	35.286	26.989	1.307	0.191
Year 2020	19.863	29.593	0.671	0.502
Grazing Intensity	-0.049	0.042	-1.173	0.241
Formicidae:Year 2020	-4.75	38.168	-0.124	0.901
Lepidoptera:Year 2020	-12.893	38.168	-0.338	0.735

Table 3. Coefficient estimates, standard errors, z values, and *p* values of variables in a linear mixed effects model that shows the relationship between the difference of peak sage-grouse usage dates and peak insect capture rate dates, and year-to-year climatic variability, grazing intensity, insect type, and the interaction between insect type and year, in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A from 2019-2020.

Figures

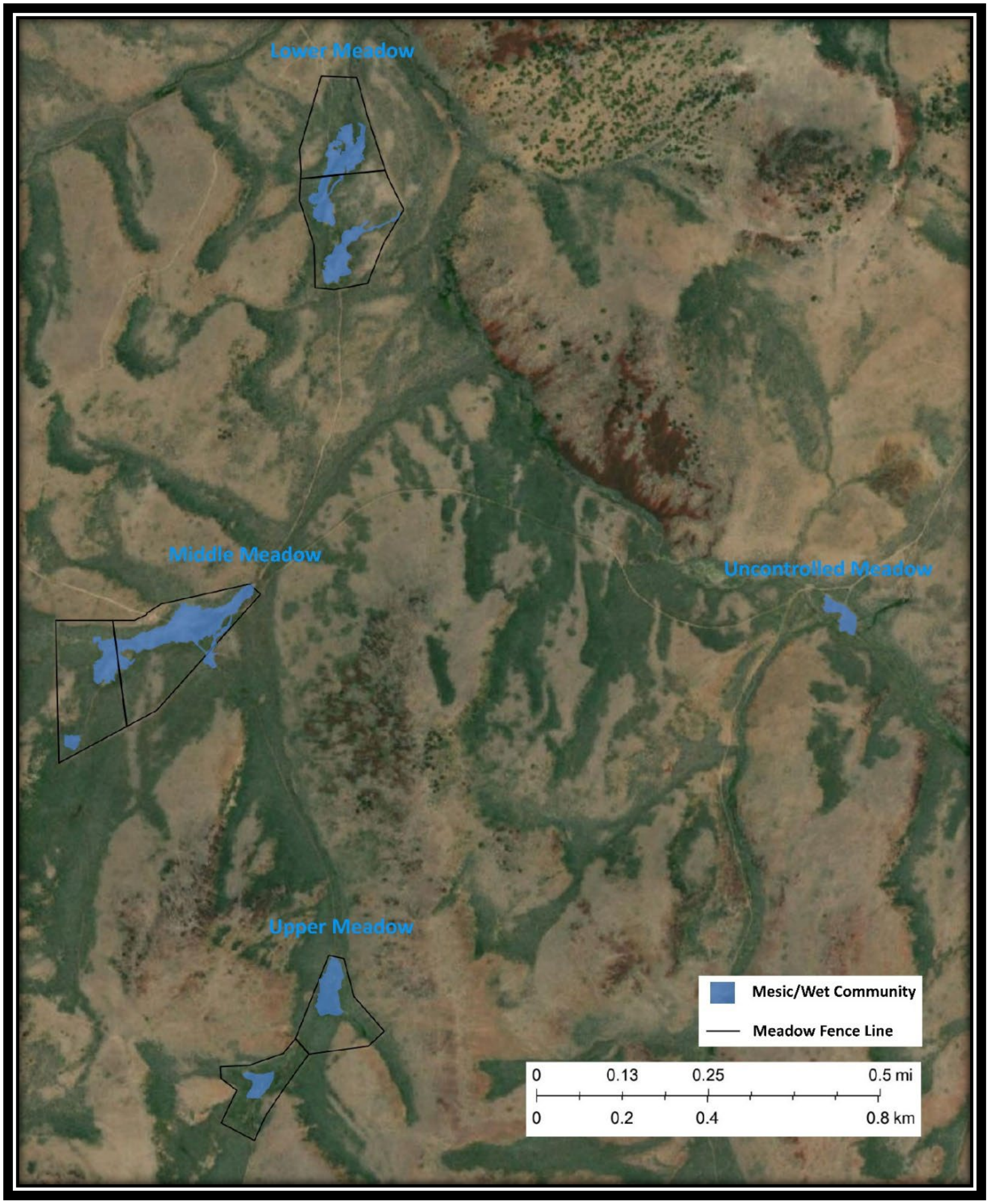

Figure 1. Layout of four meadows and their corresponding fences in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

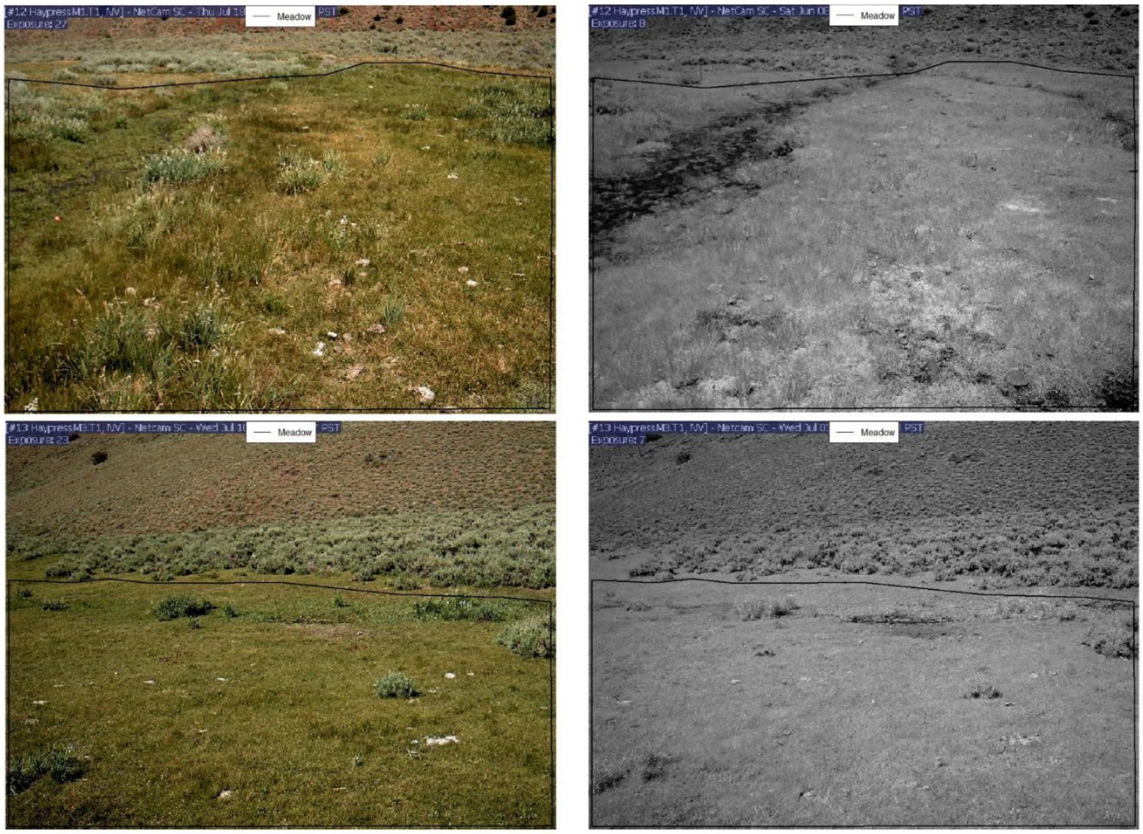

Figure 2. Examples of phenocam regions of interest (ROIs) for meadows in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

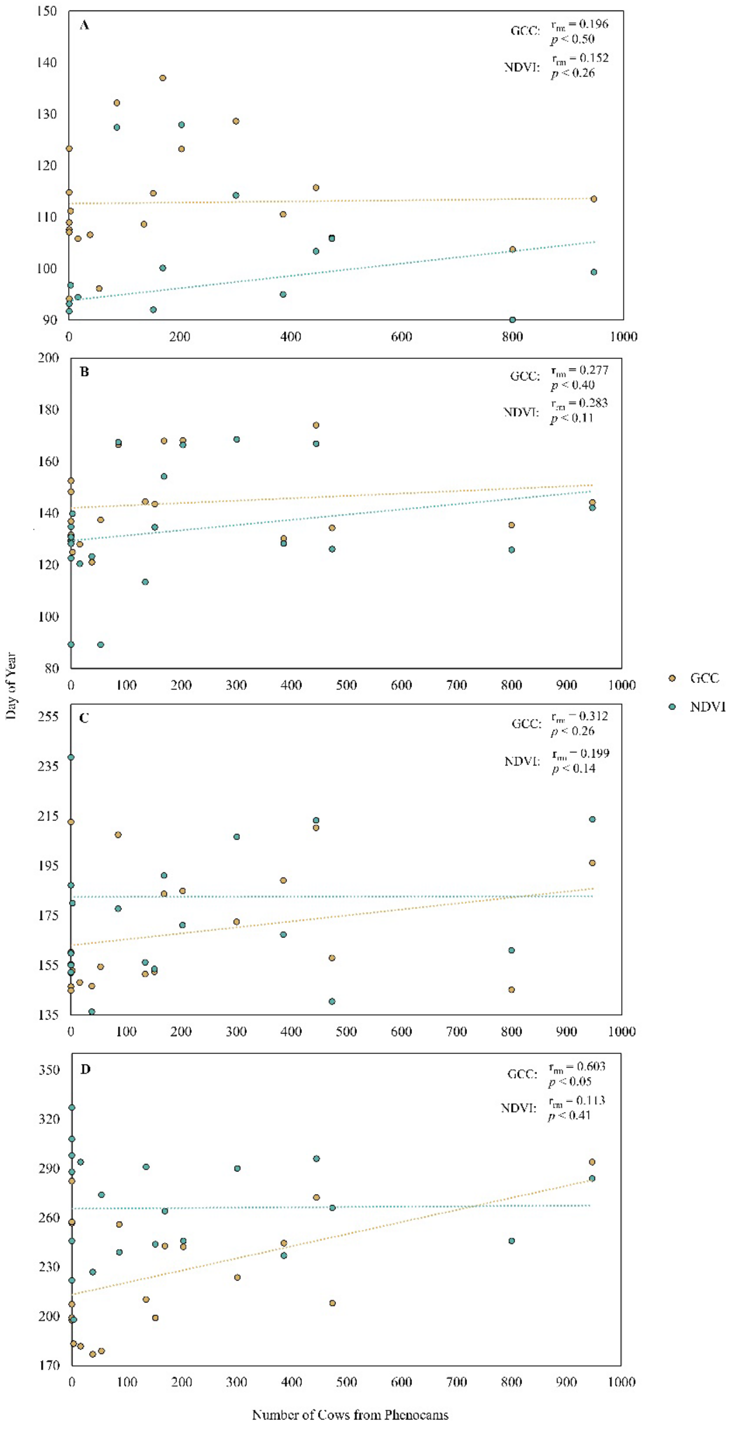

Figure 3. Phenophase day of year from phenocam GCC and NDVI plotted against grazing intensity (number of cows counted from phenocam) in seven meadow areas in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A from 2019-2021. Phenophases include, A) upturn date, B) stabilization date, C) downturn date, and D) recession date. Repeated measure correlation coefficients (r_{rm}) and *p*-values were calculated for each individual meadow across all years.

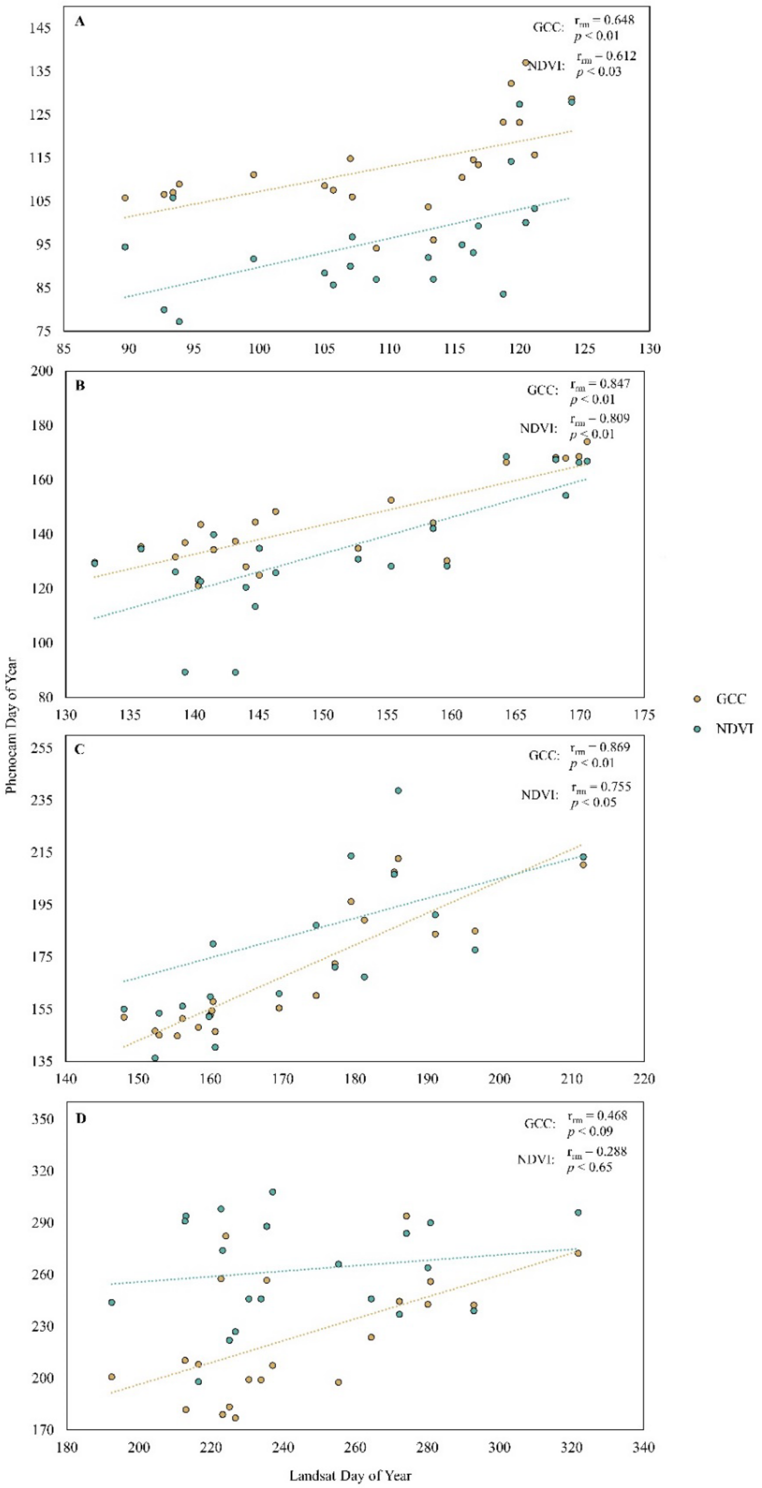

Figure 4. Phenophase day of year from phenocam GCC and NDVI plotted against Landsat NDVI in seven meadow areas in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A from 2019-2021. Phenophases include, A) upturn date, B) stabilization date, C) downturn date, and D) recession date. Repeated measure correlation coefficients (r_{rm}) and *p*-values were calculated for each individual meadow across all years.

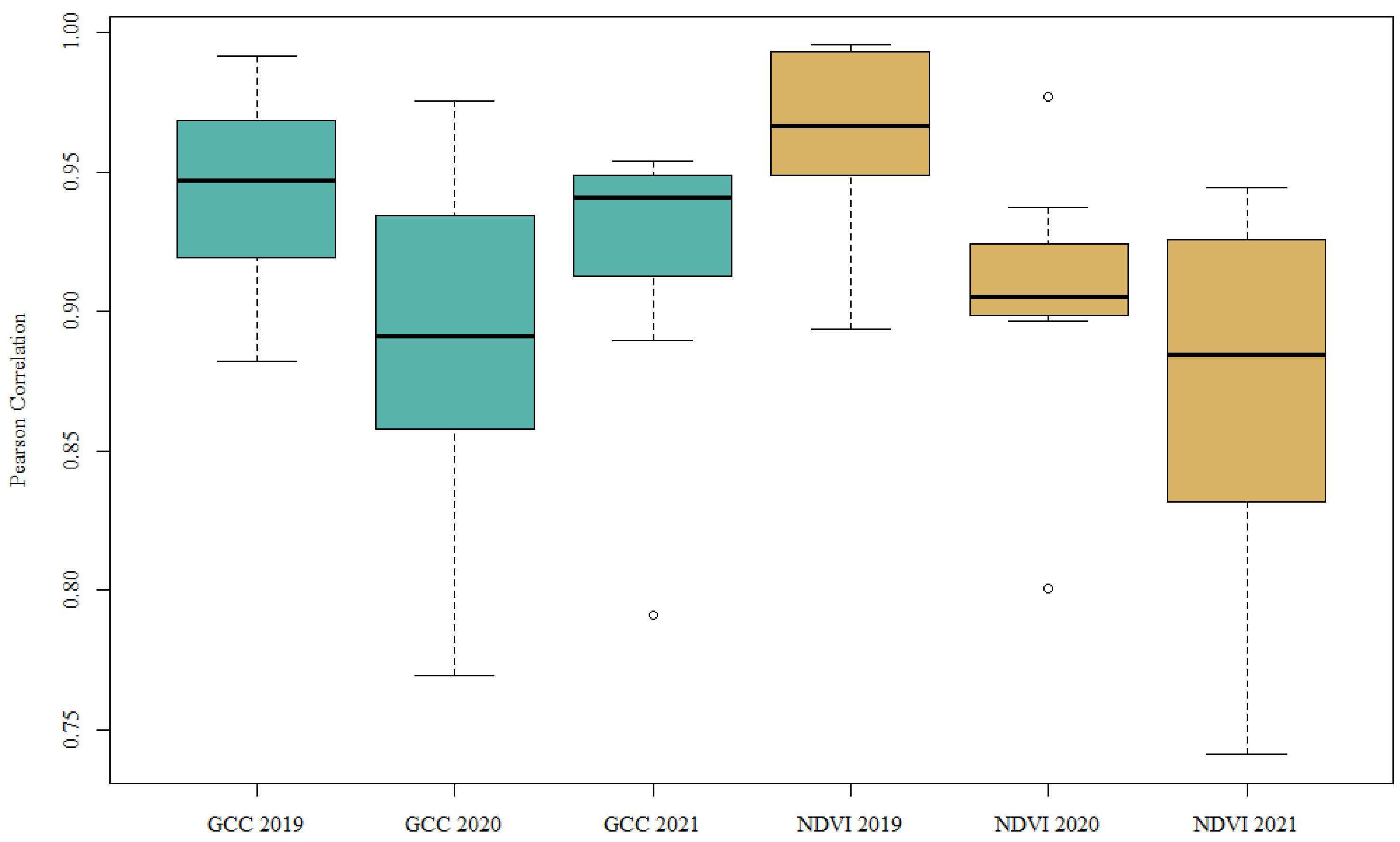

Figure 5. Boxplot that shows the Pearson's *r* correlations of Landsat NDVI and phenocam GCC/NDVI from 2019-2021 in seven meadow communities in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

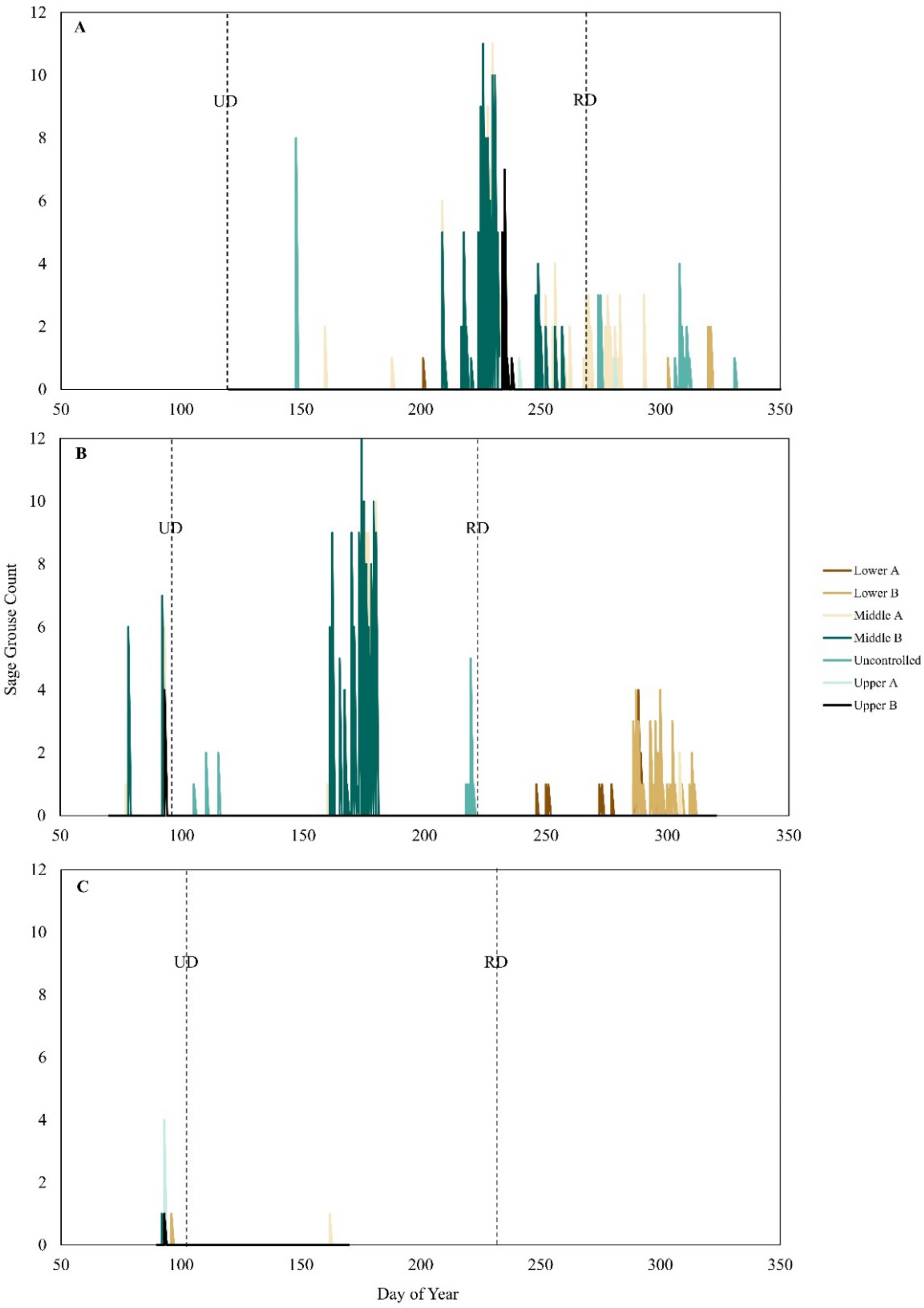

Figure 6. Number of GPS sage-grouse data points across time within 150 meters of seven meadows residing in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A. Three years of data collection occurred, A) 2019, B) 2020, and C) 2021. Average upturn dates (UD) and recession dates (RD) were calculated for each year across all meadows and metric types (Landsat Normalized Difference Vegetation Index, phenocam Green Chromatic coordinate, and phenocam Normalized Difference Vegetation Index).

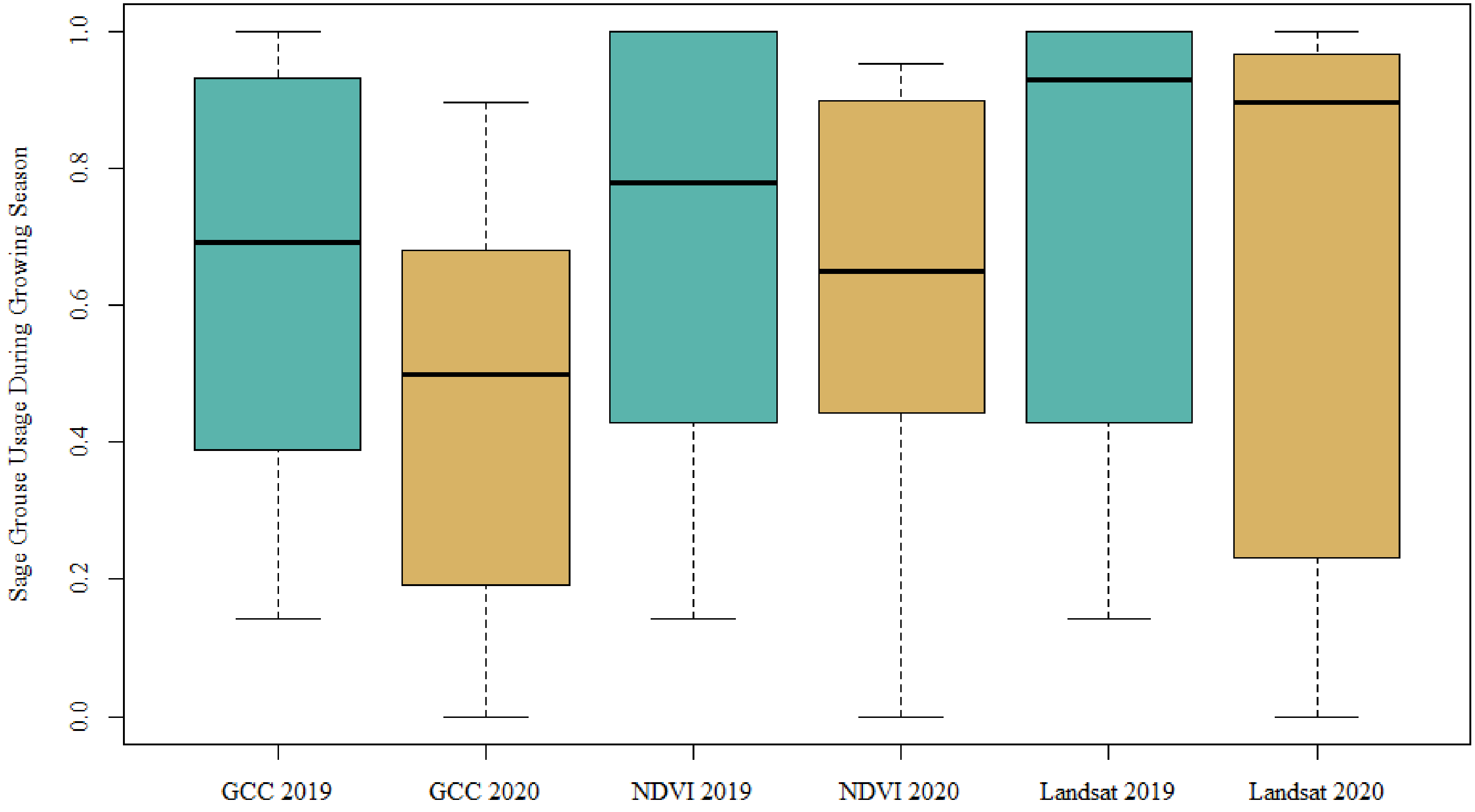

Figure 7. Boxplot that shows the ratios of GPS sage-grouse data points that fell within growing seasons (calculated from Landsat NDVI, phenocam GCC, and phenocam NDVI) and the total GPS data points in the vicinity of meadows from 2019-2020 in seven meadow communities in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S.A.

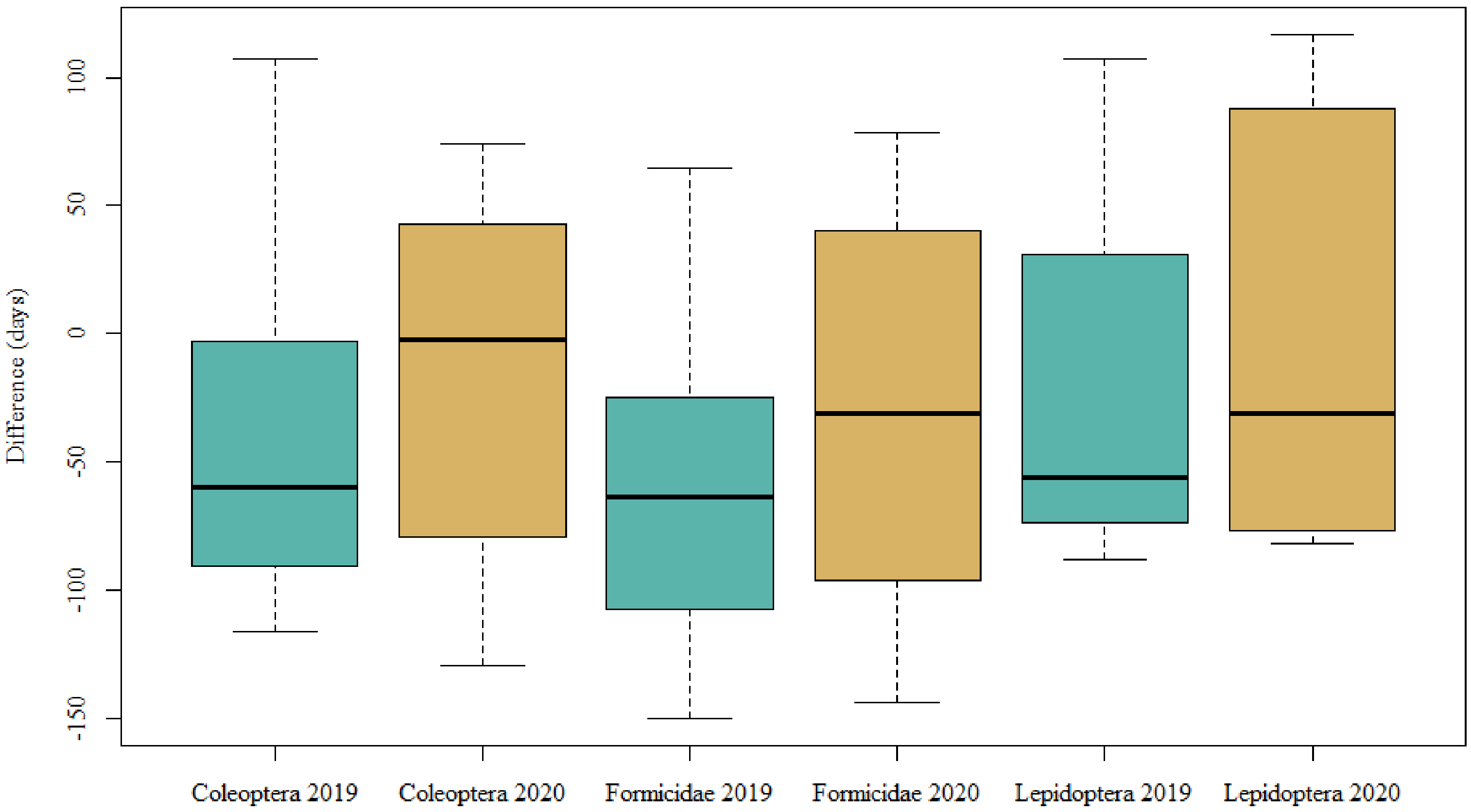

Figure 8. Boxplot that shows the difference of peak sage-grouse usage dates and peak insect capture rate dates from 2019-2020 in seven meadow communities in the Haypress Meadow complex in the Desatoya Mountains, Nevada, U.S. The difference was calculated by subtracting peak sage-grouse usage dates from peak insect capture rate dates.

CONCLUSIONS

Ultimately, the objectives of this dissertation were met. A spectrum of grazing intensities was applied to a meadow complex across highly variable years (very different precipitation and temperature regimes were recorded). Sage-grouse usage was accurately monitored and shifts in phenology were seen across a suite of remote sensing techniques. This study shows that changes in the timing of abundance of mesic resources do occur across short-term climatic time-periods and those shifts can be quite large in relation to the overall yearly growing season. Additionally, vegetative community type plays a very important role in determining how large of a shift may occur. This study encompassed some of the largest changes in precipitation seen in the meadow complex over the last decade. Understanding how years with low precipitation and high temperatures affect these meadow systems will be of increased importance in the future due to climate change.

Regarding grazing impacts, treatment effects on plant phenology were not as large as what was originally hypothesized. Shifts in phenology were only seen during the end of season, and those were only observed for one of the three remotes sensing tools used (phenocam GCC). When predicting what shifts may occur in the future with the continuation of these grazing management practices, it should be noted that these meadows have been historically utilized intensively by both cattle and horses. Those meadows that remain ungrazed may see large shifts as litter build up continues year-to-year. Sustained monitoring of these meadow systems will be paramount in understanding how consistent intensive grazing have affected these meadows' phenology and production.

This dissertation reveals multiple future research opportunities. The phenocam models could be applied to similarly outfitted meadow systems to see if on-the-ground measurements could be extrapolated. Additional meadows could be included to analyze how heavily grazed systems change in relation to properly managed environments and how well these systems emulate similar patterns to previous studies could be observed. Arthropod group collections could be broken down to the species level, showing how induvial species mirror group dynamics. Additionally, dietary and nutritional contributions from arthropod groups and forb communities to sage-grouse survivability and health could be defined.

APPENDIX A

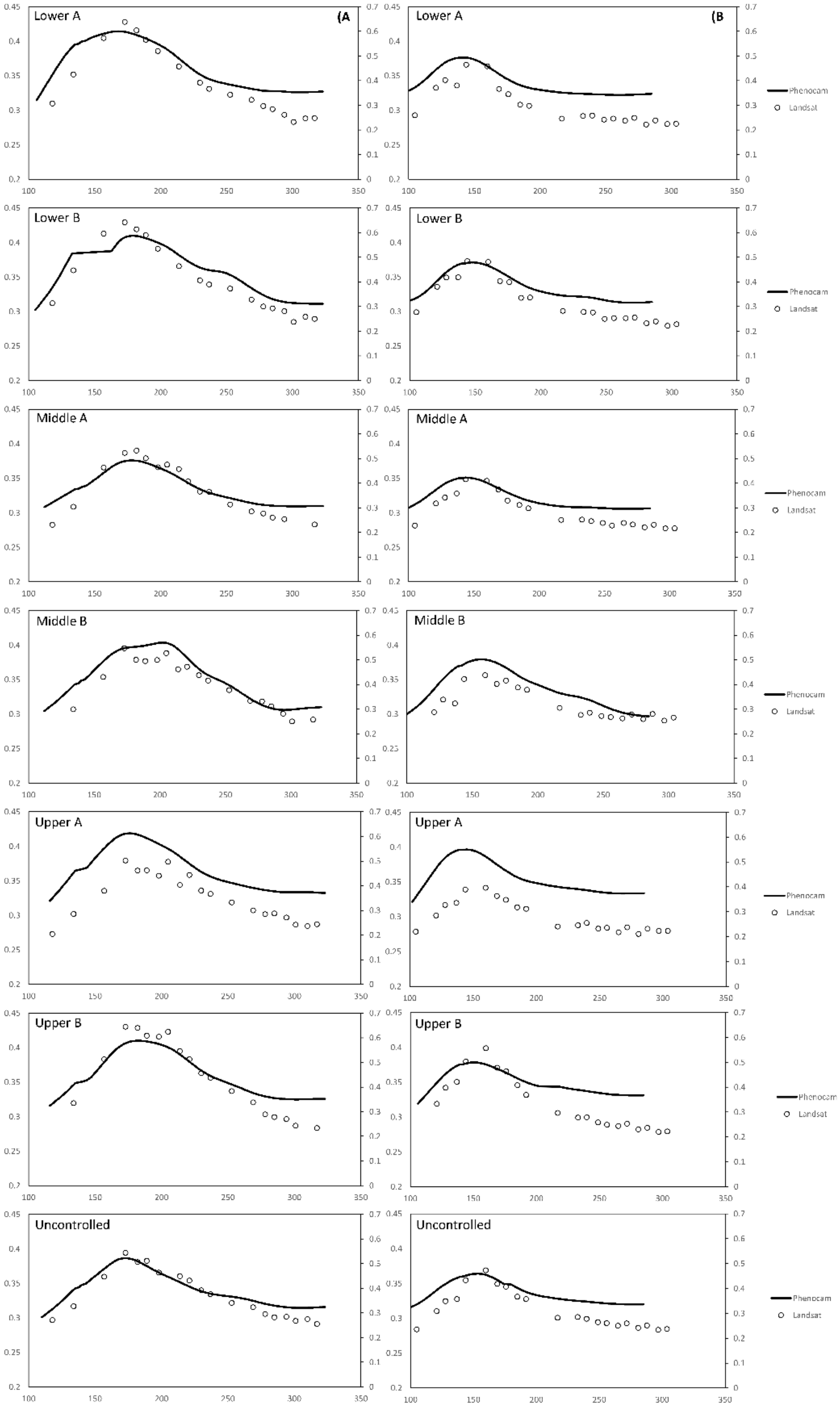

Figure S1. Phenocam derived Green Chromatic Coordinate values compared to Landsat derived Normalized Difference Vegetation Index values over a growing season for seven meadows in the Haypress Meadow complex of the Desatoya Mountains, Nevada, U.S.A. Column A represents values taken from 2019, and column B represents values taken from 2020.

APPENDIX B

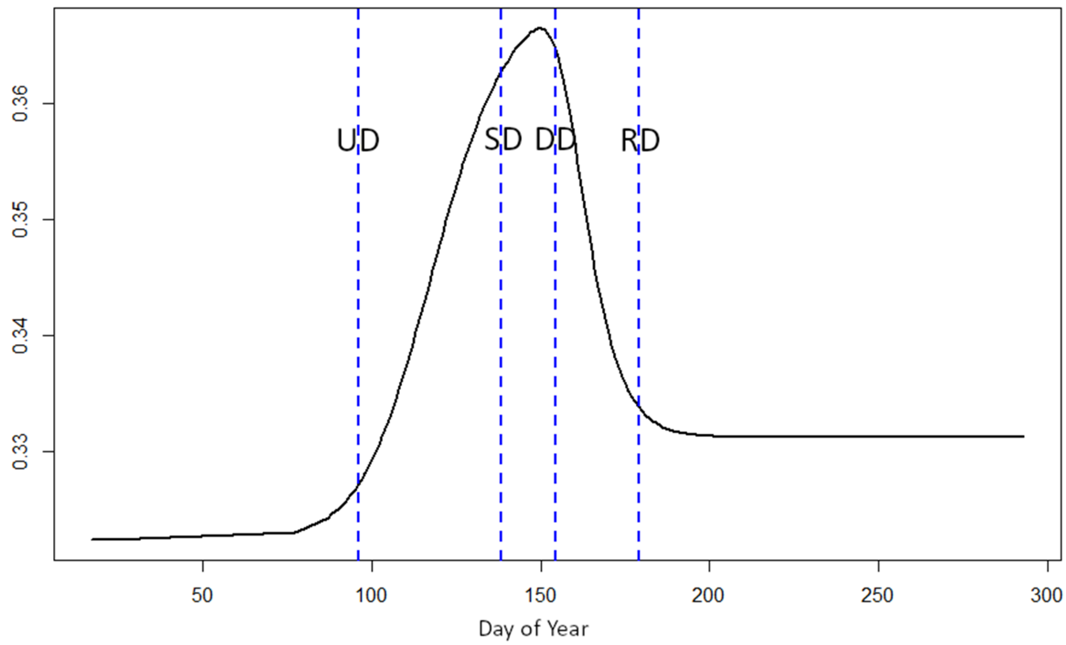

Figure S1. Example phenophase dates along a Green Chromatic Coordinate curve. UD – upturn date, SD – stabilization date, DD – downturn date, and RD – recession date.