# FINAL REPORT

## Determining prescribed fire and fuel treatment compatibility with semi-desert grassland habitat rehabilitation for the critically endangered masked bobwhite quail (*Colinus virginianus ridgwayi*)

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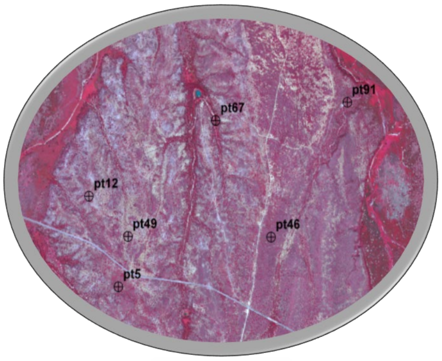

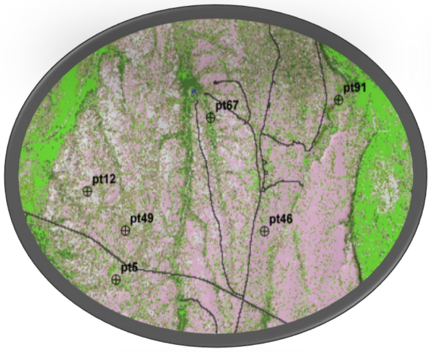

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## Project summary

The Buenos Aires National Wildlife Refuge (BANWR) in southern Arizona was established in 1985 to provide habitat for threatened and endangered plant and animal species, with an emphasis on the critically endangered masked bobwhite quail (*Colinus virginianus ridgwayi*). Livestock grazing, fire regime disruption, pronounced drought, non-native grasses and altered arid-land hydrology since the late 1800s have each played a substantial role in transforming semi-desert grasslands. Since refuge establishment and grazing cessation, prescribed fire has featured prominently as a tool for achieving desired habitat conditions. We used a designed study and quantitative methods to assess vegetation structure and composition resulting from soil, topography, and climate factors in conjunction with management approaches used to encourage quail habitat. We randomly located vegetation plots ($n$ = 239) within nine fire frequency and topographic strata to determine fire effects on habitat conditions important to masked bobwhites. We used field studies, multivariate and spatial data analysis to determine how fire and site biophysical conditions have led to present habitat conditions on BANWR.

Structural equation models (SEM) and ordination plots revealed that soil texture and climate gradients across the study area were important to determining fire-effects on habitat. From SEM, we found that fire can potentially produce short-term benefits for quail such as greater forb and herbaceous plant cover. Increased forb cover may last as little 2 or 3 years on sites quickly retaken by the non-native perennial grass *Eragrostis lehmanniana*. Leguminous shrubs and subshrubs important to masked bobwhite as food resources were in low abundance throughout the study area. Woody plants (trees and shrubs) were significantly and negatively affected by frequent fire according to SEM. Subshrubs important for both cover and food resources were more abundant on plots ($\bar{x}$ = 5.2%±5.9 cover), but were not wide spread and had a weak, but significantly negative association with site fire history. These outcomes were likely due to *Gutierrezia sarothrae*, the most common subshrub found on plots, primarily used as cover by quail. Other common seed producing leguminous subshrubs important as winter food for quail such as *Acacsia angustissima* and *Chamaecrista nictitans* were associated with drainage areas, but often absent on plots and had low correlation with fire history variables.

Plant interaction between *E. lehmanniana* and other native vegetation were critical to habitat conditions observed. We found that repeatedly burned sites were often those dominated by *E. lehmanniana* that showed higher fine-fuel concentrations, low plant diversity, and had

significantly lower habitat suitability for masked bobwhite quail. A spatial model of masked bobwhite habitat suitability indicated that areas with greater suitability ranging from 0.50 to 0.68 were located on infrequently burned sites, at the margin of fire management units. BANWR management units with the highest and most contiguously dense fine-fuel concentrations were those burned ≥4 time over a 30-year period. These areas were nearly devoid of suitable habitat for masked bobwhites and showed novel fuel conditions comprised of dense non-native grasses that can increase fire size and intensity. Non-linear models comparing fire history variables to masked bobwhite habitat suitability within BANWR management units showed a significantly negative relationship with fire frequency ($F$=14.8, $P$<0.001, $r^2 = 0.36$) and significantly positive relationship with the number of years since last burn ($F$=14.6, $P$<0.001, $r^2 = 0.35$). Conversely, locations with higher habitat suitability for quail tended to be on coarse textured soils with lower moisture holding capacity that were less suitable to *E. lehmanniana*. These sites were often associated with drainages protected from fire during prescribed fire activities that had few to no fires, likely because of low productivity and fine-fuel accumulation.

Fire will undoubtedly continue to play a role on BANWR particularly in locations where fuel-bed structure and plant composition is dominated by dense non-native grasses. Efforts to improve habitat quality for masked bobwhite quail on the refuge should consider greater protection for critical habitat areas and active management to improve shrub cover (winter forage species), connectivity and diverse foraging opportunities among sites with better habitat quality. These areas may require sufficient recovery time from previous burns, ranging from 15 to 20 years depending on site environmental factors and interannual climate variability. However, recovery timeframes are uncertain because a majority of areas with higher predicted habitat suitability (≥0.50) showed no burn history in the last 30 years. Overall, habitat suitability was low on the refuge. Small-scale management experiments are greatly needed to better determine requisite treatments that can encourage masked bobwhite habitat recovery and suitable conditions for survival and reproduction. Adaptive management approaches could benefit many aspects of quail habitat rehabilitation and prescribe fire use on BANWR. Work summarized from this project provide a foundation from which various types of management activities might help to promote and improve habitat conditions for quail, but will require improved follow-up and monitoring to verify anticipated outcomes.

## 1. Objectives

This research addresses BLM\JFSP Project Announcement No. FA-FON0013-0001 Task 6, "Compatibility of fire, fuels and rehabilitation treatments with threatened and endangered gallinaceous birds". Fire plays a key role in maintaining vegetation and fuel-bed conditions characteristic of frequent fire regimes in southwestern semi-desert grasslands (McPherson 1995). The Buenos Aires National Wildlife Refuge (BANWR) in southern Arizona (Figure 1) was established in 1985 to provide habitat for threatened and endangered plant and animal species, with an emphasis on the critically endangered masked bobwhite quail (*Colinus virginianus ridgwayi*). Fire management activities implemented at local- to landscape-scales must be compatible with specific habitat requirements for threatened and endangered gallinaceous birds. Nevertheless, prior land use, non-native plant invasions and novel fuel conditions can alter vegetation\fire dynamics and post-disturbance recovery that historically maintained semi-desert grassland conditions (D'Antonio and Vitousek 1992, Brooks et al. 2004).

To rehabilitate masked bobwhite habitat on BANWR, quantitative information were needed to assess vegetation structure and composition and site factors such as soil, climate and disturbance regimes in combination with land management approaches used to encourage suitable quail habitat. Since refuge establishment, prescribed and unmanaged (e.g. lightning or human caused) fire has featured prominently as a primary tool for achieving desired grassland conditions. Geiger and McPherson (2005) found that that cessation of livestock grazing and prescribed fire did not necessarily encourage native semi-desert grassland composition on BANWR. Moreover, land use history, site biophysical conditions, and range management activities prior refuge establishment likely mediate fire treatment outcomes aimed at habitat rehabilitation. In a review by Hernandez et al. (2006), implementing and understanding the role of habitat management was considered critical for recovering masked bobwhite populations in the US and Mexico. King (1998) also specified that retrospective studies were needed to better understand the role of prescribed burning on vegetation and recovering characteristics important to masked bobwhite quail.

This study focused on developing a quantitative assessment of fire effects and other factors influencing masked bobwhite habitat conditions on BANWR working in concurrence with refuge managers, species recovery team members, biologists, and fire management personnel. Our study objectives were to:

1) Develop field-sampling methods to determine long-term fire treatment effects on semi-desert grassland fine-fuel loads, vegetation composition and structure important to masked bobwhite quail survival and reproduction.
2) Assess how fire frequency and spatial and temporal pattern of burning can most likely achieve masked bobwhite habitat rehabilitation.
3) Compare field plots, masked bobwhite habitat conditions, and habitat suitability estimates according to fire management histories.
4) Determine how other interacting site biophysical factors such as annual variation in rainfall, soil substrates, terrain variability, and other edaphic factors may positively or adversely impact habitat conditions modified by fire.

We also sought to combine a field study of long-term fire effects on a semi-desert grasslands with other geospatial data to characterize habitat conditions at fine- to landscape-scales. This project began as a vegetation mapping exercise on BANWR prior to developing the JFSP fire-effects study. Therefore, we established field sampling and remote sensing techniques to assess habitat conditions in conjunction with JFSP project objectives to map land cover, fuel-types (e.g. grass, shrub, and tree cover), fine-fuels, and habitat suitability for the mask bobwhite. A portion of this report describes results from mapping efforts that have helped to improve understanding of fire management outcomes and quail habitat conditions on BANWR.

## 2. Background & Purpose

Masked bobwhite utilize a variety semi-desert grassland conditions throughout the year that provide cover for thermoregulation in Sonoran desert environments as well as diverse summer and winter foraging opportunities, suitable nest sites, and space to avoid or escape predators (Tomlinson 1972a, Guthrey et al. 2001, Hernandez et al. 2006, Brown et al. 2012). Semi-desert grassland ecosystems in the US and Mexico, particularly within the known historical range of the masked bobwhite, have undergone dramatic alternations in habitat composition and structure in since the late 1800s (Brown 1904, Bahre 1991). Historical and contemporary livestock grazing, fire regime disruption, pronounced drought, and altered arid-land hydrology have each played a substantial role in transforming semi-desert grasslands (Bahre and Shelton 1996). Consequently, non-native invasive plants introduced from Africa and the Mediterranean often dominate Sonoran desert grasslands (Van Devender et al. 1997). Land use, drought, and habitat changes contributed to the extirpation of masked bobwhite from the US by the early 1900s and eventual population declines on ranches in Sonora, Mexico where only a small number masked bobwhite were last detected in 2007 (Tomlinson 1972a, Brown et al. 2012).

Because of its rarity and rapid disappearance from the US historical range in southern Arizona, the masked bobwhite was one of the first species listed as endangered in the under the US Endangered Species Act in 1967 (https://ecos.fws.gov/ecp0/profile/speciesProfile?sId=3484). In the more extensive portion of its range in northern Mexico, the masked bobwhite was also listed as an endangered species in 1994 as habitat conditions and populations declined with intensified cattle grazing since the 1930s (García-Solórzano et al. 2017, Brown and Clark 2017). Brown and Clark (2017) considered that the masked bobwhite may now be functionally extinct in the wild, while others suggest that greatly diminished populations may exist in isolated portions Sonora, Mexico according to unconfirmed sightings and recent audio recordings (García-Solórzano 2017, Johnson, personal communication).

Considering its status, there is reason for pessimism about masked bobwhite recovery throughout its range. Yet we prudently considered that some former habitat areas show signs of recovery (Brown et al. 2012) and a concerted effort is being made to provide U.S. Fish and Wildlife Service staff, resources, and guidance by experts towards habitat improvement, detection in the wild, and quail re-establishment on BANWR. Among other goals, the 1995 masked bobwhite recovery plan (USFWS 1995) defines actions needed on BANWR must seek to 1) establish a viable self-sustaining population of 500 birds and 2) implement habitat management in order to maintain and increase the existing population.

With the present study, we sought to determine how fire management, site biophysical, and historical factors have shaped habitat conditions on BANWR. Our goal was to help inform where and how masked bobwhite habitat and species recovery can be achieved on BANWR and other perspective re-establishment areas. At the time of this study, no wild or captive-bred masked bobwhite quail were present in the field at BANWR because poor establishment success and suspended releases during a restructuring of the recovery program. Therefore, we established vegetation and habitat plots in place of applying avian survey techniques aimed at estimating habitat relationships through animal occurrence, density or survival (Buckland

2006). Further, we used multivariate, structural equation modeling (SEM), and machine learning approaches to develop and test assumptions about factors driving vegetation conditions on BANWR important for masked bobwhite re-establishment.

We examined fire management activities aimed at rehabilitating masked bobwhite habitat, beginning with the 1985 establishment of BANWR. In many ways, BANWR provides an important case study, where intensive century-long grazing and cessation of a historically frequent fire regime were replaced by renewed fire activity to restore habitat (Geiger and McPherson 2005). Geiger and McPherson (2005) previously observed no net effect of fire on native or non-native vegetation on BANWR from plots re-measured in 1987, 1989, 1993, 1997, and 2002. Instead, they considered precipitation and soil factors as the principal diver of changes in vegetation cover, although analyses were from a small number of 30-m vegetation transects (*n* = 36) lacking a formal study design. In addition, few studies have conducted a quantitative investigation of site biophysical and spatially explicit factors that likely influence management outcomes in semi-desert and subtropical grasslands (Cox et al. 1988, Geiger and McPherson 2005).

To the degree possible, we used qualitative information on prior land use and how climate variability is likely to mediate current and future habitat conditions. Historical livestock grazing is often cited as a primary change agent in semi-desert and subtropical grasslands (McPherson and Weltzin 2000). Other commonly cited factors include active removal of woody plants such as *Prosopis velutina,* and seeding of non-native grasses (e.g. *Eragrostis lehmaninanna* and *Cencharus ciliaris*) for erosion control and animal forage (Bock et al. 1986, Cox and Ruyle 1986, McClaren and Angell 2006). Spatial and temporal variation in precipitation patterns is also a primary ecosystem process in arid systems, interacting with land use factors (Noy-Meir 1973). Bonder and Robles (2017) found that interannual variation in precipitation coupled with drought and land use history often drive mortality and reestablishment of perennial desert grasses, a primary vegetative component of quail habitat.

Understanding the synergies between anthropogenic, and biophysical factors, often involving novel disturbance regimes, requires new modes of analysis to uncover complex causal relationships. Structural equation models (SEM) allowed us to test hypotheses regarding the effect of principal management activities and environmental variables on masked bobwhite habitat characteristics. SEMs are an intuitive graphical and quantitative representation of interactions between dependent and explanatory variables (Grace 2006). We used this and other multivariate approaches to interpret management histories, such as the use of prescribed fire relative to topo-edaphic and climate gradients, at a landscape scale.

As a basis for hypotheses regarding fire and other effects on habitat characteristics, we provide essential background information in Appendix A on masked bobwhite historical collections, habitat ecology and management as a foundation for SEMs is described in methods below. A more comprehensive early review of masked bobwhite ecology and conservation can be found in Tomlinson (1972a), while Hernandez et al. (2006) and Brown and Clark (2017) provide further historical details and an overview of masked bobwhite ecology.

We briefly describe masked bobwhite habitat conditions and food resources as follows. Goodwin and Hungerford (1977) broadly characterized habitat as consisting of dense vegetation (75% to 100% ground cover) with relatively high plant diversity. Goodwin (1982) found that reintroduced masked bobwhite commonly used bottomland habitat on BANWR. With few exceptions, Goodwin did not observe masked bobwhite in areas with less than 10 to 12 species of grasses and forbs, finding that areas supporting the best quail habitat had 18 to 20 plant species. Sites with less than 10% grass cover were avoided and preferred habitat had 22% to 30% combined grass and forb cover. Based on discussions with Goodwin, Reichenbacher and Mills (1984) provided more specific habitat attributes for masked bobwhites. They described suitable habitat as 10% to 15% woody plant cover, 12% to 15% grass cover, and 10% to 12% forb cover, with at least 450 kg/ha of grass standing crop, 300 kg/ha of forb standing crop, and 20 grass and forb species. Simms (1989) reported a similar finding for captive-released masked bobwhite on BANWR. Simms (1989) characterized habitat for masked bobwhites as consisting of 10% canopy cover of woody plants, 50% canopy cover of grass, and 15% canopy cover of forbs. Adequate diversity of grasses and forbs consisted of at least 10 species each.

Early diet studies provide insight into more specific food plants (Hernandez et al. 2006). Cottam and Knappen (1939) reported a diet consisting of 79.1% and 20.1% plant and animal material respectively, from stomach contents of 10 masked bobwhites collected in Sonora, Mexico. They found seeds from a variety of plants including acacia (*Acacia angustissima*), ground cherry (*Physalis spp.*), panic grasses (*Panicum* spp.), day flower (*Commelina elegans*), and partridge pea (*Cassia leptadenia,* or likely *Chamaecrista nictitans* using current taxonomy). Insect material consisted primarily of grasshoppers (*Orthoptera*). Tomlinson (1984) concluded that an abundance of seed-producing plants such as legumes and panic grasses, in conjunction with an abundance of insects, was an important habitat component for masked bobwhites.

## 3. Methods & Materials

### 3.1. Study site

The study area encompasses the 48,000-ha Buenos Aires National Wildlife Refuge (BANWR) that was established in 1985. BANWR is located in the Altar Valley of southern Arizona boarding with Mexico that is situated between several small mountain ranges running north and south to the Mexican border (Figure 1). The refuge was principally established to facilitate recovery of the critically endangered masked bobwhite quail within the US and northern portion of its range (USFWS 1995). BANWR itself is sub-divided into 84 separate habitat management units, of which 60 are specifically identified for prescribed burning purposes. Overlapping management units are distinct management zones developed for focusing masked bobwhite habitat restoration activities in suitable areas. The 9,300 ha masked bobwhite management zone covers priority habitat areas and drainage networks at lower elevations in the Altar Valley. An extended masked bobwhite zone encompasses an additional 8,600 ha occupying slightly more upland sites that likely contained suitable habitat conditions historically.

Climate is semiarid with low precipitation and humidity and high summer temperatures. Temperatures range from -11 ˚C in winter to 41 ˚C in summer. Summer and winter temperatures average 33 ˚C (May – Sept.) and 18 ˚C (Dec. – Feb.) respectively. Precipitation averages 42 cm annually and is bimodal with approximately 40% occurring during July and August and the remaining during winter according to the Sasabe, AZ weather station (http://www.wrcc.dri.edu/) that is located at the extreme southern end of BANWR. Anvil Ranch weather station, 19 km to the north, shows a much lower annual precipitation (29 cm) indicating an increasing north to south precipitation gradient. Worldclim bioclimatic data layers also show an increasing north to south precipitation gradient and decreasing temperature gradient, with relatively cooler and wetter conditions in the southern portion of BANWR (Figure 2). The valley floor is intersected by numerous dry washes that create level to mountainous terrain ranging for elevations between 925 m to 1,400 m. Soils are well drained and coarse to moderately fine textured Aridisols and Mollisols (Hendrix 1995).

Historical land use patterns and hydrologic changes have resulted in a mixture of native and non-native vegetation on BANWR (Robinett 1992, Geiger and McPherson 2005). *E.*

*lehmanniana* currently dominates most upland sites, in addition to other non-native grasses including *Eragrostis chlormelas* and *Eragrostis superb*. Common native grasses include *Boutaloua* spp., *Sporobolus* spp., *Aristida* spp., *Bothriochloa barbinodis* and *Digitaria californica*. Deep soils in drainages and disturbed bottomlands include *Sorghum halepense*, *Sporobulus* spp., *Amaranthus palmeri*, and *Salsola kali*. *P. velutina* is the dominant tree on BANWR, but other species such as *Celtis* spp. and *Acacia* spp. are also common. Other common woody shrubs and subshrubs include *Isocoma tenuisecta*, *Gutierrezia sarothrae*, *Caliandra eriphylla*, *Chamaecrista nictitans*, *Acacia* spp., *Mimosa* spp. and *Atriplex canecense*. The largest plant family is Asteraceae, with a marked diversity of annual and perennial forbs and subshrubs. Upper elevations between 1,200m and 1,800m areas contain areas of dense shrublands such as *Mimosa Dysocarpa* and perennial graminoids giving way to Madrean evergreen woodlands dominated by *Quercus oblongifolia*, *Juniperous deppeana* and *Pinus engelmannii.*

Since refuge establishment, prescribed fire has been used as the main management tool, in addition to some mechanical treatments to remove *P. velutina* for improving habitat conditions for the masked bobwhite and, more recently pronghorn (*Antilocarpa americana*). A majority of management units at low elevations in the central portion of the refuge have been burned at least once since 1985, with several of them being purposefully burned or by wildfires between 3 and 5 times between 1985 and 2015.

### 3.2. Data collection and consolidation

#### 3.2.1. Grassland vegetation and fine-fuels

We collected vegetation cover data from 20m x 50m rectangular plots on BANWR using stratified random sampling design during two separate periods. Vegetation plots established across the refuge prior to the JFSP project were measured between July and October of 2012 and 2013 (*n* = 207) were within vegetation strata defined by dominant life form classes (i.e. trees, shrubs and herbaceous plants), and three elevation categories roughly classed as valley bottom, foothills and Madrean physiographic environments (Figure 1). On each plot, we measured 6, 20-m point intercept transects 10m apart recording plant species and soil substrate at 0.5m intervals (*n* = 240 intercepts per plot). Species intercepted were recorded in three height classes there between 0.0 – 0.5m, 0.5m – 2.0m, and >2.0m. Plot locations were also stratified by low, medium and high productivity categories based on the normalized difference vegetation index (NDVI) from a 2011 Landsat Thematic Mapper (TM) satellite image taken during the peak growing season. The area between transects was also surveyed to record plant species occurrences for vegetation not recorded on transects.

Vegetation plots specific to the JFSP project were measured between July and October of 2014 and 2015 ($n$ = 239) and used the same measurement techniques as above, but with the addition of fine-fuel quadrats and soil sample collection. As such, we confined plots to the masked bobwhite management zone and extended management zone in the valley bottom (Figure 3). These plots were stratified based on fire history and local hill-slope position, which can influence soil physical properties and site moisture regime as well as disturbance characteristics such as fire behavior (McPherson and Weltzin 2000). Specifically, we used the USFWS Fire Atlas and a record of fire perimeters (polygons) measured between 1985 and 2015 to estimate fire frequency on BANWR. Perimeter data were converted to a raster in a geographic information system (GIS) representing the location of fires for each year that was used to calculate the number of fires occurring within a 30m grid cell over a 30-year period. Fire frequency was divided into low (0-2 fires), medium (3-4 fires) and high (≥5 fires) strata. We used 10m digital elevation model (DEM) from the USGS National Elevation Dataset to calculate topographic wetness index (TWI) values using Taudem v. 5.0 software (http://hydrology.usu.edu/taudem/taudem5/index.html) executable files in the R statistics package v. 3.1.1 (R Core Team 2013). We categorized three terrain classes from TWI values as drainage, ridges and steep slopes, and footslope classes using a detailed DEM and hillshade map developed from 2007 airborne laser altimetry (LiDAR) tiles that were limited to borderland areas (https://earthexplorer.usgs.gov/). The R statistics 'sampling' package v. 2.8 (Telí and Matei 2014) was used to create an equal number stratified random plots within fire frequency and terrain classes.

To anticipate JFSP project needs, we used double sampling approach to measure fine-fuels during 2013 to develop non-destructive measurements to improve sampling efficiency during the 2014- and 2015-field study. Therefore, grassland fuels were first sampled on $n$ = 20, 20m x 50m plots within 24, 0.5m x 0.5m quadrats spaced 5m apart along 6, 20 m transects. Destructively sampled quadrats were measured on plots during August of 2013, the month typically at or near the peak of the growing season. All herbaceous plants within quadrats were clipped to the ground and taken to a lab to be oven dried at between 60 and 70°C for at least 48 hours and weighed to the nearest 10th of a gram. We also recorded the height and percent cover of grasses, forbs, cacti, and woody plants in each quadrat. Herbaceous plant cover was visually estimated on quadrats and average canopy height was measured with a metal tape to the nearest centimeter.

Prior to plant measurements and clipping, we used an AccuPAR LP-80 ceptometer to record leaf area index (LAI) at five locations spaced 10cm apart inside the quadrat. An external sensor and the ceptometer were used together to simultaneously measure above and below canopy photosynthetically active radiation (PAR) between 400 nm and 700nm, at a resolution of 1 µmol $m^{-2}$ $s^{-1}$ within a range of 0 to 2,500 µmol $m^{-2}$ $s^{-1}$. Above and below canopy PAR measurements in addition to sun zenith angle ($z$), fraction of beam radiation ($F_b$) and a leaf distribution parameter ($X$) set to 1 for desert grasses were used to instantaneously calculate LAI. The mean LAI value per quadrat was then calculated from the five samples and stored on the ceptometer control box. Occasionally, shaded quadrat locations or those showing >10% woody plant cover were relocated to the next 1m interval along a transect to avoid shading impacts on herbaceous LAI measurements. Quadrat measures were largely taken on the east side of transects prior to noon and the west side of transects in the afternoon to avoid observer shadows over quadrats.

We then used 2013 destructive samples to parameterized herbaceous biomass regression tree models that were used to non-destructively estimate herbaceous biomass on plots measured during years 2014 and 2015 ($n$ = 229). Pending sufficient biomass model performance, all future quadrat samples used only ceptometer LAI, herbaceous plant canopy height and visual cover measurements to estimate fine-fuels with procedures described above.

All plot corners were georeferenced in the field with a Trimble GeoXT or Geo7X and post-processed to differentially correct point locations to within 1m positional accuracy using Trimble Pathfinder Office v.5.60 (Trimble Navigation Ltd. 2013).

### 3.2.2. Soils

For plots measured during 2014 and 2015, we collected 18 soil samples along plot transects at a 10cm depth to assess soil chemical and physical properties important to plant composition and structure. We cleared organic material from the soil surface prior to sampling. Soil collected on each plot was batched together for laboratory analysis of texture, nutrients, base cations (Ca, Mg, K, Na), organic material, $N0_3$, $NH_4$, pH and plant available phosphorous ($PO_4$), that are associated with hydrology, plant growth, disturbance factors and fuel characteristics on a site (Boerner 1982, Schlesinger et al. 1999). One soil bulk density measurement was sampled using the volume extraction method (Litcher and Costello 1994) at each vegetation plot, to assess potential compaction from prolonged grazing prior to refuge establishment.

Soil chemical analyses were performed by the Colorado Plateau Analytical Laboratory at Northern Arizona University. Soil samples were sieved (2mm) prior to analyses. Organic carbon was estimated from loss-on-ignition at 550°C for 5 hours in a muffle furnace; Ammonium ($NH_4$) and Nitrate ($NO_3$) KCI extraction via Lachet Colorimetry pH (1:1 soil M CaCl2). I considered the sum of $NH_4$ and $NO_3$ to be an index of the total available inorganic nitrogen. Soil particle size distribution was determined by laser diffraction with a Coulter LS 230 Particle Size Analyzer by Sedimentary Records of Environmental Change Lab at Northern Arizona University. Laser diffraction was used to determine the clay (<2 mu m), silt (2-20 mu m), sand (20-2000 mu m) fraction content represented as percentages of 5g of soil using the United States Department of Agriculture (USDA) particle size classification scheme, clay (0-2um), silt (2-50 um), and sand (50-2000 um). Soil samples were also characterized into their respective soil textural classes following the USDA soil classification system based on grain size distribution and soil separates classes for sand (Soil Survey Division Staff 1993).

### 3.2.3. Spatial data layers

We used three primary spatial data sources to characterized disturbance history, climatic conditions and site biophysical environments at plot locations in the study area. Fire history data layers were the primary source of disturbance information using a thirty-year USFWS Fire Atlas record (1985 to 2015) of fire frequency and the number of years since the last fire on a site (‘time since burn’ or TSB) calculated at a 30m grid cell size for the study area. Secondly, we used 19 Worldclim biomclimatic data layers representing temperature and precipitation from 1960 to 1990 across the study area at a 1-$km^2$ grid cell size (http://www.worldclim.org/bioclim). Bioclimatic variables were highly correlated with one another such that only precipitation for the wettest quarter during the summer growing season was used for SEMs. The 10-m DEM used to develop TWI for sampling stratification was used to generate other biophysical variables such as the total annual solar radiation for sample locations. Solar radiation ($w/m^2$) is strongly influenced by topographic and surface features such as elevation, surface orientation, and slope. ‘Global’ or total annual solar radiation ($W/m^2$) and elevation (m) values were developed using the 10-m DEM and Spatial Analyst tools in ArcMap v. 10.2.1.

In addition, we developed spatial data layers for herbaceous fine-fuel, vegetation cover and fuel-type using Random Forest regression and classification tree models (Brieman 2001) and remotely sensed data. Worldview-3 satellite imagery (8-spectral bands, 2m pixels) from 2015 representing peak-green or 'leaf-on' (August) vegetation and senescence or 'leaf-off' (June) periods were used in conjunction with georeferenced field plots to develop and validate each data layer. We used these data to describe and compare contemporary fuel and habitat conditions on BANWR. As follow-on work, we used fine-fuel and fuel-type data layers to develop appropriate fuel models following Scott and Burgan (2005) to assess potential fire behavior and estimate habitat suitability for masked bobwhite quail at a landscape scale (i.e. BANWR). Methods and results from this work is further described in Sesnie et al. (2018) and Eagleston et al. (*in revision*) however we touch on results from these two studies that helped assess vegetation conditions important for assessing masked bobwhite habitat on BANWR.

#### 3.2.4. Historical information

Where possible, we use qualitative and quantitative historical data describing management and treatment effects on vegetation prior to refuge establishment to help interpret analysis and model results and long-term changes in habitat conditions on the refuge.

### 3.3. Analysis methods

#### 3.3.1. Habitat characteristics

We predicted that habitat characteristics important to masked bobwhite recovery would covary according to fire management histories and biophysical conditions that have developed since refuge establishment. To determine how fire treatments may influence long-term habitat conditions important to masked bobwhite, we used SEMs that were appropriate for investigating complex ecological relationships (Grace et al. 2010). SEMs simultaneously account for the role of multiple factors that are likely to influence masked bobwhite habitat characteristics (Grace and Keely 2006). Conceptualized relationships between habitat conditions and each factor were first used to develop SEM 'base' and 'extended' models (Figure 4). Therefore, we examined fire effects and site biophysical factors (e.g. fire frequency, time since last burn, inorganic nitrogen, precipitation variation, and soil texture) in relation to one another and habitat conditions.

Following our conceptual model, we built baseline SEMs for each of six habitat components (i.e. plant diversity, species richness, and forb, graminoid, shrub and subshrub, and tree percent cover) using the lavaan package v. 0.6-3 (Rosseel 2012) for  in R statistical software v. 3.5.1 (R Core Team 2018). For each model, we tested model relationships first for all plant species of a given habitat characteristic and second for only plant species described as important to masked bobwhite foraging, hiding and thermal cover, according to literature on habitat preferences (Simms 1989, LaRoche and Conway 2013). We used SEM model connections shown in graphical form to indicate hypothesized unidirectional causal relationships between exogenous (external) and endogenous (internal) explanatory variables and each habitat component. This approach allowed us to analyze the net effects of disturbance, biotic and abiotic factors on specific habitat conditions. We built models by adding in all directed causal relationships specified in *a priori* models referred to as 'base models'. Abiotic predictor variables in base models models that were significantly correlated were removed from the model when correlations were > 0.90 (Grace 2006).

To interpret SEM results, we considered path coefficients (λ) mathematically equivalent to partial correlation coefficients. To estimate model fit, we evaluated the maximum likelihood $X^2$ goodness of fit index, Tucker-Lewis Index (TLI), Comparative Fit Index (CFI) and root mean square of approximation (RMSEA). When all indices showed an adequate model fit (McDonald and Ho 2002), this indicated that the hypothesized model suitably described the relationship between habitat characteristics, disturbance, and site biophysical factors. We used published guidelines to interpret model fit indicated by each of three indices (i.e. TLI, CFI and RMSEA) to determine the degree to which as model structure and covariates adequately describe factors influencing habitat conditions (Schreiber et al. 2006).

We developed slightly more complex models termed 'extended models' to compare with base models, using the difference between Akaike Information Criteria (AIC) values as an indicator of model improvement. These models were an extension of base models that used composite site variables in some cases (e.g. ratio of percent sand to silt) to help improve understanding of factors driving habitat conditions.

We also assessed differences in plant community composition and significant disturbance and biophysical factors across all plots using Bray-Curtis dissimilarity and non-metric multidimensional scaling (NMDS). For NMDS ordinations, we used the MetaMDS function in the 'vegan' package v. 2.4.4 in R statistical software (Oksanen 2017). For these analyses, we used cluster v.2.0.7-1 (Maechler et al. 2017) and 'vegan' packages in R statistical software v. 3.5.1 (R Core Team). As a maintaining diversity of grasses and forb were important to managing for masked bobwhite habitat, we evaluated the effect of dominant non-native grasses on habitat diversity by regressing *E. lehamanniana* cover against Shannon's diversity index values (Magurran 2004).

### 3.3.2. Fine-fuel and fuel-type

We assessed fine-fuel biomass and fuel-type from plots that were then used to predict landscape-scale variables used as inputs to masked bobwhite habitat suitability and fire-behavior models. For this study, we first established plot-scale methods to quantify herbaceous biomass and secondly combined these measurements with satellite imagery. We used destructively sampled biomass from all effectively sampled quadrats ($n = 431$) on 20 plots measured in August of 2013 to develop and test predictive biomass models. We evaluated which variables measured on quadrates were most important to accurately predict herbaceous biomass and which non-destructive measures could accurately estimate fine-fuel loads. Therefore, Random Forest (RF) regression tree models (Breiman 2001) were used to predict dry weight (g/m$^2$) herbaceous biomass (i.e. grasses and forbs) from explanatory variables that were LAI, percent cover and average height for each plant life-form. Random Forest classification and regression trees, often used with high-dimensional data, provided a flexible approach to both estimate fuel parameters on plots and map fuel-types from remotely sensed data that is described below.

Random Forest is a 'tree-based' machine learning method that uses multiple bootstrap samples of the data with replacement to train classification and regression models (Breiman 2001). Samples held out of training, typically one-third, were then used to evaluate model performance using the root mean squared error (RMSE) and the proportion of variance explained for regression. Overall and percent class error was used to evaluate classification model accuracy. Performance measures calculated for model iterations were aggregated at the end of training to assess error. With this study, we used the 'caret' package v. 6.0-76 (Kuhn 2017) for classification and regression model training and recursive feature elimination (RFE) available for R statistical software to develop RF plot-scale biomass models. RFE is a backwards variable selection process that progressively eliminates the least important predictors (Bazi and Meglani 2006). For developing final models, an optimized number of predictors were selected based on the lowest RMSE obtained from 10-fold cross validation. We used a model tuning algorithm to optimize the RF parameter ‘mtry’ which is the number of predictors randomly selected for each node.

The recent increase in the number and variety of private and public sector satellite remote sensing systems provide an enhanced means of estimating grassland fuel parameters. Worldview-2 and -3 sensors are capable of 1- to 4-day revisit times making appropriate timing of image capture more feasible. This is particularly beneficial in grassland ecosystems, which can exhibit strong spatial and temporal differences in plant productivity in conjunction with precipitation patterns or disturbance (Oesterheld et al. 2001, Huxman et al. 2004). For this study, we used multi-date Worldview-3 (WV3) imagery, a commercial system, captures spectral reflectance data in the coastal (400nm - 452nm), blue (448nm – 510nm), green (518nm - 586nm), yellow (590nm - 630nm), red (632nm – 692nm), red-edge (706nm – 746nm), near-infrared1 (772nm – 890nm) and near infrared2 (866nm – 954nm) ranges at a 1.24m pixel size. Therefore, RF and RFE methods described above for generating plot biomass estimates were further used to predict fine-fuels, fuel-type (e.g. grass, shrub, tree cover) and land cover from WV3 spectral bands and indices.

For developing RF fine-fuel and fuel type models with WV3 imagery, we used herbaceous biomass measurements available from 2014 and 2015 plots ($n = 239$). To improve predictions we also developed a set of principal vegetation cover data layers likely to help estimate fine-fuel biomass such as the percent cover of bare ground, woody vegetation (shrubs and trees together), trees and herbaceous plants from all plots measured between 2012 and 2015 ($n = 446$). We used error matrices to calculate overall percent accuracy and omission and commission errors (Congalton and Green 1999). We made further comparisons of fine-fuel and fuel-type models developed with alternative satellite remote sensing platforms such as Landsat Operational Land Imager (OLI). We briefly discuss comparison results that are fully described in Sesnie et al. (2018).

### 3.3.3. Habitat suitability

An objective of this study was to characterize suitable habitat conditions for masked bobwhite quail on BANWR. We proposed to use habitat suitability index (HSI) values that were developed as a set of empirical model functions using information and input from masked bobwhite experts (LaRoche and Conway 2013). However, HSI models differed strongly between experts, which created contrasting model outcomes. We instead used habitat suitability data collected on BANWR with a rapid field protocol adapted from the Northern Bobwhite Quail Habitat Evaluation application developed by the Texas A&M AgriLife Extension Service (https://wildlife.tamu.edu/mobile-apps/). Plot locations were established in the Altar Valley portion of BANWR on a 500m systematic grid as well as at random locations within drainage networks. Each plot location represented a 0.004 $km^2$ area (1 acre) and evaluated nesting cover, woody cover, food, water and the interspersion of habitat and non-habitat areas. Each habitat component was scored and combined to generate a masked bobwhite habitat suitability score between 0.0 and 1.0.

We combined field suitability estimates with WV3 satellite imagery and spatial data layers such as vegetation indices, land cover, elevation from a digital elevation model (DEM), percent slope, topographic wetness index and others. Random forest regression tree models and RFE were used to train and test habitat suitability models. We made some minimal comparisons were made with HSI models to determine differences. Wilcoxon signed rank tests were used to compare fire frequency and TSB with quail habitat suitability as well as determine potential quail release areas on BANWR.

We synthesized management histories, field data, habitat suitability and SEM model results to suggest how fire and other potential treatment types may help rehabilitate quail habitat on BANWR. We briefly discuss further data synthesis to assess fine-fuels and fuel-types on the refuge that were used to develop fire model inputs for simulating fire behavior using FLAMMAP software (Finney et al. 2006).

## 4. Results

Field data collection from 2014 and 2015 within the masked bobwhite management zone resulted in a total of $n = 239$ plots with identification for 184 species of plants, plant cover, fine-fuel estimates and soil samples. Plots represented a relatively even distribution of fire frequency classes in all but areas showing the highest fire frequency with ≥6 fires over a 30-year period that were rare (Figure 6a). The number of years since last burn for each plot location was normally distributed with some locations showing no record of fire (Figure 6b). For these locations, we assigned a value of 30 to indicate no fires had occurred in >30 years and a zero was assigned to locations with <1 year since a fire occurrence for computational purposes.

Summarized plots and habitat conditions (e.g. plant lifeforms) including all plants indicated that non-native (27.8% ±27.1) and native perennial (20.8% ±16.0) grass cover dominated the study area (Table 2a). Non-native grass cover reached as much as 95% in places which were predominately occupied by *S. halepensis* in low terrain and *E. lehmanniana* on upland sites. Forb and other herbaceous plants were the second most abundant life form in the study are, which occasionally reached nearly pure stands (>90% cover) of *A. palmeri* in shallow drainages or topographic depressions. Tree cover was low and mainly comprised or *P. velutina* that occasionally formed dense stands with as much as 50% cover in drainage areas. Subshrubs such as *I. tenuisecta* and *Gutierrezia* spp. were occasionally abundant and averaged 5.2±6.5% cover however, shrub cover was low on average (0.5%±1.9). Shrubs and subshrubs important for quail winter food such as *Acacia* spp., *Mimosa* spp., *Calliandra* spp. and *Abutilon* spp. were infrequent on plots in study area (Table 2b). Native grasses important to MBQ were somewhat common (27%±17.8), but were less widespread than non-native grasses particularly on sandy-loam sites better suited to *E. lehmanniana*. Annual forbs such as *Atriplex elegans*, *Commelina* spp., *Ipomoea* spp. and *Kallstroemia* spp., considered food sources utilized by MBQ, were also in low abundance on plots (2.8%±4.6).

Because of high levels of grassland invasion and non-native cover by *E. lehmanniana*, the average number of plant species on plot transects utilized by MBQ was low (9.1±3.52) compared to the total average number of species on plot transects (16.9±6.25). Species richness somewhat higher when all species recorded on and between transects were included, with 16.85±5.3 and 37.8±11.04 for MBQ and all native species respectively. Non-native plants potentially utilized by MBQ such as *S. halepensis* averaged 1.5±0.96 species on plots.

We considered soil measured on plots in the masked bobwhite management zone as generally sandy loam soils although sand particle size and silt content ranged widely across the study area (Table 3a). Clay content was low on all sites (5.3%±1.4), although somewhat higher on plots associated with low terrain or topographic depressions. We found that cation concentrations such as Na, $NO_3$, $NH_4$ and $PO_4$ and organic matter were not highly varied across the study area with the exception of a small number of plots (Table 3b). K, Mg and Ca ranged more widely. Soil bulk density that we used as an indicator of compaction and past grazing affects was considered relatively high (1.65 $g/cm^3$±0.25) and showed low variation across all plots, which was anticipated for fine to coarse textured sandy to sandy loam soils with low average organic material (6.64%±7.5). Soil chemical and physical properties important to quail habitat conditions are further discusses with SEM model and ordination results below.

### 4.1. Fine-fuel models

Herbaceous biomass clipped from *n* = 431 quadrats on 20 plots ranged from 0.0 g/m$^2$ to 185.4 g/m$^2$ and averaged 25.8 g/m$^2$±25.5. Fine-fuel models developed from destructive sampling indicated that ceptometer LAI, herbaceous plant cover and height measurements were good predictors of sub-plot biomass. For plot-scale fine-fuels, optimized Random Forest regression tree models developed from 2013 destructively sampled quadrats explained 84% of the variation with a root mean squared error (RMSE) of 0.97 g/m$^2$. All further fine-fuel estimates on 2014 and 2015 plots were predicted from herbaceous plant LAI, cover and height measured on quadrates. Predicted biomass for non-destructively sampled plots ranged from 56.4 kg/ha to 1,695.6 kg/ha and averaged 723.5±341.8 kg/ha (Table 2a) that were within the range of values previously reported by Marsett et al. (2005) and McClaran (2003) for Sonoran and Chihuahuan rangeland sites with a burning and grazing history. Fine-fuel biomass estimation, modeling and prediction results are further detailed in Sesnie et al. (2018).

### 4.2. Habitat conditions

To characterize existing habitat conditions in the study area, we examined vegetation by grouping plots according to plant composition, then plotting environmental, and disturbance variables along NMDS axes. A majority of the explanatory variables used were those shown to be significant from SEM results reported below. Flexible Beta cluster analysis was optimized using K-means within group sums of squares and resulted in four vegetation categories that were defined as follows:

1. Highly invaded grasslands dominated by non-native perennial grass *E. lehmanniana*
2. Mixed composition native and non-native grasslands
3. Native grasses, trees and shrubs
4. Native and non-native forbs and herbs

NMDS ordination resulted in a three-dimensional solution that converged after 20 iterations with a stress level of 0.14, indicating adequate fit. To examine principal relationships in the data, we plotted categorized sample sites along NMDS axis 1 and 2 and fit environmental and fire history vectors onto the ordination using a maximum estimated *p*-value of 0.05. Because its dominance, ordination scores were rotated by *E. lehmanniana* cover along axis 1 to interpret primary relationships. Axis 3, not shown, provided little additional information.

Ordination results indicated that site biophysical conditions, disturbance history and plant interactions strongly mediated vegetation composition in the study area (Figures 7a-f). In particular, vegetation composition followed a gradient of increasingly non-native perennial grass cover dominated by *E. lehmanniana* and decreasing species richness and diversity along axis 2 (Figures 7a,b). These conditions were linked to site factors such as increased fire frequency coupled with increased wet and dry season precipitation associated with greater fine-fuel production (Figures 7c,d). We found that plots dominated by native plant cover (e.g. trees, shrubs and grasses) to the left of axis 2 were less frequently burned and on hotter and drier low productivity sites. Native perennial and annual grasses were typically found on increasingly coarse textured soils versus non-native grasses that were associated with greater fine sand content and solar radiation (Figure 7e). Plots dominated by forbs and other herbaceous plants to the top of axis 1 were associated with low topography (e.g. higher Twi values) which had fine textured soils with greater nutrient status and clay content (Figures 7e,f). Plants such as *A. palmeri* and *S. halepensis* formed nearly pure stands on these sites with low plant diversity relative to sites with coarse textures soils (Figures 7b,e).

Our results showed that higher plant diversity important to MBQ foraging opportunities followed a decreasing trend from low to high productivity sites in the study area. More productive sites tended to be burned more frequently and became increasingly dominated by *E. lehmanniana*. Native and non-native vegetation tended to coexist on sites of low to moderate productivity determined by climate and soil conditions. Plots with greater native plant cover and lower fine-fuels were less likely to carry a fire, which, in turn, decreased prescribed burning or wildfires on these sites. From our interpretation of these data, plant competitive interactions and site invasibility were also important, rather than specific site preferences by native grass species, for example. We postulated that suitable sites made available by disturbance factors such as frequent fire allowed for rapid establishment of *E. lehmanniana* that can limit native grass cover and other plant groups less favorable or low productivity sites. Alternatively, other factors unaccounted for here such as periods of severe drought and mortality events may also have contributed to present conditions in the study site (Robinett 1992, Bodner and Robles 2017). We further explored these relationships with SEM models and other ancillary data.

### 4.3. SEM model results

Structural equation models further revealed important factors influencing habitat conditions within the study area. A majority of the vegetation characteristics shown in Table 2a and 2b were evaluated with separate SEM models. Results for models that including all plants species for each habitat characteristic (i.e. plant life form) are reported in Table 4a, and those containing plants identified as important to MBQ are reported in Table 4b.

In the following section, we characterize existing habitat conditions and explanatory variables tested with SEMs as well as their implications for maintaining or promoting masked bobwhite quail habitat on BANWR. In all cases, extended models were an improvement over base models although base models often fit the data well. Further results from base models are reported in Yurcich (2018) and only ΔAIC values from model comparisons are included in this report (Tables 4a, b). For interpretation purposes, we examined SEM results jointly to synthesize key model relationships. Extended SEM results and variable interactions are shown using path diagrams in Appendix B, Figures 1 and 2 to complement model comparisons and statistics in Table 4.

#### 4.3.1. Quail habitat diversity

We considered Shannon's diversity index a basis for assessing overall habitat conditions in the study area. Masked bobwhite quail require a diversity of plants to provide foraging opportunities throughout the year, hiding cover and nest sites (Tomlinson 1972a, LaRoche and Conway 2013). Extended plant diversity models were among our best fitting models according fit indices that had considerably lower AIC values than base models (Tables 4a,b). An important difference between extended and base models was the inclusion of plant interactions. Non-native grasses, primarily comprised of *E. lehmanniana*, showed a strong negative relationship (*Est.* -0.76, *P* = <0.001) with plant diversity in both full plant species and MBQ species only models (Tables 4a,b). Similar relationships between *E. lehmanniana* and plant diversity have been previously reported (Block and Block 1992, Geiger 2006), although our models considered plant species important to MBQ.

Plant diversity also showed a significantly positive relationship with soil texture (percent very coarse sand/percent silt) and significant negative relationship to inorganic nitrogen ($NO_3$ + $NH_4$). Plant diversity was highest for plots with coarse textured soils with poor nutrient status less favorable to *E. lehmanniana*. However, a strong positive relationship between MBQ diversity and precipitation (*Est.* 0.42, *P* <0.001) suggested that the southern portion of the study area, where *E. lehmanniana* was most extensive, also had a greater variety of plants important to MBQ. We provide further information on plant distributions and habitat conditions in the vegetation and fine-fuels mapping results sections below.

Therefore, site conditions and disturbance factors that favor *E. lehmanniana* may indirectly affect native plant diversity through competitive interactions. For example, SEMs for non-native perennial graminoids indicated a significantly positive relationship with precipitation, fire frequency and soils texture (Table 4a). More productive sites with frequent burning showed relatively high non-native grass cover and low plant diversity (Figures 7a-e). Conversely, fire frequency and the amount of time since last burn showed no significant relationship with MBQ plant diversity or native grass cover (Tables 4a,b). MBQ plant species richness also showed a significantly positive relationship to growing season precipitation and course textured soils, but no significant relationship with fire frequency (Table 4b).

These results were not unexpected as native semi-desert grasses have evolved with frequent fire, in the absence of *E. lehmanniana* (Anable et al. 1992, Bahre 1985). We interpreted these outcomes as evidence that sites with greater plant diversity tended to be on coarse textured soils with lower moisture holding capacity and fine-fuel concentrations that were less likely to be prescribe burned or invaded by *E. lehmanniana*.

### 4.3.2. Graminoids

The overall graminoids cover model (all grass species) showed good model fit and that precipitation (*Est.* 0.31, *P* < 0.001) and fire frequency (*Est.* 0.30, *P* < 0.001) were significantly positive explanatory variables (Table 4a). Topographic wetness index showed a significantly negative relationship to graminoids indicating grasses more commonly occupied upland sites. Soil texture (percent medium sand/percent silt) was not significant. These variables showed a similar relationships to non-native perennial graminoids with the exception that medium textured soil conditions was significantly positive (Table 4a). Conversely, fire frequency did not show a significant effect on native perennial grass cover (*Est.* -0.03, *P* = 0.789). The most important variable in this model and for grass species cover in MBQ models was the amount of non-native grass cover that showed a significant negative relationship (*Est.* -0.45, *P* <0.001 and *Est.* -0.55, *P* <0.001 respectively). The percentage of very coarse sand/percent silt also had a significantly positive relationship with native grass cover and for models that considered only grass species important to MBQ, although overall $R^2$ values were low (Tables 4a,b). These model outcomes further suggested that native perennial graminoids species, including those important to MBQ, were most common on sites less favorable to *E. lehmanniana*.

A majority of annual grasses in the study area were native species that were in relatively low abundance (7.8%±11.8) in comparison with perennial grasses that averaged close to 50% cover on plots. SEM models showed low $R^2$ values although model fit was adequate (Table 4a) and outcomes were similar to native perennial grasses. Neither of the fire history variables (i.e. frequency and time since the last burn) were significant. The percentage of very fine sand had a significantly positive (*Est.* 0.26, $P < 0.001$) and precipitation a significant negative (*Est.* -0.27, $P < 0.001$) relationship with native annual grasses. Topographic wetness index also at a significantly positive relationship (*Est.* 0.26, $P < 0.001$) with annual grasses indicating a tendency to be found on lower terrain. Annual grass species were not modeled separately for MBQ as very few were identified in the literature as an important to MBQ.

### 4.3.3. Forbs and herbs

SEMs for forbs and other herbaceous plants fit the data well and had a relatively high $R^2$ value ($R^2 = 0.50$). An important explanatory variable was the significant negative interaction between forbs and other graminoids (*Est.* -8.06, $P < 0.001$). Inorganic nitrogen, phosphate, clay content and fire frequency were positively related to forb cover, but growing season precipitation showed a negative relationship (Table 4a). MBQ forb models also showed a significant negative relationship to grass cover that was also negative for time since last burn indicating the importance of fire for reducing competition from grass species and promoting forb cover important to the masked bobwhite (Goodwin and Hungerford 1977, Simms 1989). Nevertheless, MBQ models contrasted from those that included all species. MBQ forbs were positively related to summer precipitation and coarse textured soils, whereas all species models were likely driven by *A. palmeri* that formed homogeneous stands in areas with low topography and higher clay content (Tables 4a,b). These relationships were also apparent from ordination plots (Figures 7a,b,e).

### 4.3.4. Woody plants and subshrubs

Tree species, highly dominated by *P. velutina* were modeled with and without the inclusion of shrubs. Shrubs were in low abundance on average (0.5%±1.9) in the study area and shrub species important to MBQ were nearly absent from plots (0.2%±0.7). Results from SEMs for trees, and trees and shrubs together were not exceptionally different from one another and extended models showed good fit to the data (Tables 4a,b). Tree cover by itself had a significantly negative relationship with grass cover and fire frequency (Table 4a). Overall, trees and shrub cover together showed a significantly positive relationship with time since the last burn, indicating recovery and recruitment by woody plants in the absence of fire. Soil texture (percent medium sand/percent silt) and growing season precipitation were also significantly positive, indicating that the more productive sites with less frequent fires supported greater woody plant cover. The two models were consistent with one another as *P. velutina* was the principal specie cover.

Only a single combined SEM was species important to MBQ because of low cover for both trees and shrubs. The MBQ model showed a significantly positive relationship to coarse textured soils with low nutrient status that was similar to results found for native perennial grasses with the exception of growing season precipitation that was not significant (Table 4b). Graminoids were negatively related to woody plants important to MBQ as with models when all species were includes. Subshrub models that included all species showed very low coefficients of determination ($R^2$ = 0.08). MBQ models performed somewhat better and fit the data well (Tables 4a,b), but also showed a low overall model coefficient of determination ($R^2$ = 0.14). From the MBQ model, fine textured soil with higher clay content (percent very fine sand/percent clay) and topography were both significantly negative explanatory variables that were hard to interpret. A negative relationship with TWI indicates that upper slopes, rather that drainage bottoms were important, however we often encountered subshrubs that are important food resources to MBQ such as *A. angustissima* and *C. nictitans* along the margin or within drainages. Conversely, the more common and widely distributed *G. sarothrae* was found mainly on upland sites (data not shown), which likely confounded models. *G. sarothrae* was found to be an important cover for released MBQ on BANWR by King (1998), but not a food resource.

#### 4.3.5. Fine fuels

We considered fine-fuels (herbaceous biomass) indirectly related to important MBQ habitat characteristics such as the amount of plant diversity and the type and amount of herbaceous plant cover in the study area. In particular, fine-fuel concentrations tended to decrease with increased plant diversity ($R^2$ = 0.29) and plant diversity significantly decreased with greater non-native grass cover ($R^2$ = 0.84). Fine-fuels comprised of >10% non-native grass cover showed a significant increasing trend ($R^2 = 0.42$) as it did when all herbaceous plant cover were included ($R^2 = 0.52$, Figures 8a-d). Therefore, SEMs for fine-fuels were significantly and positive related to the amount of native herbaceous (*Est.* 0.34, $P < 0.001$) and non-native grass (*Est.* 0.76, $P < 0.001$) on plots, indicating the important contribution of non-native perennial grass to fine-fuel concentrations. Fire frequency was not significantly related to the amount of fuel, however, sub-models indicated that fire frequency a significantly positive relationship with native (*Est.* 0.20, $P < 0.001$) and non-native herbaceous plant cover (*Est.* 0.27, $P < 0.001$, Table 4a).

### 4.4. Fire effects on habitat conditions

We developed our field study designed to examine fire and site biophysical effects on MBQ habitat components by randomly assigning plots to fire history and terrain strata. Fire effects varied by plant group and site conditions examined according to SEM models. To summarize existing habitat conditions with respect to fire history in the study area between 1985 and 2015, we used a series of box plots comparing fire frequency and time since last burn to each of the modeled habitat characteristics (Figures 9,10a-o). Notably, fire frequency greatly increased on sites with greater fine-fuels that corresponded with increased graminoid cover, particularly on sites with greater non-native and annual grass cover (Figures 9a,g,h,j). Sites with ≥3 fires over the 30-year period typically maintained approximately 20% or greater non-native grass cover (Figure 9h). The time since last burn was also typically between 3 and 8 years on sites with higher non-native grass cover (Figure 10h).

Conversely, native perennial grass cover tended to decrease slightly with greater fire frequency and sites with greater native grass cover often had ≤2 fires over the 30-year period (Figure 9i). As observed from SEM models, greater native grass cover typically occurred on less productive coarse textured soils that were less likely to carry a fire. Forbs important to masked bobwhite foraging and thermal cover generally increased on plots with greater fire frequency and tended to decrease with greater time since last burn (Figures 9k,10k). Forbs overall showed a decreasing trend with a greater number of years since the last burn (Figure 10l). Woody plants decreased in a near linear fashion with fire frequency (Figures 9m,n) and increased with a greater number of years since last burn for MBQ species (e.g. trees and shrubs important to winter food production) and overall (Figure 10m,n). Subshrubs important to MBQ also showed a slight decrease with fire frequency and increase with greater time since the last fire.

## 4.5. Mapped fuel and habitat conditions

Each of the above conditions regarding vegetation characteristics, fire history and biophysical factors show spatially explicit patterns important for understanding the present distribution of quail habitat in the study area. We extended plot data from sampled to and un-sampled areas by using 2015 WV3 satellite imagery, ancillary spatial data and field plots to map land cover (fuel-types), fine-fuels and habitat suitability for MBQ on BANWR. Some experimental methods with difference sensor types was used to enhance vegetation classification and biomass estimation. For this report, we give basic accuracy and map error estimates using best performing methods from Sesnie et al. (2018). We give additional details on methods used to estimate MBQ habitat suitability, which incorporated the spatial data layers described in the following section. We further outline fire management implications from these data in the Discussion section below.

### 4.5.1 Fine-fuel and fuel-type mapping

For estimating fine-fuels with multi-date WV3 imagery, a combined RF (Random Forest) regression tree model using ancillary spatial data (e.g. terrain and vegetation cover layers) performed substantially better than with spectral data alone. Variance explained was 47% (RMSE = 236.6) with 85 spectral predictor variables. An optimized set of predictor variables selected with RFE (Recursive feature elimination, with backwards variable selection) substantially decreased the number of predictors to 19 and increased the amount of variance explained to 65.0% (RMSE = 201.9 kg/ha, Figure 11a).

Separate approaches were taken to develop the best classification model and evaluate model error. To classify vegetation and fuel-types on BANWR, a RF classification tree model using all spectral and spatial predictors resulted in an overall classification accuracy of 82.7% from all predictors and 25% of validation samples left out of model training. Models iteratively developed using a portion of sample data with replacement to assess 'Out-of-bag' (OOB) error achieved 78.0% accuracy. A total of 67 predictors were selected out of 101using RFE to optimize the classifier. Models optimized using RFE resulted in 83.1% accuracy from separate validation samples left out of RF model training. Optimized RF models with fewer predictors resulted 79.6% overall class accuracy from OOB error assessment estimated using $n$ = 2000 classification trees. We provide error matrices and statistics for each vegetation class in Appendix C (Tables 1a,b). Most low elevation sites were dominated or co-dominated by the non-native perennial grass *E.*

*lehmanniana* with sparse native grasses (Figure 11b) such that very few samples (plots) dominated by native perennial grasses and shrubs were available for separate model training and validation data sets. Therefore, 25% of samples randomly selected for independent validation, without replacement, were not sufficiently representative for determining accuracy of native grass dominated areas (*n* = 9) or upland shrub vegetation (*n* = 4). Because vegetation data were collected from randomly assigned plots spaced at a minimum distance of 250 m apart, we believed that OOB error estimates better reflected class accuracy (Appendix C, Table 1b) than separate validation data.

Both RF fine-fuels and fuel-type models to estimate vegetation and fuel conditions for the entire study area and across BANWR. Land cover and fuel type varied across the study area and management units, but were often dominated by exotic grasses at lower elevations in the study area (Figure 11b). Fine-fuel accumulation was highest within management units (polygons) designated for burning (*n* = 60) that had greater non-native grass cover, but decreased in units with increased non-native cover (Figure 12a, b). Fuels data developed from satellite imagery and relationships between vegetation cover types were similar to fuels information from our vegetation plots show in Figures 8c and 8d. These outcomes indicated that remote sensing techniques used for developing fine-fuels data from plots and imagery were sufficient for evaluating fuel and potential fire behavior on BANWR. We converted these data to fuel models following Scott and Burgan (2005) to simulate fire behavior with the FLAMMAP fire behavior model (Finney 2006). We ultimately compared behavior model outcomes to those using LANDFIRE fuel model layers (Eagleston and Sesnie *in revision*).

#### 4.5.2. Habitat suitability mapping

Masked bobwhite quail habitat suitability was modeled using separate quail habitat assessment plots (*n* = 408) collected to evaluate habitat conditions such as escape cover, foraging and nesting opportunities. We used a large set of predictor variables (*n* = 126) such as spectral bands and vegetation indices from Worldview-3 imagery, land cover diversity metrics, topographic variables and other spatial predictors such as the proximity and density of washes that were considered important for developing habitat conditions used by MBQ (Goodwin and Hungerford 1977). We used RF models optimized using RFE to predict habitat suitability that resulted in only 26 predictor variables used in the final model. The final RF model explained 33.4% (RMSE = 0.13) of the variability in habitat suitability. We used the increase in mean squared error (MSE) when predictors were iterated out of RF regression tree models ($n = 2000$) to measure variable importance. The amount of increase by MSE with a predictor absent from the model was a good indicator a variable's contribution to variation explained (Breiman 2001). Optimized RF model runs indicated that the amount of cover by bare ground, woody plants and land cover class mixed mesquite, grass and shrubs were among the top-ten most important variables in the habitat suitability model. We interpreted the importance of the amount of bare ground and association with mixed tree and grass cover as indicative of drainage conditions that supported better MBQ habitat (Figures 13a-c).

To better understand linkages between fire history and MBQ habitat suitability, we compared the average MBQ habitat suitability to average fire frequency and time since last burn for each management unit ($n = 84$) on BANWR using non-linear regression. Fire frequency showed a significantly negative relationship ($F$=14.8, $P$<0.001) with habitat suitability and time since last burn showed a significant positive relationship ($F$=14.6, $P$<0.001) with suitability (Figure 14a, b). We found that suitability model outcomes were consistent with habitat characteristics known to be important to MBQ and the fire management history within management units on the refuge. We lastly compared habitat suitability from 2017 habitat assessment field plots ($n = 408$) in unburned areas to those burned between 1985 and 2015 using a Wilcox ranked sum test. Results showed significantly higher habitat suitability for unburned areas ($W = 8742.5$, $P$<0.001).

Our results indicated that locations without fire over the last 30 years had the greater habitat suitability. These results were principally a product of management units outside of the masked bobwhite management zone that were less frequently targeted for prescribed burning. In addition, remaining habitat in drainage areas were often avoided or protected from fire during prescribed burning activities on BANWR. Tomlinson (1972b) and Goodwin and Hungerford (1977) described habitat use by wild and released masked bobwhite in mesquite thickets, dense grass-shrub vegetation and 'edge' habitat between mesquite-lined washes and grasslands. Higher habitat suitability was most frequently encountered on assessment plots located in drainages, whereas low suitability areas were extensively found on upland sites dominated by non-native grass (Figures 13b,c).

## 5. Discussion

Our primary objective was to determine fire-effects on habitat characteristics important to masked bobwhite quail, which require a diversity of food resources and plant cover for survival and reproduction (Tomlinson 1972a, Goodwin and Hungerford 1977, Hernadez et al. 2006). We also endeavored to synthesize data collected with this and other studies to ascertain factors important to promoting favorable habitat conditions for quail. Therefore, we considered how site biophysical conditions, climate, and plant interactions might influence semi-desert grassland vegetation in conjunction management activities in the study area (Wright and Van Dyne 1996, McPherson and Weltzin 2000). Our sampling design was developed to specifically measure vegetation across all topographic positions and fire management histories.

Geiger and McPherson (2005) noted that BANWR is unique because a frequent fire regime was re-established in semi-desert grasslands after a cessation of long-term livestock grazing (Sayre 2007). According to historical records obtained from N. Sayer, only a few areas near ranch headquarters, the current refuge headquarters, were burned for range improvement during 1970s and1980s prior to refuge establishment. A total of 148 fires have been recorded inside the masked bobwhite management zone between 1985 and 2015, with 70% (*n* = 104) classified as prescribed fire. Our 2014 and 2015 vegetation plots were randomly distributed on sites ranging from no recorded fires to sites burned up to seven times (Figure 6a). Most fires were broadcast burns covering an average of 450 ha and mapped fire perimeters often followed burn unit boundaries. A total of 194 fires >1ha have been recorded for all of BANWR during this same period.

One cannot discuss fire-effects on habitat conditions for BANWR without first considering its potential impact on native and non-native perennial grasses. We found that repeatedly burned sites were often those dominated by *E. lehmanniana* that showed higher fine-fuel concentrations, lower plant diversity, and had significantly lower habitat suitability (Figures 14a,b). Areas with greater habitat suitability tended to be outside the MBQ management zone and showed little or no fire occurrence since 1985 (Figure 15a, b). Moreover, we observed that superior habitat often occurred within principal drainages that were typically protected from fire during prescribe burning according to BANWR fire managers (Figure 13a-c). Masked bobwhite released on BANWR have also shown a preference for drainage areas and edge-habitat that often contain heterogeneous grass and shrub cover (Goodwin and Hungerford 1977). Deuel (1974) indicated that radio-tracked captive-raised MBQ used a combination open areas and dense grass-shrub vegetation along washes after released. Tomlinson (1972a) also described mixed grass and shrub conditions were most important to wild birds observed in Sonora, Mexico, although he did not conduct studies of habitat use.

Precipitation and soil gradients contributed to increased cover by *E. lehmanniana* and high fine-fuel concentrations at the southern end of the study area (Figure 11a, b). SEMs indicated that explanatory variables summer precipitation, fire frequency, and the ratio of medium sand (0.25 to 0.50mm particles) to silt content each were significantly and positively related to non-native grass cover. Cox (1984) found that *E. lehmanniana* germination generally improved on sandy loam soils, and Cox and Ruyle (1986) found that areas of higher rainfall and winter temperatures were locations where *E. lehmanniana* persisted and spread. Other factors unmeasured with this study, such as periodic drought stress mortality of perennial grasses can also facilitate germination and replacement of native grasses by *E. lehmanniana* (Robinett 1992, Hamerlynck et al. 2013, Bodner and Robles 2017). McClaran (2003) notes that competitive interaction between *E. lehmanniana* and native grasses primarily occurs during the establishment stage rather than between adult plants. Roundy et al. (1992) found that reduced plant canopy from disturbances such as fire, grazing or mowing increases red light exposure in the 400nm to 700nm range (photosynthetically active radiation) that dramatically increases *E. lehmanniana* germination success. We found that *E. lehmanniana* was highly and positively correlated with the amount of annual solar radiation ($r = 0.53$) on a site that was also apparent in ordination plots (Figures 7b,e). From ours and other studies, frequent fire, intermittent drought cycles, and site factors such as soil texture, topography and precipitation gradients have likely worked together to promote extreme dominance by *E. lehmanniana* on BANWR.

Increased cover and dominance by *E. lehmanniana* alone do not necessarily define levels of habitat suitability on BANWR. Kuvlesky et al. (2002) suggested the detrimental impacts of non-native grasses such as *E. lehmanniana* and *C. ciliaris* on masked bobwhite quail are largely unstudied. King (1998) found that captive-raised masked bobwhite were more likely to utilize *E. lehmanniana* dominated areas within release areas on BANWR than those with greater native grass cover. King (1998) further suggested that remaining areas of native grass might no longer provide sufficiently dense structure needed by MBQ for thermoregulation and escape from predators (Guthrey et al. 2001).

These findings were consistent with our SEM results that indicated areas with a preponderance of native perennial grasses and, by extension, greater plant diversity were often on less productive coarse textured soils. Thus, lack of sufficient native grass cover and associated food resources within MBQ release areas is potentially limiting to quail re-establishment on BANWR. Indeed, captive birds have shown poor survival on BANWR potentially because of lack of food in areas with low plant diversity. Bock et al. (1986) found that grasshoppers, a primary food source for MBQ, were 44% less abundant on plots with exotic plant cover (e.g. *E. lehmanniana* and *C. ciliaris*) than native grassland plots. More recently, Andersen et al. (2018) determined that species richness decreased 2% and total grasshopper abundance decreased 7% for every 10% increase in *E. lehmanniana* from sweep surveys on several sites in southern Arizona. Kuvlesky et al. (2002) concluded that while MBQ may utilize sites dominated by exotic grasses, these areas are likely useful only as cover and other food producing plants or resources must be near-by (King 1998).

Deuel (1974) also considered that released MBQ surveyed with telemetry on BANWR and north of Arivaca on Rancho Seco were less food limited, and mortality was primarily a result of the conspicuousness of captive raised quail. Indeed, feeding trials revealed that MBQ ate a wide variety of seed from native and non-native grasses and forbs (Deuel 1974). He suggested that unwariness and raptor predation observed for a few of the radio-tracked birds was potentially a key mortality factor for other released birds even when sufficient escape cover was present (Deule 1975). Guthery et al. (2000) concluded that shrub or tall grass cover, a principal habitat deficiency in Arizona and Sonora, Mexico were necessary for maintaining operative temperatures for masked bobwhite quail and decreasing exposure to ground and aerial predators. Perhaps most compelling from Deuel (1974) are vegetation descriptions for release sites on and off BANWR in 1971 that identify *B. rothrockii* and *B. aristidoides* as the most abundant perennial grass species. Although non-native *S. halapense* was already common in Altar Wash, *E. lehmanniana* was not present in observations by Deuel (1974), but is now the dominant grass cover for the release site on BANWR.

We found other direct fire-effects that both positively and negatively influenced MBQ habitat conditions. Forbs beneficial to quail showed a significant positive relationship to frequent fire and greater inorganic nitrogen availability, while woody plants such as leguminous shrubs were negatively impacted. Lack of essential winter food from leguminous shrubs and other seed producing plants may be a limiting factor for quail survival during months when arthropod abundance is low (Brown and Clark 2017). Babiack (2011) found that mixed grass and shrub habitat were lacking on most sites surveyed for quail between 2007 and 2009 on BANWR and encountered no MBQ during surveys. Our data indicated that >11 years from the time of the last fire may be required to recover woody plants important to MBQ to levels above 1% cover on average (Figure 10m) as shrub cover was generally low on all plots (Table 2a).

Subshrubs important to MBQ were mainly comprised of *G. sarothrae*, which is used as primarily as cover by quail (King 1998). The second most common species observed on plots was *Chamaecrista nictitans* that is an important food source for MBQ (Tomlinson 1972a). Our SEM for MBQ subshrubs showed no significant impact from time since burned (Table 4b) although models with fire frequency showed a significant negative affect (not shown). It is possible that some species of subshrubs respond positively to fire while others do not. For example *C. nictitans* was weakly, but positively correlated with fire frequency ($r = 0.21$) and negatively correlated with time since last burn ($r = -0.24$), which was opposite for *G. sarothrae*. These opposing relationships likely lead to lower SME performance indices using all or only MBQ subshrubs species as the dependent variable (Tables 4a,b). Fire may be beneficial to species such as *C. nictitans* that averaged 1%±2.8 cover on plots and could be as high as 20% cover. *C. nictitans* also showed a significantly positive relationship with TWI and coarse textured soils that was likely associated with drainage areas sampled. Other leguminous subshrubs such as *Desmanthus cooleyi* and *A. angustissima* were less common on plots, but also tended to occur within shallow drainages. Suitable areas and disturbance factors that might encourage leguminous subshrubs require further investigation as these species are important food sources throughout the masked bobwhite's historical range (J. Levy personal communication, Tomlinson 1972a). However, none of the subshrub species mentioned above were widespread in the study area.

Additionally, we found that increased forb and other annual plant cover resulting from fire are relatively short-term, particularly on sites highly dominated by *E. lehmanniana*. *E. lehmanniana* re-establishes quickly after fire from both seed and basal sprouting such that forb or annual grass cover may only persist a few years following disturbance depending on the amount of precipitation that can vary substantially between years (Humphery and Everson 1951, Cable 1971, Bock and Bock 1992, McDonald and McPherson 2010). Cable (1967) and

Wright and Bailey (1982) suggested that perennial grasses generally recover within 3 years after fire under average rainfall conditions and in the absence of grazing. Our observations suggest that forb cover important to MBQ was greatest on plots 3 to 5 years after a burn on average that successively declined with a greater number of years since fire (Figure 10k). However, we were only able to sample recently burned areas on *n* = 8 plots (≤2 years since fire) such that MBQ forb species present post-fire may have been inadequately sampled. Plot data summarized using cover for all forb species showed a consistent decrease with an increased number of years since fire (Figure 10i). We observed sites where *E. lehmanniana* quickly re-established to near pre-fire conditions in 2 to 3 years (>50% cover, personal observation), after native forb (5%) and annual grass cover (59%) were as high as 64% and *E. lehmanniana* as low as 7% cover two months post-fire (*n* = 2 plots).

From SEMs, prescribed fire showed no direct long-term effect on native perennial or annual grasses important to quail or otherwise in the study area consistent with Geiger and McPherson (2005). This was anticipated as herbaceous plants have evolved with frequent fire in semi-desert grasslands (Bahre 1985). McGlone (2013) found only a short-term reduction in native annual grasses after fire, and a return of near pre-treatment conditions for native perennial grasses after 6 years on transects in New Mexico. Bock and Bock (1992) found that some native perennial grass species recovered slowly after fire (e.g. *Aristida* spp.), while others such as *Bouteloua* spp., *Panicum hallii* and *E. intermedia* recovered to pre-burn conditions in 4 to 7 years on plots in southern Arizona. Because we measured plots stratified by past fire history at a single point in time, we were only able to capture recent or short-term fire-effects on a few plots. For example, plots with ≤2 years since fire (*n* = 8) showed the greatest annual grass cover on average (Figures 9j, 10j). We found few references to annual grasses as an important source of MBQ food or cover (LaRoche and Conway 2013). Moreover, our results reflect longer-term outcome from fire. Cable (1965, 1967, 1973) found that most native perennial plants are susceptible to high mortality from early summer fires as growth rates are increasing. Susceptibility to post-fire mortality may be further increased during drought years when preceding wintertime precipitation is low (Bodner and Robles 2017).

Another aspect of this study was to assess fine-fuel conditions important for determining fire hazard on BANWR (Eagleston and Sesnie, *in revision*, Sesnie et al. 2018). Although sometimes limited, other data and studies provide historical context for present fuel conditions on BANWR. D. Robinett re-measured a single pre-and post-fire transect in a BANWR Airport management unit at four intervals between 1980 and 1999. It showed a trend of increased *E. lehmanniana* from 26% cover in 1981 to 46% cover in 1999 after three prescribed fires (unpublished data). Estimated herbaceous biomass increased substantially from 825 kg/ha to 2418 kg/ha between 1981 and 1999 as *E. lehmanniana* became the primary grass cover. These estimates, although showing a greater increase over time, were similar to those observed on the Santa Rita Experimental Range during the 1970s and 1980s that showed greatly increased perennial grass biomass concurrent with increased *E. lehmanniana* density (Cable and Martin 1975, Martin and Severson 1988). Our fine-fuel estimates from plots averaged 723.5±341.8 kg/ha with a maximum of 1,695.6 kg/ha on plots between 976m and 1,133m elevation that were very close to values shown as *E. lehmanniana* increased density on the Santa Rita Experimental Range (McClaran 2003).

In each case, native perennial grasses decreased as non-native perennial grasses increased. For example, pre- and post-fire transect measurements by D. Robinett showed that *Hilaria belangeri*, *Aristida purpurea*, *Boutella rothrockii* and *Boutella gracilis, Boutella rothrockii* decreased from 52% cover in 1980 to 15% cover by 1999. Shrub species *Calliandra erophylla*, that is an important food source eaten by MBQ (Tomlinson 1972a) also initially increased cover from pre-fire conditions in 1980 (8.2%) to 11.8% a few months post-fire and subsequently declined to 4% and 2% cover in 1981 and 1999 respectively. Robinett (1992) further described drought effects and soil conditions contributing to perennial grass mortality for both *E. lehmanniana* and native grasses for a grazed transect near Sasabe, AZ for years 1985 to 1988 and 1990. *E. lehmanniana* greatly increased cover over *Boutelous eriopoda* after drought conditions in while gazing utilization was relatively even for these species.

Conditions on BANWR today reflect increased fire hazard particularly in areas with near homogeneous cover by *E. lehmanniana* (Figures 11a, b). We observed plots with complete *E. lehmanniana* cover (100% of *n* = 240 plant intercepts) from measurements made during years with below average rainfall (2012 and 2013). Non-native perennial grasses averaged 28% cover for 2014 and 2015 plots with a maximum of 95% cover that were average rainfall years (Table 3). Modeled fire behavior and random ignitions used by Eagleston and Sesnie (*in revision*) showed that continuously dense stands of *E. lehmanniana* were those most capable of greater fire spread and extensive burns during periods of >90% fire weather conditions.

However, control features such as numerous unimproved roads have often prevented extensive fires on BANWR although some recent fires became large quickly in moderate to heavy fine-fuel loads and moved from BANWR to adjacent US Forest Service land (e.g. 2018 Cumero fire).

## 6. Key findings and management implications

Historical accounts establish that BANWR once supported a population of masked bobwhite quail, likely within or close to the present study area (Stephens 1885, Brewster 1885, Brown 1885). Frequent, but random fire occurrence most likely played a historical role in shaping semi-desert grassland vegetation that supported suitable habitat conditions on BANWR (Humphery 1963, Bahre 1985). These populations disappeared quickly with intensive livestock grazing and extreme drought at the turn of the century (Brown 1904). We can only presume that fire was historically a reoccurring haphazard event that could vary sufficiently to maintain the type of habitat heterogeneity, food and cover resources needed to sustain masked bobwhite quail.

Three primary conclusions are warranted from our quantitative observations and those from previous studies and data. The first is that prescribed fire may help to develop some habitat characteristics important to masked bobwhite quail such as increased forb and annual plant cover. These benefits are likely short-term. Non-native perennial grass recovers quickly after fire on favorable sites, such that increased food and thermal cover by forbs may persist for only a few years. Subshrubs beneficial to MBQ may be enhanced by or are neutral to fire where disturbance may have a transitory or even positive effect. However, novel fuel conditions present on BANWR may no longer be conducive to low intensity fires needed to maintain subshrubs and other woody plants beneficial to quail. This aspect of pre-fire conditions and near-term post-fire recovery deserves greater attention and monitoring as recommended by King (1998).

Secondly, passive use of fire initiated in 1985, primarily broadcast burning as a stand-alone treatment, did not demonstrated positive outcomes for overall habitat suitability on BANWR. Mapped land cover and habitat suitability from 2015 satellite imagery indicated that most remaining habitat resides along the margin of the masked bobwhite management zone where fires, prescribed or otherwise, were less frequent or absent (Figures 15a,b). Areas of higher suitability within the masked bobwhite management zone often included narrow portions of drainage areas previously protected from prescribed fires. Mixed grass and shrub habitat conditions may indeed be a later successional stage that requires a longer fire return interval, one or more standard deviations from a 7- to 10-year interval considered for desert grasslands (Bahre 1985, McPherson and Weltzin 2000). McPherson and Weltzin (2000) indicate that many desert woody plants may take up to 10 years to produce seed, however this may differ for subshrub species beneficial to quail such as *A. angustissima* that can potentially produce seed twice a year within a few years of germination (Lloyd-Reilley 2011). Trees and leguminous woody shrubs identified as important to MBQ were not widely distributed and had very low cover in the study area, with the exception of some drainage areas. We recommend that these areas, with further on-the-ground verification, become the focus of habitat restoration activities for MBQ.

Strategic and targeted management activities to extend and connect areas presently showing higher habitat suitability would likely benefit future quail releases. Efforts such as enrichment planting of leguminous shrubs, subshrubs or other plants with high seed producing capability (e.g. *A. angustisima* and *C. nictitans*) for foraging or improved thermal and escape cover can be used to help improve winter survival when arthropods are scarce. BANWR itself is at the upper end of elevation range know to have historically supported masked bobwhite quail (240m to 1,060m, Brown et al. 2012). Much of the area with greater habitat suitability we observed was at elevations between 1,100m and 1,250m. These conditions might be limiting to MBQ should temperature extremes, humidity or other factors diminish survival and reproductive success for released quail (Tomlinson 1972b, Guthery et al. 2000, Guthery et al. 2001). Buffered areas of mapped habitat suitability to seek to further target specific locations for habitat improvement that can potentially increase connectivity and quail movement between lower and higher elevation sites (Figure 15a).

Lastly, we have noted that increased fine-fuel concentrations particularly within dense stands of *E. lehmanniana* may pose high fire hazard. Fire behavior simulations by Eagleston and Sesnie (*in revision*) confirm that such conditions can result in larger fires on BANWR without suppression efforts. An area of higher habitat suitability was, in fact recently burned by the 2018 Cumero wildfire that occurred within previously unburned or infrequently burned portions of the Canoa management unit. Conversely, we observed that areas with high and most contiguously dense fuel concentrations on BANWR were those burned up to 5 times over 30 years, which is approximately a 6-year fire return interval in certain cases (Figure 15b). These areas were nearly devoid of suitable habitat for MBQ with the exception of a few minor patches within drainages (Figure 15a). We concluded that frequent prescribed fire has, in places, worked in opposition to fuel hazard reduction that has potentially led to enhanced germination environments for *E. lehmanniana* and novel fuel conditions (D'antonio and Vitousek 1992, Brooks et al. 2004). This was particularly apparent in the southern portion of BANWR that showed soil, topography and precipitation conditions highly favorable to *E. lehmanniana* establishment and spread. These areas were primary comprised of native perennial grasses prior to refuge establishment according to qualitative information obtained (J. Goodwin and D. Robinett, personal communication). These insights were consistent with other unpublished data and reports from this area (Deuel 1974). Drought effects during the late 1970s followed by above average rainfall in the early 1980s has also contributed to these transformations (Robinett 1992), that have very likely worked in conjunction with frequent fire.

Homogenous areas of dense *E. lehmanniana* with little or no habitat value for quail may very well be suitable sites for experimentation and grassland rehabilitation. Under these circumstances, small-scale experimental areas (≤1 or 2 ha) using fire in combination with other management activities such as mechanical soil treatments followed by seeding or planting with native perennial grasses should be considered (Hernandez et al. 2006). These efforts would help to devise more effective treatment regimens that reduce hazardous fuel and increase plant diversity with clearly defined pre- and post-treatment monitoring protocols that adequately assess outcomes (King 1998, McPherson and Weltzin 2000). On sites where rapid recovery by *E. lehmanniana* is inevitable, particularly in burned areas prior summer rainfall, experimental treatment techniques may consider, for example, treatments such as 'out of season' burning followed by post-fire site preparation at times of reduced productivity that are subsequently seeded or planted.

Fire will undoubtedly continue to play a role on BANWR particularly in locations where fuel-bed structure and composition are presently dominated by dense non-native grasses. Efforts to improve habitat quality for masked bobwhite quail on the refuge should consider protection for critical habitat areas and active management to improve cover, connectivity and diverse foraging opportunities among these sites. These areas may require sufficient recovery time from previous burns that could range from 15 to 20 years depending on site environmental factors and interannual climate variability. However, this is an uncertainty without an improved understanding of pre- and post-treatment conditions and outcomes from specific activities designed to encourage improved habitat conditions. McPherson and Weltzin (2000) provide an excellent set of recommendations and considerations for developing management relevant experiments in desert and semi-desert grassland environments that are particularly appropriate here.

## 7. Ongoing and future research needs

There are a number of follow-up activities needed or are in progress that stem from this project listed below with a brief description.

- Small scale-management experiments to improve habitat conditions for quail

We believe that small-scale management experiments are needed and would greatly benefit many aspects of quail habitat rehabilitation activities and prescribe fire use on BANWR. Work summarized from this project provides foundation from which various types of activities may help promote and improve habitat conditions for quail, but require follow-up and verification monitoring. These activities could begin with testable hypothesis followed by treatments designed to produce specific outcomes for quail habitat improvement.

- Fire behavior modelling

This work is essentially completed and is in the process of revision for publication in the International Journal of Wildland Fire (Eagleston and Sesnie, *in revision*). Fuel models developed from fine-fuel biomass and fuel-type digital data layers from this project were used to compare FLAMMAP fire simulations run with LANDFIRE fuel models. These data revealed statistically different fire behavior results. Model simulations run at 90% fire weather conditions using our data layers more closely resembled actual fire behavior and severity from recent fires on BANWR. Our fuel models and behavior simulations showed realistic fire size and fire line intensity as opposed to LANDFIRE model simulations that showed very little fire spread. LANDFIRE fuel models did not accurately represent semidesert fuel conditions known to carry fire in these environments.

- Refinement of structural equation models

While we consider the relationships interpreted from structural equations finalized in this report, there is room for improvement. A few of these improvements will include development of models that more precisely determine the effect of ‘latent’ or unobserved variables on other observed variables important to developing quail habitat conditions. Some model revisions will also seek to parse critical food resources from other habitat features that may help to clarify some model relationships (e.g. subshrub models).

- Improved habitat suitability models for BANWR

Masked bobwhite quail habitat suitability modeling efforts with this project would benefit from three-dimensional data that better describes vegetation and habitat structure. Airborne LiDAR data for much of the study area is presently being processed for integration into suitability models.

- Current and future climate evaluations for BANWR and Sonora, Mexico study sites.

We have mentioned little in this report about climate factors and their importance to masked bobwhite quail re-establishment on BANWR. However, the prognosis for future hotter and drier climatic conditions in the desert southwest will continue to impact habitat conditions and the physiologic responses by quail (e.g. heat stress impacts on reproduction). These conditions are currently being assessed as part of a project entitled *"Forecasting resource availability for wildlife populations in arid grasslands under future climate extremes"*

funded by the Southwest Climate Science Center. The project is developing phenology-based models and future climate scenarios for other species of quail on BANWR (i.e. *Callipepla squamata and C. gambelii*). We have developed some preliminary climate data analyses in support of these activities and ongoing masked bobwhite re-establishment effort on BANWR.

- Applying high resolution satellite imagery and mapping techniques with other techniques to detect masked bobwhite quail in Sonora, Mexico

Field sampling and land cover, fuels and habitat mapping methods developed with this study can be applied in other locations. We are currently using vegetation data collected on study sites in Sonora, Mexico paired with Sentinel-2 and Worldview-2 and Worldview-3 satellite data to map conditions for those study sites. Mapped data are being combined with novel quail detection techniques and a study design incorporating automated recording units (ARU).

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## Tables

Table 1. Land cover class descriptions and abbreviations used for mapping fuel-types on Buenos Aires National Wildlife Refuge using multidate leaf-on and leaf-off 2015 Worldview-3 imagery.

| Class name | Class Abbrev. | Description |
|---|---|---|
| Shadow | Sh | Image shadows from buildings, topography, trees and clouds |
| Tree cover | Tr | Lower elevation dense tree cover |
| Upland shrubs | Upsh | Upper elevation dense shrubs |
| Urban/developed | Ur | Roads, buildings and other human infrastructure |
| Open water | Wa | Tanks, ponds and perennial waters |
| Mesquite/native grass/shrub mix | Mgrw | Mixed mesquite trees, grass and shrubs |
| Bare ground | Ba | Rock, soil and other un-vegetated areas |
| Exotic grass | Xgr | Primarily *Eragrostis lehmanniana* |
| Forb and herblands | Fh | Dense herbaceous plants and forbs |
| Native grass | Ngr | Mainly sparse native grasses |
| Madrean oak-juniper | Oj | Upper elevation oak and juniper woodlands |

Table 2. Vegetation characteristics from $n = 239$ plots measured between July and October of 2014 and 2015 that included a) all plant species summarized by life form and b) only plant species important to masked bobwhite quail summarized by life form.

a) All plant species and fine-fuel biomass

| **Habitat characteristic** | **Mean (%)** | **SD** | **Min (%)** | **Max (%)** |
|---|---|---|---|---|
| Forbs/Herbs | 13.0 | 16.4 | 0.0 | 98.3 |
| Graminoids | 54.1 | 24.7 | 0.0 | 100.0 |
| Shrubs | 0.5 | 1.9 | 0.0 | 17.9 |
| Subshrubs | 5.2 | 6.5 | 0.0 | 38.8 |
| Native annual grass | 7.8 | 11.8 | 0.0 | 60.0 |
| Non-native annual grass | 0.0 | 0.1 | 0.0 | 0.4 |
| Native perennial grass | 20.8 | 16.0 | 0.0 | 77.5 |
| Non-native perennial grass | 27.8 | 27.1 | 0.0 | 95.4 |
| Succulents | 0.1 | 0.5 | 0.0 | 7.1 |
| Trees | 11.0 | 11.3 | 0.0 | 59.6 |
| Vines | 0.5 | 2.3 | 0.0 | 33.3 |
| Bare soil | 17.8 | 14.8 | 0.0 | 71.3 |
| Native plants | 55.7 | 24.3 | 3.3 | 100.0 |
| Non-native plants | 27.7 | 26.7 | 0.0 | 92.5 |
| **Fine-fuel** | **Mean (kg/ha)** | **SD** | **Min (kg/ha)** | **Max (kg/ha)** |
| Herbaceous biomass | 723.5 | 341.8 | 56.4 | 1695.6 |

b) MBQ plant species only

| **Habitat characteristic** | **Mean (%)** | **SD** | **Min (%)** | **Max (%)** |
|---|---|---|---|---|
| Forbs/Herbs | 2.8 | 4.6 | 0.0 | 46.3 |
| Graminoids | 27.4 | 17.8 | 0.0 | 75.0 |
| Shrubs | 0.2 | 0.7 | 0.0 | 5.4 |
| Subshrubs | 5.2 | 5.9 | 0.0 | 29.6 |
| Succulents | 0.0 | 0.4 | 0.0 | 5.8 |
| Trees | 1.0 | 1.2 | 0.0 | 6.7 |
| Vines | 0.1 | 0.3 | 0.0 | 2.5 |

Table 3. Summarized a) soil physical and b) chemical properties from 18 x 10cm depth samples taken from *n* = 239 plots on Buenos Aires National Wildlife Refuge during 2014 and 2015.

a) Soil physical properties

| **Soil texture** | **Mean (%)** | **SD** | **Min (%)** | **Max (%)** |
|---|---|---|---|---|
| Clay | 5.31 | 1.39 | 1.95 | 10.99 |
| Silt | 40.79 | 13.17 | 9.92 | 86.63 |
| Very fine sand | 13.38 | 3.86 | 2.37 | 30.71 |
| Fine sand | 13.03 | 4.25 | 0.00 | 23.93 |
| Medium sand | 9.31 | 4.24 | 0.00 | 23.28 |
| Coarse sand | 13.04 | 5.63 | 0.00 | 27.48 |
| Very coarse sand | 5.14 | 3.46 | 0.00 | 18.12 |
| Combined sand | 53.90 | 13.93 | 2.38 | 88.13 |

b) Soil chemical properties

| **Soil chemistry** | **Mean (µg/ml)** | **SD** | **Min (µg/ml)** | **Max (µg/ml)** |
|---|---|---|---|---|
| PO4 | 3.95 | 4.40 | 0.00 | 31.90 |
| NO3 | 8.93 | 9.40 | 0.00 | 58.20 |
| NH4 | 0.97 | 1.48 | 0.00 | 16.40 |
| Inorganic N | 9.90 | 9.22 | 0.00 | 59.10 |
| K | 168.18 | 125.41 | 0.00 | 1327.00 |
| Mg | 168.83 | 76.72 | 0.00 | 452.41 |
| Ca | 1044.01 | 605.99 | 111.40 | 3845.02 |
| Na | 8.66 | 12.28 | 0.00 | 169.84 |
| SOM | 6.64 | 7.54 | -3.67 | 110.30 |
| Bulk density | 1.65 | 0.25 | 0.57 | 2.34 |

Table 4. Structural equation model (SEM) dependent and explanatory variables, abbreviations, units of measure and their source.

| Variable abbrev. | Description | Units | Reference/source |
|---|---|---|---|
| Ppt.wet | Precipitation of wettest quarter (summer) | mm | http://www.worldclim.org/bioclim |
| Freq | Fire frequency | Count | USFWS Fire Atlas |
| Tsb | Time since last burn | Years | USFWS Fire Atlas |
| N | $NO_3 + NH_4$ | µg/ml | Field and lab |
| P | $PO_4$ | µg/ml | Field and lab |
| Som | Soil organic mater | % | Field and lab |
| Silt | Silt | % | Field and lab |
| VC.sand | Very coarse sand | % | Field and lab |
| VF.sand | Very fine sand | % | Field and lab |
| Comb.sand | Combined sand | % | Field and lab |
| Txt.1 | Soil texture composite 1 | Sqrt(Medium sand(%)/silt(%)) | Field and lab |
| Txt.2 | Soil texture composite 2 | Sqrt(Very coarse sand (%)/silt(%)) | Field and lab |
| Txt.3 | Soil texture composite 3 | Sqrt(Very fine sand (%)/silt(%)) | Field and lab |
| Txt.4 | Soil texture composite 4 | Sqrt(Very fine sand (%)/clay(%)) | |
| Txt.5 | Soil texture composite 5 | Sqrt(silt (%)/clay(%)) | Field and lab |
| Twi | Topographic wetness index | Index | NED 10m DEM |
| Woody | Trees + shrubs | % | Field |
| NN.grass | Non-native perennial grasses | % | Field |
| N.grass | Native perennial grasses | % | Field |
| An.grass | Annual grasses | % | Field |
| Sha.diversity | Shannon diversity | Index | Field |
| Sp.number | Number of species | Count | Field |
| Graminoids | Graminoids | % | Field |
| Forbs | Forbs | % | Field |
| Herb.cov | Forbs + grass | % | Field |
| Woody | Trees + shrubs | % | Field |
| Subshrubs | Subshrubs | % | Field |
| Fine.fuel | Herbaceous biomass | kg/ha | Field |

Table 5. Structural equation model (SEM) comparisons contrasting the base SEMs with extended SEMs that included composite variables and increased model complexity for a) all plant species and b) MBQ plant species only.

a) All plant species and life forms

| **Habitat var.** | **Base model** | **Final model** | [1]***P* from *$x^2$*** | [2]**ΔAIC** | [3]**CFI** | [4]**TLI** | [5]**RMSEA** | **Reg. Models** | ***$R^2$*** | **Expl. Var.** | [6]**Est.** | P(>\|z\|) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plant diversity | Sha.diversity ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | Sha.diversity ~ NN.Grass + tsb + freq + N + txt<br>N ~ NN.Grass + txt + som<br>freq ~ ppt.wet + txt + NN.Grass<br>tsb ~ freq + NN.Grass | <0.001 | 2185.0 | 0.94 | 0.86 | 0.13 | Sha.diversity | 0.58 | NN.Grass | -0.76 | <0.001 |
| | | | | | | | | | | Tsb | 0.05 | 0.472 |
| | | | | | | | | | | Freq | 0.17 | 0.025 |
| | | | | | | | | | | N | -0.33 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.17 | <0.001 |
| | | | | | | | | N | 0.15 | NN.Grass | -0.9 | 0.002 |
| | | | | | | | | | | Txt.2 | -0.05 | 0.473 |
| | | | | | | | | | | Som | 0.30 | <0.001 |
| | | | | | | | | Freq | 0.21 | Ppt.wet | 0.37 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.09 | 0.192 |
| | | | | | | | | | | NN.Grass | 0.17 | 0.009 |
| | | | | | | | | Tsb | 0.66 | Freq | -0.84 | <0.001 |
| | | | | | | | | | | NN.Grass | 0.10 | 0.013 |
| Number plant species | Sp.number ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | Sp.number ~ NN.Grass + freq + N + som + txt + ppt.wet<br>N ~ txt + som<br>freq ~ ppt.wet + NN.Grass | 0.003 | 504.1 | 0.93 | 0.78 | 0.10 | Sp.number | 0.31 | NN.Grass | -0.49 | <0.001 |
| | | | | | | | | | | Freq | 0.11 | 0.064 |
| | | | | | | | | | | N | -0.24 | <0.001 |
| | | | | | | | | | | Som | -0.08 | 0.195 |
| | | | | | | | | | | Txt.2 | 0.23 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.22 | 0.001 |
| | | | | | | | | N | 0.11 | Txt.2 | -0.03 | 0.646 |
| | | | | | | | | | | Som | 0.33 | <0.001 |
| | | | | | | | | Freq | 0.20 | Ppt.wet | 0.34 | <0.001 |
| | | | | | | | | | | NN.Grass | 0.17 | 0.006 |
| Graminoids | Graminoids ~ silt + twi + ppt.wet + freq + tsb<br>silt ~ twi<br>freq ~ ppt.wet + silt<br>tsb ~ freq | Graminoids ~ txt + twi + freq + ppt.wet<br>txt ~ twi + ppt.wet<br>freq ~ ppt.wet + txt<br>tsb ~ freq | <0.001 | 4.14 | 0.96 | 0.85 | 0.14 | Graminoids | 0.32 | Txt.1 | 0.13 | 0.023 |
| | | | | | | | | | | Twi | -0.25 | <0.001 |
| | | | | | | | | | | Freq | 0.30 | 0.001 |
| | | | | | | | | | | Tsb | 0.08 | 0.386 |
| | | | | | | | | | | Ppt.wet | 0.31 | <0.001 |
| | | | | | | | | Txt | 0.06 | Twi | 0.01 | 0.899 |
| | | | | | | | | | | Ppt.wet | -0.24 | <0.001 |
| | | | | | | | | Freq | 0.23 | Ppt.wet | 0.48 | <0.001 |
| | | | | | | | | | | Txt.1 | 0.24 | <0.001 |
| | | | | | | | | Tsb | 0.65 | Freq | -0.81 | <0.001 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Forbs | Forbs ~ N + silt + twi + ppt.wet + freq + tsb<br>N ~ silt + ppt.wet<br>freq ~ ppt.wet + silt<br>tsb ~ freq<br>silt ~ twi | Forbs ~ Graminoids + freq + clay + N + P + ppt.wet<br>freq ~ Graminoids + ppt.wet<br>N ~ freq + txt + P + twi + ppt.wet + som | <0.001 | 1206.56 | 0.93 | 0.84 | 0.10 | Forbs | 0.50 | Graminoids | -8.06 | <0.001 |
| | | | | | | | | | | Freq | 4.17 | <0.001 |
| | | | | | | | | | | Clay | 3.79 | <0.001 |
| | | | | | | | | | | N | 6.90 | <0.001 |
| | | | | | | | | | | P | 3.64 | <0.001 |
| | | | | | | | | | | Ppt.wet | -2.16 | <0.031 |
| | | | | | | | | Freq | 0.23 | Graminoids | 4.17 | <0.001 |
| | | | | | | | | | | Ppt.wet | 4.82 | <0.001 |
| | | | | | | | | N | 0.33 | Freq | -0.54 | 0.589 |
| | | | | | | | | | | Txt.5 | 4.79 | <0.001 |
| | | | | | | | | | | P | 0.662 | 0.587 |
| | | | | | | | | | | Twi | 3.91 | <0.001 |
| | | | | | | | | | | Ppt.wet | -1.59 | 0.112 |
| | | | | | | | | | | Som | 4.03 | <0.001 |
| Trees and shrubs | Woody ~ N + comb.sand + twi + ppt.wet + freq + tsb<br>N ~ comb.sand + ppt.wet<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq<br>comb.sand ~ twi | Woody ~ Graminoids + tsb + txt + N + P + ppt.wet<br>tsb ~ Graminoids + freq + ppt.wet<br>N ~ txt + ppt.wet + twi + som | <0.001 | 1390.3 | 0.93 | 0.84 | 0.12 | Woody | 0.47 | Graminoids | -0.48 | <0.001 |
| | | | | | | | | | | Tsb | 0.30 | <0.001 |
| | | | | | | | | | | Txt.1 | 0.42 | <0.001 |
| | | | | | | | | | | N | -0.15 | 0.003 |
| | | | | | | | | | | P | 0.20 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.32 | <0.001 |
| | | | | | | | | Tsb | 0.68 | Graminoids | 0.01 | 0.912 |
| | | | | | | | | | | Freq | -0.88 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.17 | <0.001 |
| | | | | | | | | N | 0.27 | Txt.1 | -0.16 | 0.009 |
| | | | | | | | | | | Ppt.wet | -0.19 | 0.001 |
| | | | | | | | | | | Twi | 0.32 | <0.001 |
| | | | | | | | | | | Som | 0.23 | <0.001 |
| Trees | Tree ~ N + comb.sand + twi + ppt.wet + freq + tsb<br>N ~ comb.sand + ppt.wet<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq<br>comb.sand ~ twi | Trees ~ Graminoids + freq + txt + N + P + ppt.wet<br>freq ~ Graminoids + ppt.wet<br>tsb ~ freq + ppt.wet<br>N ~ txt + ppt.wet + twi + P + som | <0.001 | 845.8 | 0.95 | 0.88 | 0.10 | Tree | 0.38 | Graminoids | -0.48 | <0.001 |
| | | | | | | | | | | Freq | -0.28 | <0.001 |
| | | | | | | | | | | Txt.1 | 0.38 | <0.001 |
| | | | | | | | | | | N | -0.17 | 0.003 |
| | | | | | | | | | | P | 0.19 | 0.002 |
| | | | | | | | | | | Ppt.wet | 0.39 | <0.001 |
| | | | | | | | | Freq | 0.07 | Graminoids | 0.26 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.30 | <0.001 |
| | | | | | | | | Tsb | 0.65 | Freq | -0.88 | 0.039 |
| | | | | | | | | | | Ppt.wet | 0.17 | 0.018 |
| | | | | | | | | N | 0.28 | Txt.1 | -0.12 | <0.001 |
| | | | | | | | | | | Ppt.wet | -0.14 | <0.001 |
| | | | | | | | | | | Twi | 0.23 | <0.001 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | P | 0.27 | <0.001 |
| | | | | | | | | | | Som | 0.20 | <0.001 |
| Subshrubs | Subshrubs ~ tsb + comb.sand + twi + ppt.wet + N<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq<br>comb.sand ~ twi<br>N ~ comb.sand + ppt.wet | Subshrubs ~ tsb + txt + ppt.wet + N + P + twi<br>tsb ~ freq + ppt.wet<br>N ~ txt + ppt.wet + twi + som | <0.001 | 1352.7 | 0.94 | 0.86 | 0.10 | Subshrubs | 0.08 | Tsb | -1.80 | 0.071 |
| | | | | | | | | | | Txt.3 | -2.71 | 0.007 |
| | | | | | | | | | | Ppt.wet | 1.76 | 0.077 |
| | | | | | | | | | | N | -0.30 | 0.752 |
| | | | | | | | | | | P | -2.34 | 0.014 |
| | | | | | | | | | | Twi | -0.01 | 0.993 |
| | | | | | | | | Tsb | 0.68 | Freq | -21.7 | <0.001 |
| | | | | | | | | | | Ppt.wet | 4.18 | <0.001 |
| | | | | | | | | N | 0.25 | Txt.4 | -1.70 | 0.090 |
| | | | | | | | | | | Ppt.wet | -1.54 | 0.124 |
| | | | | | | | | | | Twi | 5.65 | <0.001 |
| | | | | | | | | | | Som | 5.05 | <0.001 |
| Fine fuel | Fuel ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | Fuel ~ Nherb.cov + NN.Grass + freq + txt + N + P + ppt.wet<br>freq ~ NN.Grass + Nherb.cov + ppt.wet<br>tsb ~ freq + ppt.wet<br>N ~ txt + ppt.wet + twi + P + som | <0.001 | 259.9 | 0.93 | 0.86 | 0.11 | Fuel | 0.59 | Nherb.cov | 0.33 | <0.001 |
| | | | | | | | | | | NN.Grass | 0.74 | <0.001 |
| | | | | | | | | | | Freq | -0.03 | 0.525 |
| | | | | | | | | | | Txt.1 | -0.03 | 0.501 |
| | | | | | | | | | | N | 0.28 | <0.001 |
| | | | | | | | | | | P | -0.07 | 0.141 |
| | | | | | | | | | | Ppt.wet | 0.23 | <0.001 |
| | | | | | | | | Freq | 0.25 | Nherb.cov | 0.20 | 0.003 |
| | | | | | | | | | | NN.grass | 0.27 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.37 | <0.001 |
| | | | | | | | | Tsb | 0.68 | Freq | -0.88 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.17 | <0.001 |
| | | | | | | | | N | 0.33 | Txt.1 | -0.12 | 0.039 |
| | | | | | | | | | | Ppt.wet | -0.14 | <0.018 |
| | | | | | | | | | | Twi | 0.23 | <0.001 |
| | | | | | | | | | | P | 0.27 | <0.001 |
| | | | | | | | | | | Som | 0.20 | <0.001 |
| Non-native perennial gramioids | NN.grass ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand | NN.grass ~ N.Grass + freq + tsb + txt + twi + ppt.wet + N<br>freq ~ ppt.wet + txt<br>tsb ~ freq | 0.001 | 712.6 | 0.96 | 0.91 | 0.10 | NN.grass | 0.40 | N.Grass | -0.34 | <0.001 |
| | | | | | | | | | | Freq | 0.24 | 0.008 |
| | | | | | | | | | | Tsb | 0.16 | 0.060 |
| | | | | | | | | | | Txt.1 | 0.16 | 0.004 |
| | | | | | | | | | | Twi | -0.25 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.37 | <0.001 |
| | | | | | | | | | | N | -0.06 | 0.228 |
| | | | | | | | | Freq | 0.23 | Ppt.wet | 0.48 | <0.001 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | tsb ~ freq | | | | | | | | | Txt.1 | 0.24 | <0.001 |
| | | | | | | | | Tsb | 0.65 | Freq | -0.81 | <0.001 |
| Native perennial graminoids | N.grass ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | N.grass ~ NN.Grass + freq + tsb + vc.sand + twi + ppt.wet<br>freq ~ ppt.wet + m.sand<br>tsb ~ freq | <0.001 | 713.3 | 0.92 | 0.83 | 0.12 | N.grass | 0.20 | NN.Grass | -0.44 | <0.001 |
| | | | | | | | | | | Freq | -0.03 | 0.789 |
| | | | | | | | | | | Tsb | -0.01 | 0.931 |
| | | | | | | | | | | VC.sand | 0.17 | 0.01 |
| | | | | | | | | | | Twi | -0.18 | 0.003 |
| | | | | | | | | | | Ppt.wet | 0.29 | <0.001 |
| | | | | | | | | Freq | 0.22 | Ppt.wet | 0.47 | <0.00 |
| | | | | | | | | | | M.sand | 0.21 | <0.001 |
| | | | | | | | | Tsb | 0.65 | Freq | -0.81 | <0.001 |
| Annual graminoids | An.grass ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | An.grass ~ Graminoids + freq + tsb + vf.sand + twi + ppt.wet<br>freq ~ ppt.wet + txt<br>tsb ~ freq | 0.001 | 720.9 | 0.93 | 0.85 | 0.12 | An.grass | 0.17 | Graminoids | 0.18 | 0.01 |
| | | | | | | | | | | Freq | -0.09 | 0.39 |
| | | | | | | | | | | Tsb | -0.10 | 0.36 |
| | | | | | | | | | | VF.sand | 0.26 | <0.001 |
| | | | | | | | | | | Twi | 0.26 | <0.001 |
| | | | | | | | | | | Ppt.wet | -0.27 | <0.001 |
| | | | | | | | | Freq | 0.33 | Ppt.wet | 0.40 | <0.001 |
| | | | | | | | | | | Txt.3 | 0.40 | <0.001 |
| | | | | | | | | Tsb | 0.65 | Freq | 0.81 | <0.001 |

b) Masked bobwhite quail (MBQ) plant species identified in the literature and by experts as an important food resource, thermal of escape cover from predators or nesting habitat

| **Habitat var.** | **Base model** | **Extended model** | **[1]*P* from $x^2$** | **[2]ΔAIC** | **[3]CFI** | **[4]TLI** | **[5]RMSEA** | **Reg. Models** | **$R^2$** | **Expl. Var.** | **[6]Est.** | **P(>\|z\|)** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| MBQ plant diversity | Sha.diversity ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | Sha.diversity ~ NN.Grass + tsb + freq + N + txt + ppt.wet<br>N ~ NN.Grass + txt + som<br>freq ~ ppt.wet + txt + NN.Grass<br>tsb ~ freq + NN.Grass | 0.004 | 54.1 | 0.97 | 0.91 | 0.09 | Sh.diversity | 0.26 | NN.Grass | -0.18 | 0.006 |
| | | | | | | | | | | Tsb | -0.001 | 0.992 |
| | | | | | | | | | | Freq | -0.08 | 0.443 |
| | | | | | | | | | | N | -0.29 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.28 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.41 | <0.001 |
| | | | | | | | | N | 0.15 | NN.Grass | -0.19 | 0.002 |
| | | | | | | | | | | Txt.2 | -0.05 | 0.473 |
| | | | | | | | | | | Som | 0.30 | <0.001 |
| | | | | | | | | Freq | 0.21 | Ppt.wet | 0.37 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.08 | 0.192 |
| | | | | | | | | | | NN.Grass | 0.17 | 0.009 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | Tsb | 0.66 | Freq | -0.84 | <0.001 |
| | | | | | | | | | | NN.Grass | 0.10 | 0.013 |
| MBQ number plants species | Sp.number ~ comb.sand + twi + ppt.wet + freq + tsb<br>comb.sand ~ twi<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq | Sp.number ~ NN.Grass + freq + N + som + txt + ppt.wet<br>N ~ txt + som<br>freq ~ NN.Grass + ppt.wet | 0.003 | 491.9 | 0.92 | 0.77 | 0.10 | Sp.number | 0.29 | NN.Grass | -0.39 | <0.001 |
| | | | | | | | | | | Freq | -0.01 | 0.931 |
| | | | | | | | | | | N | -0.27 | <0.001 |
| | | | | | | | | | | Som | -0.06 | 0.328 |
| | | | | | | | | | | Txt.2 | 0.20 | 0.001 |
| | | | | | | | | | | Ppt.wet | 0.40 | <0.001 |
| | | | | | | | | N | 0.11 | Txt.2 | -0.03 | 0.646 |
| | | | | | | | | | | Som | 0.33 | <0.001 |
| | | | | | | | | Freq | 0.20 | NN.Grass | 0.17 | 0.006 |
| | | | | | | | | | | Ppt.wet | 0.34 | <0.001 |
| MBQ graminoids | Graminoids ~ silt + twi + ppt.wet + freq + tsb<br>silt ~ twi<br>freq ~ ppt.wet + silt<br>tsb ~ freq | Graminoids ~ NN.Grass + txt + twi + freq + ppt.wet<br>txt ~ twi + ppt.wet<br>freq ~ ppt.wet + txt<br>tsb ~ freq | <0.001 | 53.7 | 0.93 | 0.82 | 0.05 | Graminoids | 0.26 | NN.Grass | -0.55 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.11 | 0.050 |
| | | | | | | | | | | Twi | -0.03 | 0.603 |
| | | | | | | | | | | Freq | 0.06 | 0.353 |
| | | | | | | | | | | Ppt.wet | 0.14 | 0.040 |
| | | | | | | | | Txt | 0.07 | Twi.2 | 0.03 | 0.614 |
| | | | | | | | | | | Ppt.wet | -0.25 | <0.001 |
| | | | | | | | | Freq | 0.18 | Ppt.wet | 0.44 | <0.001 |
| | | | | | | | | | | Txt.2 | 0.09 | 0.134 |
| | | | | | | | | Tsb | 0.63 | Freq | -0.81 | <0.001 |
| MBQ forbs | Forbs ~ N + silt + twi + ppt.wet + freq + tsb<br>N ~ silt + ppt.wet<br>freq ~ ppt.wet + silt<br>tsb ~ freq<br>silt ~ twi | Forbs ~ Graminoids + tsb + twi + ppt.wet + N + txt<br>freq ~ ppt.wet + txt<br>tsb ~ Graminoids + freq<br>txt ~ twi<br>N ~ txt + ppt.wet + som | <0.001 | 94.6 | 0.86 | 0.73 | 0.14 | Forbs | 0.28 | Graminoids | -0.24 | <0.001 |
| | | | | | | | | | | Tsb | -0.19 | 0.001 |
| | | | | | | | | | | Twi | 0.26 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.17 | 0.008 |
| | | | | | | | | | | N | -0.15 | 0.021 |
| | | | | | | | | | | Txt.2 | 4.48 | <0.001 |
| | | | | | | | | Freq | 0.18 | Txt.2 | 0.29 | <0.001 |
| | | | | | | | | | | Ppt.wet | 0.39 | <0.001 |
| | | | | | | | | | | Txt.2 | -0.14 | 0.019 |
| | | | | | | | | Tsb | 0.66 | Graminoids | 0.07 | 0.073 |
| | | | | | | | | | | Freq | -0.83 | <0.001 |
| | | | | | | | | Txt | 0.13 | Twi | 0.36 | <0.001 |
| | | | | | | | | N | 0.23 | Txt.2 | 0.37 | <0.001 |
| | | | | | | | | | | Ppt.wet | -0.12 | 0.032 |
| | | | | | | | | | | Som | 0.24 | <0.001 |

|  |  |  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| MBQ trees and shrubs | Woody ~ N + comb.sand + twi + ppt.wet + freq + tsb<br>N ~ comb.sand + ppt.wet<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq<br>comb.sand ~ twi | Woody ~ Graminoids + tsb + txt + N + P + ppt.wet<br>tsb ~ Graminoids + freq + ppt.wet<br>N ~ txt + ppt.wet + twi + P + som | 0.561 | 1379.3 | 1.0 | 1.0 | 0.00 | Woody | 0.22 | Graminoids | -0.22 | 0.002 |
|  |  |  |  |  |  |  |  |  |  | Tsb | 0.05 | 0.414 |
|  |  |  |  |  |  |  |  |  |  | Txt.2 | 0.32 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | N | -0.14 | 0.030 |
|  |  |  |  |  |  |  |  |  |  | P | 0.09 | 0.201 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | -0.10 | 0.146 |
|  |  |  |  |  |  |  |  | Tsb | 0.68 | Graminoids | 0.02 | 0.727 |
|  |  |  |  |  |  |  |  |  |  | Freq | -0.88 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | 0.16 | <0.001 |
|  |  |  |  |  |  |  |  | N | 0.32 | Txt.2 | -0.08 | 0.162 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | -0.13 | 0.031 |
|  |  |  |  |  |  |  |  |  |  | Twi | 0.23 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | P | 0.29 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | Som | 0.21 | <0.001 |
| MBQ subshrubs | Subshrubs ~ tsb + comb.sand + twi + ppt.wet + N<br>freq ~ ppt.wet + comb.sand<br>tsb ~ freq<br>comb.sand ~ twi<br>N ~ comb.sand + ppt.wet | Subshrubs ~ tsb + txt + ppt.wet + N + P + twi<br>tsb ~ freq + ppt.wet<br>N ~ txt + ppt.wet + twi + som | <0.001 | 1350.2 | 0.93 | 0.83 | 0.12 | Subshrubs | 0.14 | Tsb | 1.59 | 0.112 |
|  |  |  |  |  |  |  |  |  |  | Txt.4 | -2.36 | 0.019 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | 3.39 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | N | -0.40 | 0.687 |
|  |  |  |  |  |  |  |  |  |  | P | -2.01 | 0.045 |
|  |  |  |  |  |  |  |  |  |  | Twi | -2.013 | 0.044 |
|  |  |  |  |  |  |  |  | Tsb | 0.68 | Freq | -21.7 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | 4.18 | <0.001 |
|  |  |  |  |  |  |  |  | N | 0.25 | Txt.4 | -1.70 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | Ppt.wet | -1.54 | 0.124 |
|  |  |  |  |  |  |  |  |  |  | Twi | 5.65 | <0.001 |
|  |  |  |  |  |  |  |  |  |  | Som | 5.05 | <0.001 |

[1] Chi-square *p*-value from maximum likelihood model fit statistic that assumes model fit is no greater than if by chance, [2] Delta AIC values comparing base mode to the final extended model, [3] Comparative Fit Index, [4] Tucker-Lewis Fit Index, [5] Root mean squared error of approximation, and [6] Parameter estimates that are most equivalent to correlation coefficients.

**Figures**

Figure 1. Study area on a) Buenos Aires National Wildlife Refuge within the masked bobwhite management zone and all 2012 to 2015 plot locations b) historical range of the masked bobwhite quail that covers the US and Mexico. Semi-desert grassland habitat was mapped from masked bobwhite quail habitat suitability models developed by S.E. Sesnie from remotely sensed data and historical masked bobwhite locations (unpublished data).

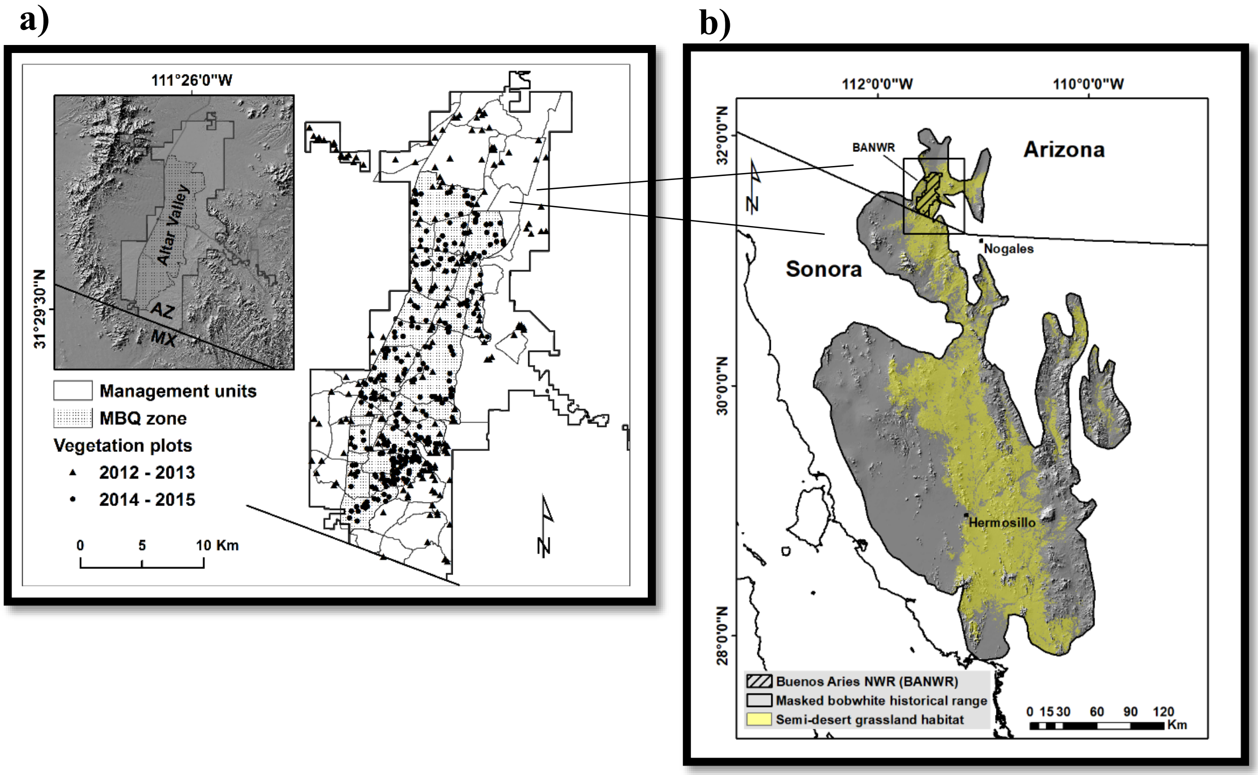


Figure 2. Worldclim bioclimatic layers for a) average total precipitation during the wettest quarter of the year and b) mean temperature during the wettest quarter of the year. Gray lines are management units and red lines are major paved roads on BANWR.

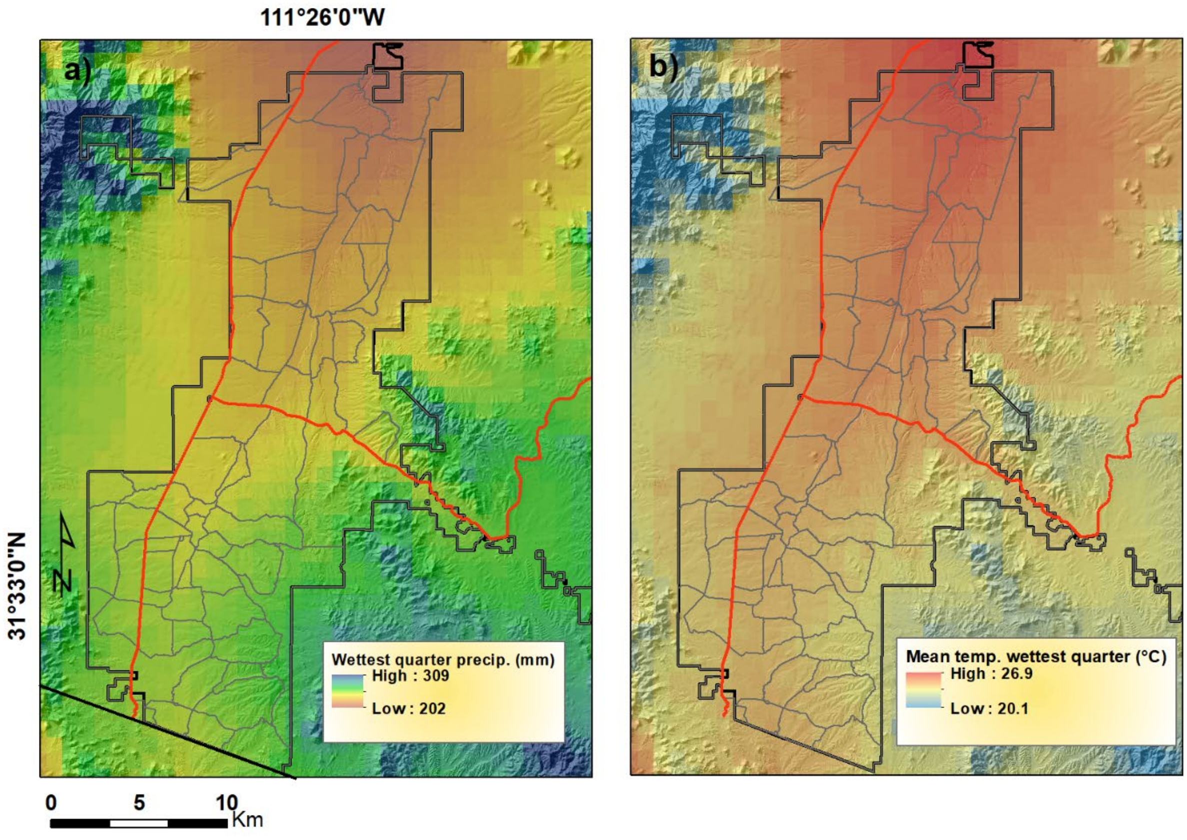


Figure 3. Vegetation plots sampled within the Buenos Aires National Wildlife Refuge masked bobwhite management zone during the primary growing season (July to October) in 2014 and 2015. Plots ($n$ = 239) were randomly selected at locations >50m from an unimproved road and within three fire frequency and terrain strata. Plots were spaced at a minimum of 250m apart.

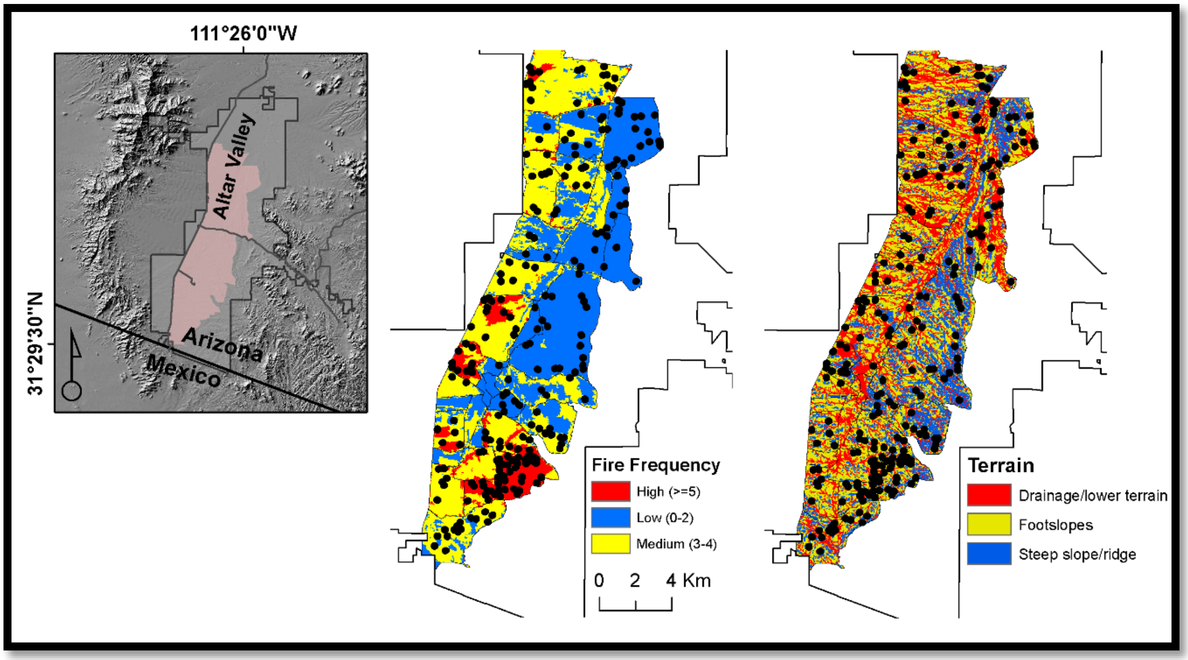


Figure 4. Conceptual model indicating a set of factors expected to influence masked bobwhite quail habitat and vegetation conditions within the study area. Separate structural equation models (SEM) were developed for each plant life-form (e.g. shrub, tree, grass, forb) and habitat characteristics examined following the conceptual diagram. Specific explanatory variables were selected based on those hypothesized to impact cover for each habitat characteristic examined.

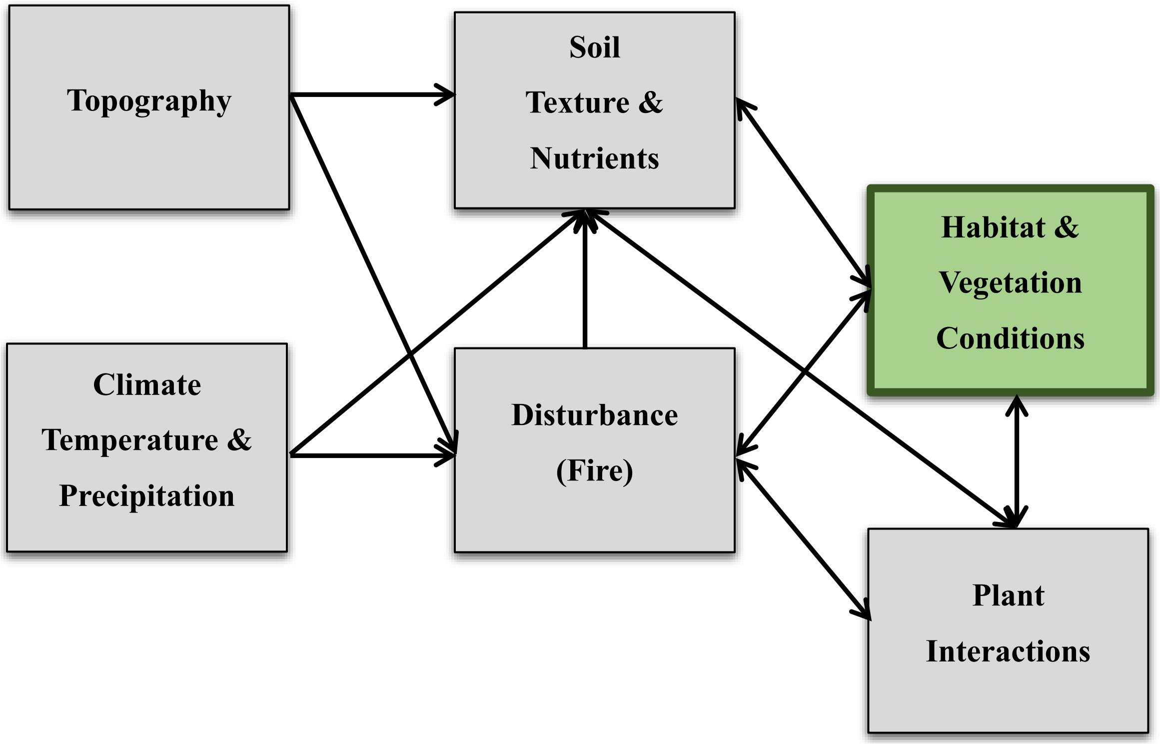


Figure 5. Generalized workflow used for developing plot and landscape-scale fine-fuels, fuel-type, and land cover data developed using Random Forest (RF) and Recursive Feature Extraction (RFE) machine learning techniques. We further integrated land cover data and Worldview-3 multispectral and other spatial data into habitat suitability models for masked bobwhite quail.

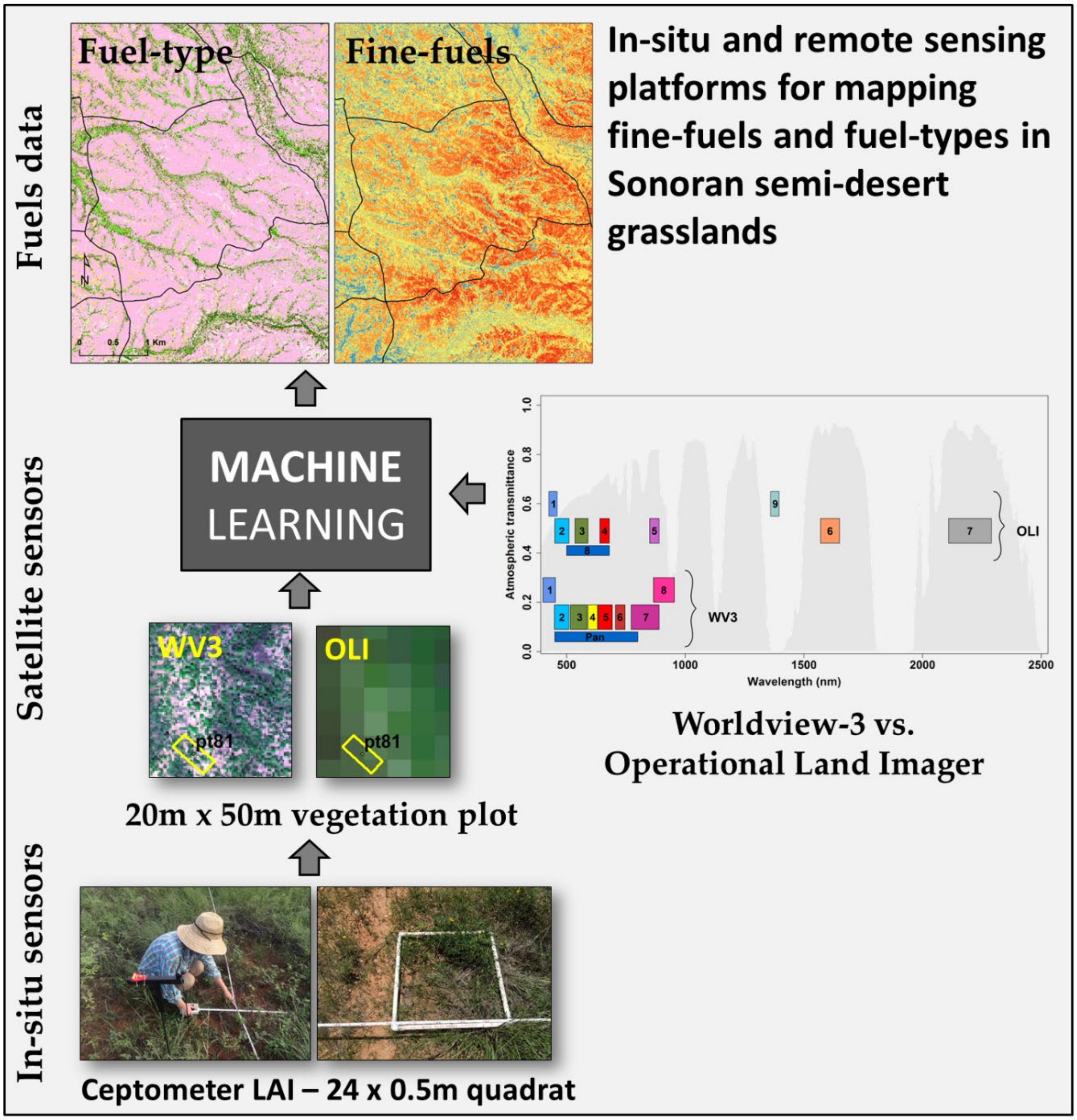


Figure 6. The number of plots measured in the study area according to a) fire frequency and b) the number of years since the last fire occurrence between 1985 and 2015. Locations with no fire occurrence were given a value of 30, to indicate locations with no fire history since refuge establishment.

Figure 7. NMDS results for plant species on plots overlaid with a) vegetation life forms, b) plant diversity and species richness, c) fire history variables between 1985 and 2015, d) bioclimatic variables for temperature (max) and precipitation (dry and wet season), e) soil texture, particle sizes and topography, and f) soil chemistry, organic matter and bulk density. Plant abbreviations are *Amaranthus palmari* (AMPA), *Sorghum halepense* (SOHA), and *E. lehmanniana* (ERLE).

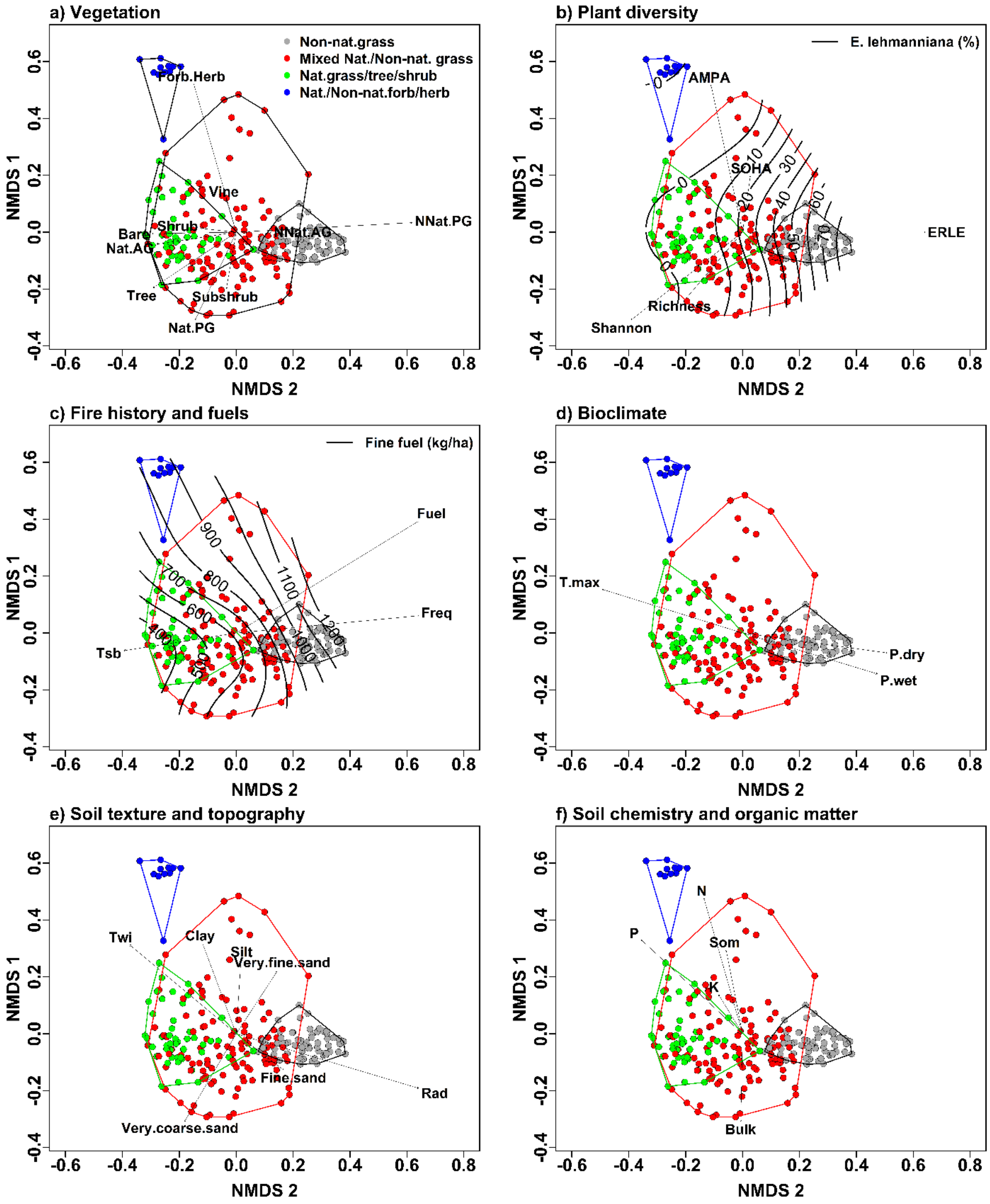
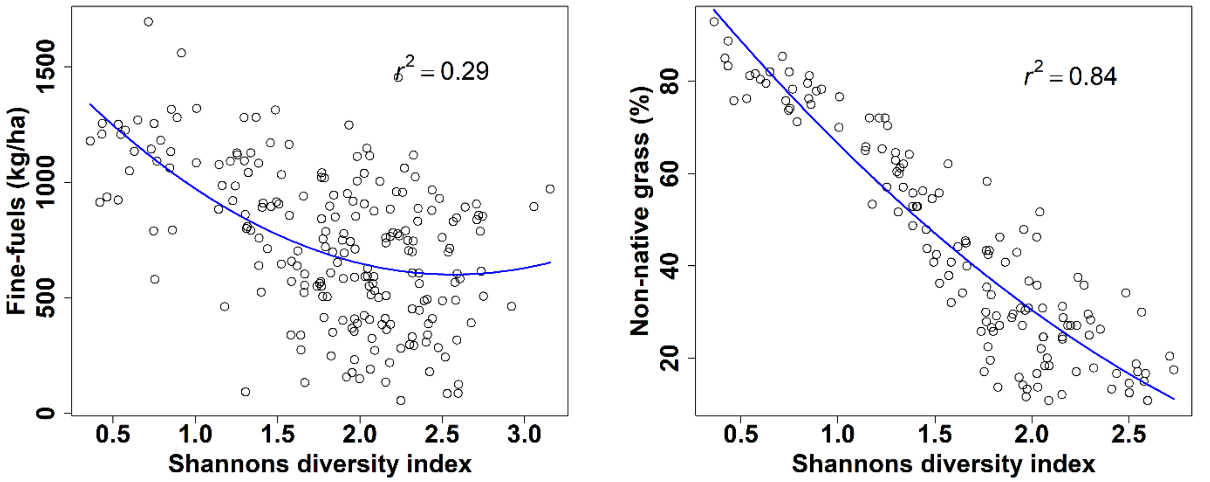
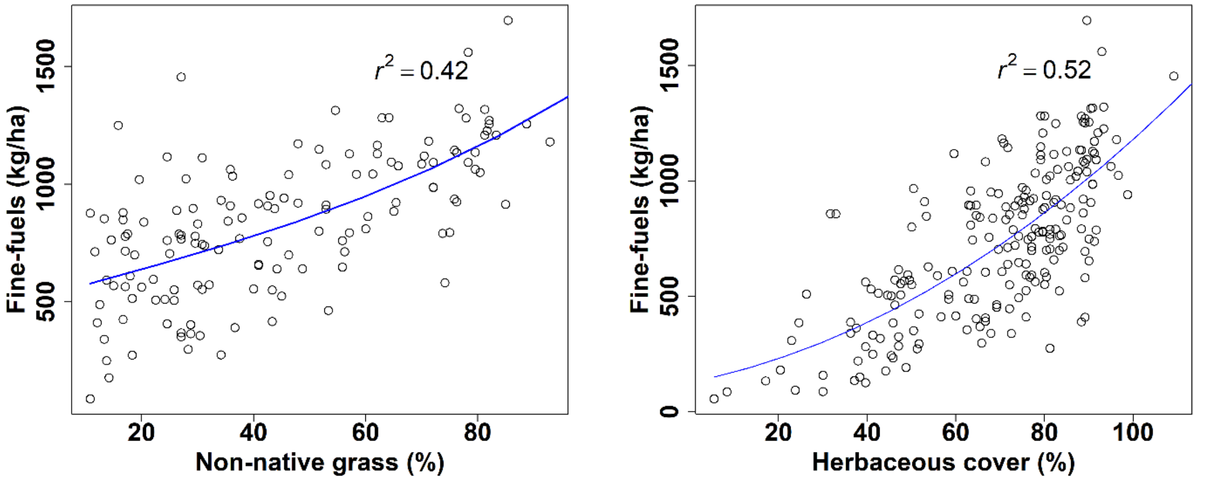


Figure 8. Non-linear relationships between a) fine-fuel and plant diversity, b) non-native grass cover >10 % and diversity, c) fine-fuel and non-native grass cover >10% and d) fine-fuel and all native and non-native herbaceous plant cover.



a) Plant diversity vs fuels

b) Plant diversity vs non-native grass (>10%)



c) Non-native grass (>10%) vs fuels

d) Herbaceous cover vs fuels

Figure 9. Habitat characteristics and fine-fuels from *n* = 239 plots within the masked bobwhite management zone on BANWR summarized by fire frequency.

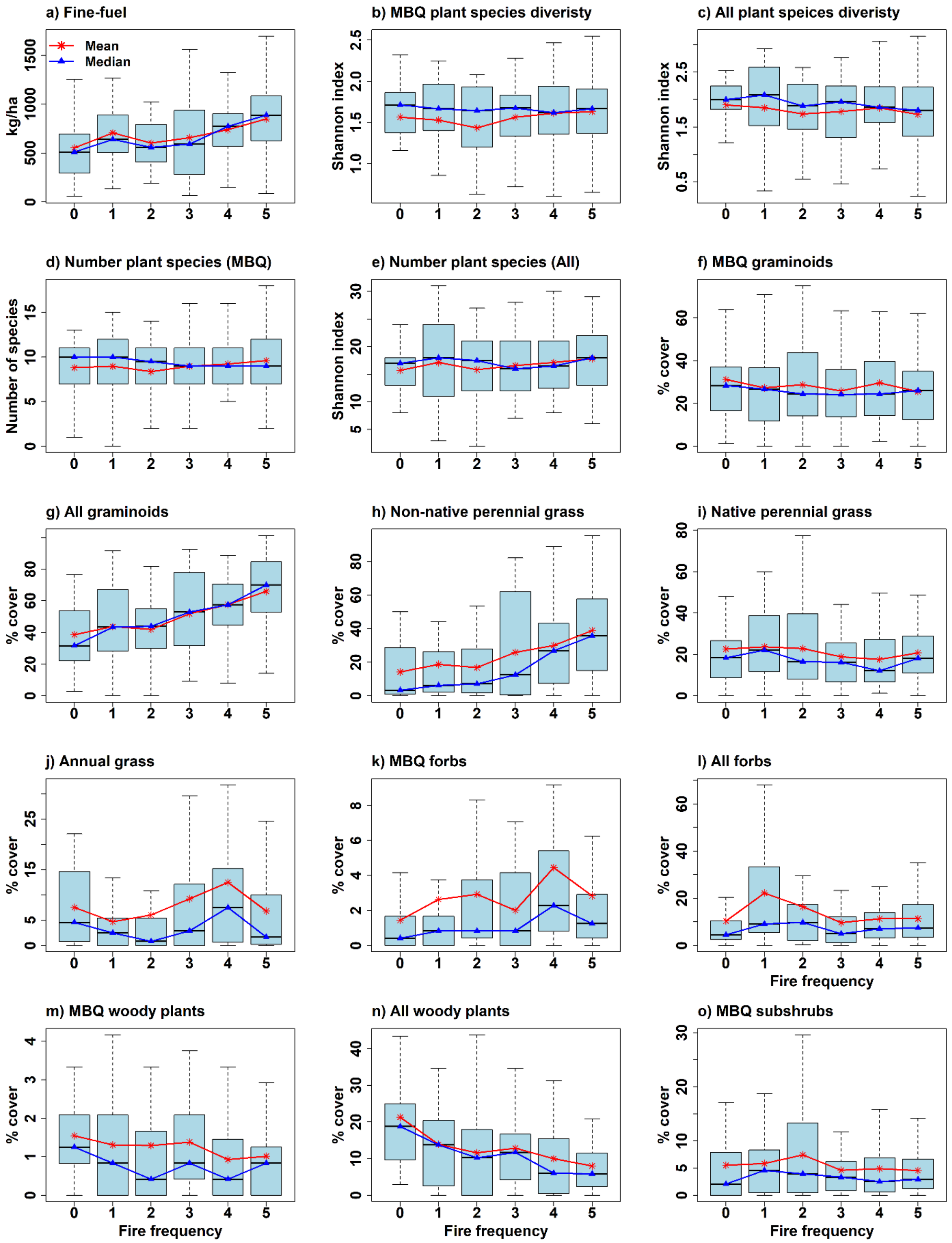


Figure 10. Habitat characteristics and fine-fuels from *n* = 239 plots within the masked bobwhite management zone on BANWR summarized by time since the last burn.

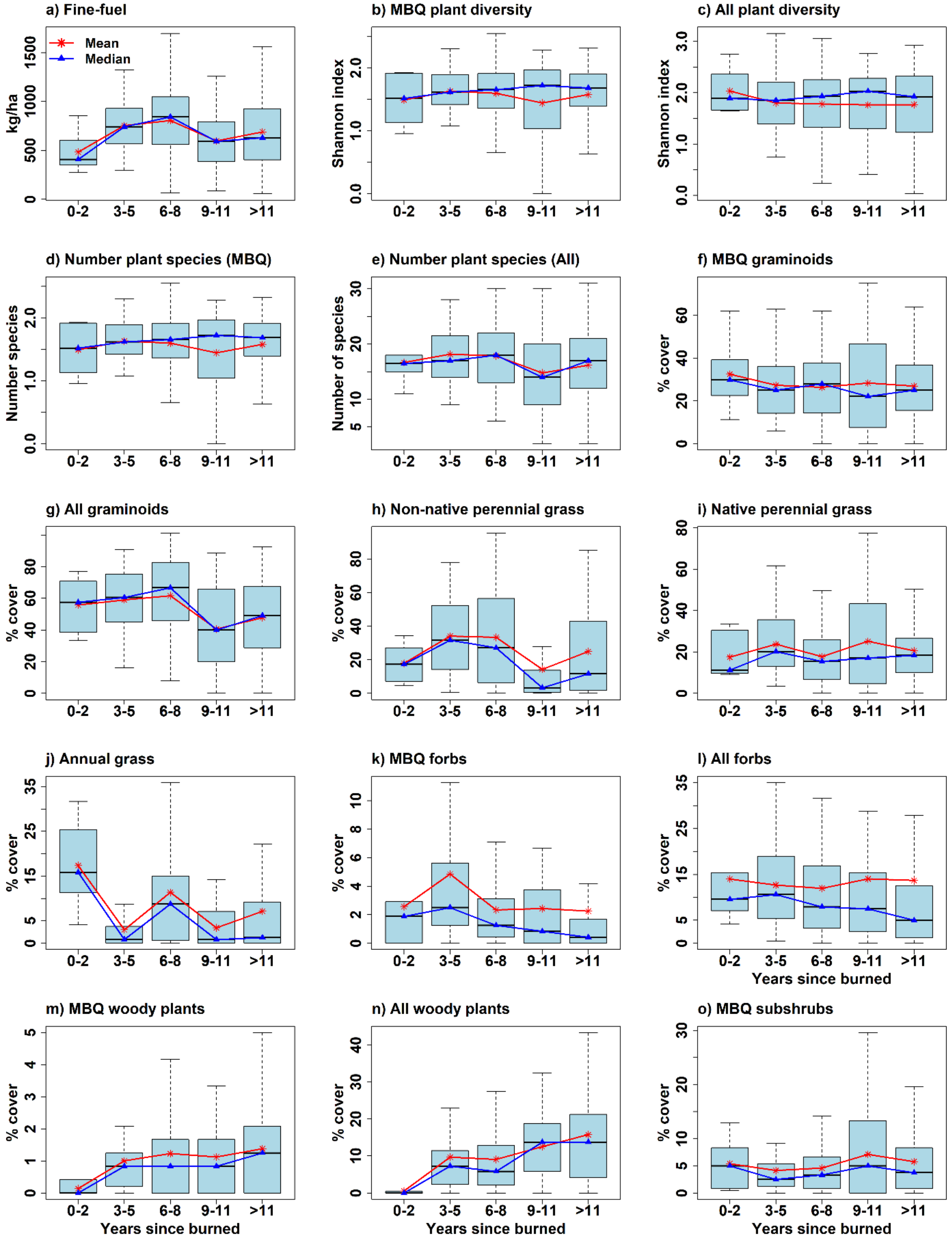


Figure 11. Multi-date Worldview-3 imagery from 2015 was used to map a) fine-fuel using herbaceous biomass plots measured on BANWR between 2014 and 2015 and b) land cover and fuel-types using vegetation plot measured between 2012 and 2015.

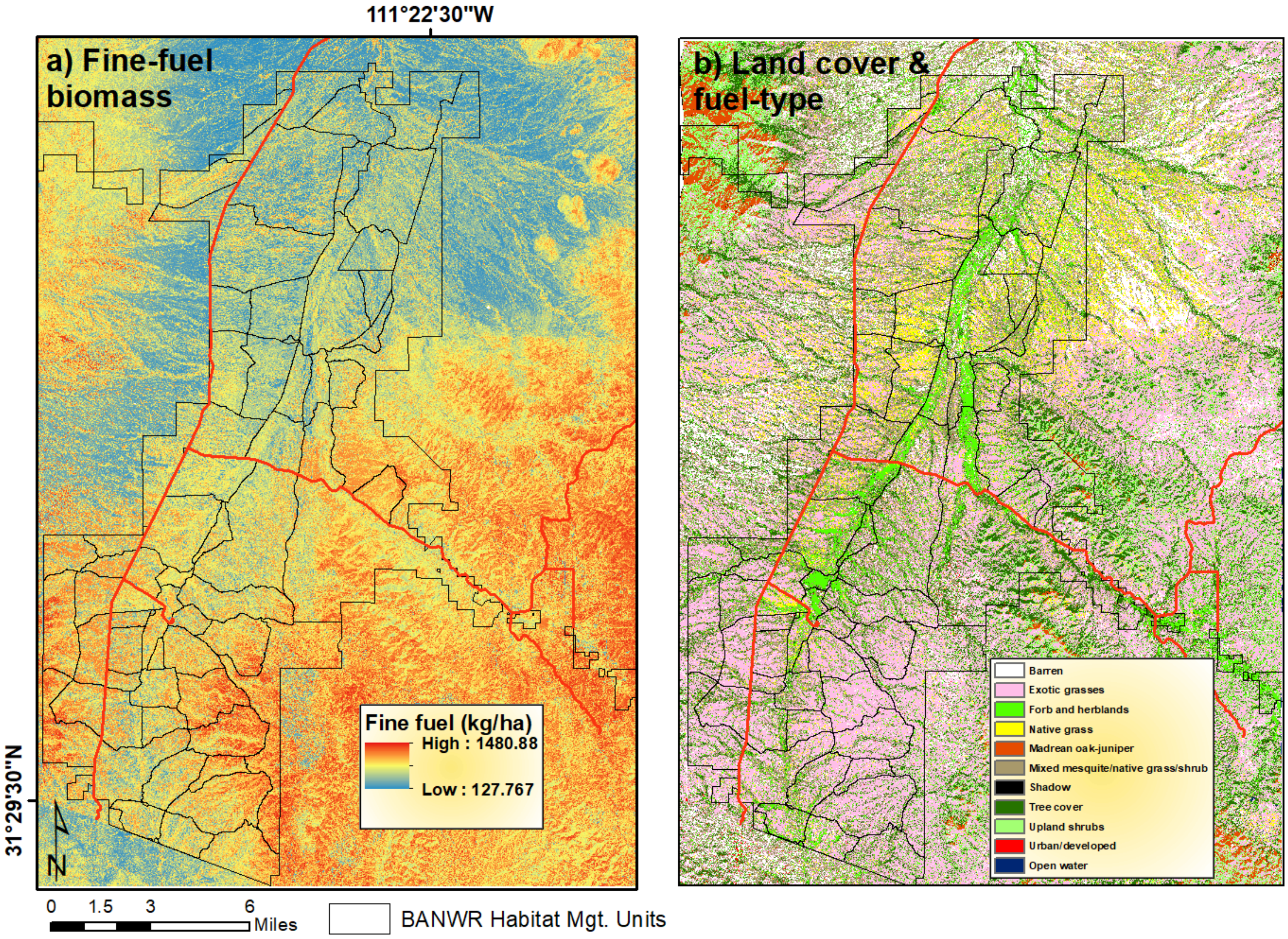
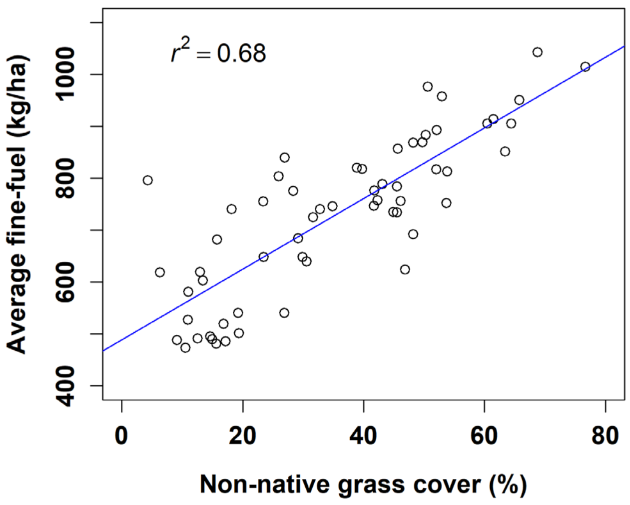


Figure 12. Fine-fuel and fuel-type predictions summarized for each management unit (*n* = 60) designated for prescribed burning on BANWR comparing a) average fine-fuel accumulation with non-native grass cover and b) average fine-fuel accumulation from native grass cover. Fuels data layers were developed from 2015 multi-date Worldview-3 imagery.


a) Fine-fuel vs. non-native grass

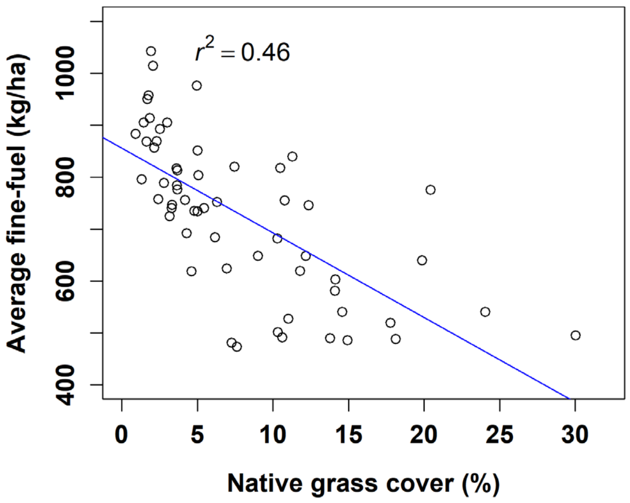
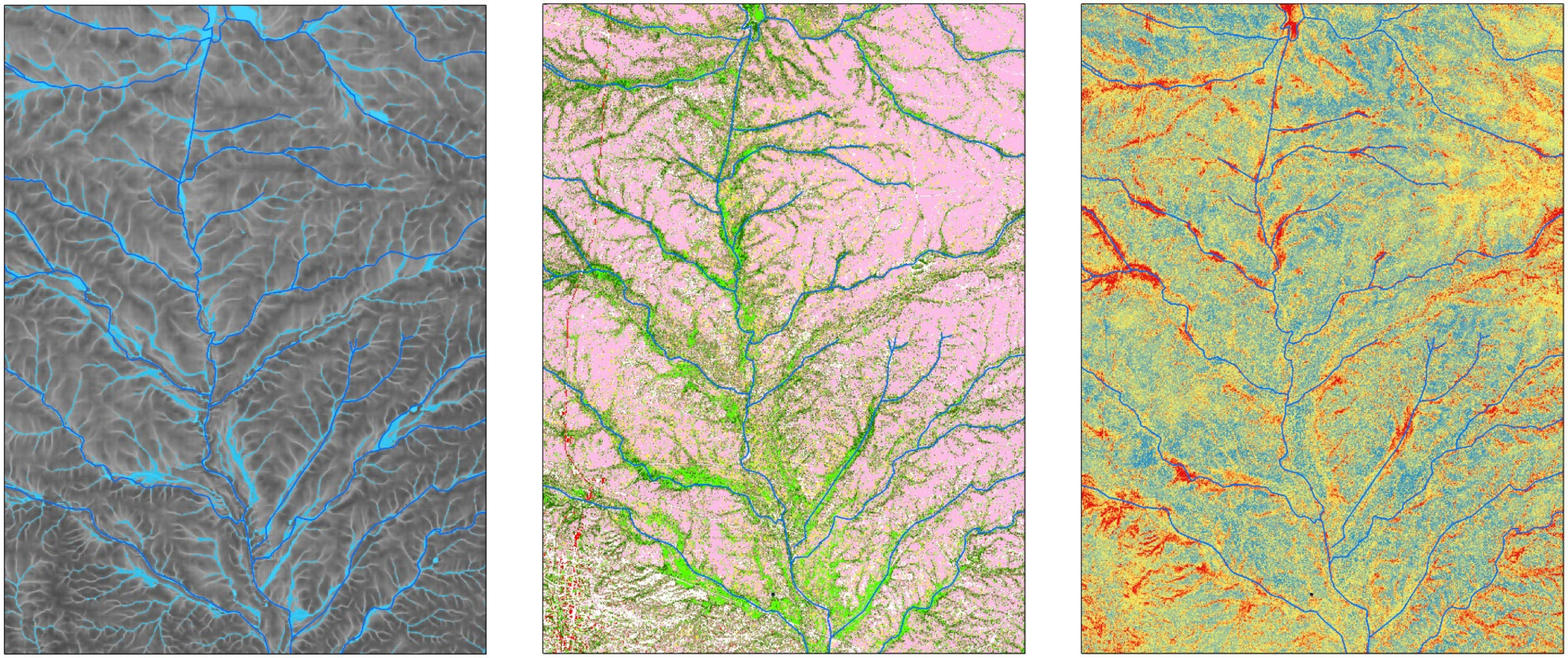

b) Fine-fuel vs. native grass

Figure 13. Topographic and vegetation conditions associated with higher habitat suitability for masked bobwhite quail. Data layers such as a) drainage networks mapped from topographic wetness index (TWI, light blue) and digitize drainages (dark blue) and b) mixed tree, native grass and shrub vegetation were strongly associated with c) areas of higher habitat suitability (yellow to red colors). More homogeneous upland areas invaded by *E. lehmanniana* showed low habitat suitability.



a) Principal drainage networks (light & dark blue)

b) Mixed vegetation along drainages versus non-native grasses in the uplands (pink).

c) Predicted higher (yellow to red) & lower (blue) habitat suitability

Figure 14. Comparison of average masked bobwhite quail habitat suitability and fire history variables for each management unit on the BANWR (*n* = 84). Average habitat suitability values for each unit were compared with a) average fire frequency and b) average time since the last burn across each unit. Habitat suitability data from 2017 field assessments (*n* = 408) indicated that areas burned (≥1 fire) between 1985 and 2015 had significantly lower masked bobwhite habitat suitability (*W* = 8742.5, *P*<0.001).

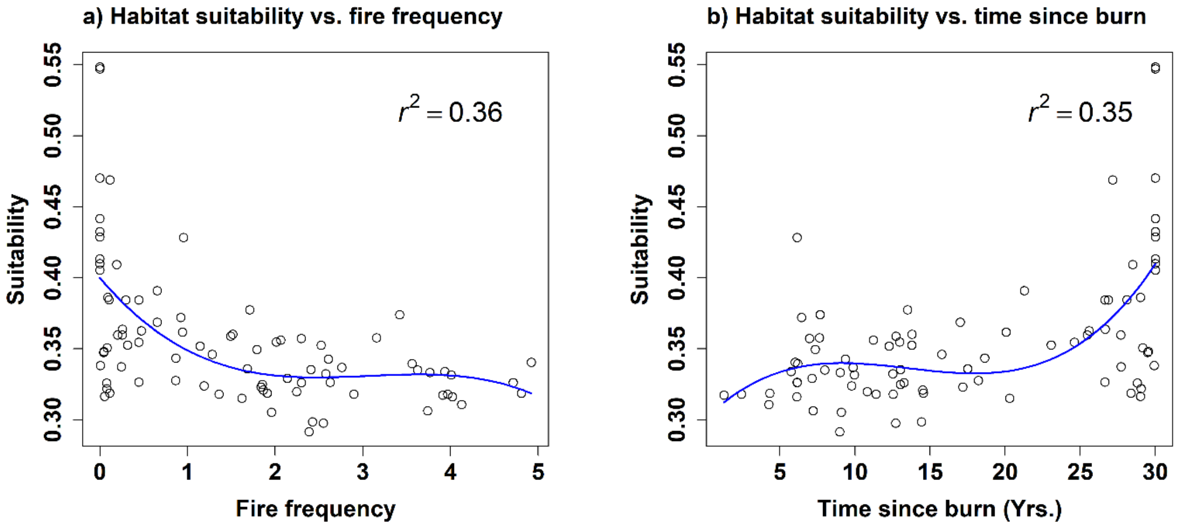
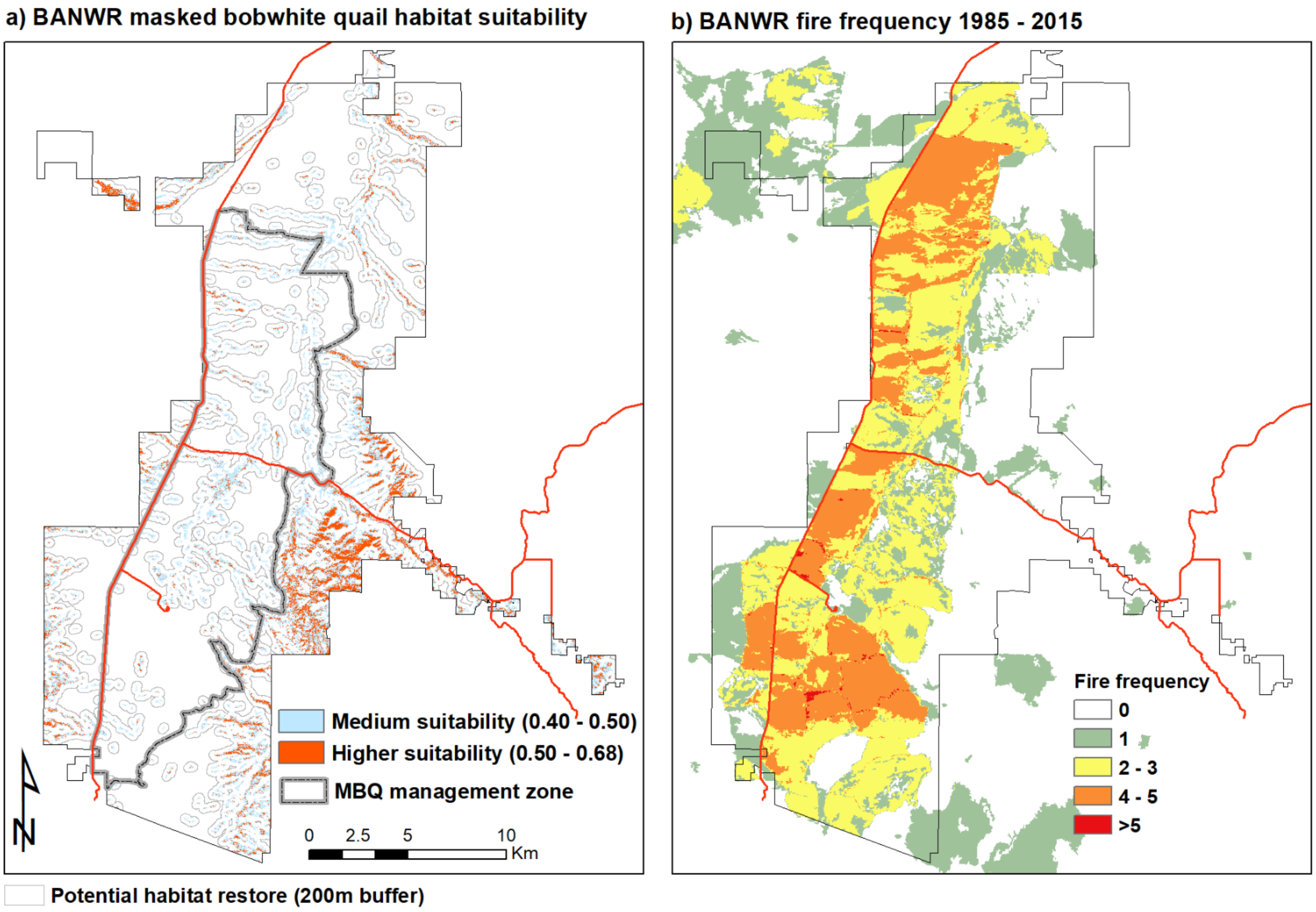


Figure 15. Mapped a) masked bobwhite habitat suitability for values between 0.4 and 0.50 (medium) and >0.50 (higher) from 2015 satellite imagery and other spatial data and b) fire frequency between 1985 and 2015 on BANWR. A 200m buffers around each habitat are potential areas for habitat rehabilitation, protection from fire and locations to improve connectivity.


a) BANWR masked bobwhite quail habitat suitability

b) BANWR fire frequency 1985 - 2015

## Appendices

**Appendix A**. Overview of masked bobwhite quail historical collections, habitat ecology and management approaches from Yurcich (2018).

### 1.1. Historical collections

Naturalists were intrigued by the masked bobwhite's rarity and restricted range in the US and Mexico, which generated over 250 museum collections between 1885 and 1993 (Brown et al. 2012). Tomlinson (1972a) and more recently Brown et al. (2012) summarized collections and their geographic locations that show the historical distribution of masked bobwhite and habitat affiliation with semi-desert and subtropical grasslands. Brown et al. (2012) revisited several collection sites in the US and in the Mexican state of Sonora. In general, collections were from subtropical desert environments below 1,060m elevation, according to Köppen-Geiger climate subdivisions (Thornwaite 1931). All sites were categorized with a mean annual rainfall of >350mm, with 250mm occurring from June to September, and mean annual temperatures of 18°C or above. A total of 14 masked bobwhite collections were taken in Arizona from the Altar Valley and Santa Cruz Valley (Brown et al 2012). Habitat information specific to the Altar Valley, Arizona, population indicated that most masked bobwhite occurred at or below 1060m and occupied level plains and mesas (Brown et al. 2012). Collections suggest that BANWR, located in the Altar Valley at an average elevation of approximately 1000m for the valley floor, was at the upper elevation range for masked bobwhite.

Historical and recent photographs by Tomlinson (1972a) and Brown et al. (2012) also help characterize past and present vegetation conditions, some of which show recovered native plant communities following the removal of livestock grazing. However, animal collection periods were short-term and sites were haphazardly selected, without a specific sampling design. Therefore, collection locations may not represent the full range of conditions once occupied by masked bobwhite. Nevertheless, specimens collected over time represent an important source of information for better understanding the range of conditions that once supported habitat structure and composition important to recovering masked bobwhite populations.

### 1.2. Life history and habitat ecology

The masked bobwhite is a species representative of productive semi-desert grasslands with a diversity of graminoids, forbs, and leguminous shrubs (Tomlenson 1972a). Masked bobwhite are known to occur on sites of level terrain and shallow drainages at moderate elevations (240m-

1,060m) with abundant grass cover and seed producing plants, a high diversity of forbs and interspersed woody plant cover (Goodwin and Hungerford 1977, Hernández et al. 2006, Brown et al. 2012). Historically, masked bobwhite distributions and habitat were described as ‘isolated’ relative to that of other sympatric species such as *Callipepla gambelii* and *C. squamata* (Aldrich and Duvall 1955). Goodwin and Hungerford (1977) considered masked bobwhite preferred habitat conditions as ‘intermediate’ between *C. gambelii* and *C. squamata* habitats and ‘edge-effect’ environments situated between species rich grasslands and dense grass-shrub pockets.

Goodwin and Hungerford (1977) more broadly characterized habitat as consisting of dense vegetation (75% to 100% ground cover) with relatively high plant diversity. Goodwin (1982) found that reintroduced masked bobwhite exclusively used bottomland habitat of drainages on BANWR. With few exceptions, Goodwin did not observe masked bobwhite in areas with less than 10 to 12 species of grasses and forbs, finding the areas supporting the best quail habitat have 18 to 20 plant species. Sites with less than 10% grass cover were avoided and preferred habitat had 22% to 30% combined grass and forb cover. Based on discussions with Goodwin, Reichenbacher and Mills (1984) provided more specific attributes of suitable habitat for masked bobwhites. They described suitable habitat as 10% to 15% woody plant cover, 12% to 15% grass cover, and 10% to 12% forb cover, with at least 450 kg/ha of grass standing crop, 300 kg/ha of forb standing crop, and 20 grass and forb species. Simms (1989) reported a similar finding for captive-released masked bobwhite on BANWR. Simms (1989) characterized habitat for masked bobwhites as consisting of 10% canopy cover of woody plants, 50% canopy cover of grass, and 15% canopy cover of forbs. Adequate diversity of grasses and forbs consisted of at least 10 species each.

Guthery et al. (2000) reported that the greatest habitat deficiency for masked bobwhites in Arizona was the quantity of herbaceous cover, followed by canopy cover of woody vegetation. Their study concluded that masked bobwhite preferred sites with woody cover with low hanging branches, and that reduced height and density led to increases in aerial predation and operative ground temperatures. Vegetation cover plays a key role in protecting quail from predation and heat (King 1998). Cover and visual obstruction provided by dense herbaceous vegetation and woody plants were important in habitats selected by captive-released masked bobwhite on BANWR (King 1998). Hernandez et al. (2006) suggested that the greatest habitat deficiency for the masked bobwhite in Arizona is decreased cover of native herbaceous plants. Interviews with

9 experts on masked bobwhite quail ecology conducted by LaRoche and Conway (2012) revealed that a lack of leguminous shrubs reduced winter forage, which is a likely limiting factor at the BANWR. King (1998) also concluded that important shrub components that produce reliable winter food (i.e. seeds) and cover were missing from BANWR.

Early diet studies provide insight into more specific food plants (Hernandez et al. 2006). Cottam and Knappen (1939) reported a diet consisting of 79.1% and 20.1% plant and animal material respectively, from stomach contents of 10 masked bobwhites collected in Sonora, Mexico during the month of October. They found seeds from a variety of plants including acacia (*Acacia angustissima*), ground cherry (*Physalis spp.),* panic grasses (*Panicum spp.),* day flower (*Commelina elegans*), and partridge pea (*Cassia leptadenia*). Insect material consisted primarily of grasshoppers (*Orthoptera*). Tomlinson (1984) concluded that an abundance of seed-producing plants such as legumes and panic grasses, in conjunction with an abundance of insects, was an important habitat component for masked bobwhites. Nevertheless, we know of no winter diet studies to confirm food preferences during this period when insect populations decrease.

Tomlinson (1972b) and Guthery et al. (2000, 2001) indicated that summer temperature, precipitation and humidity are critical to masked bobwhite survival and breeding. Tomlinson (1972b) found that a minimum daytime temperature of 13°C and average relative humidity of 25% are needed for active vocalization, which is important to breeding success. Masked bobwhite calling (70 days) and nesting season (90 days) are short relative to sympatric species found on BANWR. Summer precipitation events directly coincide with the beginning of masked bobwhite breeding (Hernandez et al. 2006, Tomlinson 1972a). Changes in summer precipitation impacting vegetation production may reduce chances of survival for masked bobwhite. The average annual rainfall within the geographic distribution of the historical range of masked bobwhites is 37 cm (Sonora) and 36 cm (Altar Valley, Arizona., USA), with about 75% occurring during the summer months (Jul–Sep; Tomlinson 1984, Camou et al. 1998). Camou et al. (1998) indicated that drought periods show substantially less average annual rainfall (<25 cm) on sites in Sonora, Mexico, but masked bobwhite populations persisted. Thus, masked bobwhites have evolved to inhabit a semiarid environment, with a pronounced seasonal peak in precipitation with intermittent drought periods and their life history reflects such evolution (Brown 1989).

### 1.3. Re-establishment and recovery

In 1937, populations of masked bobwhite were declining in Mexico as cattle ranching became more prevalent in the state of Sonora (Tomlinson 1972a). Efforts were made to translocate wild masked bobwhite captured in Mexico to the US for reestablishing populations during the 1940s and 1950s (Ligon 1952). These efforts were unsuccessful, with managers often releasing wild or captive-raised birds in locations outside of their historical habitat and without pre-conditioning to new environments (Brown et al. 2017). In some cases, released birds disappeared after a few months or did not survive through the winter period (Hernandez et al. 2006, Brown et al. 2017). By the 1950s the masked bobwhite was considered to be nearly extinct in the wild (USFWS 1995). As a result, the masked bobwhite was listed as endangered in 1968 with the passage of the Endangered Species Conservation Act (Hernandez et al. 2006). Masked bobwhite maintained an endangered status with the passage of the Endangered Species Act of 1973 (USFWS 1995). In 1969, the USFWS, in cooperation with the Arizona Game and Fish Department (AGFD), began a search for suitable release sites within the historic range in Arizona. In 1970, four experimental release sites were selected in the Altar Valley, AZ. Habitat conditions for most sites were marginal, because elevations were too high and grazing continued to occur (USFWS 1995).

Previous studies reported that pen-reared quail could be successfully reestablished if released in quality habitat (Capenter et al 1991, Stoddard 1931). Between 1970 and 1974 quail were raised to maturity in captivity at the Patuxent Wildlife Research Center in Maryland, USA, and released in Arizona without conditioning to the wild (Carpenter et al. 1991). In 1975, a 745 ha site was selected, and experimental releases were undertaken by the USFWS between 1975-78, on the Las Delicias Ranch in Arizona, adjacent to what is now BANWR (Ellis et al. 1978). This site was abandoned after early releases indicated that birds preferred floodplains and washes. Another 465 ha bottomland habitat area on the Buenos Aires Ranch was leased from the Victorio Land and Cattle Company (Brown et al. 2017).

Rigorous pre-release conditioning of pen-reared quail increased survival and annual reproduction in several cases (Ellis et al. 1978). Ellis et al. (1978) and Brown et al. (2017) provide further details on preconditioning techniques. Pre-conditioning quail produced birds that overwintered for consecutive years beginning in 1976, and the wild population included 74 calling males by 1979. Natural reproduction occurred in 1977, but early successes were not long lasting, as grazing later intensified within reintroduction areas (J. Goodwin, personal communication). Subsequently, it was determined that sizeable habitat areas managed specifically for masked bobwhite habitat were needed to increase the likelihood of long-term success. Also noteworthy was that after two summers with below average precipitation, surveys showed only a few birds remaining on these release sites (USFWS 1995). Ellis et al. (1978) point out that a primary limiting factor for establishing and sustaining a wild population within experimental areas was a lack of suitable habitat.

In 1985, the USFWS purchased the Buenos Aires Ranch to establish a large management area within the historical range of masked bobwhite to protect it from overgrazing while improving habitat conditions for the masked bobwhite. Grazing was discontinued upon BANWR establishment, and the reintroduction of fire became a priority tool for promoting quail habitat (USFWS 2003). From 1984 to 2012 more than 21,000 captive-raised birds were released on BANWR (Cohan et al. 2012). Subsequent studies by King (1998) and Babiak (2011) concluded that habitat conditions, particularly those needed for winter survival, were lacking on many parts of the refuge. While this has been a persistent concern, no detailed study assessing vegetation and habitat conditions has been conducted on BANWR.

Captive release efforts were discontinued in 2005, pending an effort to critically evaluate the effectiveness of the program (Hernadez et al. 2006). In 2010 BANWR reestablished the release program with the release of 74 individuals from the captive flock (USFWS 2014). However, by 2012, it was determined that no wild population persisted on the refuge (Brown et al 2012). Presently, all masked bobwhite known in the US are restricted to the captive breeding facility at BANWR and more recently established facilities at Africam Safari located in Puebla, Mexico, and Sutton Avian Research Center in Bartelsville, Oklahoma.

Wild masked bobwhite were known to exist on a few ranches in Sonora, Mexico, but no birds have been sighted since 2007 (Brown and Clark 2017). Following a several-year period without credible detections in Mexico, organized field surveys by the USFWS were discontinued in 2011. New automated recording and detection methods are currently being developed to sample locations in Mexico such as El Carrizo and San Dario Ranches near Benjamin Hill, Sonora where populations were last known to occur.

### 1.4. BANWR management history

Just prior to BANWRs establishment, prescribed fire was beginning to be used in conjunction with re-seeding of perennial grasses, to enhance forage production for cattle and reduce the abundance of mesquite (*P. velutina*). Sayer (2006) notes that Victorio Land and Cattle Company also invested heavily in water infrastructure, primarily drinkers for cattle, but also to maintain grass production in principal drainages with the use of spreader dams. Nearly all land with moderate topography and mesquite trees on the valley floor had been chained and grubbed during the 1970s according to Buenos Aires Ranch records (provided by N. Sayer). Brush piles from mesquite clearing were scattered throughout the valley that released masked bobwhite quickly learned to use for hiding cover and shade (J. Goodwin, personal communication). Reseeding was also extensive, but records indicate that this was expensive and primarily occurred north of ranch headquarters, which later became the main office and visitor center for BANWR. Seed mixes for both uplands and drainages typically contained native and non-native grasses. Some rangeland seeding treatments were also applied during this period, in the northern portion of the refuge with a mix of *Panicum antidotale*, *Sorghum halepense*, *Leptochloa dubia*, and *E. lehmanniana* primarily in drainage bottoms (unpublished data). Upland areas in the northern portion of BANWR were also apparently seeded with a mix of *Eragrostis chloromelas*, *Eragrostis intermedia*, *E. lemanniana* and *L. dubia* that covered approximately 7,500 ha and *E. lehamanniana* was at least 50% of seeds per pound. Ranch records indicate that prescribed burning was also conducted north of headquarters, but that neither seeding nor prescribed fire resulted in marked improvement in forage production. It is likely that overstocking of cattle and below average rainfall during much of the 1970s played a key role in unsuccessful range improvements.

Historically, fire in addition to precipitation was a principal driver of spatial and temporal variation in semi-desert vegetation that likely helped to maintain bobwhite habitat (Tomlinson 1972a, McPherson and Weltzin 1997). While only imprecise management records exist prior to the 1970s, it is unlikely that fire played a significant role in shaping grassland conditions in the Altar Valley after the introduction of cattle. Locations with dense grassland fuel that formerly would have carried fires were often quickly cleared by livestock grazing (Brown 1885, Tomlinson 1972a). The interruption of historical fire regimes has contributed to pronounced changes in vegetation structure, composition and function in native desert grasslands (Bahre 1991). Knowledge of the historical role of fire suggested that its reintroduction to desert grasslands would be an important tool for restoring masked bobwhite habitat (Brown and Ellis 1975), and long-term conservation of semi-desert grasslands.

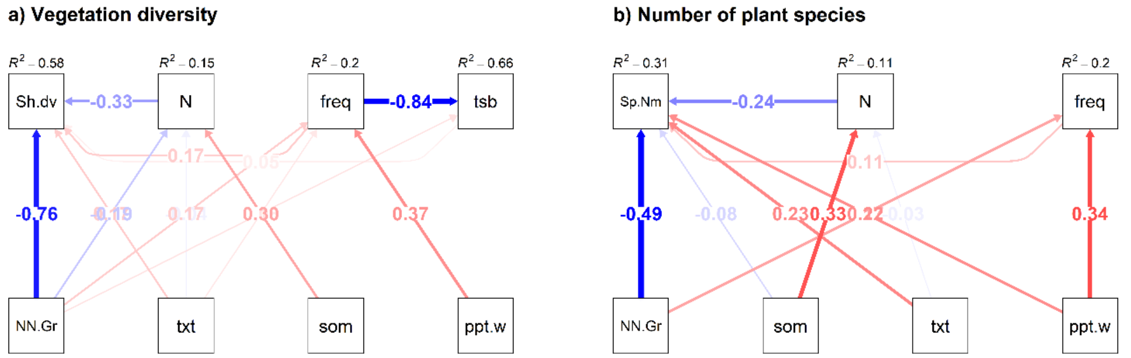
Fire regimes in southeastern Arizona grasslands were relatively frequent and covered extensive areas (Wright and Bailey 1982, Bahre 1985). Pre-settlement fires in some locations possibly occurred at intervals of 10 years or less, and were typically ignited by dry lightning in late June or July (McPherson 1995). However, the isolated nature of habitat historically occupied by masked bobwhite were possibly the result of deviations from average fire-return intervals. Leguminous and other shrubs utilized for winter forage and summer hiding cover are later successional woody species, which require longer time intervals to develop to maturity (Bock and Bock 1992). Alternatively, fast moving ground fires with lower residence time and severity may have had only a transitory effect on woody plants, burning mainly herbaceous material (Bock and Bock 1992). Post-fire effects on habitat composition and structure are also important. Tomlinson (1972a) found that masked bobwhite preferred stands of dense grasses and forbs situated in close proximity to trees and brush. Goodwin and Hungerford (1977) later pointed out that masked bobwhite that persisted in the Altar Valley during the 1970s used grass-shrub pockets and edge habitat along mesquite lined washes.

Since 1985, prescribed fire has been widely applied to rehabilitate grasslands on BANWR (Geiger and McPherson 2005, Villarreal et al. 2016). Although wildfires do occur, prescribed fire is used as the principal management tool to restore and maintain habitat for the masked bobwhite quail on BANWR (USFWS 2003). Prescribed fires typically occur in late spring to more closely correspond with the historical fire season, mimicking conditions in which native species evolved (Brooks and McPherson 2008). Yet prescribed fire has been implemented on some sites with greater frequency than what is has been revealed by the reconstruction of historical fire regimes, with some locations burned at intervals of every 3 to 5 years, while others were not burned at all (Babiak 2011).

Prescribed fires in semi-desert grasslands have been known to reduce woody plant cover of both trees and shrubs that have expanded beyond the drainages (Cable 1973, Brown and Ellis 1975). However, prior land use make predicting prescribed fire effects on habitat components important to masked bobwhite uncertain. For example, frequent fires may inhibit leguminous shrub development, a key component to masked bobwhite habitat (Tomlinson 1972a). Woody plant cover intermixed with grasses and forbs typical of masked bobwhite habitat, may also be negatively impacted by high-frequency burning (McPherson 1995). Relatively large burns just prior to nesting can increase distances to unburned, dense vegetation and make large areas uninhabitable for masked bobwhites (Wright and Bailey 1982, McPherson 1995). Conversely, post-fire responses in semi-desert grasslands suggest that short-term increases in seed-producing grasses and forbs (Bock and Bock 1992, Bock and Block 2005) may temporarily benefit masked bobwhites.

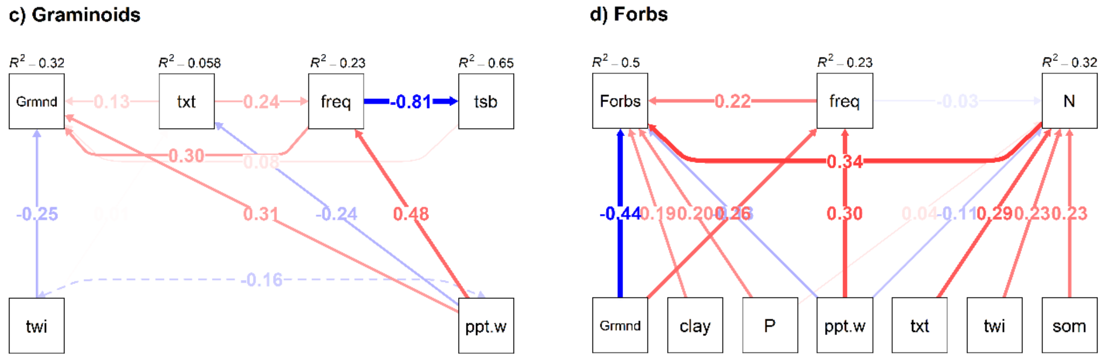
**Appendix B**. Structural equation model (SEM) path diagrams for each model of factors influencing or related to masked bobwhite quail habitat characteristics.

Figure 1. Structural equation model relationships for each masked bobwhite quail habitat characteristic showing model $R^2$ values and path coefficients. Models shown are for a) vegetation diversity (Shannon's H), b) number of plant species on plots, c) all graminoids, d) forbs and herbs, e) tree and shrubs, f) fine-fuel biomass, g) native perennial graminoids, h) non-native perennial graminoids, i) annual gramioinds, and k) subshrubs.



a) Vegetation diversity

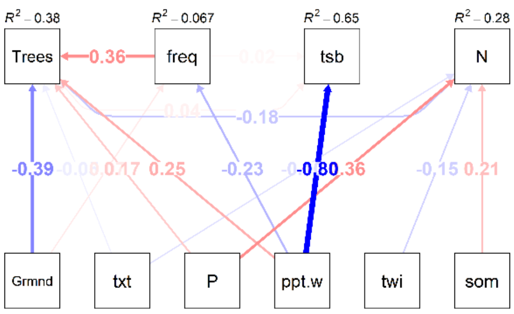
b) Number of plant species



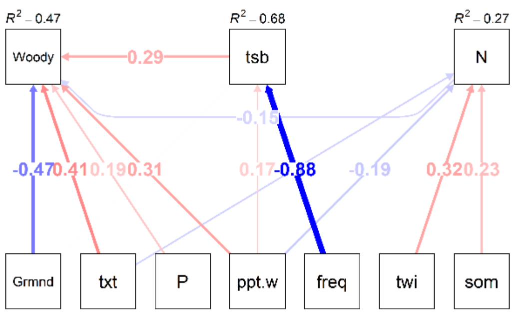
c) Graminoids

d) Forbs

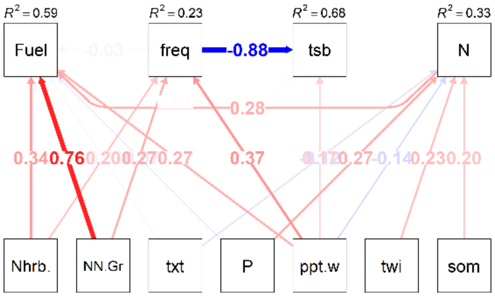
f) Trees



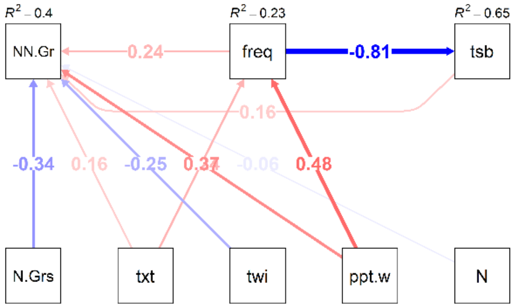
e) Trees and shrubs



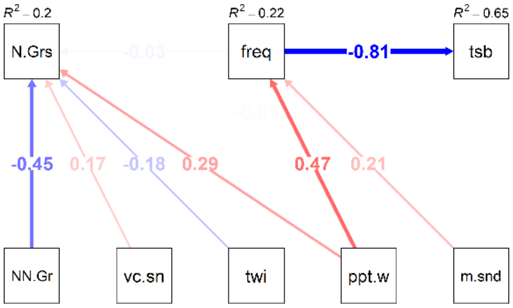
g) Fine-fuel



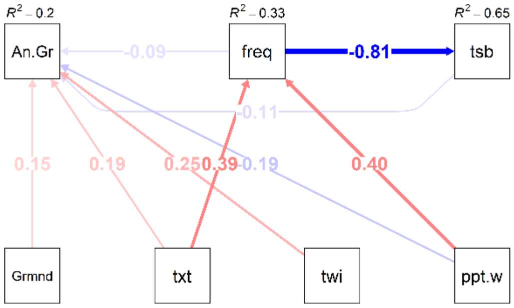
h) Non-native perennial graminoids



i) Native perennial graminoids



j) Annual graminoids



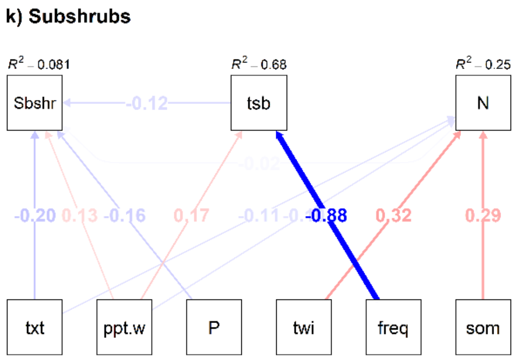
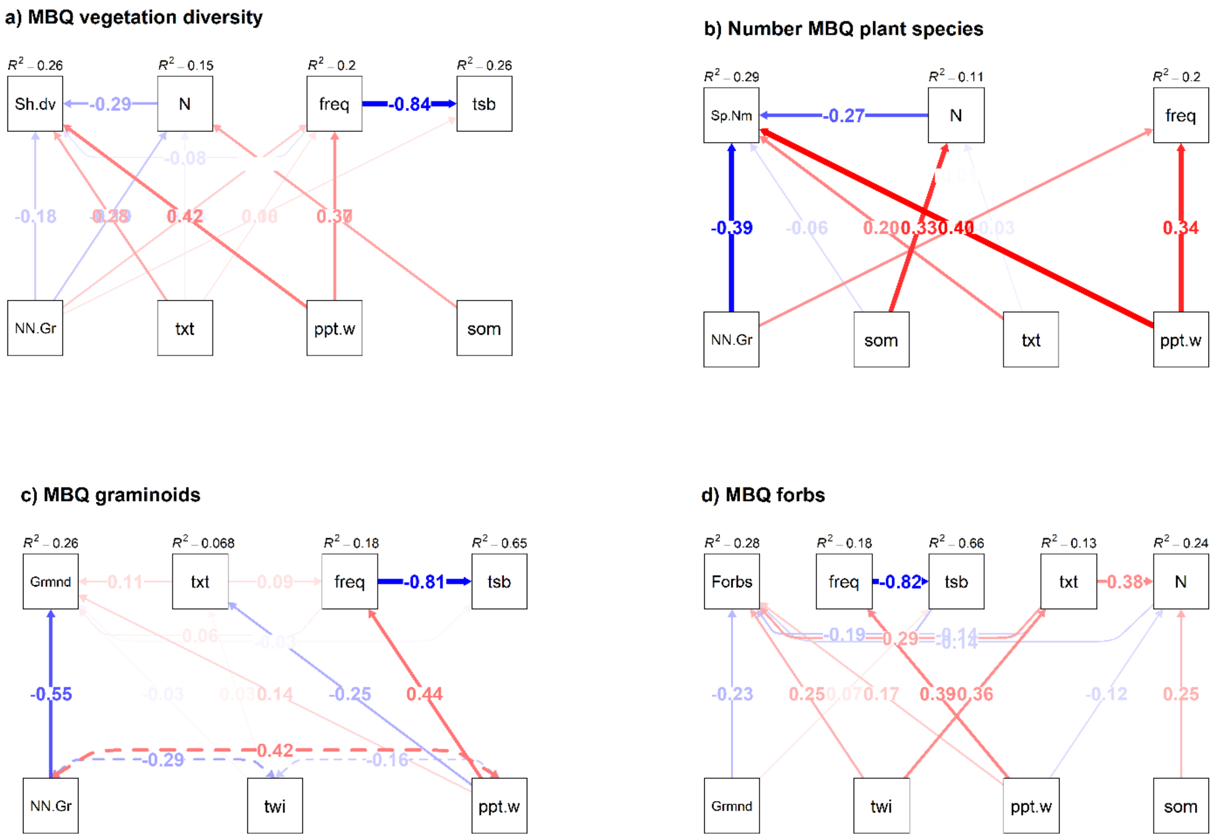
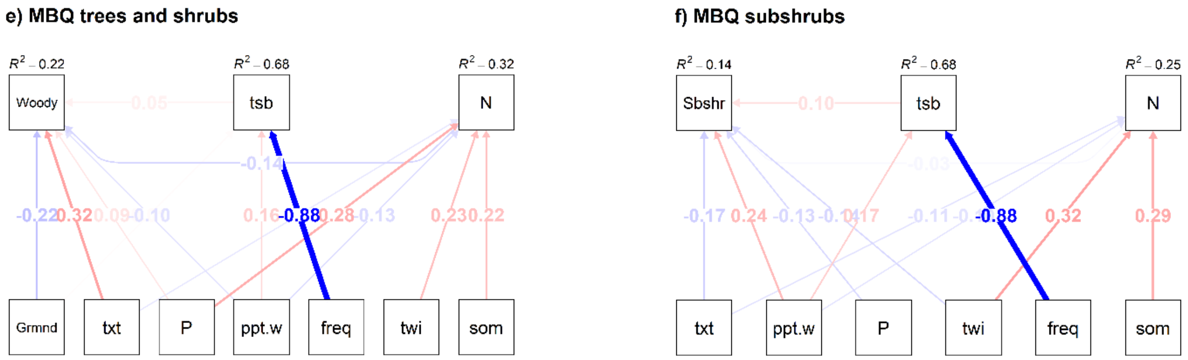


Figure 2. Structural equation model relationships using plant species important to masked bobwhite quail and each habitat characteristic showing model $R^2$ values and path coefficients. Models shown are for a) vegetation diversity (Shannon's H), b) number of plant species on plots, c) graminoids, d) forbs and herbs, e) tree and shrubs and g) subshrubs.




e) MBQ trees and shrubs
$R^2 = 0.22$
Woody
$R^2 = 0.68$
tsb
$R^2 = 0.32$
N
0.05
-0.14
-0.22
0.32
0.09
-0.10
0.16
-0.88
0.28
-0.13
0.23
0.22
Grmnd
txt
P
ppt.w
freq
twi
som
f) MBQ subshrubs
$R^2 = 0.14$
Sbshr
$R^2 = 0.68$
tsb
$R^2 = 0.25$
N
0.10
-0.03
-0.17
0.24
-0.13
-0.10
0.17
-0.11
-0.88
0.32
0.29
txt
ppt.w
P
twi
freq
som

**Appendix C.** Worldview-3 2015 satellite image classification error matrices for fuel- and land cover-types.

Table 1. Classification error matrices from a) separate testing and training sample data and b) all data bootstrapped error assessment using data left ‘out of bag’ (OOB) at each Random Forest classification model iteration. Users, producers and overall accuracy are summarized for each classification routine, however the final classifier used all sample data.

| | **a)** | | | | | | **True** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Class** | **Ba** | **Xgr** | **Fh** | **Ngr** | **Oj** | **Mgrw** | **Sh** | **Tr** | **Upsh** | **Ur** | **Wa** | **Users acc. (%)** | **Producers acc. (%)** |
| **Predicted** | **Ba** | **42** | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 4 | 0 | 89.4 | 95.5 |
| | **Xgr** | 0 | **53** | 0 | 1 | 0 | 3 | 0 | 0 | 1 | 0 | 0 | 91.4 | 85.5 |
| | **Fh** | 0 | 1 | **18** | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 90.0 | 72.0 |
| | **Ngr** | 1 | 4 | 1 | **2** | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 22.2 | 22.2 |
| | **Oj** | 0 | 0 | 0 | 0 | **11** | 0 | 0 | 0 | 0 | 0 | 0 | 100.0 | 78.6 |
| | **Mgrw** | 0 | 4 | 5 | 6 | 0 | **20** | 0 | 4 | 1 | 0 | 0 | 50.0 | 83.3 |
| | **Sh** | 0 | 0 | 0 | 0 | 2 | 0 | **4** | 0 | 0 | 0 | 0 | 66.7 | 100.0 |
| | **Tr** | 0 | 0 | 1 | 0 | 1 | 0 | 0 | **25** | 0 | 0 | 0 | 92.6 | 80.6 |
| | **Upsh** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | **2** | 0 | 0 | 100.0 | 50.0 |
| | **Ur** | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | **27** | 0 | 96.4 | 87.1 |
| | **Wa** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | **12** | 100.0 | 100.0 |
| | | | | | | | | | | | | | **Overall acc. %** | **83.1** |

| | **b)** | | | | | | **True** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Class** | **Ba** | **Xgr** | **Fh** | **Ngr** | **Oj** | **Mgrw** | **Sh** | **Tr** | **Upsh** | **Ur** | **Wa** | **Users acc. (%)** | **Producers acc. (%)** |
| **Predicted** | **Ba** | **170** | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 11 | 0 | 92.9 | 89.5 |
| | **Xgr** | 0 | **182** | 0 | 8 | 0 | 17 | 0 | 0 | 2 | 0 | 0 | 87.1 | 76.8 |
| | **Fh** | 0 | 0 | **60** | 1 | 0 | 13 | 0 | 3 | 0 | 0 | 0 | 77.9 | 80.0 |
| | **Ngr** | 0 | 23 | 3 | **15** | 0 | 24 | 0 | 0 | 0 | 0 | 0 | 23.1 | 48.4 |
| | **Oj** | 0 | 1 | 0 | 0 | **41** | 0 | 1 | 0 | 2 | 0 | 0 | 91.1 | 91.1 |
| | **Mgrw** | 2 | 24 | 8 | 7 | 0 | **69** | 0 | 7 | 0 | 0 | 0 | 59.0 | 50.0 |
| | **Sh** | 0 | 0 | 0 | 0 | 3 | 0 | **16** | 0 | 0 | 0 | 0 | 84.2 | 88.9 |
| | **Tr** | 1 | 0 | 3 | 0 | 0 | 9 | 1 | **95** | 0 | 2 | 0 | 85.6 | 87.2 |
| | **Upsh** | 0 | 7 | 0 | 0 | 1 | 5 | 0 | 3 | **9** | 0 | 0 | 36.0 | 69.2 |
| | **Ur** | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | **117** | 0 | 87.3 | 90.0 |
| | **Wa** | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | **54** | 98.2 | 100.0 |
| | | | | | | | | | | | | | **Overall acc. (%)** | **79.6** |

**Appendix D.** Project Accomplishments

Publications, theses, reports and proceedings

1. Sesnie, S.E., H. Eagleston, L. Johnson, and E. Yurcich. 2018. In-situ and remote sensing platforms for mapping fine-fuels and fuel-types in Sonoran semi-desert grasslands. Remote Sensing 10:1358; https://doi.org/10.3390/rs10091358.
2. Eagleston, H. and S. Sesnie. *Submitted.* Developing alternative fuel models with World View-3 imagery to estimate fire behavior patterns in a semi-desert grassland, Arizona USA. International Journal of Wildland Fire. ASPRS Annual Conference and International Lidar Mapping Forum Proceedings. Denver, CO January 27 – 31, 2019.
3. Eagleston, H. and S. Sesnie. *In revision*. Developing alternative fuel models with World View-3 imagery to estimate fire behavior patterns in a semi-desert grassland. International Journal of Wildland Fire.
4. Yurcich, E. 2018. Prescribed fire effects on habitat components important to the critically endangered masked bobwhite quail (Colinus virginianus ridgewayi) using a 30-year Landsat derived fire history. MS Thesis, School of Earth and Sustainability, Northern Arizona University, Flagstaff AZ. 112p.
5. Eagleston, H. and S.E. Sesnie. DOI Remote Sensing Activities 2016. Alternative fuel models to estimate fire behavior patterns. On-line report: https://eros.usgs.gov/doi-remote-sensing-activities/2017/fws/alternative-fuel-models-estimate-fire-behavior-patterns
6. Sesnie, S.E. and H. Eagleston. DOI Remote Sensing Activities 2016. WorldView-3 data informs masked bobwhite quail management. DOI Remote Sensing Working Group. On-line report: https://eros.usgs.gov/doi-remote-sensing-activities/2016/fws/worldview-3-data-informs-masked-bobwhite-quail-management
7. Sesnie, S.E. 2015. DOI Remote Sensing Activities 2015. Estimating prescribed fire effects on semidesert vegetation composition and structure.. DOI Remote Sensing Working Group. On-line report: https://eros.usgs.gov/doi-remote-sensing-activities/2015/fws/estimating-prescribed-fire-effects-semidesert-vegetation-composition-and-structure
8. Sesnie, S.E. DOI Remote Sensing Activities 2013. Field leaf area index (LAI) measurements for estimating fuel loads. U.S. Department of Interior. DOI Remote Sensing Activities 2013. DOI Remote Sensing Working Group. On-line report: https://eros.usgs.gov/doi-remote-sensing-activities/2013/fws/field-leaf-area-index-lai-measurements-estimating-fuel-loads

Presentations

1. Yurcich, E. Estimating Prescribed Fire Effects on Semidesert Vegetation Components Important to the Masked Bobwhite Quail (Colinus Virginanus Ridgwayi) Using a 30 Year Landsat Derived Fire History Data. 2018. The Wildlife Society's 25th Annual Conference. October 7-11, 2018.
2. Yurcich, E., S.E. Sesnie, and T.D. Sisk. 2016. Estimating prescribed fire effects on semidesert vegetation composition and structure using 30-year Landsat derived fire history data. Beyond hazardous fuels: managing fire for social, economic, and ecological benefits. Tucson, AZ. November 28 – December 2
3. Sesnie, S.E., E. Yurcich, H.E. Eagleston, and L. Johnson. 2016 Determining prescribed fire and fuel treatment compatibility with semidesert grassland habitat rehabilitation for the critically endangered masked bobwhite quail (*Colinus virginianus ridgwayi*)**.** Masked Bobwhite Endangered Species Recovery Team meeting. May 20, Benjamin Hill, Mexico.
4. Sesnie, S.E., E. Yurcich, H.E. Eagleston, and L. Johnson. 2016. Determining prescribed fire and fuel treatment compatibility with semidesert grassland habitat rehabilitation for the critically endangered masked bobwhite quail (*Colinus virginianus ridgwayi*. Masked Bobwhite Endangered Species Recovery Team meeting. March 3, Tucson, AZ.
5. Yurcich, E., S.E. Sesnie, and T.D. Sisk. 2015. Estimating prescribed fire effects on semidesert vegetation composition and structure using 30-year Landsat derived fire history data. Biennial Conference of Science and Management on the Colorado Plateau & Southwest Region. October 5 – 8, Flagstaff, AZ
6. Sesnie, S.E., L.J. Zachmann, B.G. Dickson, E. Yurcich, J.M. Rundall, and L. Johnson. 2015. Multiscaled approaches to southwestern arid lands vegetation monitoring, modeling, and management. Biennial Conference of Science and Management on the Colorado Plateau & Southwest Region. October 5 – 8, Flagstaff, AZ
7. Sesnie, S.E., M. M. Altemus, and L. Johnson. 2014. Field ceptometer LAI measurements for monitoring semidesert grassland biomass, fuel loads, and wildlife habitat conditions with remotely sensed data. ASPRS Rio Grande Chapter Annual Spring Meeting, April 5th, 2014. University of New Mexico, Albuquerque, NM.

8. Sesnie, S.E. 2015. Presentation to the masked bobwhite quail recovery team: Mapping habitat suitability for masked bobwhite quail (*Colinus virginianus ridgwayi*) using MODIS phenology products. February 18th, Tucson, AZ.

Media

1. ArcGIS story map; Plight of the Masked Bobwhite https://www.arcgis.com/apps/MapJournal/index.html?appid=637adcdb1ae74448aecbf5d35a4db7d4
2. ArcGIS story map; Determining Prescribed Fire Compatibility with Masked Bobwhite Quail Rehabilitation https://usbr.maps.arcgis.com/apps/MapSeries/index.html?appid=6df14f948d4d4cd68b4d61b4e4764807
3. Monitoring and Adaptive Management Case Study Handout; Determining Prescribed Fire Compatibility with Masked Bobwhite Quail Rehabilitation https://lccnetwork.org/sites/default/files/Resources/Determining%20Prescribed%20Fire%20Compatibility%20with%20Masked%20Bobwhite%20Quail%20Habiat%20Rehabilitation_1.pdf