EIGHTY-TWO YEARS OF CHANGE IN PLANT FUNCTIONAL TYPE,
COMPOSITION, AND COVER FOR AN ARIZONA INTERIOR CHAPARRAL
COMMUNITY

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ABSTRACT

Understanding plant community response to environmental change is one of the greatest challenges ecologists face. Understanding how plant communities respond to climate or land-use change is complicated in diverse communities where it is logistically difficult to assess the response of every species. Grouping species into functional groups and then examining functional group response to changes is one way to reduce complexity of diverse communities and enhance our understanding of overall community dynamics. Plant functional types (PFT) characterizes groups of plants according to their ecological function. For instance, the PFT approach can group species by taxonomic groups or broad life-form characteristics (e.g., trees, shrubs, herbs, graminoids) or more specific plant traits, such as plant size, shade tolerance, or photosynthetic pathways (C_3 vs. C_4). Such PFT approaches are also relatively easy to measure and allow species with similar life history strategies to be grouped together. Few studies have documented shifts in PFTs over time because long-term data from permanent plots are limited.

We analyzed a long-term ecological data set from the Sierra Ancha Experimental Forest (SAEF) in central Arizona to evaluate how PFTs, species richness, and plant basal cover have changed among 1935, 1952, and 2017 in an interior chaparral community. Twenty-four permanent 1-m^2 chart quadrats established on the SAEF Natural Drainages study site were mapped in 1935 and then remeasured in 1952 and 2017. We assessed changes in plant communities by examining changes in the abundance of PFTs (C_3 vs. C_4 graminoids, forbs, halfshrubs, and shrubs).

Over the 82-year time period, plant basal cover significantly increased by approximately 26%, and species richness increased by an average of 0.5 species per

square meter. The relative basal cover of C_4 graminoids increased ~15% and C_3 graminoids increased ~1% with half of the C_3 graminoid increase attributed to the introduction of *Bromus rubens,* a non-native winter annual. Forbs and shrubs increased ~ 4%, while halfshrubs increased ~2%. Between 1935 and 1952, shrubs increased by 7% driven by increases in *Arctostaphylos spp.* (manzanita) and *Ceanothus greggii* (desert ceanothus). Only the increase in C_4 graminoids and overall basal cover were statistically significant. An herbicide treatment, which was applied to all shrubs on two of the Natural Drainages watersheds in 1954, resulted in a decrease and long-lasting impact on all shrubs and also resulted in a 16% increase in C4 graminoids, and a 16% increase in overall basal cover. We acknowledge the scale constraints of our study in that the shrub increase and decrease on the treated watersheds was not adequately captured on the 1-m2 quadrats. Early land management activities and herbicide treatments, and their subsequent legacy effects can be viewed through the lens of the historical and contemporary data sets presented here.

Key words: plant functional type, long-term data set, chart quadrat, land cover change, land use legacy, interior chaparral, Sierra Ancha Experimental Forest

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Introduction

Understanding plant community responses to environmental change is one of the greatest challenges ecologists face. Climatic drivers and land-use activities play a key role in structuring plant communities (Kelly and Goulden 2008; Gonzalez et al. 2010; Lawler et al. 2014). Classification and analysis of diverse plant communities in response to climate or land-use change often requires that the large number of plant species in a region be reduced to a smaller number of categories for easier analyses and interpretation. Plant functional types (PFT) characterizes groups of plants according to their ecological function (Walker 1992; Noble and Gitay 1996). The PFT approach can use taxonomic groups and/or broad, life-form characteristics (e.g., trees, shrubs, herbs, graminoids, etc.) and/or more specific plant traits, such as plant size (short vs. tall), shade tolerance, and photosynthetic pathways, etc. Such PFT approaches are also relatively easy to measure characteristics of plant communities that relate to plant species survival strategies. Few studies have documented shifts in PFTs over time because long-term data from permanent plots are limited. In this study, we analyzed a long-term ecological data set from the Sierra Ancha Experimental Forest (SAEF) Natural Drainages study site in central Arizona to evaluate how PFTs, species richness, and plant basal cover have changed among years 1935, 1952, and 2017 in an interior chaparral community. The research questions were to determine how: 1) plant functional types (C_3 vs. C_4 graminoids, forbs, halfshrubs, and shrubs); 2) plant species composition; and 3) basal cover by species have changed over time using basal cover data collected on historical and contemporary 1-m^2 chart quadrats.

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Background

The Sierra Ancha Experimental Forest (SAEF), which contains the Natural Drainages study site described in this paper, is located in the the Central Arizona Highlands (Fig. 1). The Central Arizona Highlands are situated in a unique climatic, geologic, and vegetation zone between the Colorado Plateau and the Sonoran Desert Regions (Ffolliott 1999). The SAEF is located within about 16 km of the Roosevelt Dam near Roosevelt, Arizona. The Roosevelt Dam was built between 1909 and 1911 and confines the Salt River below the confluence of Tonto Creek to form Roosevelt Reservoir. This impoundment, together with other dams and reservoirs, provides reliable water to the Phoenix metropolitan area.

Between 1909 and 1925, watershed and range managers became concerned with the erosion and sediments from the Salt River watersheds because those sediments began to accumulate in the Roosevelt Reservoir, reducing the reservoir's holding capacity (Olberding 2018). In 1920, a series of range livestock exclosure plots were established by the US Forest Service, called Plot 1, 2 and 3, to determine the impact of livestock grazing on herbaceous and woody vegetation cover on watersheds surrounding the reservoir (USDA Forest Service 1938). In 1925, the US Forest Service established additional range plots, called the Summit Plots. These plots were established to study the effects of vegetation recovery, mechanical stabilization, and cover changes on stormflow and sediment yields (Rich and Reynolds 1963; Gottfried et al. 1999). In 1932, the US Forest Service dedicated a research area in the nearby Roosevelt and Salt River Watersheds known as the Parker Creek Experimental Forest (USDA Forest Service 1932).

Charles Cooperrider, a USFS grazing inspector, was charged with leading the newly created Southwest Forest and Range Experiment Station (SWFRES) watershed

management program in 1930 (Olberding 2018). Cooperrider searched for a location in the National Forest System where water was more valuable than timber and forage. At this time, large amounts of sediment were accumulating at a rapid rate behind the Roosevelt Dam. Range scientists noticed this sedimentation build up coincided with large amounts of overgrazing occurring on nearby rangelands. Based on scientific reports that stated two-thirds of the natural vegetation cover had been disturbed in this area, Cooperrider headed off to view this site of 724 hectares and later established it as the Parker Creek Experimental Forest and Range Influences Station (Olberding 2018). An additional 4,601 ha were set-aside in 1938 and it was renamed the Sierra Ancha Experimental Forest (SAEF). One of the first watershed studies established within SAEF in 1934 was Natural Drainages, which is the focus of this study. Today, SAEF encompasses 5,325 ha. Scientists at SAEF continued to monitor existing older vegetation studies and established many new watershed studies that examined livestock grazing, herbicide, and logging impacts on watershed vegetation, streamflow, soil erosion, floods, and sedimentation.

Eight vegetation types exist on SAEF ranging from the high elevation mixed conifer and ponderosa pine forests to the low elevation desert grassland and desert shrub (Pase and Johnson 1968). In the mid-elevations of SAEF are the oak woodlands and chaparral vegetation types. The chaparral vegetation type covers 57% of the SAEF (Pase and Johnson 1968). Arizona interior chaparral vegetation types in the Central Arizona Highlands, similar to other chaparral and desert grasslands of the Southwest, was severely degraded by 1900 due to decades of heavy and year-round livestock grazing (Croxen 1926). Livestock overgrazing and fire suppression are considered to be the

primary reasons for the encroachment of 'brush' cover in the Southwest including the Sierra Ancha region (Leopold 1924; Cable 1975; Archer et al. 1995).

      The results of these early range studies, established on SAEF and the surrounding watersheds, provided the basis for establishing long-term watershed studies in the 1950s-1970s to fully evaluate water-yield responses to brush control in chaparral and other vegetation types (Baker 1999). Studies used herbicide alone (Cable 1957; Lillie et al. 1964) or both herbicide and prescribed fire (Lindenmuth and Glendening 1962; Lillie et al. 1964; Pase and Glendening 1965; Pase and Lindenmuth 1971) to control shrubs and trees.

      Natural Drainages, an area of about 21 ha within SAEF (Fig 1), was one of the first watershed-level studies used in the Central Arizona Highlands to evaluate the effect of grazing on vegetative change and on stream flow and sediment yield (Rich and Reynolds 1963; Baker 1999; Gottfried et al. 1999). The four chaparral-covered watersheds were established in 1934 and the understory herbaceous vegetation was quantified beginning in 1935 (Appendix A, Fig. 1). In 1938, a livestock grazing experiment was established that tested the effect of grazing intensity upon vegetation, runoff, and erosion. Beginning in 1939, Drainages A and D were grazed with cattle and horses. Drainages B and C were designated as controls and were not grazed. Grazers and other animals were permitted to move into the drainages from adjacent ranges until 1945. Thereafter, grazers were confined to their respective drainages. Stocking rates averaged 4.3 acres per animal month, with a variation from 2.5 to 7.5 acres on Drainage D. On Drainage A, stocking rates averaged 4.9 acres per animal month (Rich and Reynolds 1963). These studies were terminated in 1952 when it was determined that the intensities

of grazing used in this livestock grazing experiment had no effect on total water yield or sediment trapped in the weir ponds (Rich and Reynolds 1963).

Another major study at Natural Drainages, established in 1954 by F.W. Pond, investigated the response of understory vegetation to chemical control of chaparral (Pond 1964; Ingebo 1972; Ingebo and Hibbert 1974). This research effort was spurred by the water users of southern Arizona to augment streamflow by manipulating vegetation in Central Arizona in the 1950s. All shrubs and trees on Drainages A and C were sprayed with herbicides while Drainages B and D were not sprayed and used as the controls. During the summer of 1954, the basal 6 inches of each living shrub and tree on Drainage C was sprayed with a 6.6 percent solution of 2,4-D and 2,4,5-T in diesel oil until the outer bark was saturated. Surviving shrubs and trees on Drainage C were resprayed in 1956 and 1958. Trees and shrubs on Drainage A were sprayed in a similar manner in 1955 and resprayed in 1957. Halfshrubs were not sprayed (Pond 1964). Before and after treatment photos show the visual effects the herbicide had on shrubs (Figs. 2 & 9). Livestock were excluded from all four drainages between 1955 until the end of the herbicide study in 1959. Between 1959 and 2017, no studies were carried out on Natural Drainages.

In 1935, 24 permanent 1-m^2 chart quadrats were established within the Natural Drainages study site. Here we analyze this long-term chart quadrat data set to evaluate how plant functional types, species richness, and basal cover have changed among years 1935, 1952, and 2017 and between past herbicide treatments in this interior chaparral community. Quantifying vegetation change over these longer time spans may yield insights into how climate and land-use impact these plant communities.

Materials and Methods

The Sierra Ancha Experimental Forest (SAEF) is an area of approximately 5,325 hectares located within the Tonto National Forest in central Arizona on the western slope of the Sierra Ancha Mountains USA (Fig. 1). Precipitation falls mostly as rain during the winter months (November-March) and during the summer monsoon season (July-August). This study focuses on a study site within SAEF named "Natural Drainages". The four Natural Drainages watersheds were established in 1934 and are located in the chaparral vegetation zone of the SAEF (Fig. 2). Each watershed ranges from 9 to 19 ha in size and elevation ranges from 1,369 to 1,444 m, with an average elevation of 1407 m. (Fig. 3). The vegetation at the study site consists of interior chaparral vegetation species including *Quercus turbinella* (turbinella oak)*, Rhus aromatica* (skunkbush sumac), *Arctostaphylos spp.* (Pringle's and pointleaf manzanita), *Ceanothus greggii* (desert ceanothus), and *Nolina microcarpa* (nolina or sacahuiste). *Juniperis monosperma* (one-seed juniper) and *Pinus edulis* (pinyon pine) are tree species that are scattered throughout the study site. Major halfshrubs present include *Eriogonum wrightii* (Wright's buckwheat) and *Menodora scabra* (rough menodora). Major graminoids present include *Bouteloua spp.* (grama) and *Aristida spp.* (three-awn) species (Pond 1964; Brown et al. 1979; Schussman 2006). The plant species list, for species located within the 1-m^2 quadrats within the Natural Drainages study site, with their scientific and common names, and species codes, are listed in Appendix C, Table 1. The parent material is diabase and quartzite rock. Precipitation averages around 512 mm per year. Average

temperature is 16 °C, with maximum temperatures that can reach 40 °C and the minimum temperatures can reach -10 °C.

Field methods

Beginning in 1935, herbaceous, halfshrub, and shrub vegetation species and basal cover were quantified using 1-m^2 quadrats. Twenty-four 1-m^2 quadrats were established throughout the Natural Drainages study site. Six quadrats were placed in each of four drainages (Fig. 3). Quadrats were located on both quartzite and diabase soils (twelve quadrats on quartzite and twelve on diabase soils). No quadrats were placed in areas of dense chaparral shrubs that were little used by livestock (Rich and Reynolds 1963). All 24 quadrats were permanently marked in each corner using metal angle iron and mapped in 1935-36, 1940-42, 1952, and 1955 using the chart quadrat method (Weaver and Clements 1938) using the pantograph designed by R.R. Hill (Hill 1920) (Fig. 4). The chart quadrat is a type of permanent plot that provides a quantitative approach to understanding vegetation change, plant demography, and plant competition. The chart quadrat method was developed by J. E. Weaver and F. E. Clements in the early twentieth century (Weaver and Clements 1938) and has been used extensively by plant ecologists and rangeland scientists ever since.

The historical quadrat datasheets are located in the US Forest Service Rocky Mountain Research Station's (RMRS) Sierra Ancha Experimental Forest archives (https://www.fs.fed.us/rmrs/projects/100-years-vegetation-change-sierra-ancha-experimental-forest) in Flagstaff, Arizona. The historical quadrats were mapped by different range scientist among the years analyzed (Table 1). All 24 quadrats were

relocated in the field and mapped in the fall of 2017 using the methods established by the early range scientists and analyses methods followed Laughlin et al. 2011 (Figs. 5, 6, and Appendix D, Figures 1-6). Basal cover of all plant species, within the 1-m^2 quadrat, were traced using acetate sheets and paper datasheets then digitized in the lab (see Data Analyses section below). All 24 quadrats were re-mapped between November 5 and 29, 2017.

Data Analyses

We digitized all of the quadrat datasheets from every year they were mapped, including 2017, using ArcMap 10.4 software (ESRI, Redlands, CA). For this study, basal cover percentages for each species per quadrat were compiled into Excel charts for the years 1935, 1952, and 2017. Each species was assigned to one of the five plant functional types (PFT) (cool-season or C_3 graminoids, warm-season or C_4 graminoids, forbs, halfshrubs, and shrubs), and overall basal cover was calculated per functional type per quadrat for each of the years analyzed (Fig. 7). Halfshrubs are also known as subshrubs (or chamaephytes) and are defined as low-growing shrubs usually under 0.5 m tall, never exceeding 1 m tall at maturity, with weak woody stems, and often die back to their woody root crown each year (USDA, NRCS 2019). Tukey's Honestly Significant Difference (HSD) test was used to compare all possible pairs of means ($\alpha = 0.05$). We assessed species importance (Mueller-Dombois and Ellenberg 1974) by summing relative density and relative basal cover and scaling the result between 0 and 100 (Fig. 8). To determine the impact of the 1950s herbicide treatment, we used a two-sample unequal

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variance *t*-test to determine significant differences between the treated and control

drainages (Figs. 9-12).

Results

Change in percent basal cover of each PFT among years 1935, 1952, and 2017

were analyzed (Fig. 7). All five PFTs increased between 1935 and 2017 although only the

change in C_4 graminoids, C_3 graminoids, and overall basal cover had a statistically

significant increase (Fig. 7). C_4 graminoids increased ~ 15%, driven primarily by an

increase in *Bouteloua curtipendula* (sideoats grama) and *B. hirsuta* (hairy grama). C_3

graminoids increased ~ 1% (Fig. 8). Over half of the C_3 graminoid increase is attributed

to *Bromus rubens* (red brome), a non-native winter annual. Forbs increased ~ 4%, driven

primarily by an increase in *Artemisia dracunculus* (tarragon) (data not shown).

Halfshrubs increased ~ 2%, driven primarily by an increase in *Eriogonum wrightii*

(Wright's buckwheat). Shrubs increased ~ 4%, driven primarily by an increase in

Arctostaphylos spp. (manzanita). Overall plant basal cover increased ~ 26% between

1935 and 2017 (Fig. 7). Species richness between 1935 and 2017 increased by an average

of 0.5 species per 1-m^2. *Bouteloua curtipendula* (sideoats grama) and *B. hirsuta* (hairy

grama) were the only species included in the top five importance value (IV) rating for all

three years (Fig. 8). Noticeably absent in the top five IV ratings in 1952 and 2017 is

Bouteloua eriopoda (black grama).

Changes in PFTs, species richness, and overall basal cover between the treated

drainages and the control drainages were also analyzed (Figs. 10-13). The C_4 graminoid

basal cover increased ~ 10% on the treated drainages and ~ 5% on the control drainages

from 1952 to 2017 (Fig. 10). Halfshrub basal cover increased ~2% on the treated

drainages and decreased ~1% on the control drainages from 1952 to 2017 (data not shown). Shrub basal cover increased ~ 7% on the treated drainages between 1935 and 1952 but decreased by ~ 6% between 1952 and 2017 (Fig. 11). The shrub basal cover on the control drainages did not change between 1935 and 1952 but increased ~ 4% between 1952 and 2017 (Fig. 11). The overall basal cover on the treated drainages increased from ~ 3% in 1935 to ~ 9% in 1952, and then to ~ 18% in 2017 (Fig. 12). The overall basal cover on the control drainages increased from ~ 2% in 1935 to ~ 3% in 1952, and then to ~ 13% in 2017 (Fig. 12). Species richness on the treated drainages decreased an average of ~ 0.3 species per m^2 from 1935 to 1952 and increased an average of ~ 1 species per m^2 from 1952 to 2017 (Fig. 13). Species richness on the control drainages decreased an average of ~ 0.3 species per m^2 from 1935 to 1952 and increased an average of 0.5 species per m^2 from 1952 to 2017 (Fig. 13). C_3 graminoids were not analyzed for the herbicide treatment, because there were no C_3 graminoids present in our 1-m^2 quadrats in 1935 or 1952. The early research scientists could have been biased and only measured forage species (which would not include the C_3 graminoids) on the historical datasheets, or there were actually no C_3 graminoid species present in our 1-m^2 quadrats in 1935 and 1952. No evidence was found to reach a conclusion on either idea.   Forb basal cover among the drainages before the herbicide treatment highly varied so the herbicide effects on this PFT could not be accurately assessed.

Precipitation and temperature trends for the study site were analyzed with the best resolution and data sets available. The PRISM climate data shows that there is a slight upward trend in mean annual temperature (MAT) and a slight decreasing trend in mean annual precipitation (MAP) since 1895 (Appendix E, Fig. 1). We overlaid the PRISM

MAP data with raw MAP data we found in the RMRS archives for the study site, and there are differences, but they are minimal. The PRISM climate data also shows that there was minimal variation of MAP among years 1935, 1952, and 2017 (Table 1).

Discussion

The greatest change observed in the Arizona interior chaparral vegetation at the Natural Drainages study site from 1935 until 2017, at the scale of the 1-m^2 quadrat, was an increase in herbaceous and halfshrub basal cover. The factors contributing to this basal cover increase are confounded and could be due to release from heavy livestock grazing in the early 1900s, shrub cover decrease from herbicide treatment on two watersheds in the 1950s, or precipitation and temperature patterns over the past 100 years.

The significant increase in warm-season (C_4) graminoid basal cover was driven by the species *Bouteloua curtipendula* (sideoats grama) and *Bouteloua hirsuta* (hairy grama). These were the same dominant and persistent (cover and density) species in the 1930s, 1950s, and 2017. More C_4 graminoid species have established in the study site since 1952, including *Bouteloua aristoides* (needle grama) and *Panicum hirticaule* (Mexican panicgrass). However, *B. curtipendula* and *B. hirsuta* are still the dominant species in 2017, and have contributed significantly more to the basal cover increase. The reason for the increase in native annual graminoids we observe could be that the annual graminoids were not mapped by early range scientists, because they had a bias method that only measured major perennial forage species.

Bouteloua curtipendula (sideoats grama) is highly productive and provides valuable forage for all classes of livestock and wildlife in both winter and summer

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(Hassell and Oak 1987). *B. curtipendula* is also highly palatable to livestock during early and late spring and summer, and provides fair forage value when mature (Wasser 1982). *Bouteloua hirsuta* (hairy grama), on the other hand, is considered good quality forage, but it is not of primary importance for livestock (Parker and Martin 1952). Hairy grama also has low palatability for livestock, in part due to the awn on the spike comb (Steiger 1930). The forage value and palatability of these two dominant species indicate that heavy grazing would decrease sideoats grama cover. Hairy grama would initially tolerate grazing but could be used as a "last resort" if other valuable and palatable species decrease in the area. Sideoats grama is very drought resistant (Albertson 1937; Weaver 1968; Newell and Moline 1978) and can increase rapidly on prairie damaged by extreme drought (Weaver 1968). Hairy grama also resists drought (Stubbendieck et al. 1985) and suffers very little mortality under high temperatures and low water (Mueller and Weaver 1942). The drought-resistant qualities of sideoats grama and hairy grama most likely contributed to their persistent dominance in this plant community for at least 82 years. *Bouteloua eriopoda* (black grama) decreased in abundance and density between 1935 and 1952. We also observed no recordings of black grama in any of our $1\text{-}m^2$ quadrats in 2017. This decrease and eventual disappearance of black grama from our quadrats could be attributed to multiple factors. Future studies and analysis will be needed to look further into this observation.

The significant increase in C_4 graminoids between 1952 and 2017 is likely attributed to the herbicide treatment on all shrub and tree species that occurred in 1954 (Pond 1964). Pond's 1964 study determined the response of understory vegetation to chemical control of chaparral at the Natural Drainages study site and he found that grass

production was "twice as great" on drainages that were sprayed than the control drainages and that this significant difference was due to increased production of *B. curtipendula* (sideoats grama). Our results show that C_4 graminoid basal cover on the treated (sprayed) drainages increased ~5% more than on the control drainages and that *B. curtipendula* increased 4-fold between 1952 (two years prior chemical treatment) and 2017. Shrub basal cover also decreased on the treated drainages, while the shrub cover increased on the control drainages between 1952 and 2017. Overall basal cover on the treated drainages increased ~6% more, and species richness increased an average of 0.5 species per m^2 more than on the control drainages between 1952 and 2017. Results from Pond (1964), combined with our longer temporal analysis, suggest that the herbicide treatment had a legacy effect on the landscape.

The significant increase in cool-season (C_3) graminoids between 1952 and 2017 was attributed primarily to the introduction of *Bromus rubens* (red brome), a non-native winter annual. Red brome is an invader of Arizona desert scrub (McClaran and Brady 1994) and is common in open canopies of Arizona chaparral subjected to heavy grazing (Bolander 1982). In relatively dry areas of the Southwest, red brome may displace native species during wetter years (Banner 1992). However, relatively drier winters and wetter summers may slow the red brome invasion (Betancourt 1996). The presence and increased cover of red brome should be managed as this species has the potential to increase the size and frequency of wildfires (Phillips 1992; Brooks 2004; Horn 2013).

Forb and halfshrub basal cover also increased between 1935 and 2017. *Artemisia dracunculus* (tarragon) was the forb that increased the most between 1952 and 2017 and is now a component in the plant community. Tarragon provides valuable forage for

domestic sheep but has little value for cattle in western states (Stubbendieck et al. 2003).

Other grazing and drought resistant information is lacking. *Eriogonum wrightii* (Wright's

buckwheat) increased in cover between 1935 and 1952 but decreased between 1952 and

2017. Wright's buckwheat has always been a major component in this plant community

and is good forage for whitetail deer, especially during drought (Anthony 1976). Other

grazing and drought resistant information are lacking.

Precipitation and temperature trends in the region have an effect on vegetation

composition and cover over time. The most extreme drought of the past 400 years in the

Southwest occurred in the mid–twentieth century (1942–57). This drought resulted in

broadscale plant dieoffs in shrublands, woodlands, and forests and accelerated shrub

invasion of grasslands (Swetnam and Betancourt 1998). Modern human impacts,

specifically livestock grazing and fire suppression, may have exacerbated the ecological

consequences of the 1950s drought, however, additional study is needed to disentangle

the interacting roles of land use and climate.

The PRISM data show a slight increasing trend in MAT and a slight decreasing

trend in MAP since 1895. Also, the MAP among the mapped years (1935, 1952, and

2017) varied slightly from one another (Table 1). This observation suggests that

precipitation may have had minimal effect on the change in vegetation communities and

abundance throughout the 82-year period. However, the range of total precipitation

throughout the year within 1935 and 1952 is spread out over many more months than

within 2017 (Appendix C. Fig. 1), which suggests increased variation of interannual

precipitation regimes observed in other dryland regions (Knapp et al. 2015; Gherardi and

Sala 2018). As mentioned previously, the invasive graminoid red brome can thrive in wet

years, but becomes constrained in drier winters and wetter summers (Banner 1992; Betancourt 1996). The effect that varied interannual precipitation has on red brome populations has been studied (Hereford et al. 2006; Snyder and Tartowski 2006).

The southwestern US is predicted to become drier and experience more severe droughts in the future (Cayan et al. 2010). Based off the drought-resistant qualities of sideoats grama and hairy grama, and their dominance among this 82-year analysis, we predict that both of these species will persist into the foreseeable future.

We acknowledge that the scale used for our study has its constraints. While the 1-m^2 quadrats capture herbaceous vegetation change, this scale does not totally capture the increase in shrub basal cover throughout our study site. The 1-m^2 scale was used so that data among all years would be easily compared to one another.

Management Implications

Wildfire risk

Climate change in the western United States has dramatically increased the number of large forest wildfires during the past 35 years (Westerling et al. 2006). The expansion of the US wildland-urban interface has also increased the number and severity of wildfires, especially in the southwestern US (Radeloff et al. 2018). In Arizona, the wildland-urban interface currently encompasses 294,517 km^2 and is predicted to expand by approximately 30% by 2030 (Theobald and Romme 2007). An expansion of the wildland-urban interface means that there will be more wildfires due to human ignition, and wildfires will pose an increasing risk to lives and homes. Arizona interior chaparral is flammable in nature and has potential for extreme fire behavior. Land around homes within the wildland-urban interface must be carefully managed both to mitigate structural

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losses from wildfire and to protect adjacent wildlands from fires originating on residential or commercial property. Arizona Extension Office recently suggested that residents create wildfire defensible space (Schalau and Twaronite 2010). Creating this survivable space includes "finding the right balance between reducing fuels and maintaining a diverse plant community that protects the soil from erosion and provides suitable habitat for desired wildlife species" (Schalau and Twaronite 2010). Wright's buckwheat (*Eriogonum wrightii*) and many native warm-season (C_4) grasses are considered excellent choices for wildfire survivable space in chaparral, because it mimics the open, post-fire stage in chaparral succession. Our study shows that *Eriogonum wrightii* and many C_4 grasses are increasing in cover in our interior chaparral study site. This suggests, based on Schalau and Twaronite (2010), that wildfire risk should be decreasing in our study site. However, chaparral shrub cover is increasing overall in the region, at the study site, and even in the drainages that were treated. (Huebner et al. 1999; Sankey et al. 2019).

Woody plant encroachment

A shift in plant functional groups from herbaceous plants (especially perennial graminoids) to shrubs can have important ecological and economical impacts within SAEF and throughout the interior chaparral vegetation type, as shrub encroachment has been documented to decrease cover of grasses, reducing the availability of critical forage species (Archer et al. 2017). Shrub encroachment has also been shown to reduce water yield (Hibbert 1983). We documented an increase in shrubs on our study site, but the small size of the 1-m^2 quadrats did not capture the magnitude of the woody plant increase.

Woody plant encroachment (WPE) is also occurring in the region due to 80 years of no fire (Sankey et al. 2019). However, multiple interacting drivers, not just fire suppression, can also cause WPE. Mean annual precipitation can set an upper limit to woody plant cover, but local disturbances such as fire, livestock overgrazing, and soil properties constrain this potential. Seasonality, interannual variation, and intensity of precipitation events can also explain the rate and extent of woody plant expansion (Archer et al. 2017). WPE in this region has created elevated levels of woody cover/fuels that could potentially increase the fire extent and frequency of this interior chaparral vegetation type. The culmination of human impacts in this region over the last century have contributed to increasing areas of contiguous fuel loading that has reduced the fuel patchiness observed in natural interior chaparral landscapes. Land managers in this region should consider decreasing woody fuel loads. Severe wildfires due to increased woody fuel loads may have a large detrimental effect on watersheds in the region.

Establishment and spread of invasive herbaceous species

The presence and increased cover of the invasive *Bromus rubens* (red brome) graminoid should be managed as this species has the potential to increase the size and frequency of wildfires. The introduction and expansion of exotic annual grasses in arid ecosystems are one the most widespread and ecologically problematic invasions land managers face (D'Antonio and Vitousek 1992). Red brome is a Eurasian originating annual graminoid species and is associated with increased fire extent and frequency in the Mojave Deserts (Brooks 2004; Horn 2013). Invasive annual grasses, including red brome, are expected to spread across the deserts of the western U.S. due to climate change

(Abatzoglou and Kolden 2011). Red brome has invaded the semi-desert grassland regions around the Natural Drainages study site (Lata 2019). The increase in the exotic, annual grasses, including red brome and Mediterranean grass (*Shismus spp.*), in this region contributed to the increased severity of the Woodbury fire that burned over 123,000 acres in June and July of 2019 (Lata 2019). A variety of management methods should be used to control the spread of red brome in this region. Actively controlling these invasive annual grasses early in their establishment phase will save an untold amount of time and money in the long run.

Herbicide treatment

Chemical control of interior chaparral woody vegetation is a land management option that has been used previously (Cable 1957; Pond 1964; Perry 1967; Murphy and Leonard 1974; Browning and Archer 2011). Long-term studies on the legacy effects of these chemical treatments are lacking. Using a combination of historical and contemporary data and technology, effects of the 1954 chemical treatment on PFT composition, can be quantified. If results are promising, we suggest land managers re-invest and investigate using current-day herbicides as a tool for reducing woody fuel loading in interior chaparral landscapes. These suggestions depend largely on the management objectives of the landscape in question.

Grazing management

The overall basal cover increase throughout the study site could be attributed to the exclusion of heavy livestock grazing beginning with the creation of the Natural

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Drainage experiments in 1934. Total grass cover in grama-dominated (*Bouteloua*) southwestern grasslands has shown to dramatically increase within livestock grazing exclosures (Bock and Bock 1993; Valone and Sauter 2005). Livestock grazing is considered a key component of shrub encroachment, although most evidence is anecdotal or confounded by other factors (Browning and Archer 2011). Contrary to widely held assumptions, protection from livestock since 1932 not only failed to deter woody-plant proliferation, but also actually promoted it relative to grazed areas. Results suggest (1) that thresholds for grassland resistance to shrub encroachment had been crossed by the 1930s, and (2) fire management rather than grazing management may be key to maintaining grassland physiognomy in this bioclimatic region (Archer et al. 2017).

Targeted grazing could also be used as a tool to reduce the spread of invasive annual grasses. There has been a lot of research published recently on the use of grazers to help control the spread of *Bromus tectorum* (cheatgrass) (Young and Clements 2007; Diamond et al. 2009; Schmelzer et al. 2014; Davies et al. 2016). Cheatgrass and red brome are very similar ecologically and historically and are separated mostly by an elevational gradient (Reid et al. 2008). Similar grazing techniques used to control cheatgrass could be implemented in areas where red brome is highly invasive. Grazing red brome while it is green, and not already seeded out, can reduce plant size and density. In areas that have been moderately grazed, *Bouteloua curtipendula* (sideoats grama) and other perennial grasses are common, while in areas that have had poor grazing management, the non-native *Bromus rubens* (red brome) and other annual grasses are more common (Bolander 1982). Proper timing, using a low-density approach, and close management of livestock grazing can also minimize impact to non-target species (USFS

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2014).

Conclusion

Long-term and permanent chart quadrats provide strong empirical and quantitative data that can be used to analyze vegetation change over long temporal scales. Although chart quadrats can be limited spatially, the temporal scale and species-specific detail gained from these plots are unmatched in ecological research. We quantified changes in plant functional types in an Arizona interior chaparral community over 80 years and the long-term impact of past management activities. These permanent data sets can also be used for fine-scale plant demography studies and to support rangeland monitoring. Although the 1-m^2 quadrats captured the increase in herbaceous and halfshrub cover, and an increase in the shrub PFT was shown, the sampling footprint is too small on a 1-m^2 quadrat to capture the magnitude of the shrub cover increase over time (shrub reestablishment or increase in individual shrub sizes). The use of these long-term data sets in rangeland monitoring is increasingly needed to better inform land managers, policy makers, and other stakeholders about vegetation changes related to land-use and climate change. The need to continue these long-term studies is also imperative to increase the predictive power of future vegetation change models. History can give us valuable insight into the future. These chart quadrats give us a lens to look into the past and can improve our lens to also look into the future.

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Table 1. The year and month 1-m^2 quadrats were mapped along with the mean annual precipitation (MAP) and mean annual temperature (MAT) within that year for the SAEF Natural Drainages study site.

Years mapped	Months mapped	MAP (mm)	MAT (°C)
1935	Aug-Dec	844.3	14.2
1952	Late July	754.1	14.2
2017	November	512.3	15.9
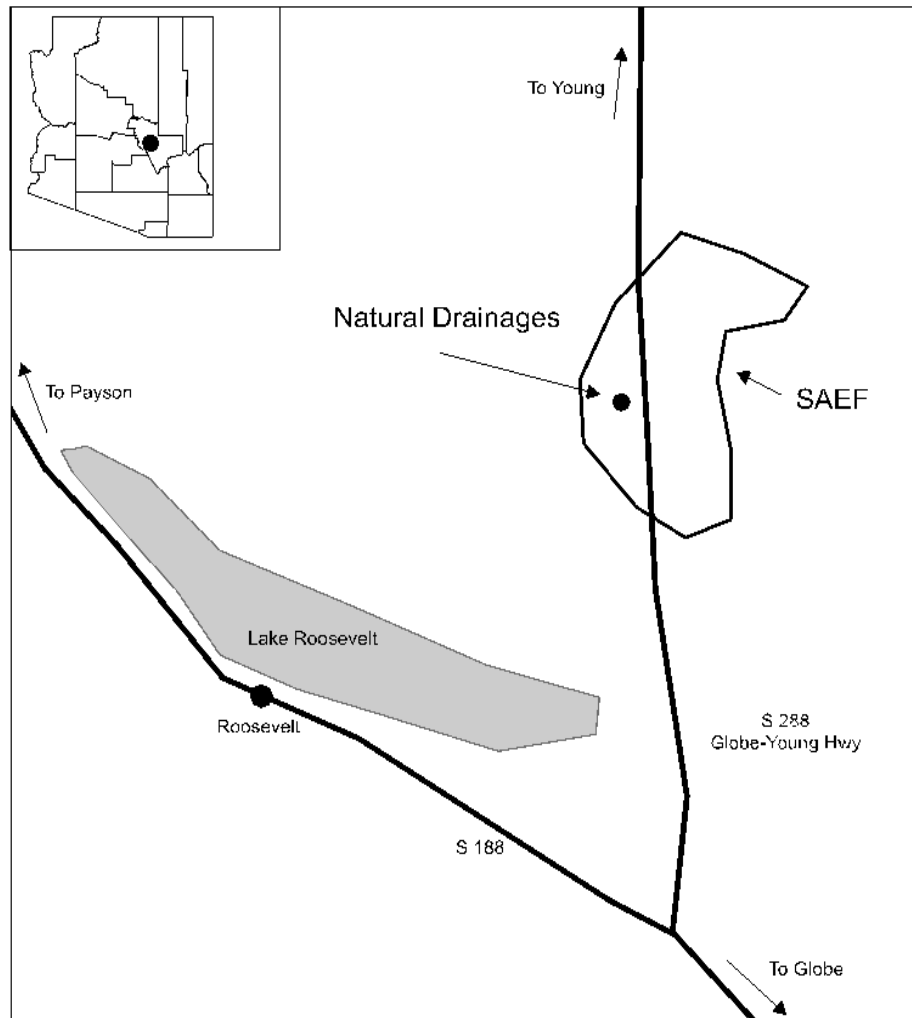
Figure 1. Location map of Sierra Ancha Experimental Forest (SAEF) and Natural Drainages study site.
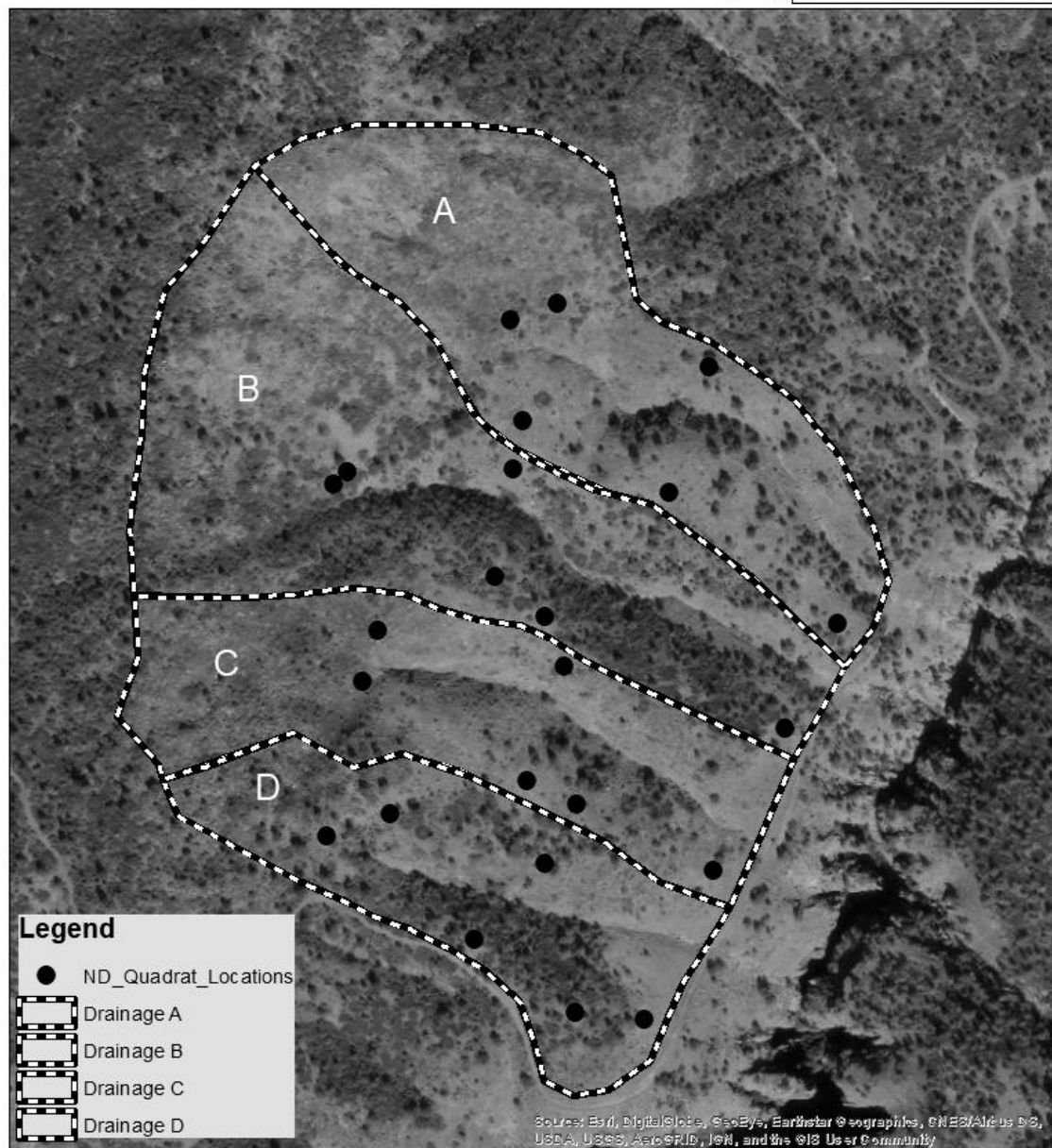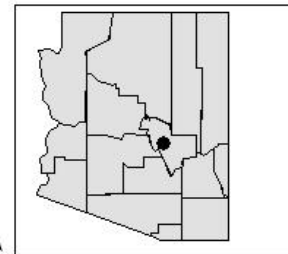
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Figure 2. Repeat photographs of Natural Drainages study site: 1935 (top by W.J. Cribbs); 1960 (middle by H.G. Reynolds), and 2016 (bottom by J.M. Leonard).
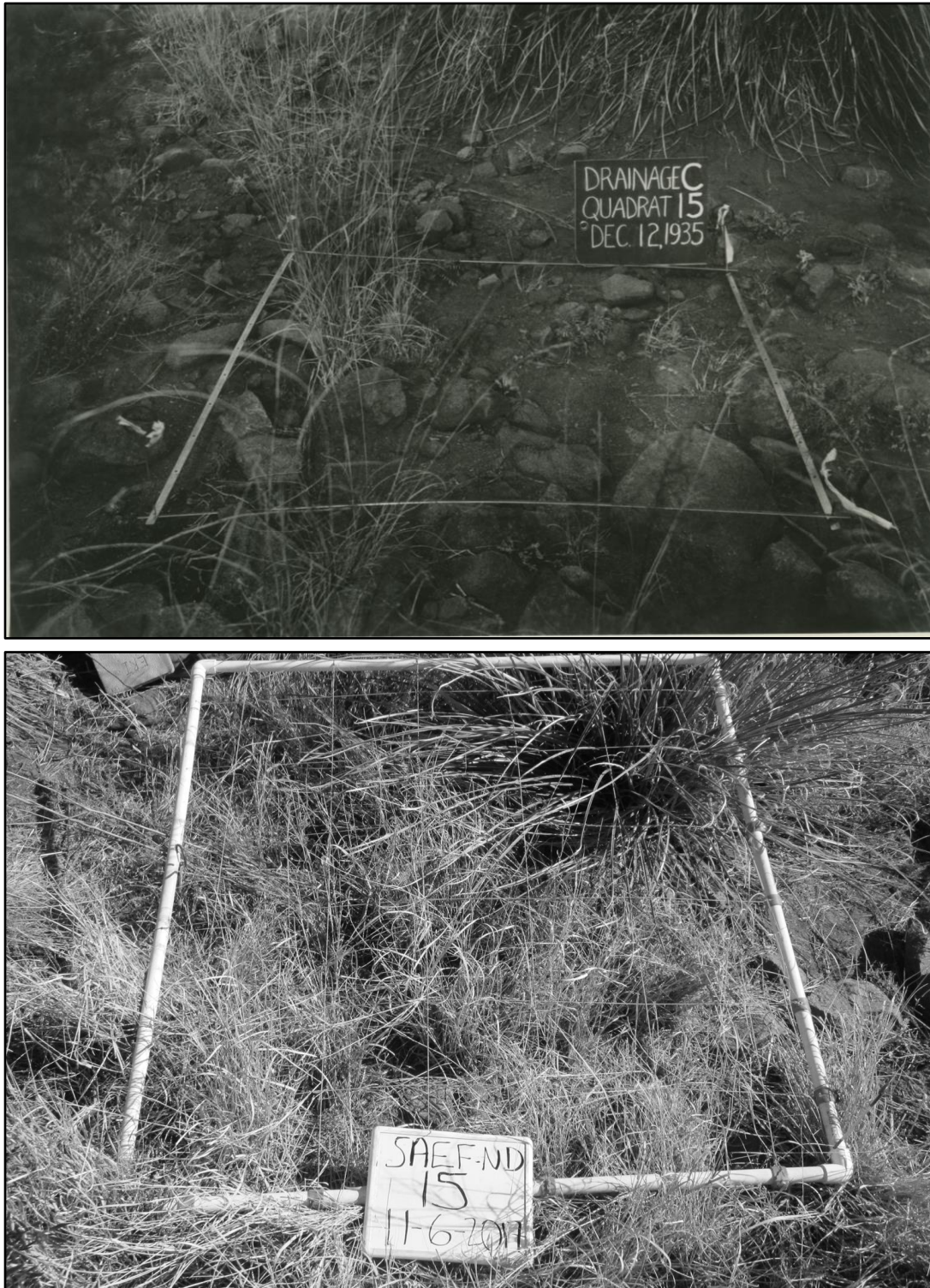
Figure 3. Layout map of Natural Drainages including arrangement and location of all drainages and 1-m^2 quadrats (n=24).

Figure 4. Grazing examiner M.W. Talbot and Ranger Sherman operating R.R. Hill's pantograph charting vegetation quadrats on the Tonto NF (1920). US Forest Service photograph 162775.
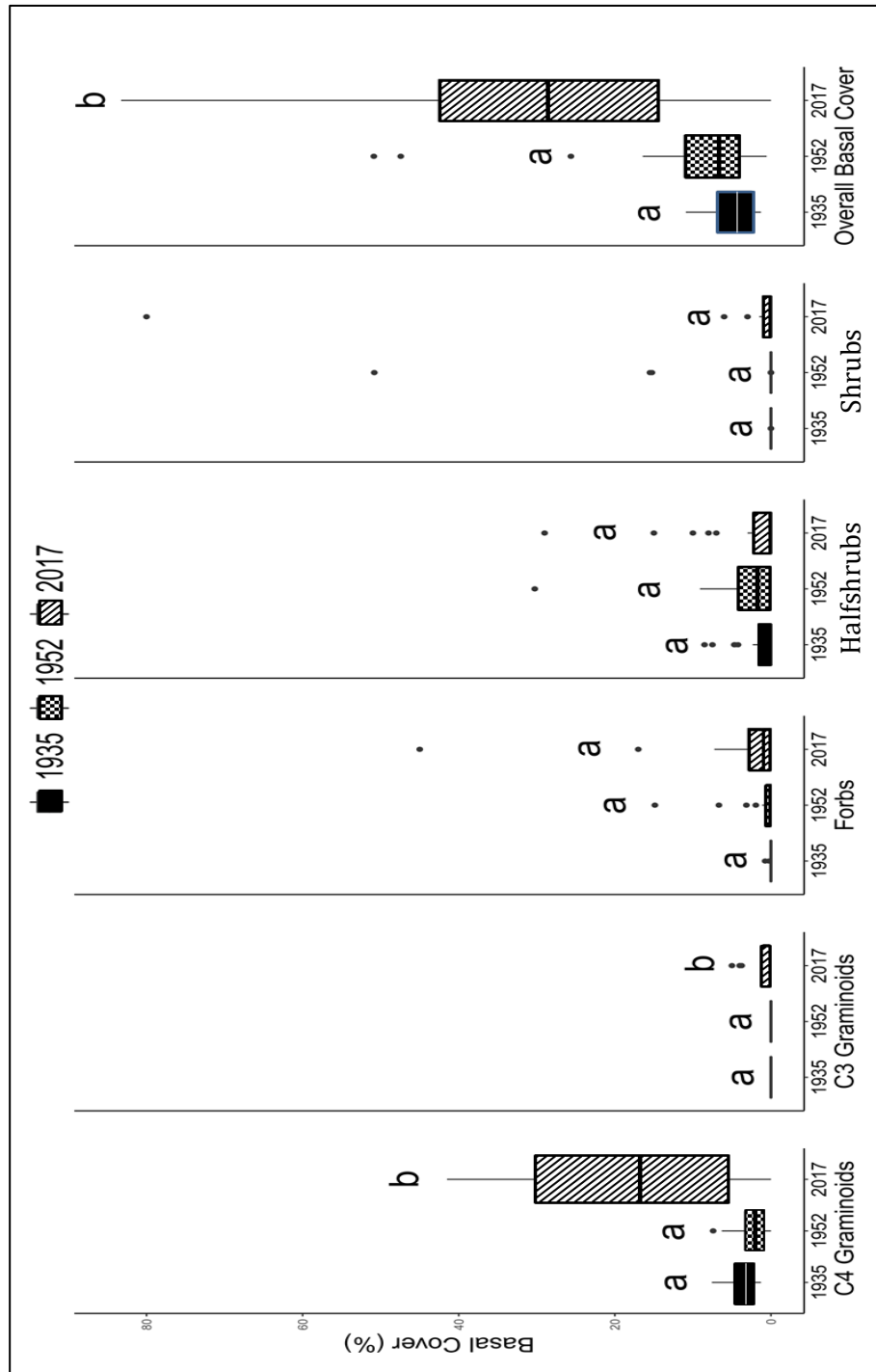
Figure 5. Photograph of quadrat #15 in 1935 (above) vs. 2017 (bottom) at SAEF Natural Drainages study site.
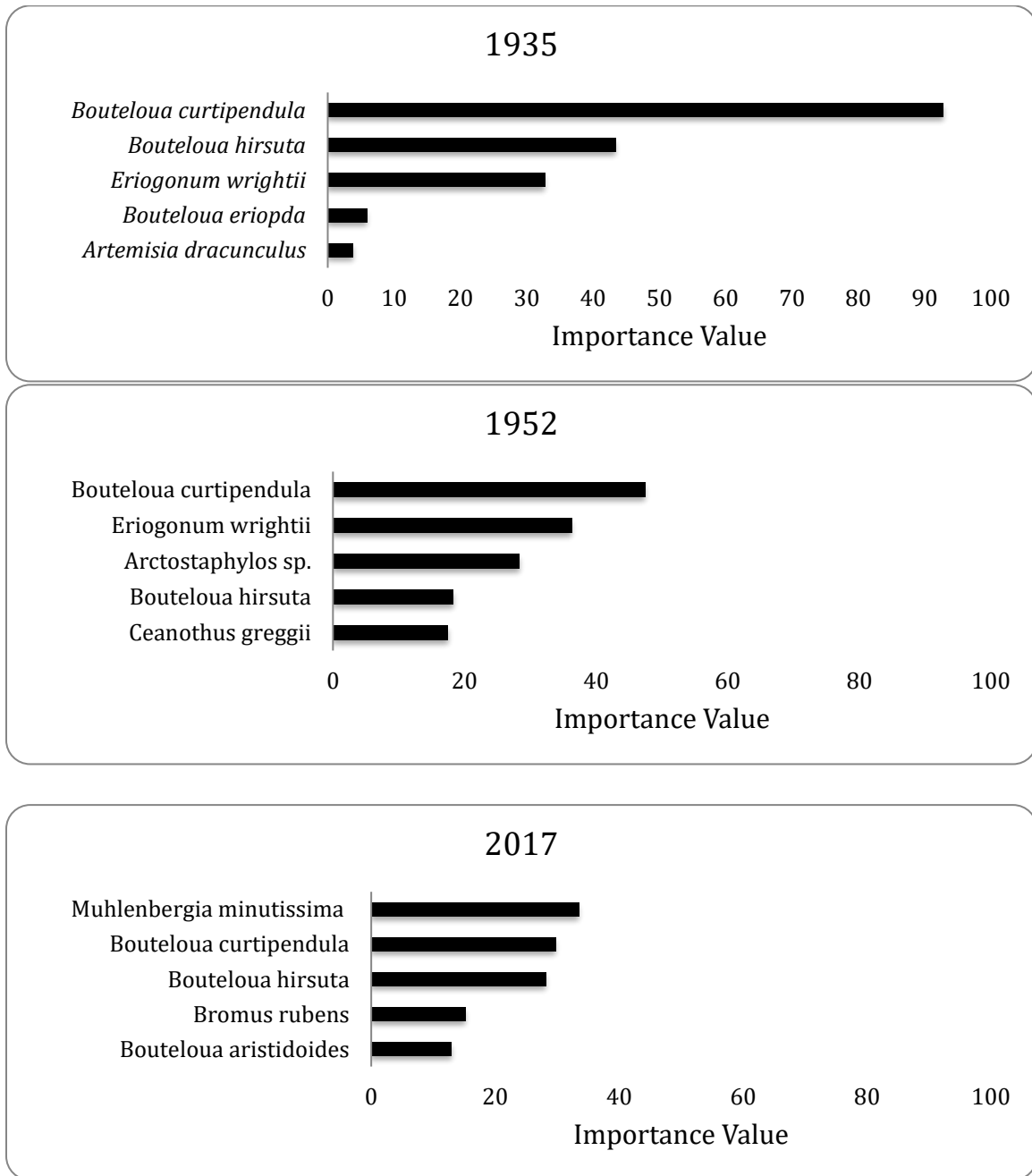
Figure 6. Researchers W. M. Gibson and S. M. Massed mapping vegetation quadrats in Natural Drainages (October 2018).

Figure 7. Changes in percent basal cover of five plant functional types (PFTs) and overall cover among years 1935, 1952, and 2017 *(n = 24)*. Means sharing a letter are not statistically different (P>0.05; Tukey's HSD).
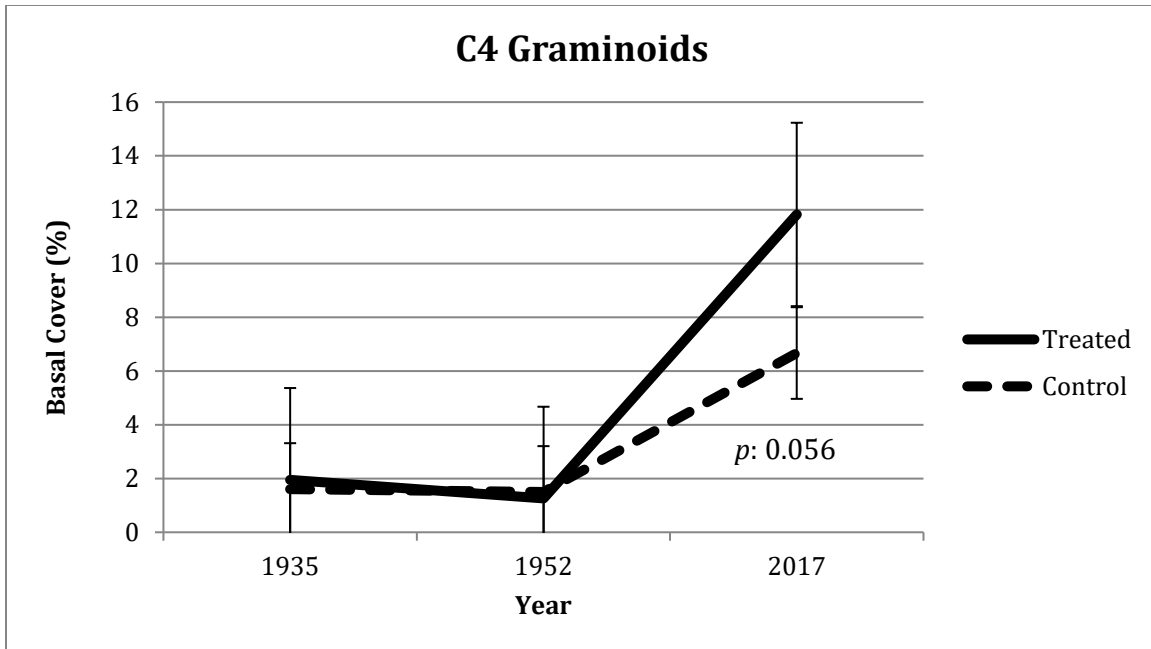
Figure 8. Importance values (IV) for the top 5 species among 1935, 1952, and 2017 at the SAEF Natural Drainages study site.

Figure 9. Photographs before (1954; above) and after (1956; below) the herbicide treatment on two different drainages. Drainage B (control; left) and Drainage A (treated; right).

Figure 10. C$_4$ graminoid basal cover (%) change among years 1935, 1952, and 2017 between treated (Drainage A and C) and control (Drainage B and D) drainages with standard error bars. P-values were calculated from a paired two-sample *t*-test. The initial herbicide treatment occurred in 1954.
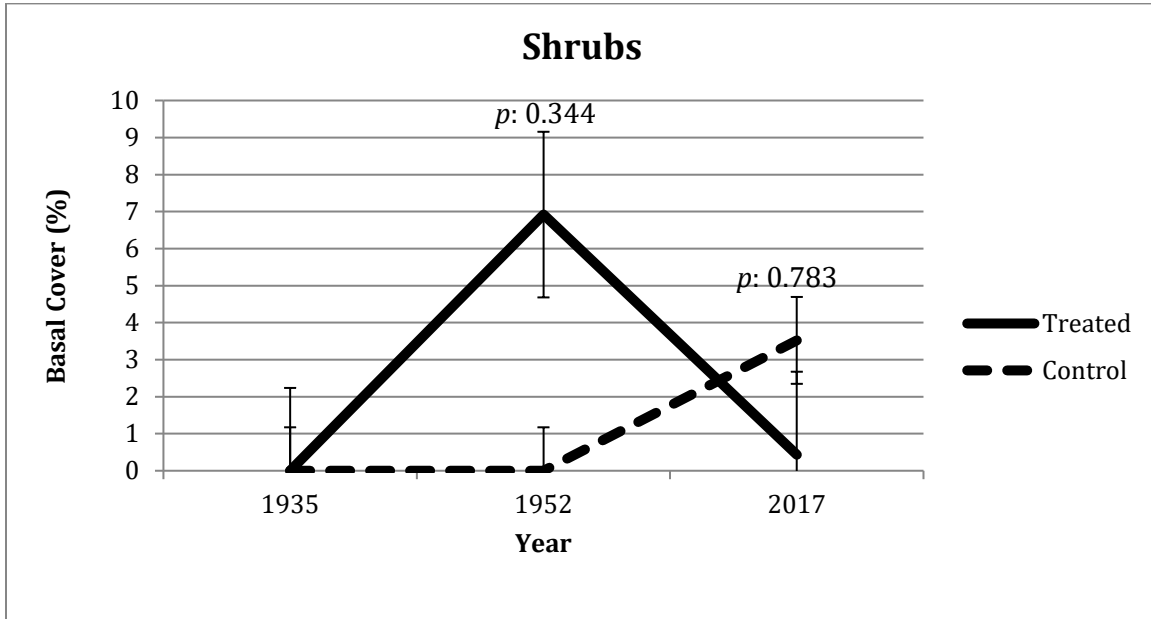
Figure 11. Shrub basal cover (%) change among years 1935, 1952, and 2017 between treated (Drainage A and C) and control (Drainage B and D) drainages with standard error bars. P-values were calculated from a paired two-sample *t*-test. The initial herbicide treatment occurred in 1954.
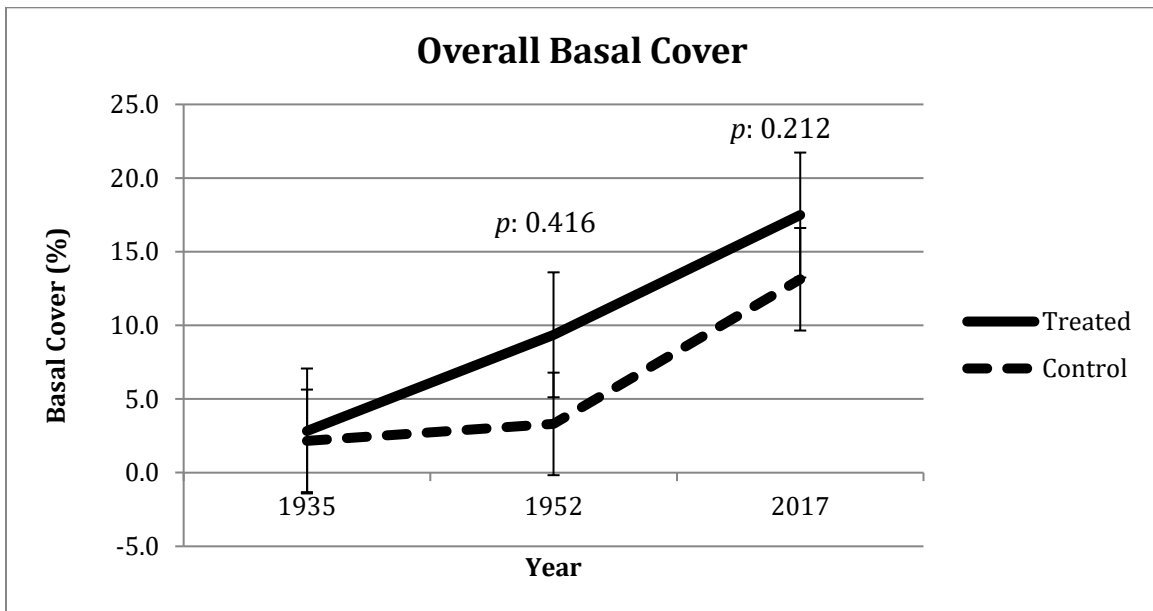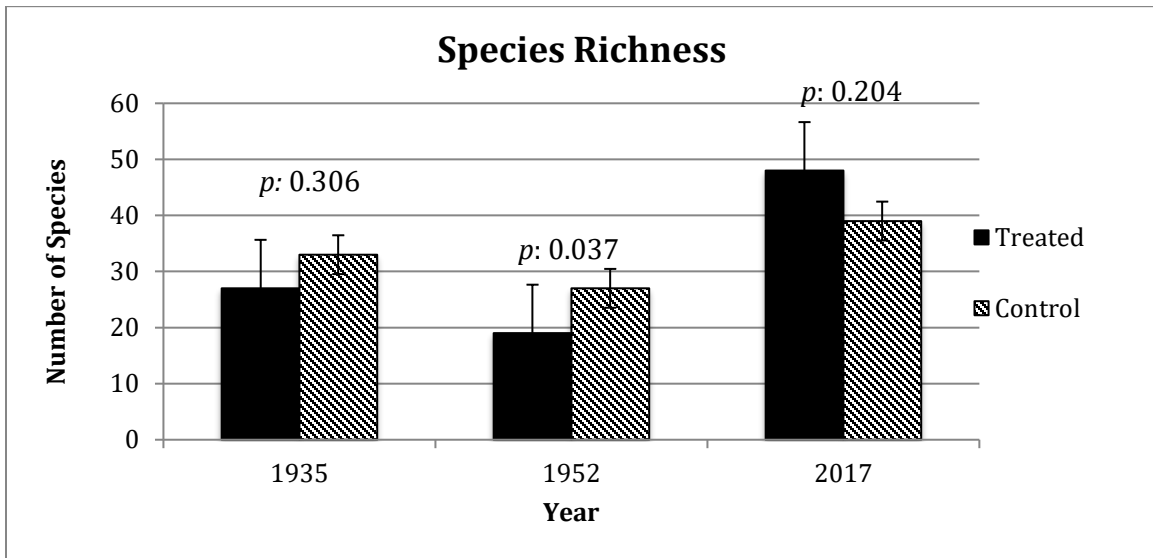
Figure 12. Overall basal cover (%) change among years 1935, 1952, and 2017 between treated (Drainage A and C) and control (Drainage B and D) drainages with standard error bars. P-values were calculated from a paired two-sample *t*-test. The initial herbicide treatment occurred in 1954.

Figure 13. Species richness change among years 1935, 1952, and 2017 between treated (Drainage A and C) and control (Drainage B and D) drainages with standard error bars. P-values were calculated from a paired two-sample *t*-test. The initial herbicide treatment occurred in 1954.
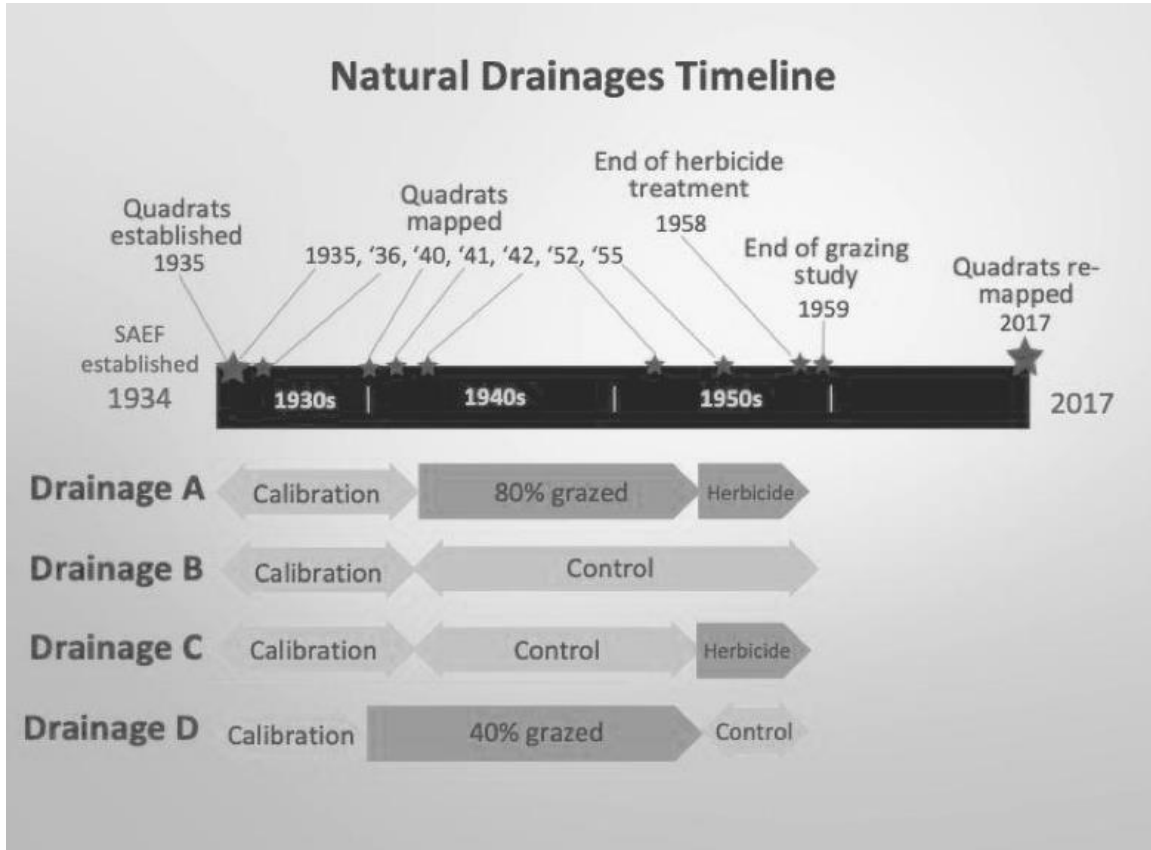
Appendix A. Figure 1. Natural Drainages timeline of experiments (1934-2017).

Appendix B, Table 1. Natural Drainages plant species list from 1-m2 quadrats only including 6-letter code, scientific and common name, photosynthetic pathway, nativity, and growth form. Plant scientific and common names follow SEINET database: (http://swbiodiversity.org/seinet/index.php)

Species code	Genus	Species	Family	Common Name	C3/C4	P/A[1]	Native/ Nonnative	Growth Form
ABUPAR	Abutilon	parvulum	Malvaceae	dwarf Indian mallow	C3	P	Native	Forb
ACMRIG	Acmispon	rigidus	Fabaceae	broom deerweed	C3	P	Native	Forb
AGAPAR	Agave	parryi	Agavaceae	Parry's agave	CAM	P	Native	Forb
ARCTSP[2]	Arctostaphylos	spp.#	Ericaceae	manzanita	C3	P	Native	Shrub
ARIADS	Aristida	adscensionis	Poaceae	sixweeks threeawn	C4	A	Native	Graminoid
ARIHAM	Aristida	hamulosa	Poaceae	Allred spidergrass	C4	P	Native	Graminoid
ARIPUR	Aristida	purpurea	Poaceae	purple threeawn	C4	P	Native	Graminoid
ARTDRA	Artemisia	dracunculus	Asteraceae	tarragon	C3	P	Native	Forb
BOTBAR	Bothriochloa	barbinodis	Poaceae	cane bluestem	C4	P	Native	Graminoid
BOTSAC	Bothriochloa	saccharoides	Poaceae	feathergrass or silver bluestem	C4	P	Native	Graminoid
BOUARI	Bouteloua	aristidoides	Poaceae	needle grama	C4	A	Native	Graminoid
BOUCUR	Bouteloua	curtipendula	Poaceae	sideoats grama	C4	P	Native	Graminoid
BOUERI	Bouteloua	eriopoda	Poaceae	black grama	C4	P	Native	Graminoid
BOUHIR	Bouteloua	hirsuta	Poaceae	hairy grama	C4	P	Native	Graminoid
BRORUB	Bromus	rubens	Poaceae	red brome	C3	A	Introduced	Graminoid
CEAGRE	Ceanothus	greggii	Rhamnaceae	desert ceanothus	C3	P	Native	Shrub
CYLSPI	Cylindropuntia	spinosior	Cactaceae	walking stick cactus	CAM	P	Native	Cactus
ELYELY	Elymus	elymoides	Poaceae	squirreltail	C3	P	Native	Graminoid
ERIDIF	Eriastrum	diffusum	Polemoniaceae	miniature woollystar	C3	A	Native	Forb

ERIANG	Eriodictyon	angustifolium	Hydrophyllaceae	narrowleaf yerba santa	C3	P	Native	Shrub
ERIABE	Eriogonum	abertianum	Polygonaceae	Abert's buckwheat	C3	A	Native	Forb
ERIWRI	Eriogonum	wrightii	Polygonaceae	bastardsage or white buckwheat	C3	P	Native	Halfshrub[3]
ESCVIV	Escobaria	vivipara	Cactaceae	spinystar cactus or pincuchion cactus	CAM	P	Native	Cactus
EUPH SP	Euphorbia	spp.	Euphorbiaceae	spurge	C4	A	Native	Forb
GLABIP	Glandularia	bipinnatifida var. ciliata	Verbenaceae	Davis mountain mock vervain	C3	P	Native	Forb
GUTSAR	Gutierrezia	sarothrae	Asteraceae	broom snakeweed	C3	P	Native	Halfshrub[3]
HYMWRI	Hymenothrix	wrightii	Asteraceae	Wright's thimblehead	C3	P	Native	Forb
LEPDUB	Leptochloa	dubia	Poaceae	green sprangletop	C4	P	Native	Graminoid
LYCSET	Lycurus	setosus	Poaceae	bristly wolf's-tail	C4	P	Native	Graminoid
MENSCA	Menodora	scabra	Oleaceae	rough menodora	C3	P	Native	Halfshrub[3]
MIMBIU	Mimosa	aculeaticarpa var. biuncifera	Fabaceae	catclaw mimosa	C3	P	Native	Shrub
MOLCER	Mollugo	cerviana	Molluginaceae	threadstem carpetweed	C4	A	Introduced	Forb
MUHMIN	Muhlenbergia	minutissima	Poaceae	annual muhly	C4	A	Native	Graminoid
NOLMIC	Nolina	microcarpa	Liliaceae	nolina; sacahuiste	C3	P	Native	Halfshrub[3]
OPUNSP	Opuntia	spp.	Cactaceae	prickly pear cactus	CAM	-	Unknown	Cactus
PANHIR	Panicum	hirticaule	Poaceae	Mexican panicgrass	C4	A	Native	Graminoid
PECPAP	Pectis	papposa	Asteraceae	chinchweed	C4	A	Native	Forb
PENTHU	Penstemon	thurberi	Plantaginaceae	Thurber's beardtongue	C3	P	Native	Halfshrub[3]
PLAPAT	Plantago	patagonica	Plantaginaceae	wooly plantain	C3	A	Native	Forb
PSECAN	Pseudognaphalium	canescens	Asteraceae	Wright's cudweed	C3	Both	Native	Forb

| SALV SP	Salvia	spp.	Lamiaceae	sage	C3	P	Native	Forb
SCHCIR	Schizachyrium	cirratum	Poaceae	Texas bluestem	C4	P	Native	Graminoid
SPOCRY	Sporobolus	cryptandrus	Poaceae	sand dropseed	C4	P	Native	Graminoid
VULOCT	Vulpia	octoflora	Poaceae	sixweeks fescue	C3	A	Native	Graminoid
XANGRA	Xanthisma	gracile	Asteraceae	tansyaster	C3	A	Native	Forb

[1]**P/A** = perennial vs. annual

[2]Arctostaphylos spp. = combined *A. pungens* and *A. pringlei*

[3]Halfshrub = subshrub
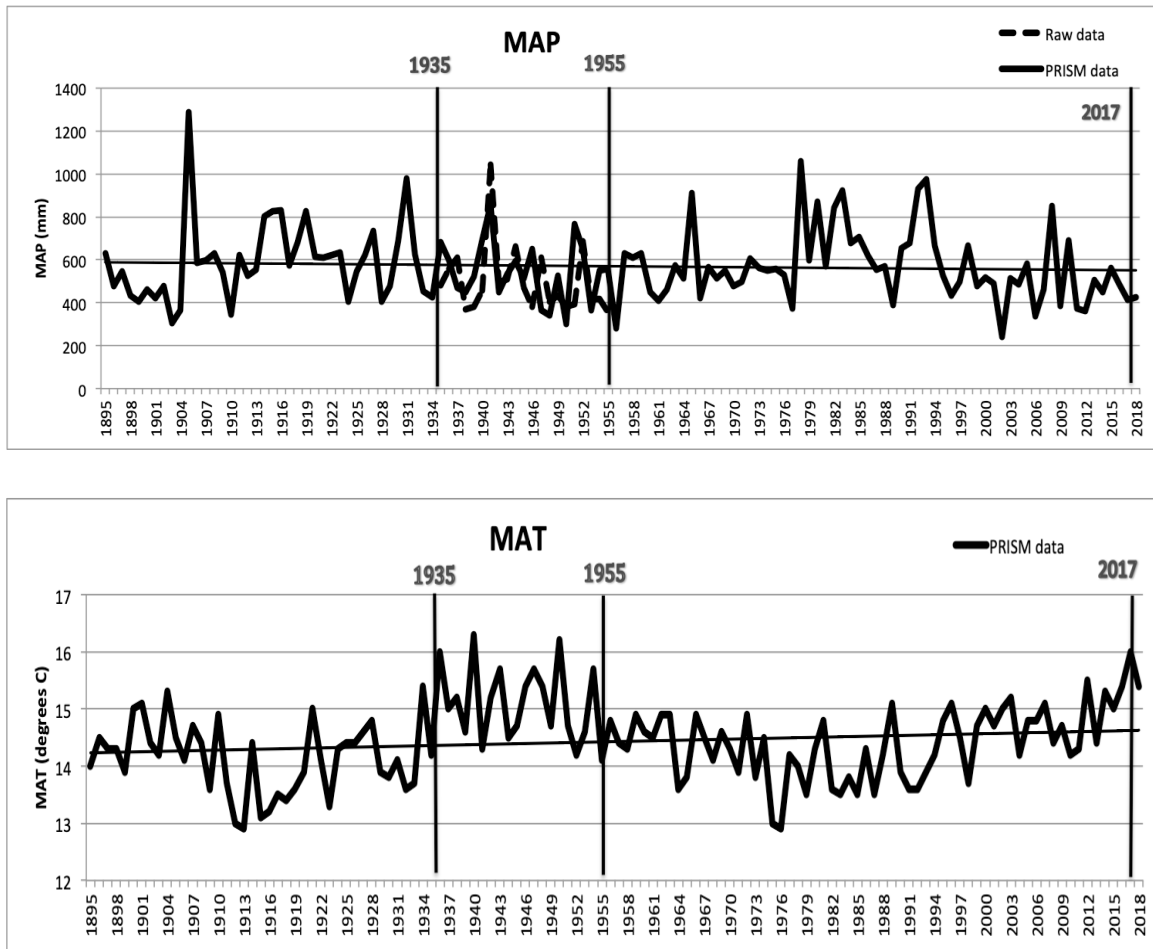
Appendix C. Figure 1. Natural Drainages climate data: mean annual precipitation (MAP) and mean annual temperature (MAT). Solid lines are PRISM data and dashed lines are raw climate data.
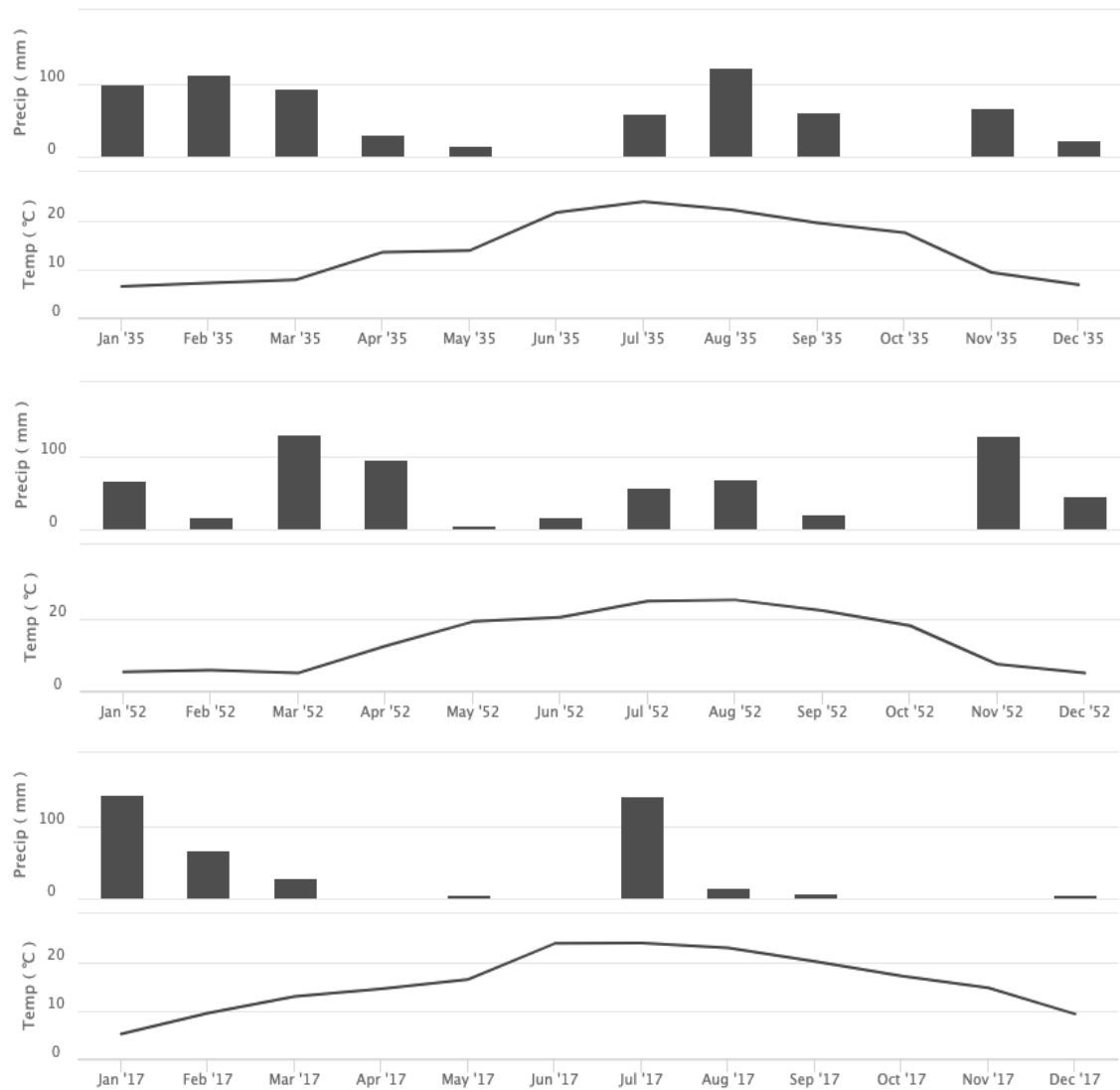
Appendix D. Figure 1. Study site mean monthly precipitation and mean monthly temperature in 1935 (top), 1952 (middle), and 2017 (bottom). Copyright © 2019, PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu Map created 09/23/2019.
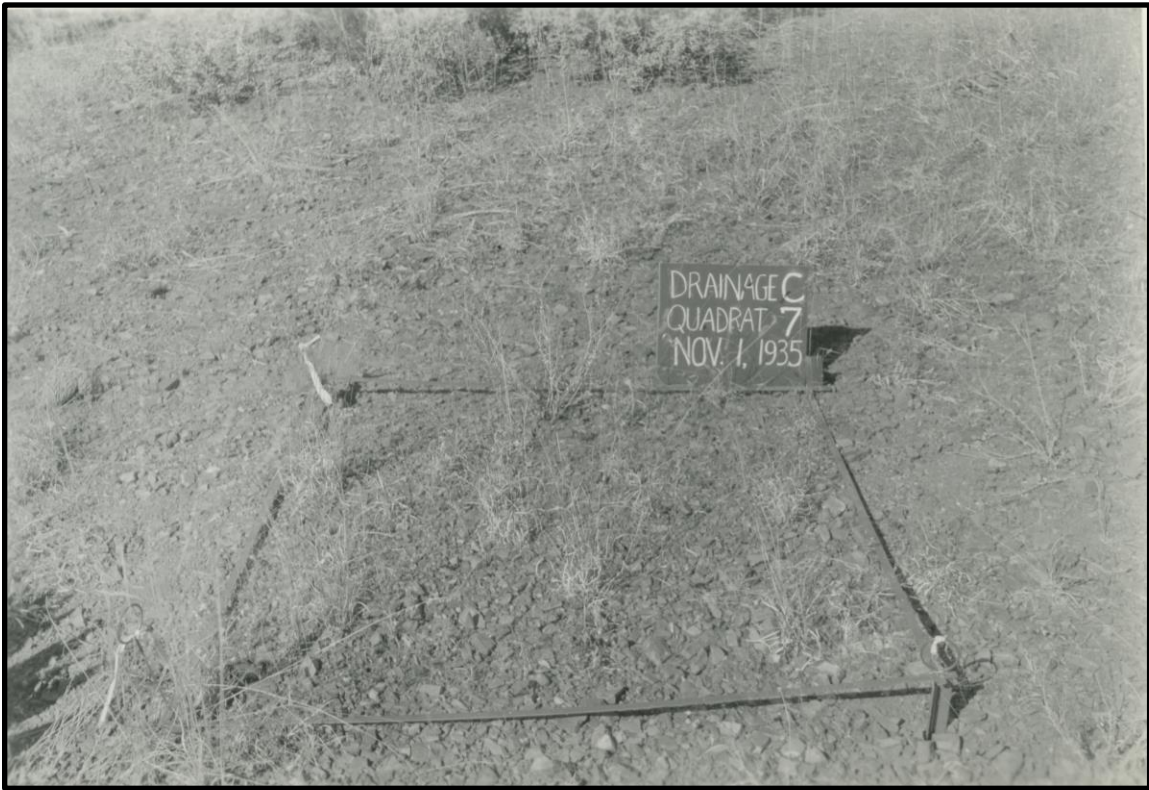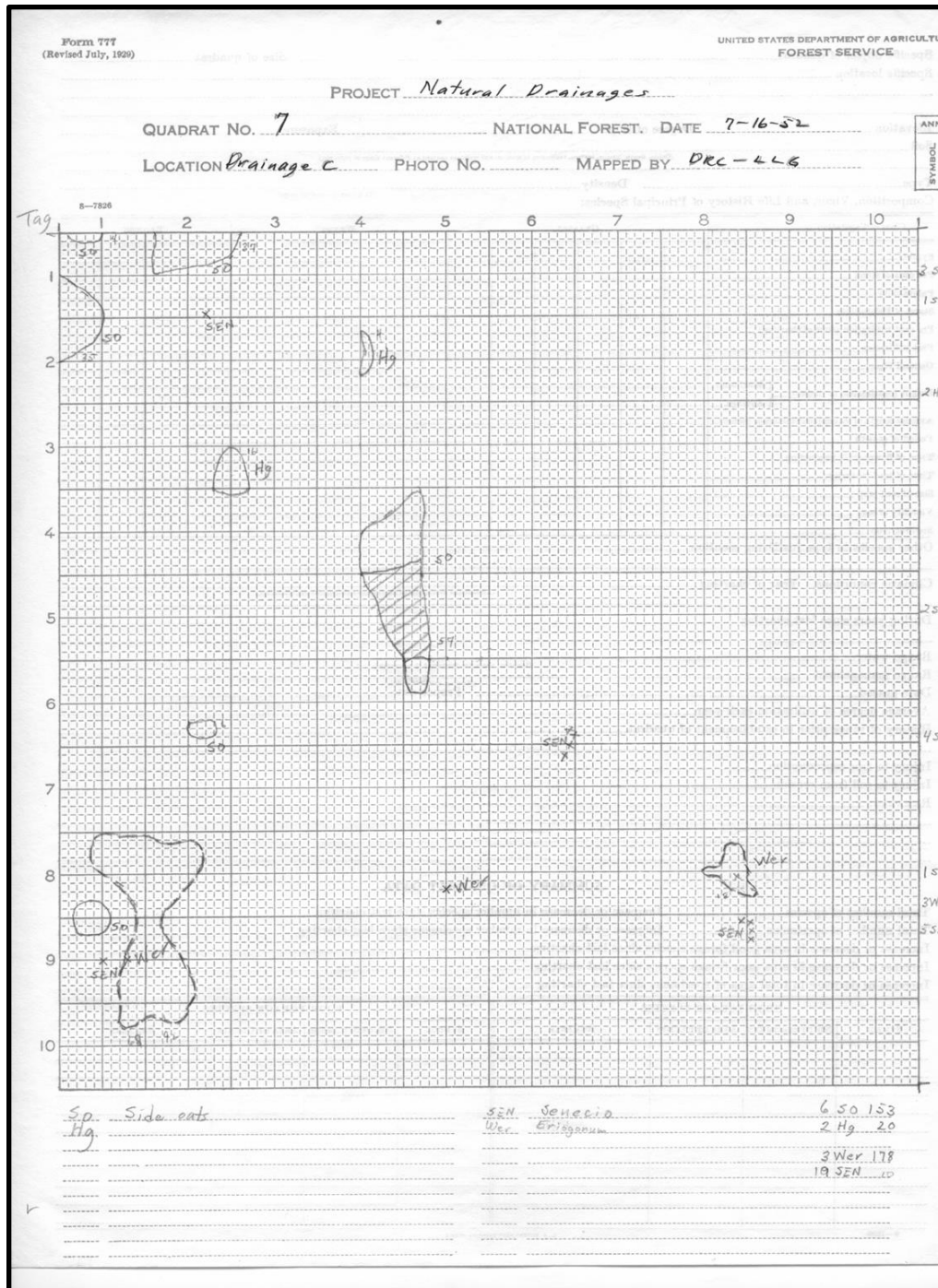
Appendix E. Figure 1. Photograph of 1-m^2 chart quadrat #7 in 1935. Photo credit: E.L. Little. The chart quadrat vegetation map in 1935 is located in Appendix E, Figure. 2.
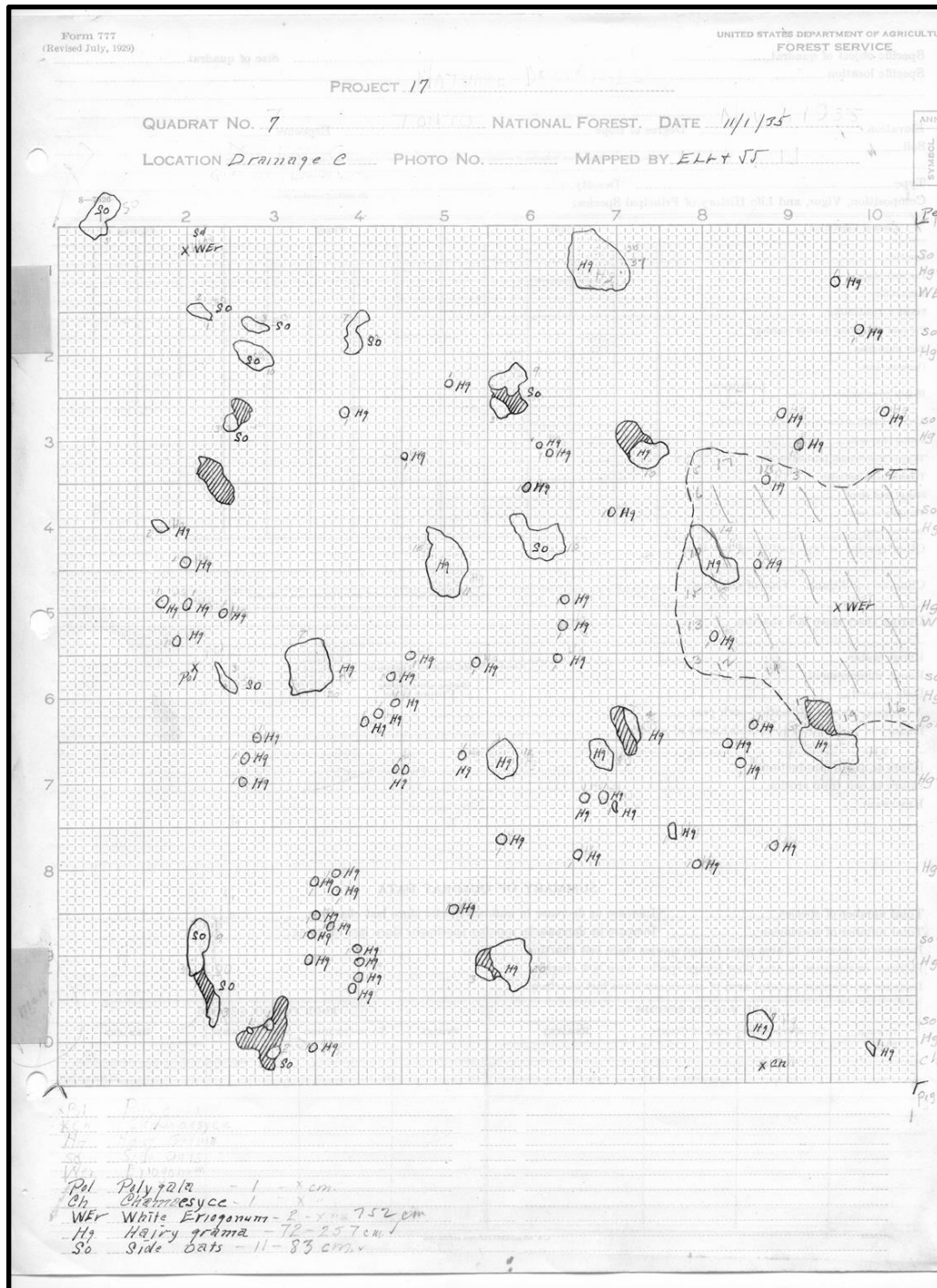
Appendix E. Figure 2. Chart quadrat #7 vegetation map in 1935. The photograph of quadrat in 1935 is located in Appendix E, Figure 1.

Appendix E. Figure 3. Photograph of 1-m^2 chart quadrat #7 in 1952. Photo credit: H.G. Reynolds. The chart quadrat vegetation map in 1952 is located in Appendix E, Figure. 4.

Appendix E. Figure 4. Quadrat #7 vegetation map in 1952. The photograph of quadrat in 1952 is located in Appendix E, Figure 3.
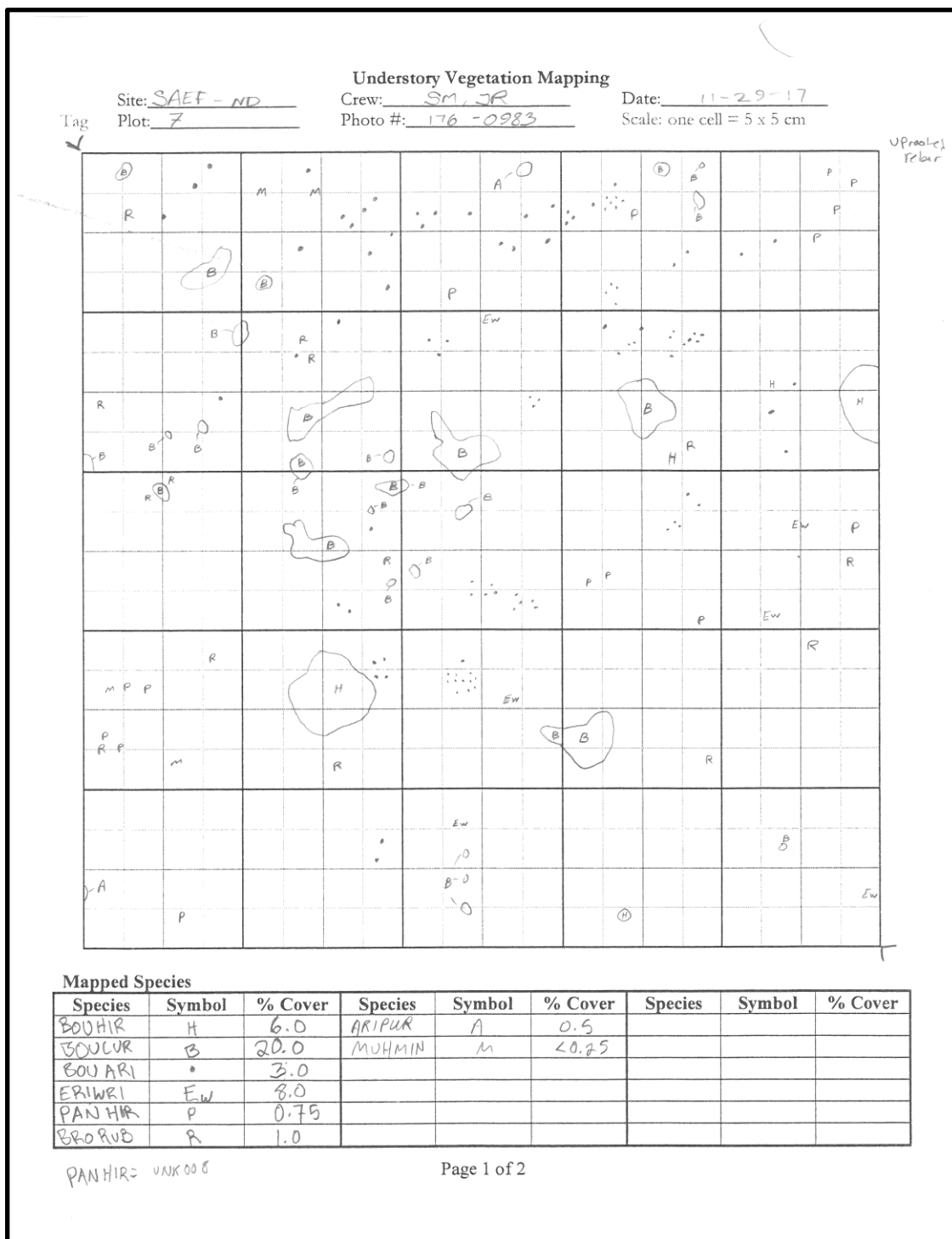
Appendix E. Figure 5. Photograph of 1-m^2 chart quadrat #7 in 2017. Photo credit: S.M. Massed. The chart quadrat vegetation map in 2017 is located in Appendix E, Figure. 6.

Understory Vegetation Mapping

Site: SAEF - ND   Crew: SM, JR   Date: 11-29-17
Tag   Plot: 7   Photo #: 176 -0983   Scale: one cell = 5 x 5 cm

Uprooted rebar

Mapped Species

Species	Symbol	% Cover	Species	Symbol	% Cover	Species	Symbol	% Cover
BOUHIR	H	6.0	ARIPUR	A	0.5			
BOULUR	B	20.0	MUHMIN	M	<0.75			
BOU ARI	•	3.0						
ERIWRI	Ew	8.0						
PANHIR	P	0.75						
BRORUB	R	1.0						

PANHIR= UNK 008   Page 1 of 2

Appendix E. Figure 6. Quadrat #7 vegetation map in 2017. The photograph of quadrat in 2017 is located in Appendix E, Figure 5.

Appendix F. Figure 1. Drainage C. Water flow on base rock drainage. May 1935. W.J. Cribbs.

Appendix F. Figure 2. Drainage D. Southeast side at end of grazing period. Oct. 3rd, 1942. B.A. Hendricks.

Appendix F. Figure 3. Drainage C. Northeast side looking west. Oct. 3rd, 1942. BA Hendricks.

Appendix F. Figure 4. Upper portion of Natural Drainage area. Diabase soil-type. May 1935. W.J. Cribbs.