US Climate Alliance Grant Program for NWL Research 2020

Quantifying Carbon Sequestration in Nevada's Rangelands

Final Report

Credit: L. Provencher/TNC, 2014; Non-native annual species grasslands and forblands of central Nevada

Credits: L. Provencher/TNC, 2012; Mid-successional crested wheatgrass seeding in black sagebrush from Hamlin Valley UT

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March 31st, 2022

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Recommended citation: Provencher L, Byer S, Badik K. 2022. Quantifying Carbon Sequestration in Nevada's Rangelands. Final Report to the Nevada Division of Natural Heritage, American Forests, and US Climate Alliance Grant Program for NWL Research 2020. The Nature Conservancy, Reno, NV. 52 pp.

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SUMMARY

The goal of this study is to show that degraded sagebrush (*Artemisia* spp.) shrublands dominated by non-native annual grasses and forbs (NNAGF) might be a feasible source for additional carbon sequestration as a result of seeding. Typically, productive forests are considered preferable targets for carbon sequestration, but not arid shrublands. The practical interest of this study is to determine if seeding to improve sagebrush shrublands for wildlife, soil stability, and/or livestock forage can also provide the justification to create a carbon mitigation program. The Area of Interest (AOI) for mapping included: a) Great Basin ecoregion in Nevada, Utah, and eastern California, b) the southern Columbia Plateau ecoregion in Nevada, northeast California, southern Idaho and southeast Oregon, and c) the Nevada portion of the Mojave Desert.

To measure the contributions of seeding perennial grass and woody species in sagebrush dominated by NNAGF, we needed three components: a) A new map of non-native annual species cover for a large geography for which past simulation results were applicable and where we could delineate areas dominated by NNAGF for calculation of total added carbon sequestration resulting from seeding; b) previously developed state-and-transition simulation models (STSM) that could be adapted to simulate and budget cost-effective seeding in shrublands (mostly sagebrush) dominated by NNAGF; and c) a carbon stock-and-flow sub-model coupled to the disturbances of the STSM. Therefore, three products were created: (1) A NNAGF species cover index map spanning a 485,623-km^2 (120 million-acre) geography; (2) estimated net amount of carbon stored per acre through restoration of NNAGF to perennial vegetation, and (3) estimated cost of storing net carbon through restoration per map pixel.

Mapping

- The map of the AOI was obtained from analysis of Sentinel-2 imagery that was captured by twin satellites with 13 spectral bands at 10, 20, or 60-m resolutions (https://sentinel.esa.int/web/sentinel/user-guides/sentinel-2-msi).
- We compared Normalized Vegetation Difference Index (NDVI) from the spring and summer. In a pixel with annual species; the NDVI should be much higher in the spring than in summer. A pixel primarily containing moisture-resilient sagebrush or perennial species should not appreciably change in NDVI between seasons. Therefore, the large difference of NDVI between spring and summer at a pixel represented the dramatic senescence of annual species that typically occurs at the end of spring and early summer (May/June).
- Published metrics of seasonal differences were calculated to represent phenological or ecological processes that helped identify areas where annual species were present. Climate data from the Idaho GRIDMET climate products (ultimately derived from 4km PRISM data: https://developers.google.com/earth-engine/datasets/catalog/IDAHO_EPSCOR_GRIDMET#description) were used to produce precipitation metrics as potential predictors. Additional abiotic factors such as lithography, elevation, aspect, slope, and the 10-m resolution Continuous Heat-Insolation Load Index (CHILI), which is a 10-m dataset derived from USGS digital elevation models available through Google Earth Engine, were included.
- A statistical Random Forest model was used to identify pixels of different NNAGF cover and run in Google Earth Engine.
- Two sources of training data were used to validate the Random Forest model.
    - The Nature Conservancy's (TNC) local remote sensing maps that were heavily ground-verified. TNC training data were obtained from two projects; one in central Nevada with the Cortez Range in its center and the other from southwest Utah centered on Pine Valley in Beaver and

Millard Counties. These two landscapes were retained because they post-dated the launch of the Sentinel 2 satellites. NNAGF median cover was obtained from each of the TNC cover classes to be used as training data.
    - Bureau of Land Management's (BLM) Assessment, Inventory, and Monitoring (AIM) point data that include estimated cover of non-native annual species were used to train and validate the Random Forest model. Only plots sampled after the launch of the Sentinel 2 satellites could be used. The AIM dataset was filtered to observations made in 2020 and limited to the extent of the project area. The AIM data were primarily collected outside the state of Nevada in 2020 with an exception in the northwest portion of the state. A total of 5,080 training points were available, although these points were not evenly distributed across the project area.
- The Random Forest model was trained with 70% of the data (3,595) and 30% were held out for validation (1,485).

State-and-Transition Simulation Modeling

- Existing state-and-transition simulation models (STSM) from three landscapes spanning climates from northern Nevada's southern Columbia Plateau ecoregion to southwest Utah's southeast Great Basin ecoregion were re-used for carbon modeling. The models from the IL Ranch of north-central Nevada (southern Columbia Plateau ecoregion), TS-Horseshoe Ranch of central Nevada (Great Basin ecoregion), and Pine Valley-Mountain Home Range project of the southeast Great Basin ecoregion in southwest Utah provided the latitudinal gradient to estimate the net carbon stored in restored rangelands under different land management experiences and cost.
- Two spatial management scenarios were defined in the STSM in each landscape: Custodial (no seeding and custodial livestock grazing and fire management) and Seeding management. Seeding management only included seeding actions in sagebrush, mountain shrub, grasslands with a sagebrush component, and incised riparian floodplain that converted to sagebrush and basin wildrye (*Leymus cinereus*) systems. Seed species could be introduced, native, or a mix of both. Constraints on seeding management were already defined in the models when these were originally developed with partners. That is, seed mixes were previously developed with partners and were not a focus of this study, but rather used to assess carbon sequestration if successfully used. Additionally, the TS-Horseshoe Ranch was simulated with too frequent fire, as a fortuitous modeling accident, and historic fire return intervals.
- In each landscape's STSM, a carbon continuous-dynamics stock-and-flow sub-model was populated with stock and flow estimates from the US Geological Survey's carbon models for CONUS such that ecological disturbances in the STSM model (e.g., fire) simultaneously changed fluxes and stocks in the stock-and-flow sub-model. Seeding actions also changed stocks and flows.
- STSMs were simulated for 30 years into the future starting in the year each model was originally simulated: 2015 for the IL and TS-Horseshoe Ranches and 2018 for the Pine Valley-Mountain Home Range landscape. High levels of seeding were implemented for 20 years and the simulation continued without management for another 10 years after cessation of seeding.

Findings

- The 485,623-km^2 (120 million-acre) map of NNAGF cover was partitioned into three climatic zones to which each STSM was applied for carbon estimation:
    - About 18,047 km^2 of NNAGF dominant vegetation was estimated in the IL Ranch, which is located in northern Nevada's southern Columbia Plateau ecoregion, as candidate for

seeding that met four conditions: (a) The pixel belonged to sagebrush systems or systems with a significant sagebrush component during succession; (b) the pixel was on a slope <15% for seeding with tractor pulled equipment (this constraint only applied to California, Idaho, Nevada, and Oregon); (c) the pixel was at or above the 25.4 cm (10 inch) precipitation zone; and (d) patches of adjacent and touching pixels (i.e., a patch) had to be at least 202 ha (500 acres) for financially feasible operations (i.e., a contractor will not bid on seeding less than 202 ha, although we know from experience that many contractors do not bid on seedings less than 2,023 ha [5,000 acres]).

- Nevada and California's Great Basin ecoregion corresponded to the central and southern Basin and Range geologic province not generally influenced by the North American monsoon and where the TS-Horseshoe Ranch was situated. About 5,177 km^2 of NNAGF dominant vegetation was estimated to be appropriate for seeding following the same four criteria as above.
- Utah's Great Basin ecoregion in the northern and southern Basin and Range geologic province that was influenced by the North American monsoon also contained the Pine Valley-Mountain Home Range landscape. While the very eastern edge of Nevada is technically influenced by monsoonal precipitation, estimation of carbon was limited to Utah for ease of computation. About 6,890 km^2 of NNAGF dominant vegetation was estimated to be appropriate for seeding, but the slope constraint was increased to 30% as a rough chain pulled by two bulldozers can be used in Utah to imprint seed into the soil and improve germination success and survival.

- Net biome productivity (NBP) at each site represents the net carbon flux in the system where positive values indicated carbon sinks from the atmosphere and negative values were sources of carbon to the atmosphere. NBP reported in this study ranged from -35 to 84 g $C \cdot m^{-2} \cdot yr^{-1}$ without seeding. NBP was 55% to 93% stored in the soil stock. NBP values were smaller than the maximum values reported for other more productive systems but within the range of wet to forested systems reported in the literature. Comparison to a national model for the Piedmont ecoregion indicated that rangelands compared favorably to Appalachian forests for carbon sequestration (25 g $C \cdot m^{-2} \cdot yr^{-1}$).

- The TS-Horseshoe Ranch with frequent fire was the exception by being a carbon source of -35.3 g $C \cdot m^{-2} \cdot yr^{-1}$, which was highly dominated by shrublands dominated by NNAGF prone to frequent fires and former seedings with many invaded by NNAGF species. The same model with normal "historic" fire was positive, thus a sink of carbon from the atmosphere.

- The modeled carbon sink difference (NBP difference between Seeding and Custodial management scenarios) due to seeding estimated in this study ranged from 0.61 g $C \cdot m^{-2} \cdot yr^{-1}$ (TS-Horseshoe Ranch with historic fire) and 0.74 g $C \cdot m^{-2} \cdot yr^{-1}$ (IL Ranch) to 19.9 g $C \cdot m^{-2} \cdot yr^{-1}$ (Pine Valley-Mountain Home Range). In the case of the Pine Valley-Mountain Home Range, seeding could result in 104 g $C \cdot m^{-2} \cdot yr^{-1}$ sequestered total, mostly as a stable form in the soil. Seeding would lessen the carbon source to the atmosphere of the TS-Horseshoe Ranch with frequent fire to -15 g $C \cdot m^{-2} \cdot yr^{-1}$ (compared to -35 g $C \cdot m^{-2} \cdot yr^{-1}$), which is a significant offset.

- An interesting result of models reported here was that the extent of NNAGF dominated areas appeared to positively correlate with the magnitude of the carbon sink difference per unit area. In other words, greater NNAGF area offered the greatest opportunity for uplift from restoration as it

would be difficult to improve the ecological condition of a system already close to the reference condition. Being closer to the reference condition usually means the system has a higher representation of more mature vegetation classes that, presumably, have grown a more substantial root system contributing carbon to the soil. We believe that the IL Ranch fits that description as it was both a sink of carbon without the added effect of seeding and contribution to carbon storage could be further increased, albeit weakly, by seeding NNAGFs.

- A critical aspect of the concept of carbon sequestration uplift was, and will be, the cost per unit area of seeding. The cheapest average cost of seeding was in Utah, which converted to about $\$66 \cdot ha^{-1}$ ($\$163 \cdot ac^{-1}$), whereas the highest cost was $\$69 \cdot ha^{-1}$ ($\$170 \cdot ac^{-1}$) at the TS-Horseshoe Ranch. These small differences in cost resulted in large financial differences when extrapolated to large areas of the AOI.

- Lowering the cost of seeding is critical to achieve large-scale seedings and sizable carbon sequestration. Carbon sequestration in rangelands appeared to be a viable strategy because, for example, spending $1.6 million to seed 4,047 ha (10,000 acres) of perennial species to replace undesirable NNAGF and sequester an additional 800 metric Tons of C yr^{-1} (800,000 Kg yr^{-1}) was in line with current range improvement costs.
- For the three parts of the AOI outside of the Mojave Desert, feasible and realistic seeding of perennial grass and shrub species in sagebrush communities converted to NNAGF-dominant vegetation by past fires resulted, respectively, in 136,132 (Utah), 3,196 (southern Great Basin ecoregion of Nevada and California with historic fire), and 12,743 (southern Columbia Plateau ecoregion of Nevada, Utah, California, southern Idaho, and southeast Oregon) metric Tons of C yr^{-1}.

1. INTRODUCTION

In the USA, forests have traditionally been the focus of discussion regarding carbon sequestration due to their high rates of aboveground productivity (Liu et al. 2012, 2014; Griscom et al. 2017; Mykleby et al. 2017; Fargione et al. 2018, Reed et al. 2020). New research, however, has shifted the conversation to explore other aspects of carbon dynamics. These concepts include identifying pools of carbon that are irreplaceable when lost and focusing on more stable pools of carbon (Köchy et al. 2015; Nahlik and Fennessy 2016; Reed et al. 2020). Soil carbon is generally more stable and buffered from loss due to logging, development, or high severity fires (Meyer et al. 2012; Minasny et al. 2017). Modeling work has shown that sites in California dominated by grasses may store more carbon long-term than trees due to carbon loss from fires (Dass et al. 2018).

Rangelands are often ignored in the discussion of carbon storage as these systems are considerably less productive (i.e., slow rate of carbon update in biomass; Svejcar et al. 2008; Meyer 2012; Thomey et al. 2014). However, arid and semi-arid ecosystems occupy approximately 30% of the global terrestrial land and thus represent one of the largest pools of total organic carbon (Thomey et al. 2014). Additionally, several studies have explored how conversion of native communities to more monotypic invaded sites after fires may reduce carbon storage within the intermountain West (Bradley et al. 2006, Austreng 2012, Nagy et al. 2020). Once established, these monotypic invaded sites burn again one order of magnitude more frequently than uninvaded sites and maintain their dominance (Chambers et al. 2014; Bradley et al. 2018). Restoring invaded sites may provide an opportunity to increase carbon storage capacity (Bradley et al. 2006, Nagy et al. 2020). Bradley et al. (2018) estimated that roughly a third of the Great Basin had cheatgrass cover exceeding 15% (~31,500 km^2), which was the threshold for changing fire behavior.

In 2017, The Nature Conservancy (TNC) in Nevada combined two concepts in a pilot study using existing state-and-transition simulation models that were developed for Greater Sage-Grouse and range management in north-central Nevada in the southern Columbia Plateau ecoregion to ask: Can the restoration of rangelands from degraded, non-native annual species grasslands and forblands created by past fires (NNAGF; mostly *Bromus tectorum, Bromus rubrum*, *Taeniatherum caput-medusae, Erodium cicutarium*, and *Descurainia pinnata*) to perennial grasses and shrubs, such as sagebrush (*Artemisia* spp.), store significant amounts of soil carbon and reduce future fire risk compared to doing nothing (Appendix A)? To answer the question, TNC simulated rangeland vegetation dynamics (including fire) and carbon stock-and-flow dynamics by comparing a no-management scenario to two treatment scenarios. One treatment consists of fuel breaks located adjacent to roads and the second treatment combined fuel breaks and seeding sagebrush communities dominated by NNAGF with perennial introduced grass species and sagebrush seed (Pilot study in Appendix A). The fire return interval of perennial species seedings was assumed to be longer (ranging between 250-500 years depending on the site) than that of shrublands dominated by NNAGF (5-10 years), which affected carbon dynamics.

Results of the pilot study showed that the fuel break combined with seeding scenario had the highest carbon uptake in the last 10 years of the simulation (average of 19,649,813 Kg of carbon), followed by the fuel break only (18,848,537 Kg of carbon) and minimum (no management) scenarios (17,991,574 Kg of carbon). As expected, the majority of the carbon lost from the terrestrial stocks to the atmosphere was due to fire on the landscape. Also, about 70% of the total carbon was stored in the soil. As little data are available to differentiate grass species within the region especially within the soil carbon stock, these differences were due to the reduced fire activity caused by seedings and increased presence of woody species. This research limitation to soil carbon dynamics persists to this day.

Results showed that a net difference of 1,632,933 Kg of carbon due to the seedings leading to native woody species establishment and changing fire behavior in this limited geography could be replicated in many other rangelands dominated by NNAGF of the Intermountain West where restoration is feasible and cost-effective, such as sagebrush systems above 20.3 cm (8 inch) of precipitation. The message was that while rangelands may not store carbon as standing biomass like temperate forests, the vastness of degraded rangelands and the stability of soil carbon might compensate for lower productivity.

1.1. Objectives

We proposed to create three products: (1) Updated NNAGF species cover index map spanning a 485,623-km^2 (120 million-acre) geography; (2) estimated net amount of carbon stored per unit area through restoration of NNAGF to perennial vegetation, and (3) estimated cost of storing net carbon through restoration per unit area. To measure the contributions of the restoration of NNAGF to seedings of perennial grasses and shrubs, we needed three components: a) A new map of NNAGF species cover for a large geography for which past simulation results were applicable and where we could delineate areas dominated by NNAGF for calculation of total carbon stored; b) previously developed state-and-transition simulation models (STSM) that could be adapted to simulate and budget cost-effective seeding in shrublands (mostly sagebrush) dominated by NNAGF; and c) a carbon stock-and-flow sub-model coupled to the ecological disturbances simulated in the STSM.

The Nevada Division of Natural Heritage's prior landmark map of non-native species cover index was from 2005 (actually, pre-2005 Landsat imagery was used; Peterson 2005, 2006) and needed to be updated as frequent large fires and wet years causing strong recruitment of non-native annual species occurred since 2005. Also, we developed a new map of annual species at a higher resolution than is currently available through the US Geological Survey (USGS, 30-m Landsat-based map; Pastick et al. 2020) and based on locally acquired data by TNC from Nevada and the rest of the Great Basin. Other maps of non-native annual species have been made available within the project area. USGS produced a near real-time estimate map of annual exotic herbaceous cover that extends throughout most of our project area but does not map southern Nevada (Pastick et al. 2020). These maps are available at 30-m resolution starting in 2015 (https://data.usgs.gov/datacatalog/data/USGS:5f0e030782ce21d4c4053ec2). The Rangeland Analysis Platform (RAP) developed by the Natural Resources Conservation Service (NRCS), the Bureau of Land Management (BLM), and the University of Montana hosts map products of plant functional group cover including annual forbs and grasses throughout the western United States (https://rangelands.app/). These maps are available annually from 1984 to 2020 at the 30-m resolution across our entire project area (Allred et al. 2021). Because TNC had developed STSMs for landscapes in the southern Columbia Plateau ecoregion, central and southern Great Basin ecoregion, and the Mojave Desert ecoregion, the new map would encompass all or parts of those ecoregions.

2. METHODS

2.1. Study Area

The Area of Interest (AOI) included: a) Great Basin ecoregion in Nevada, Utah, and eastern California; b) the southern Columbia Plateau ecoregion in southern Idaho, southeast Oregon, and northeast California; and c) the Nevada portion of the Mojave Basin and Range (a.k.a., Mojave Desert, Fig. 1). Being such a large project area, the climate and ecology varied appreciably latitudinally. For the carbon dynamics analysis, only the areas in the Columbia Plateau and Great Basin were modelled. Success rates of native perennial grass species seedings in the Mojave Desert are less than 5% in southern Nevada; therefore, seeding in the Mojave Desert is currently not viable at the landscape scale.

2.2. Overview of Analysis

Four discrete steps accomplished the project goals: a) Map non-native annual species cover in the AOI; b) delineate the sagebrush shrublands dominated by non-native annual species occupying precipitation zones and slopes where seedings can be successfully accomplished; c) using STSMs, estimate the net carbon stored and cost of seeding per unit area in degraded sagebrush ecosystems (i.e., Columbia Plateau and Great Basin); and d) estimate and map the entire amount of soil carbon stored and total cost of restoration in the Columbia Plateau and Great Basin ecoregions.

Figure 1. Project area (485,623-km² or 119,994,008.31 acres) shown to include all Environmental Protection Agency (EPA) Level 3 ecoregions within Nevada as well as the Southern Columbia Plateau and Central Basin and Range regions. Rangelands are shaded darker than the surrounding landscapes.

2.3. Imagery Analysis

Sentinel-2 imagery was captured by twin satellites with 13 spectral bands at 10-m, 20-m, or 60-m resolutions (https://sentinel.esa.int/web/sentinel/user-guides/sentinel-2-msi). The Sentinel-2 mission began producing images in summer of 2017 and has a revisit rate on the same geo-reference area every five days. Images from Sentinel-2 were available through Google Earth Engine, including bands with data quality information produced with each image. Sentinel-2 images contain bands that could be combined into metrics that represent vegetation productivity over time. Normalized Difference Vegetation Index (NDVI) was a combination of the reflected red and near-infrared (NIR) light from the Earth's surface (Tucker 1979). Healthy, productive vegetation reflects more NIR light and absorbs more red light; therefore, pixels with a high NDVI value indicate highly productive vegetation. NDVI is essentially a measure of 'greenness' or photosynthetic activity. Sentinel-2 bands could be combined in several ways

to represent various characteristics of the land surface, including vegetation phenology, moisture, or soil reflectance.

In the cold desert of the Great Basin and Columbia Plateau ecoregions where vegetation production is low and growth is slow, large seasonal bursts of productivity are unusual among native species. Annual species have a distinct signature in this geography since they experience quick growth in the early spring when moisture is available and while native plants are still dormant, and then quickly senesce during the early summer (Peterson 2006; Boyte et al. 2016). At the transition from spring to summer as moisture rapidly declines, native, perennial, herbaceous species are greener and more productive than their non-native counterparts. We used seasonal remote sensing metrics of vegetation productivity to identify locations that exhibited this behavior.

A common method of identifying annual species in dry regions is to compare NDVI from the spring and summer (Peterson 2006; Kokaly et al. 2011; Boyte et al. 2016). In a pixel with annual species, the NDVI should be far higher in the spring than in summer. A pixel primarily containing moisture-resilient sagebrush or perennial species should not change in NDVI as much as annual species between seasons. Therefore, the large difference of NDVI between spring and summer at a pixel represented the dramatic senescence of annual species that typically occurs at the end of spring and early summer (May/June, Fig. 2).

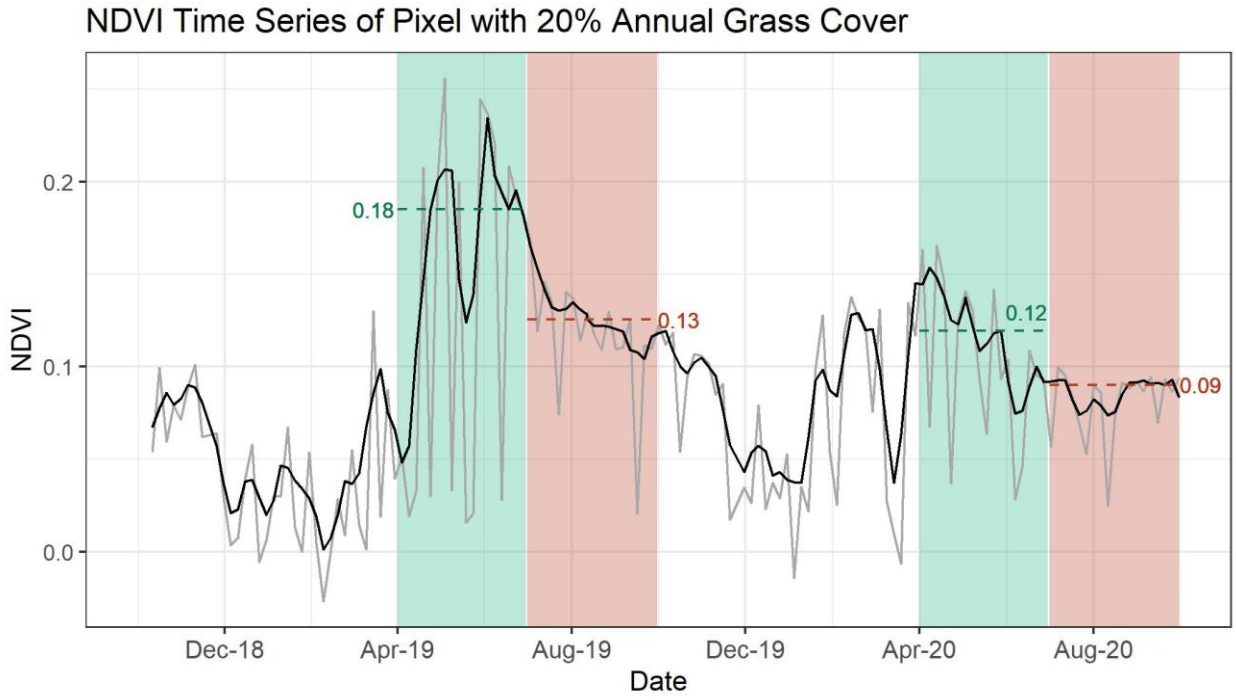

Figure 2. Two years of NDVI values at a location with 20% median annual grass cover. The raw NDVI data are shown in light grey. A smoothed time series is shown in black using the Savitzky-Golay smoothing, which performs a local polynomial regression on a series of values which are treated as being equally spaced to determine the smoothed value for each point. The shaded areas illustrate the difference between spring growth (green) and summer senescence (red) represented by NDVI to identify annual species in high-desert regions. Located in Central Nevada, from the Cortez project area, median spring NDVI at this location is greater than median summer NDVI in both 2019 and 2020 as indicated by the dashed lines.

As previously mentioned, the climate and ecology varied appreciably across latitudes within the large project area. As such, phenological differences are expected from non-native annual species between the southern and northern portions of the project area. Creating a seasonal median from multiple images provides a buffer for these seasonal latitudinal differences (Fig. 2). Compared to northern areas, non-native annual grass species in southern areas may begin their growing season earlier (April or even March). Consequently, these areas will likely exhibit senescence earlier as the temperature rises rapidly, and moisture availability declines. The seasonal median metrics capture this variation in phenology throughout the project area.

Other vegetation metrics derived from one or more spectral bands were included in the model. A full list of metrics created and used in the Random Forest model are available in Table 1. Seasonal differences of these metrics were also calculated as these represent phenological or ecological processes that helped identify areas where annual species were present. Climate data from the Idaho GRIDMET climate products (ultimately derived from 4-km PRISM data: https://developers.google.com/earth-engine/datasets/catalog/IDAHO_EPSCOR_GRIDMET#description) were used to produce precipitation metrics as potential predictors to the Random Forest model. Temperature data have been shown to be unimportant in mapping non-native annual species and were not included in this analysis (Pilliod et al. 2017).

Table 1. All input variables to the regression Random Forest Model. '$\bar{x}$' refers to median value rather than mean in this model. Continuous Heat-Insolation Load Index (CHILI) is a measure of the heat load based on topographic characteristics. Legend: spr = spring, sum = summer, fal = fall, win = winter, falpy = fall precipitation of the previous year.

SPATIAL VARIABLES	SEASON	SEASONAL DIFFERENCES	RESOLUTION (METERS)
SENTINEL-2 BANDS			
GREEN (BAND 3)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
RED (BAND 4)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
VEGETATION RED-EDGE (BAND 5)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
VEGETATION RED-EDGE (BAND 6)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
VEGETATION RED-EDGE (BAND 7)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
NIR (BAND 8)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
NARROW NIR (BAND 8A)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
WATER VAPOR (BAND 9)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	60
SHORTWAVE INFRARED (BAND 10)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	60
SHORTWAVE INFRARED (BAND 11)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
SHORTWAVE INFRARED (BAND 12)	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	20
VEGETATION INDICES			
NDVI	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
EVI	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
SAVI	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
MSAVI	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
TCG	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{fal}$	$\bar{x}_{spr} - \bar{x}_{sum}$, $\bar{x}_{sum} - \bar{x}_{fal}$	10
TOPOGRAPHY/SOILS			
ELEVATION	**NA**	**NA**	10
CHILI	**NA**	**NA**	10

LITHOLOGY	**NA**	**NA**	90
PRECIPITATION	$\bar{x}_{spr}$, $\bar{x}_{sum}$, $\bar{x}_{falpy}$, $\bar{x}_{win}$	$\bar{x}_{spr} - \bar{x}_{falpy}$	4000

TNC primarily used remote sensing metrics that represent the timing and magnitude of vegetation productivity. Metrics included in this model were also chosen based on variable importance identified by Jones et al. (2018) for mapping annual species. Not all metrics used by Jones et al. (2018) were computed for this analysis to create Sentinel-2 products; however, we considered Jones et al.'s (2018) top 10 metrics identified as more important and retained elevation and precipitation based on experience.

We retained five remote sensing metrics used in Jones et al. (2018) developed from Sentinel-2: NDVI, Enhanced Vegetation Index (EVI), Soil Adjusted Vegetation Index (SAVI), Modified SAVI (MSAVI), and Tasseled-cap greenness (TCg). The EVI is similar to NDVI and represents vegetation productivity, but it is more sensitive to areas with dense vegetation and reduces background noise, atmospheric noise, and saturation because of the inclusion of the blue band (https://www.indexdatabase.de/). The SAVI accounts for soil brightness where vegetation is sparse by using the ratio between the red and NIR bands like NDVI but includes a brightness correction factor. The MSAVI estimates vegetation productivity but uses a different formula that minimizes influence from bare soil effects and also uses the red and NIR bands. TCg measures how green the reflected pixel is based on the tasseled-cap transformation function on all reflectance bands (Nedkov 2017).

Seasonal metrics of precipitation were created as model inputs because variations in the timing and amount of precipitation have a strong effect on annual grass phenology and composition (Pilliod et al. 2017). Non-native annual species are particularly more reactive to precipitation dynamics than native vegetation (Boyte et al. 2015). More precipitation in the winter and early growing season (spring) benefits the non-native annual species and increases their cover in the growing season (Pilliod et al. 2017). The sooner that precipitation events stop in the late spring/summer, the quicker the annuals will senesce. Precipitation in the fall before the growing season may initiate germination of non-native annual species before native species for the following growing season (Pilliod et al. 2017, Horn et al. 2017). Seasonal differences in precipitation were also created (spring minus summer, summer minus previous fall) as in Jones et al. (2018).

Elevation, aspect, and slope are known to control vegetation composition and, thus, annual grass distribution and cover that when combined with phenology can be used for remote sensing (Chambers et al. 2014, Boyte et al. 2015). The Continuous Heat-Insolation Load Index (CHILI) is a 10-m dataset derived from USGS digital elevation models available through Google Earth Engine. CHILI combines slope, aspect, and latitude into a measure of heat load, a strong predictor of evapotranspiration and, consequently, vegetation distributions (Theobald et al. 2015). Resistance to disturbance and subsequent invasion by cheatgrass can be determined by these topographic variables (Chambers et al. 2014). In the case of aspect, plant communities on northern aspects are more resilient to disturbance than southern aspects. Higher-elevation communities (i.e., colder/wetter sites) are also typically more resistant to non-native, annual species invasion than lower elevation communities (i.e., warmer/dryer sites; Chambers et al. 2014).

Lithology, or soil parent class, data were used due to the strong relationship between soil texture and chemistry of the mapped classes and ecological response (Theobald et al. 2014). Certain soil

characteristics increase the likelihood of invasion by non-native annual species; resistance to invasion is generally lowest on coarse dry soils (Chambers et al. 2014). The lithology dataset was not as fine-scaled as other national soil products such as the SSURGO and STATSGO databases, but the 20 soil classes in this dataset captured the necessary soil information for this coarse analysis (https://developers.google.com/earth-engine/datasets/catalog/CSP_ERGo_1_0_US_lithology).

2.3.1. Training data

2.3.1.1. TNC Data

TNC training data were obtained from two remote sensing projects; one in central Nevada (hereafter, Cortez) and the other from southwest Utah (hereafter, Pine Valley-Mountain Home or PVMH; Fig. 3). Descriptions of modeling methods and remote sensing are found in Provencher et al. (2021a), whereas reports for each site are found in Provencher et al. (2017) for Cortez and Provencher et al. (2019) for PVMH. These were the only sites that were mapped by TNC after the launch of Sentinel satellites; therefore, all landscapes mapped before the launch of Sentinel satellites could not be used. An important aspect of TNC's training data was that vegetation classes were in categories defined by percent cover *ranges* for native or non-native species group, shrubs, and trees. Using the vegetation description for each class, ranges of cover (e.g., 5%-15% cover of non-native annual species) were converted to median values, and open-ended cover values (e.g., >5% cover of non-native annual species) were assigned a most likely value that was frequently observed in the field (15% cover of non-native annual species). All TNC data were assigned median percent-cover values of annual grass and forbs based on their vegetation system and class. In addition to considering training data with detectable annual grass cover, we included some vegetation classes without annual grass cover from both landscapes in the training data as it was judged just as important to identify locations without annual grass as it is to estimate cover in areas with annual species.

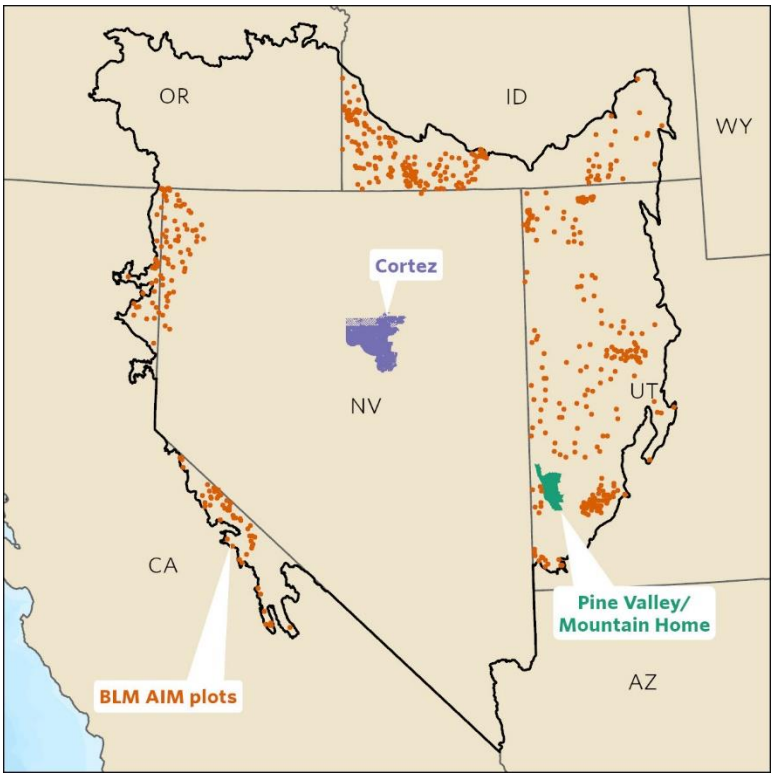

Figure 3. Map of annual grass cover data collected in 2020 used to train the Random Forest model.

The Cortez area was approximately situated off Highway 278 to the east and the Roberts Mountains to the south in Elko, Eureka, and Lander Counties and the Shoshone Range to the West (Fig. 4). The area is bordered to the north by the Dry Hills and encompasses to the south the northern tip of the Toiyabe Range, Red Mountains, and the northern part of Carico Valley. The project area spans about 335,442 ha (828,894 ac). Each study area contains typical rangelands; however, the valley floor on the Crescent Valley half of the area is substantially lower than the Horse Creek Valley area to the east. The Cortez Range, Simpson Park Range, Shoshone Range, Sulphur Spring Range, and Dry Hills are primarily volcanic and north-south trending, whereas the Roberts Mountains are dominated by carbonate rocks and have a more circular shape than classic north-south trending basin and range formations.

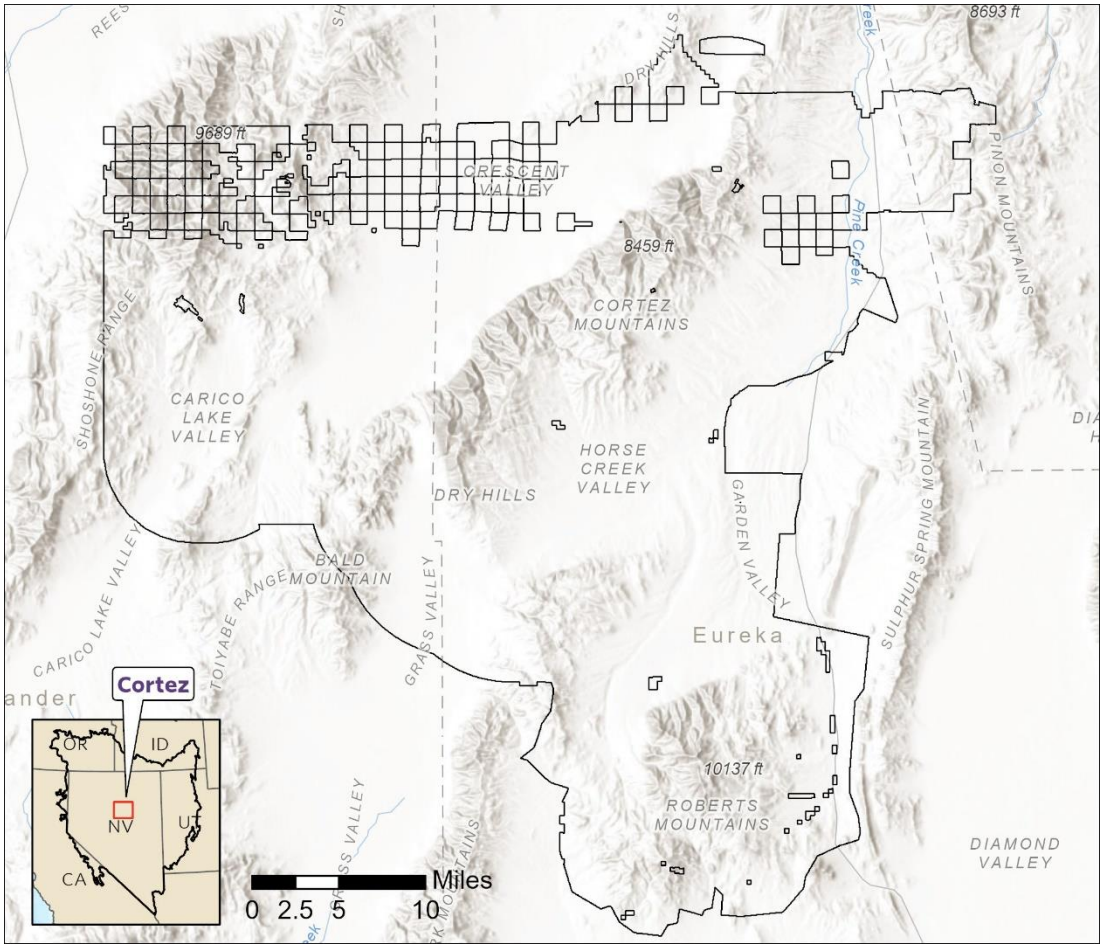

Figure 4. Detailed map of the Cortez area mapped in July 2020.

The PVMH landscape was comprised of the areas known as Pine Valley (Beaver and Millard Counties), the Mountain Home Range, Indian Peak Range, and the western flank of the southern half of the Wah Wah Range in southwest Utah, 48 km (30 miles) southeast of Great Basin National Park, covering an area of about 129,095 ha (319,000 acres; Fig. 5). The vegetation is typical of the southeastern Great Basin ecoregion, dominated by sagebrush shrublands and pinyon-juniper woodlands but containing monsoonal dependent communities with such species as ponderosa pine (*Pinus ponderosa*), Engelmann spruce (*Picea engelmannii*), and Stanbury's cliffrose (*Purshuia stansburiana*).

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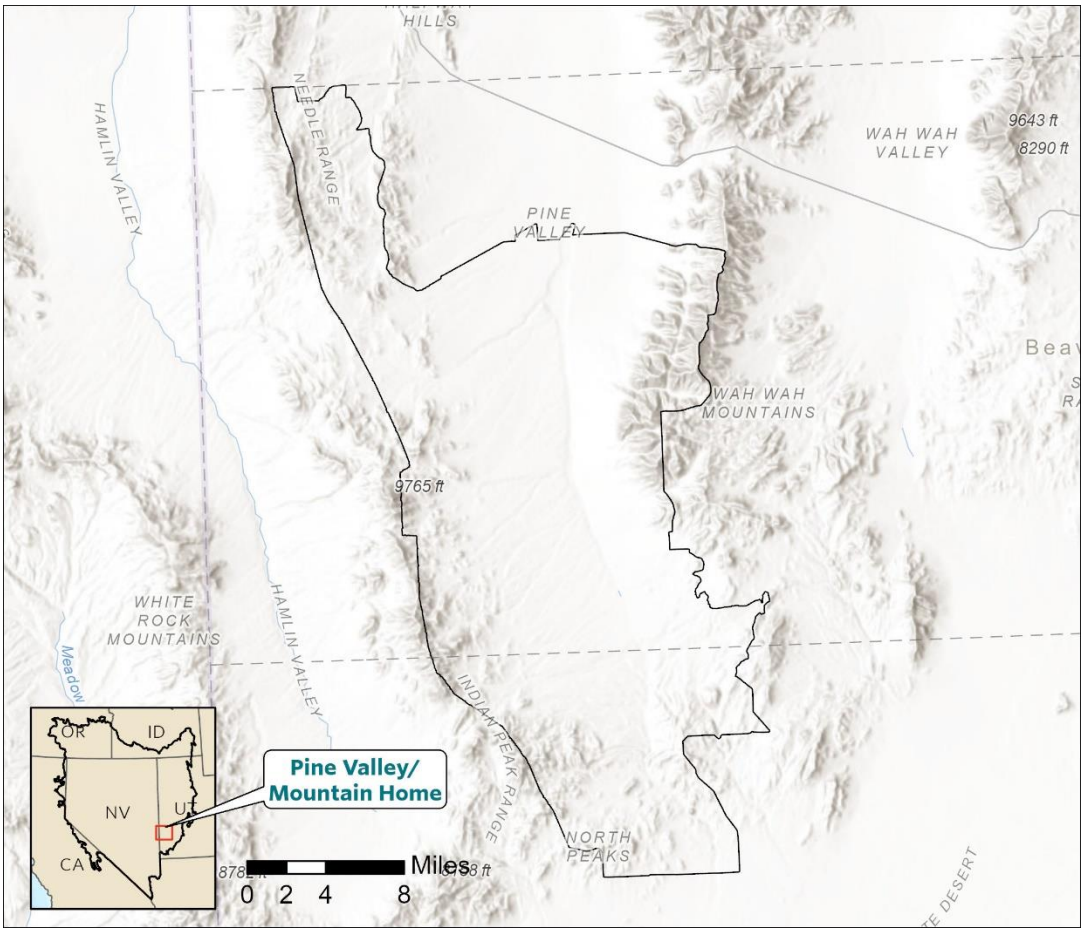


Figure 5. Detailed map of the Pine Valley-Mountain Home area mapped in July 2020.

Rather than using each pixel as a training point from the vegetation rasters, large areas of continuous annual grass and forb cover that contained a series of "large patches" were identified as training plots. The vegetation system and class rasters were converted to polygons. Adjacent polygons with the same vegetation system and class were dissolved into an individual feature. Any feature with a total area < 1000-m^2 were removed to ensure that the training data were using large, contiguous patches. Even after filtering for large patches, too many training sites remained for processing in Google Earth Engine; therefore, 5% of the patches were randomly selected to use as training data. Arcpy's Feature to Point tool (Esri Inc. ArcGIS Pro 2.7.1. Redlands, CA: Esri Inc. 2020. Software) was then used to convert the polygons to points and assigned the points their patch's median annual grass percent cover values.

Raw NDVI time series were assessed to determine whether the early-season phenology of annual species was detectable at locations with confirmed annual grass or forb presence. Figure 2 shows the rapid increase in productivity in April 2020 followed by a decline in productivity throughout the summer 2020 at this location with observed 20% median cover of annual grass and/or forb. This location was the center of a large patch of Wyoming big sagebrush on semi-desert soil (*A. tridentata* ssp. *wyomingensis*; 20.3-cm to 25.4-cm [8 – 10 in.] precipitation zone) vegetation with annual species present (location:

39.33°N 116.63°W, SYSXCLA Code: 10802100; Provencher et al. 2017). This plot was in the southeastern portion of Crescent Valley and has experienced past disturbances leading to annual grass invasions.

2.3.1.2 BLM AIM DATA

Point data with associated estimated cover of non-native annual species were used in addition to TNC data to train and validate the Random Forest model. The BLM Assessment, Inventory, and Monitoring (AIM) TerrADat dataset was filtered to observations made in 2020 and limited to the extent of the project area. The TerrADat data were primarily collected outside the state of Nevada in 2020 with an exception in the northwest portion of the state (Fig. 3). Only data collected after the launch of Sentinel satellites could be used, which excluded most plots in Nevada. Plot locations within scheduled areas were determined randomly – crews go to determined locations and perform a line-point-intercept protocol along transects to record the presence of plants, rock material, or bare ground (Toevs et al. 2010). Estimates of functional plant group cover, including annual species, were derived from these observations (Allred et al. 2021; Jones et al. 2018).

More information about the TerrADat dataset can be found at https://aim.landscapetoolbox.org/wp-content/uploads/2015/08/Monitoring-Manual-Volume-II.pdf. The AH_AnnualGrassCover field was used to represent median cover in the training data. The year 2020 was used to obtain training data. The BLM dataset was merged with TNC datasets and loaded into Google Earth Engine as training data.

A total of 5,080 training points were available, although these points were not evenly distributed across the project area (Fig. 3). The Random Forest model was trained with 70% of the data (3,595) and 30% were held out for validation (1,485). The distribution of annual grass cover values ranges from 0% to 91% with a median value of 1% (skewed by large number of 0% cover training points, particularly from the TNC data; Figure 6).

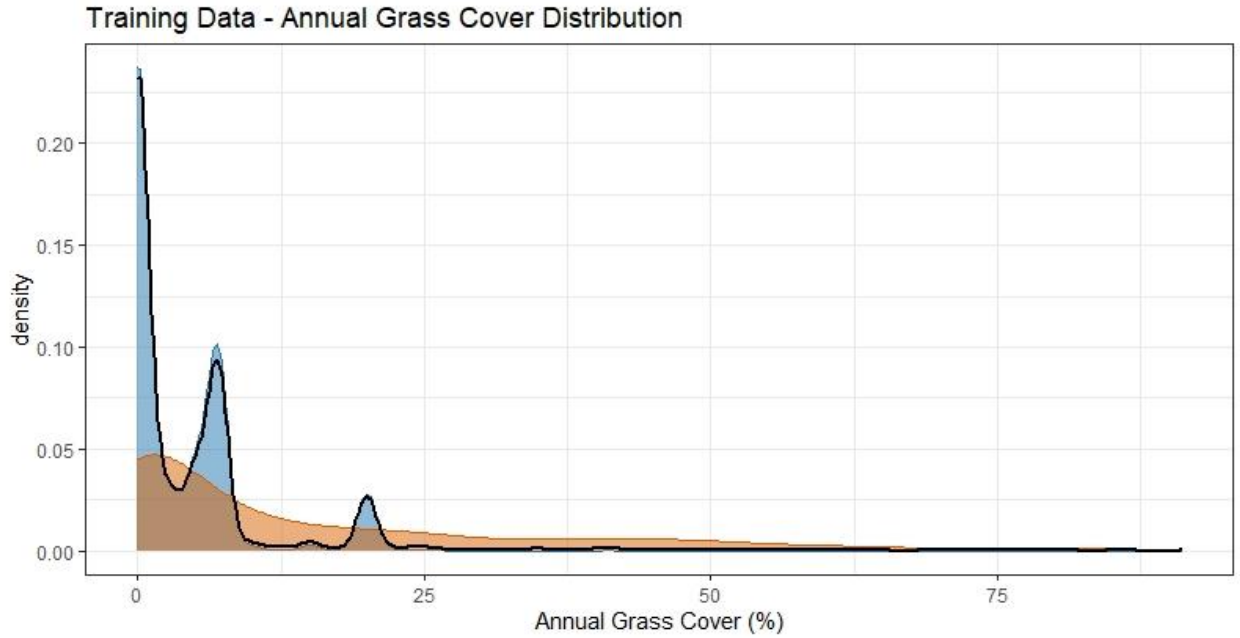


Figure 6. Distribution of median annual grass cover values (%) from TNC-collected data in two remote sensing projects (blue) and BLM AIM data (orange). The density curve of all combined training data is shown by the black line. TNC's training data was classified in categorical vegetation classes of the state-and transition models defined by percent cover ranges for native or non-native species group, shrubs, and trees. Using the vegetation description for each class, ranges of cover (e.g., 5%-15% cover

of non-native annual species) were converted to median values, and open-ended cover values (e.g., >5% cover of non-native annual species) were assigned a most likely value that was frequently observed in the field (15% cover of non-native annual species) that correspond to the peaks in the figure.

2.4 State-and-Transition Simulation Models

STSMs are computerized boxes-and-arrows models that represent the vegetation dynamics of different ecological systems each categorized as states (the mutually exclusive vegetation classes within each ecological system) that experience transitions that are either ecological (e.g., fire) or intentional (e.g., chainsaw thinning) disturbances in a landscape (Daniel et al. 2016). General methodology of STSMs are described in Daniel et al. (2016) and Provencher et al. (2016, 2021a). The comparative description of conceptual and simulation models was reviewed by Provencher et al. (2016). All simulations were conducted in the ST-Sim/Syncrosim model platform created by ApexRMS Ltd. (Daniel et al. 2016; www.apexrms.com).

The models from the IL Ranch of north-central Nevada (southern Columbia Plateau ecoregion; Provencher et al. 2016), TS-Horseshoe Ranch of central Nevada (northern Great Basin ecoregion; Provencher et al. 2016), and Pine Valley-Mountain Home project of the southeast Great Basin ecoregion in southwest Utah (Provencher et al. 2019, 2021b) provided the latitudinal gradient to estimate the net carbon stored of restored rangelands under different land management experiences and cost. Sagebrush shrublands were extensive in these rangeland landscapes. Provencher et al. (2021a) described STSMs from southwest Utah's Great Basin ecoregion that were similar to the models of the three retained landscapes; therefore, method details will not be repeated here. While the three landscapes were completed in different years, they share the same structure and disturbance regimes, albeit with local parameter adjustments.

While several management actions were modeled in Provencher et al. (2021a) that may contribute to carbon storage, we focused only on those actions that sought to reestablish perennial grass and shrub species into sagebrush shrublands or systems where sagebrush is a sub-dominant component (e.g., mountain shrub, sub-xeric grassland, and incised floodplains converted to sagebrush/Great Basin wildrye [*Leymus cinereus*] systems) dominated by NNAGF (Table 2). Seeding could be combined with other actions required to increase the success of the seeding. For sagebrush, only black (*A. nova*), low (*A. arbuscula*; occasionally), and big sagebrush subspecies (*A. tridentata*) systems received treatments within the AOI.

Table 2. Cost per hectare (acre) of seeding actions applied in the IL Ranch (NV), TS-Horseshoe Ranch (NV), and Pine Valley-Mountain Home(UT). Only those actions were used in this project's simulations. Cost estimates were from Bureau of Land Management staff.

Action Name	System	IL Ranch	TS-Horseshoe Ranch	Pine Valley-Mountain Home
Herbicide-Plateau+Seed	All sagebrush	$\$420 \cdot ha^{-1}/\$170 \cdot ac^{-1}$	$\$420 \cdot ha^{-1}/\$170 \cdot ac^{-1}$	$\$289 \cdot ha^{-1}/\$117 \cdot ac^{-1}$
Herbicide-Plateau+ Native-Seed	All sagebrush	$\$729 \cdot ha^{-1}/\$295 \cdot ac^{-1}$	$\$729 \cdot ha^{-1}/\$295 \cdot ac^{-1}$	
Chaining+Native-Seed	All sagebrush			$\$506 \cdot ha^{-1}/\$205 \cdot ac^{-1}$

Chaining+Plateau+Native-Seed	Black sagebrush			$573·ha^{-1}/$232·ac^{-1}$
Chaining+Plateau+Seed	Black sagebrush			$395·ha^{-1}/$160·ac^{-1}$
	Montane sagebrush steppe			$474·ha^{-1}/$192·ac^{-1}$
	Wyoming big sagebrush			$474·ha^{-1}/$192·ac^{-1}$
Chaining+Seed	All sagebrush			$408·ha^{-1}/$165·ac^{-1}$

Modeled planned annual seeding rates per landscape reflected levels to force high levels of restoration and change in fire regimes that would be measurable by the end of simulation. In the PVMH landscape, 20,242 ha (50,000 acres) per year were allowed (up to these numbers, although the ST-Sim software would be limited by available NNAGF areas) in all treatments and in all land ownerships from 2023 to 2042 (simulations started in 2018). For the TS-Horseshoe Ranch, 20,242 ha (50,000 acres) per year and 10,141 ha (25,000 acres) per year, respectively, were allowed for Herbicide-Plateau+Seed and Herbicide-Plateau+Native-Seed treatments in all ownerships from 2019 to 2040 (simulations started in 2014). For the IL Ranch, which was less dominated by non-native annual species, 6,070 ha (15,000 acres) per year and 10,117 ha (25,000 acres) per year per ownership (predominantly public), respectively, were allowed to implement the Herbicide-Plateau+Native-Seed and Herbicide-Plateau+Seed treatments from 2019 to 2040 on BLM lands. Keeping in line with TNC's original pilot study and already completed seeding by the Bureau of Land Management after the 2018 Martin Fire, these treatments were also allowed in early-successional classes of mixed annual and (native) perennial grass species. This class was not treated in other landscapes.

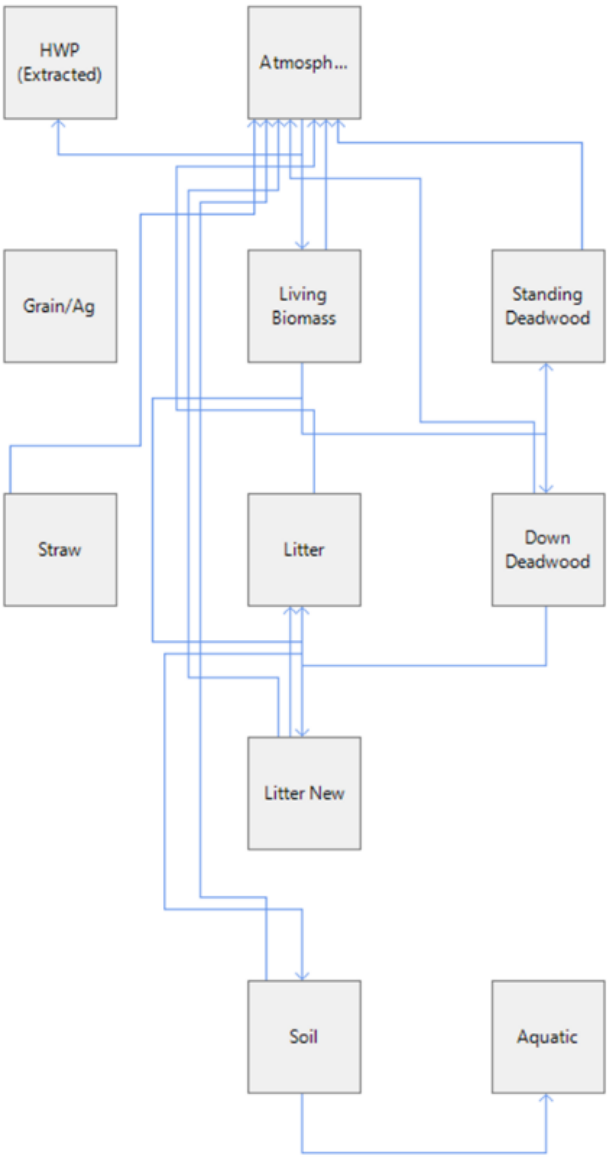


Figure 7. Carbon stock-and-flow model used in this analysis. Model parameters were adapted from Sleeter et al. (2015). Note, no flow was modeled to the "Aquatic," "GrainAg," HWP (Extracted)," and "Straw" stocks.

Modeling the carbon dynamics followed the methodology in Sleeter et al. (2015) and Daniel et al. (2018). This method utilizes a stock-and-flow sub-model of the Syncrosim software used as an advanced feature in ST-Sim simulation models, where the total carbon storage of a system is divided into several stocks, such as atmosphere, living biomass, litter, and soil (Fig. 7). Moreover, storage or release of carbon is synchronized with the natural processes in the non-carbon STSMs. Flows represent the processes that move carbon from one stock to another and similar to "transitions" in the STSM. Flows can either be automatic, such as plant growth, decomposition, or soil emissions, or event-based, such as land use change or fire. As flows occur, the model tracks the proportion of carbon that is moved from one stock to others. The amount released was determined by assigning a severity rank to the removal (complete release to the atmosphere, partial, dead biomass standing, slow decomposition, etc.) and a quantity of carbon per acre of each broad vegetation type. Of the carbon lost from the living biomass stock during a fire, a portion will be emitted into the atmosphere and a portion will be converted to standing dead biomass. As with the STSM, external factors such as climate can be input into the simulation to model climate impacts on carbon storage.

Carbon flux rates were derived from estimates by Sleeter et al. (2018). These are provided by leading lifeform (grassland, shrubland, forest) and ecoregion. Each ecological system and state class in our model was classified as one of these life forms and the corresponding flux rates from Sleeter et al. (2018) were used. In addition to this, net primary production (NPP) multipliers were estimated and applied by ecosystem based on expert opinion, and by projected precipitation estimates for each site following Del Grosso et al. (2008).

Carbon fluxes for livestock and feral horse grazing were obtained by converting the amount of annual forage consumed in Animal Unit Months (AUMs) per ecological system and per vegetation class into lbs/acre as 1 AUM = (26 lbs per cow-calf pair per day multiplied by 30 days per month). The annual production would be to multiply 1 AUM by 12 for the 12 months per year. For horses, AUMs consumed

were already multiplied by 1.25 using standard assumptions. When grazing occurred in a cell, the corresponding amount of carbon was removed from the living biomass with an additional 10% removed from the litter. In shrubland state classes, 90% of the carbon removed from living biomass was lost to emissions, with the remaining 10% lost to mortality. In all other situations, all carbon lost to grazing was lost as emissions.

Initial litter, living biomass, and down deadwood carbon values were similarly assigned by lifeform based on Sleeter et al. (2018). In contrast, initial soil carbon values were estimated by ecological system and age using a spin-up simulation. The spin-up consisted of simulating each ecosystem for a 3000-year burn-in period followed by 250 years of undisturbed growth. During the burn-in period, a representative native state class for each ecosystem was simulated under growth, succession, and replacement fire, with fires applied deterministically based on the expected fire return interval for each ecosystem. This allowed the soil carbon estimates to stabilize under the expected fire regime. After one final replacement fire, the simulated cells were then allowed to grow undisturbed for 250 years. The time series of soil carbon values from this undisturbed growth was used to assign initial soil carbon to future simulations by ecosystem and age. Initial ages for future simulations were based on the midpoint age for each state class with both minimum and maximum age bounds and the minimum age for those with no upper bound.

Two management scenarios were developed: Custodial management (i.e., minimal management intervention but did include livestock grazing and assumed fire suppression) and Seeding management (i.e., implementation of management actions to remove NNAGFs and establish perennial species) and were run for 20 years. Only NNAGF sites were candidates for seeding, except for the early-successional class made of mixed NNAGF and native perennial grass species for the IL Ranch. These were mostly in sagebrush, basin wildrye, and mountain shrub (e.g., Utah serviceberry) ecological systems above the 10-inch precipitation zone (i.e., upland soils and moister). For each landscape, the first 5 years of the Seeding scenario were identical to those of the Custodial scenario to allow the spread of fires and build-up of NNAGFs, then on the 6th year seeding started for the next 20 years at very high rates to exhaust the availability of the NNAGF class. No seeding occurred during the last 5 years. The IL Ranch and TS-Horseshoe Ranch simulations started in 2015 with seeding implemented in 2020 and ending in 2040. The simulations ended in 2045. The PVMH landscape was simulated from 2018 to 2047 where seeding started in 2023 and ended in 2043.

Each scenario was replicated 10 times using climate as the source of variability among replicates as in the original simulations (Provencher et al. 2016, 2019). For the PVMH landscape, the climate scenario was the statistical forecasting of PRISM data (http://www.prism.oregonstate.edu/historical/; Daly et al. 2008) with a stochastic weather generator (Verdin et al. 2014) as described in Provencher et al. (2021a). The older IL and TS-Horseshoe Ranches simulations used a different climate methodology by resampled average sub-regional Standard Precipitation Index (a drought index, see Provencher et al. 2021a) to derive ecological disturbance (e.g., severe drought, fire, annual species invasion, and so on) variability around mean parameter values and flood values from USGS gauge data (older method described in Provencher et al. 2016).

An additional fortuitous complication occurred with the TS-Horseshoe Ranch. Our original simulation of carbon stock and flows was conducted with frequent fire set at 10 times the normal rates for all vegetation types. We re-simulated the same model but reestablished the “historic” fire regime that was part of the original project (Provencher et al. 2016). Because this error was informative for carbon dynamics, we decided to present and discuss both sets of results.

Carbon sequestered and cost of restoration were reported in different periods. Annual carbon sequestered due to seeding was the Net Biome Productivity (NBP) difference between the Seeding management and the Custodial management scenarios per year within each replicate (10 replicates per scenario). The unit of carbon was g $C \cdot m^{-2} \cdot yr^{-1}$ for pixel-level calculations, and then converted to metric $tons \cdot yr^{-1}$ for regional extrapolation. The amount of carbon retained to measure the effect of seeding was the temporal sum of these annual differences for 10 years after seedings started succession from early successional to mid-successional phases. At the beginning of this transition, seedings were mostly completed and vegetation had time to establish and mature. New fires in other areas of treatable systems invaded by NNAGF could trigger additional seedings; however, most seedings would be completed within the first 10 years of simulations. The sum of NBP differences was divided by 10 to obtain a per-year estimate. The cost of restoration was per square meter of seeded area (including failed seedings) and evaluated over the entire period of implementation because the full seeding effort contributed to NBP differences.

Two types of sub-regional estimations of NBP differences were conducted for each of the three geographies assessed in the AOI. First, the amount of NBP difference that entered the system (positive values) in each geography was obtained by multiplying the average g $C \cdot m^{-2} \cdot yr^{-1}$ per landscape (i.e., PVMH, IL Ranch, or TS-Horseshoe Ranch) by the area (in m^2) of NNAGF that could be feasibly seeded in the AOI where each landscape's climate applies. The estimate was per year. Similarly, the sub-regional cost of restoration was estimated. Because these are large numbers (the AOI is 485,623-km^2 [120 million acres]) that may not reflect a realistic level of effort, a generic ranch-level estimate was also provided where we assumed that a 121,457-ha (300,000-acre) private-public lands ranch would contain 4,046.8 hectares (10,000 acres) of NNAGF that could be seeded. The cost of seeding per m^2 was also multiplied by the equivalent of 4,046.8 hectares in m^2.

3. RESULTS and DISCUSSION

3.1. Map Description

The 10-m^2 resolution map produced by applying the Random Forest model to the available Sentinel-2 imagery found continuous values that range from 0% to 61.1% annual species cover with a mean of 12.6%. The map was masked using the Coterminous US Rangelands dataset to remove urban/developed areas, open water, forests, and agricultural land (Reeves and Mitchell 2011). The root mean squared error (RMSE) of the available validation data (n = 1,240) was 9.29% and the mean absolute error (MAE) was 5.83% (Fig. 8). Some validation data points (n = 245) were removed from the accuracy assessment after being masked by the rangeland dataset.

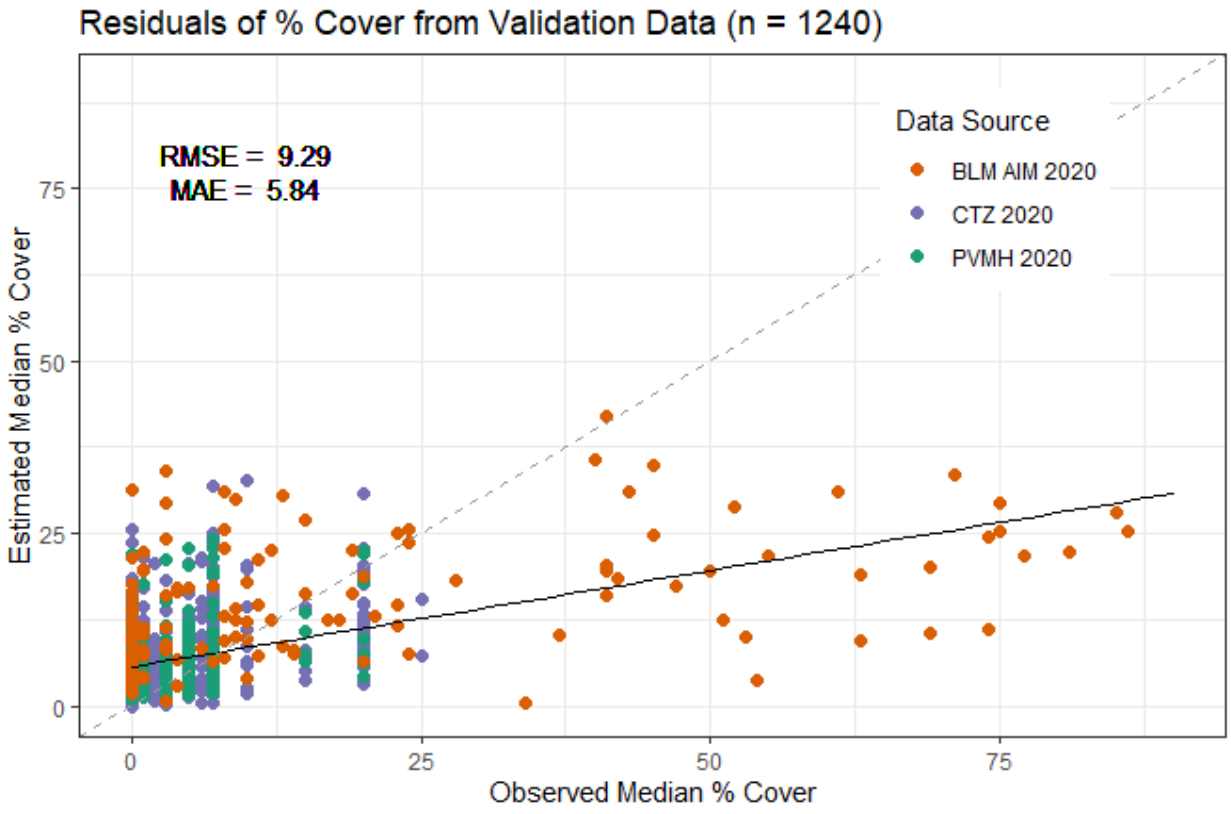


Figure 8. Residuals of the available validation data. Residuals show that the TNC Random Forest model is more likely to underestimate annual grass cover where it is high (> 30%).

Annual species cover was mapped either using original continuous percent cover values as well as classified into bins of cover range (Fig. 9). Overall, higher percent cover values were observed in the northern basin and range towards the Snake River plains, around the Great Salt Lake, and in the Mojave Desert northeast of Las Vegas, NV. As expected, relatively higher percent cover values were found inside recently burned areas compared to the surrounding unburned areas (Fig. 10). Higher-elevation areas were mapped as having less cover than low elevation areas, a consistent observation from other studies (Chambers et al. 2014). Another common pattern was high cover of annual species around towns, major roads, and agricultural areas. These spaces are more likely to experience disturbance, leading to greater presence of non-native annual species (Chambers et al. 2014).

The model and map produced from this method should not be used as definite proof of the presence or concentration of specific non-native annuals, such as cheatgrass, western tansy mustard (*Descurainia pinnata*), or redstem filaree (*Erodium cicutarium*). The model effectively uses imagery and metrics to determine where vegetation phenology is like that of non-native annual species in Nevada and the Great Basin. We acknowledged that the model may be selecting similar phenological patterns from other plants such as *Poa secunda*, a widespread native perennial grass. Some variants of *P. secunda* may have a matching phenology to non-native annual species – a short growing season that begins in the spring and ends in the early summer as the species senesces (Peterson 2005). A large presence of *P. secunda*

may lead to overestimates of the non-native annual species we seek to map. Without site visits, it is difficult to say where these errors may occur or how severe they might be.

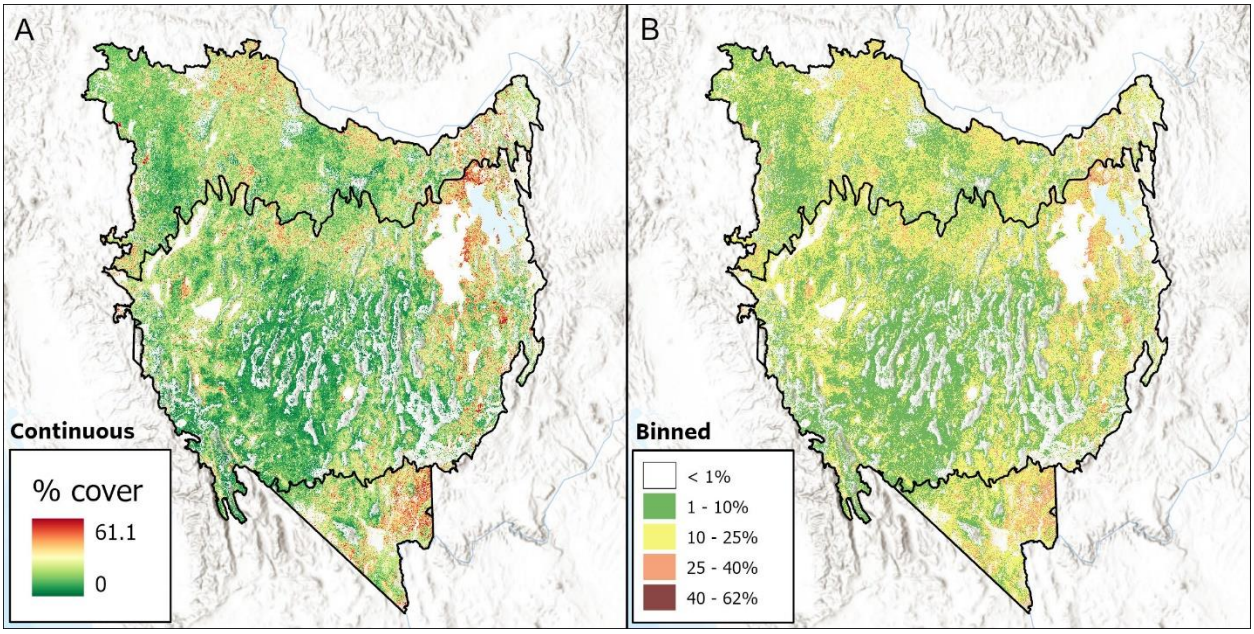


Figure 9. Estimated continuous percent cover (A) and binned cover (B) of non-native annual species.

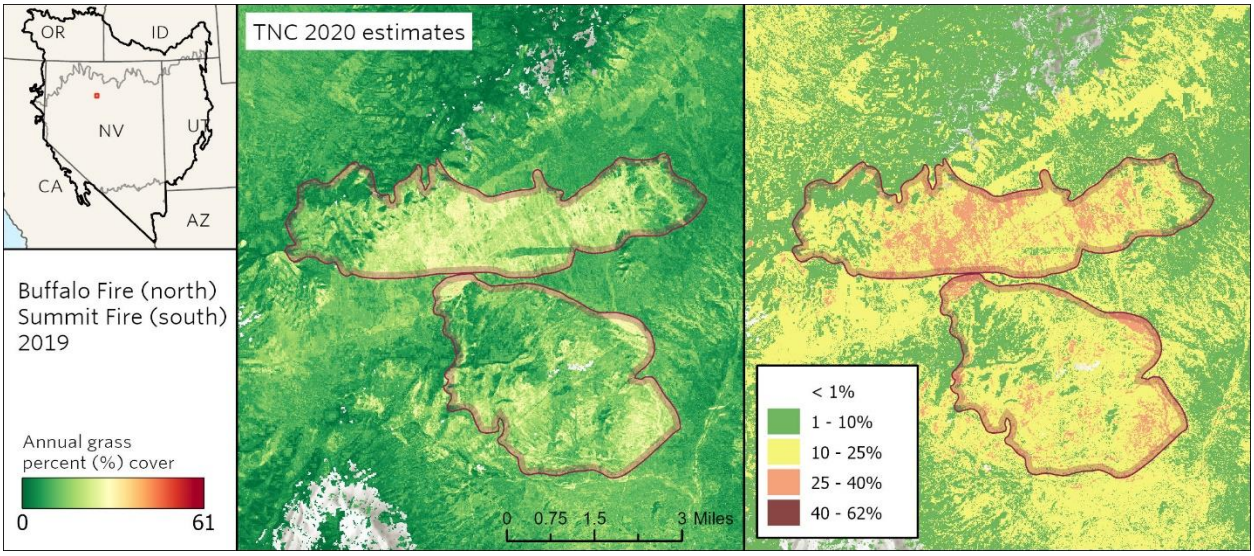


Figure 10. 2020 estimates of annual grass cover in continuous (left) and binned (right) displays are shown to be higher within recently burned areas than the surrounding landscape. The Buffalo (north) and Summit (south) fires burned on BLM land approximately 27 miles southeast of Winnemucca, NV in 2019.

3.2. Comparison to Similar Datasets

The project area, mean and median percent-cover of annual species from the TNC 2020 map estimates were 12.6% and 11.3%, respectively. Mean and median for the 2020 USGS near real-time (NRT) map were 11.9% and 10%, respectively. Mean and median cover for the 2020 RAP map were 11.35% and 7%, respectively. Comparing the maps visually reveals how each model estimates annual grass cover in different parts of the project area (Fig. 11).

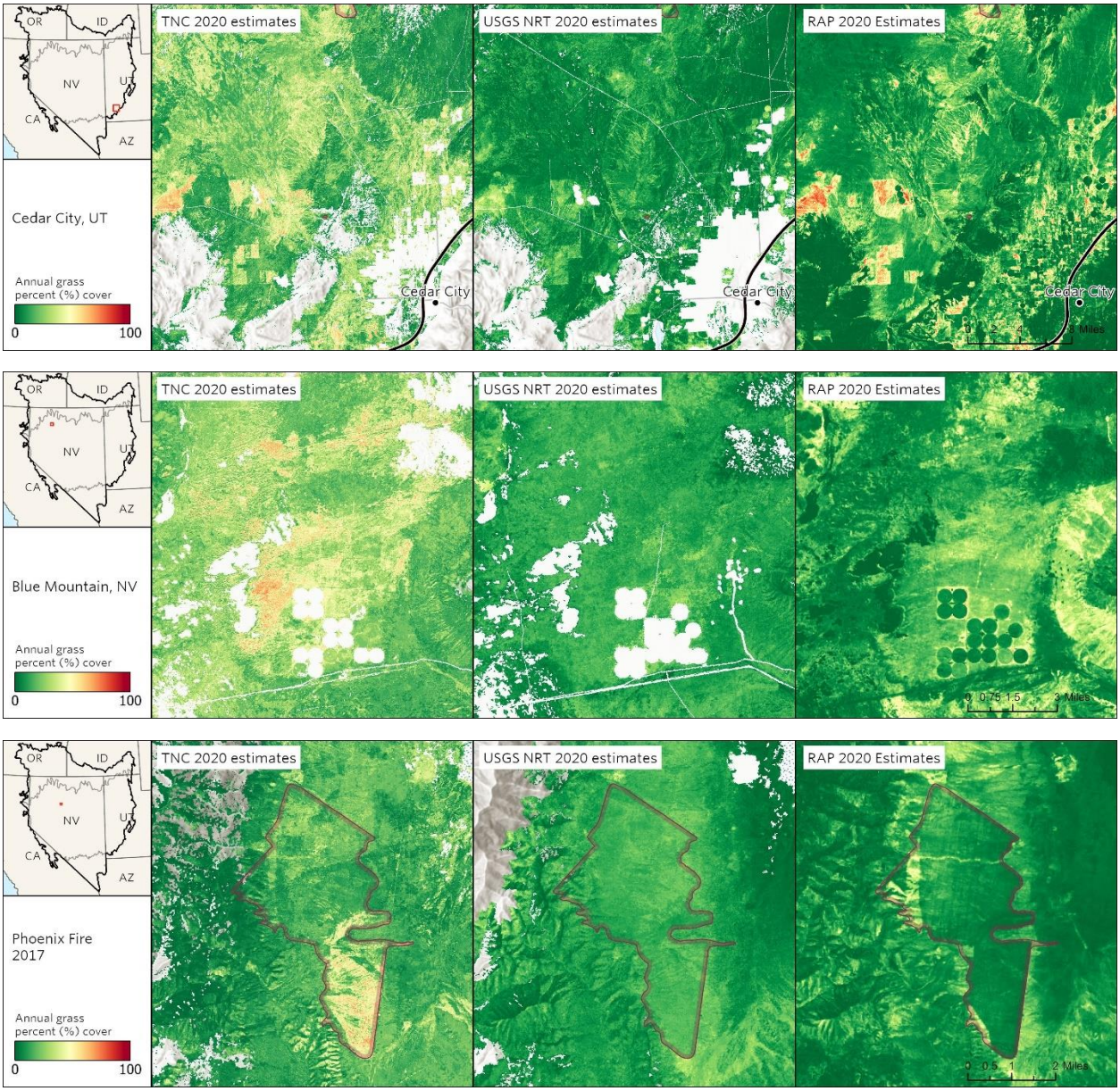


Figure 11. Comparing 2020 annual grass cover estimates across products in different areas. Top – Northwest of Cedar City, Utah. Middle – West of Blue Mountain, NV in a low desert surrounding an agricultural area. Bottom – Burn area from the 2017 Phoenix fire south of Winnemucca, NV.

3.3 Variable Importance

The Random Forest regression model computed variable importance for each input variable as an output. This value was relative within the input variables such that the relative impact among variables on results could be ranked. The top 10 variables by importance values are shown in Fig. 12, but remaining variables are found in Table 1. By far the variable with the most explanatory power was elevation (Fig. 12). Fig. 12 displays the SHapley Additive exPlanations (SHAP) value for each metric. SHAP values can be calculated for any tree-based model and describe each metric's relative contribution to the modeled independent variable (Lundberg and Lee 2017). Elevation was a very important predictor in the model, likely because of the large elevation range throughout the project area. Boyte et al. (2015) found elevation to have a less important role in their model than other variables, but they attributed this to their model being restricted to areas below 2,000-m. Our project area ranges in elevation from 145-m to 4,306-m, and training data ranged in elevation from 1,220-m to 3,045-m.

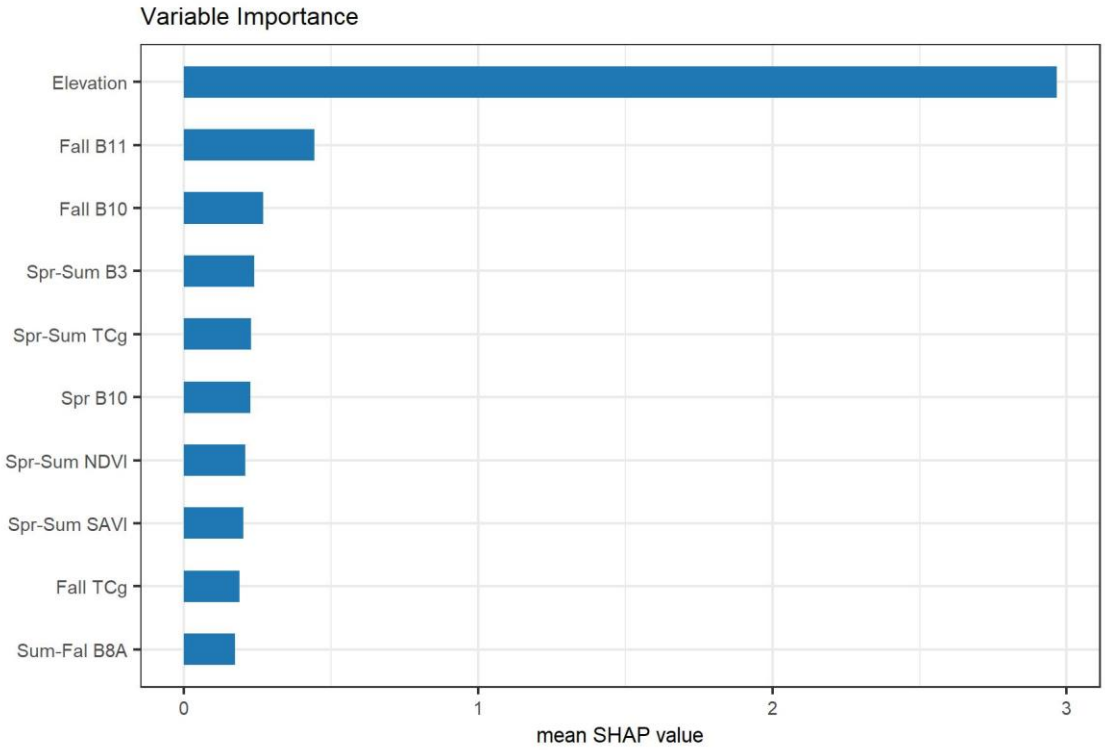


Figure 12. Top 10 variables with most impact on model outputs based on SHapley Additive exPlanations (SHAP). Legend: spr = spring, sum = summer, fal = fall, B3, B8A, B10, B11 are, respectively, mid-infrared Band 3, narrow near-infrared band 8A, mid-infrared Band 10, mid-infrared Band 11, NDVI = Normalized Difference Vegetation Index, SAVI = Soil Adjusted Vegetation Index, TCg = Tasseled-cap greenness.

Shortwave infrared (SWIR) bands 10 and 11 from fall and spring Sentinel-2 turned out to be relatively important variables (Fig. 12). More study would be needed to determine why these might be important variables. It was possible that the SWIR bands, which can be used to display vegetation density or bare soil, have picked up the presence or absence of annual species on the ground. Not surprisingly, metrics that measured the difference between spring and summer vegetation productivity or greenness had a greater impact on model results (Spr-Sum B3, Spr-Sum TCg, Spr-Sum NDVI, etc.).

3.4 Future Applications

The model allowed us to create a map of non-native annual species cover. The map product will be used to determine the acres of treatable land in the Great Basin and Columbia Plateau with annual species and thus, inform management actions and carbon sequestration potential (next sections). Additionally, we can retain the RF model made using the 2020 inputs and training data, then apply it to future years with concurrent inputs as Sentinel-2 continues to capture imagery. As of this writing, inputs for 2021 were only available through summer 2021. This model can be used to estimate annual species in future years, though further study into this model's applications and limits should be pursued.

3.5 Estimating Net Carbon Stored from Restored Rangelands

The 10-year period (which gave seedings the opportunity to establish woody species) of NBP differences among scenarios varied among landscapes. For the IL and TS-Horseshoe Ranches, respectively, these periods were from 2030 to 2039 (Fig. 13) and 2032 to 2042 (Fig. 14 A&B). For the PVMH landscape, this period was from years 2037 to 2046 (Fig. 15).

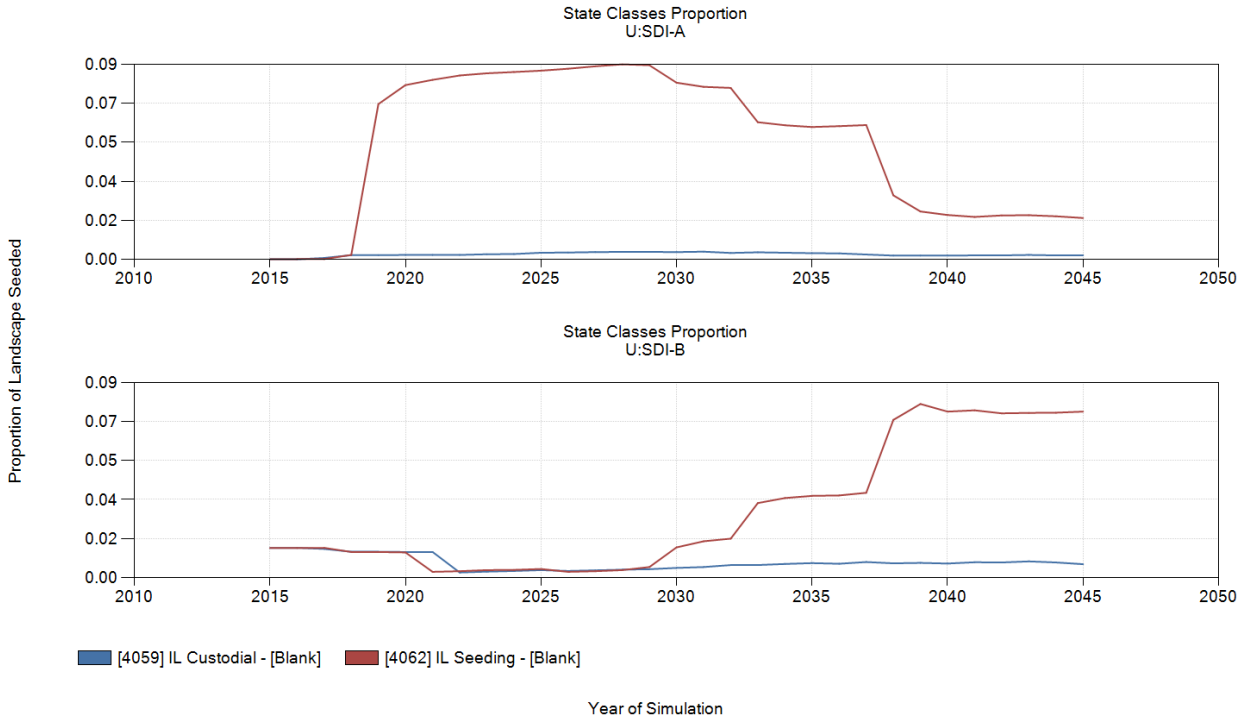
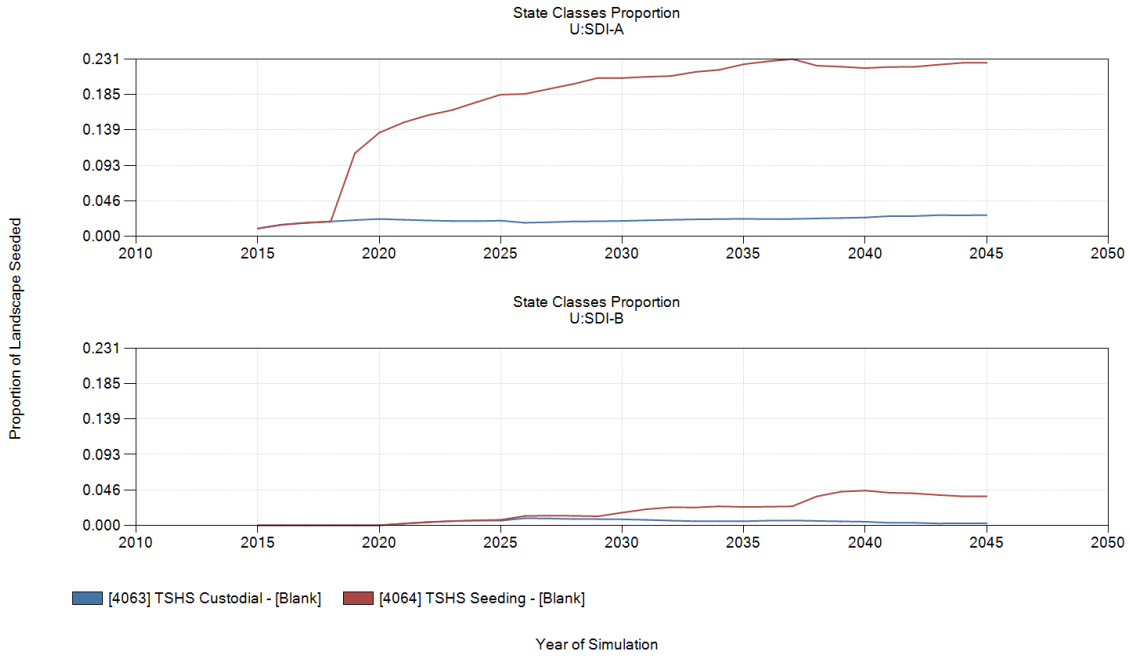


Figure 13. Proportion of landscape seeded in U:SDI-A class (e.g., early successional class; top graph) and U:SDI-B (e.g., mid-successional class; bottom graph for the Custodial and Seeding management scenarios on the IL Ranch. Note the increase of the U:SDI-B class in 2030 in the Seeding management scenario represents the transition from early successional class (U:SDI-A) dominated by perennial grass to mid-successional class (U:SDI-B) co-dominated by perennial grass and woody shrubs.

A-Frequent Fire



B-Normal ("Historic") Fire

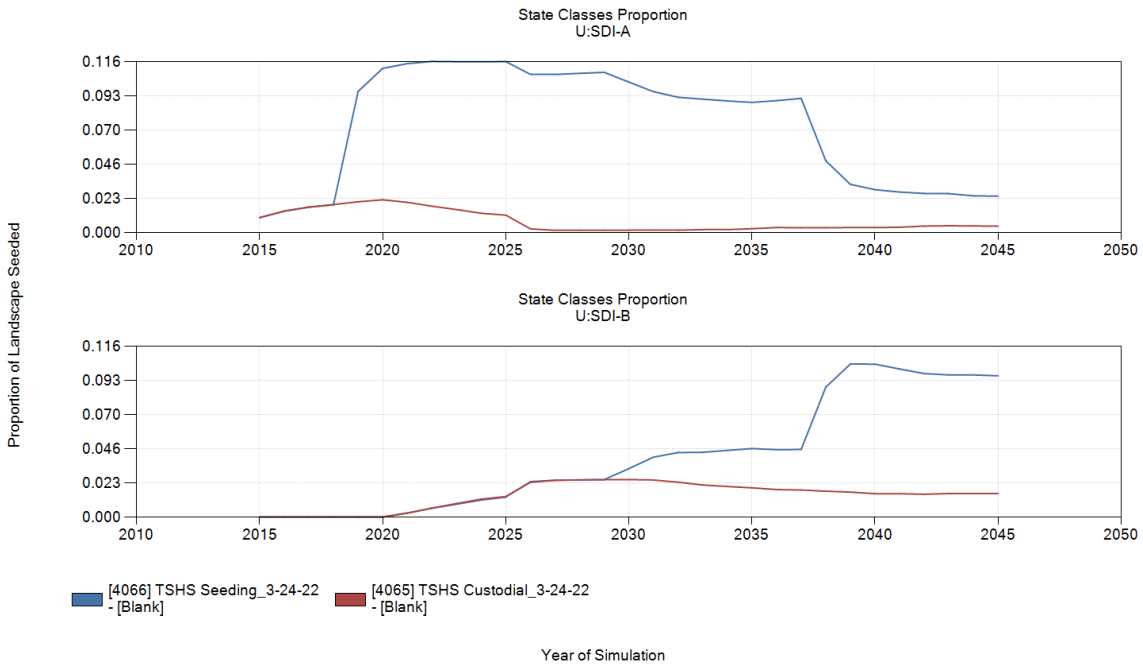


Figure 14 A & B. Proportion of landscape seeded in U:SDI-A class (e.g., early successional class; top graph of panels A and B) and U:SDI-B (e.g., mid-successional class; bottom graph of panels A and B for the Custodial and Seeding management scenarios on the TS Horseshoe Ranch for a frequent fire regime (Panel A) and normal fire regime reflecting the continuation of historic fire patterns (Panel B). Note the increase of the U:SDI-B class in 2032 in the Seeding management scenario represents the transition from early successional class (U:SDI-A) dominated by perennial grass to mid-successional class (U:SDI-B) co-dominated by perennial grass and woody shrubs.

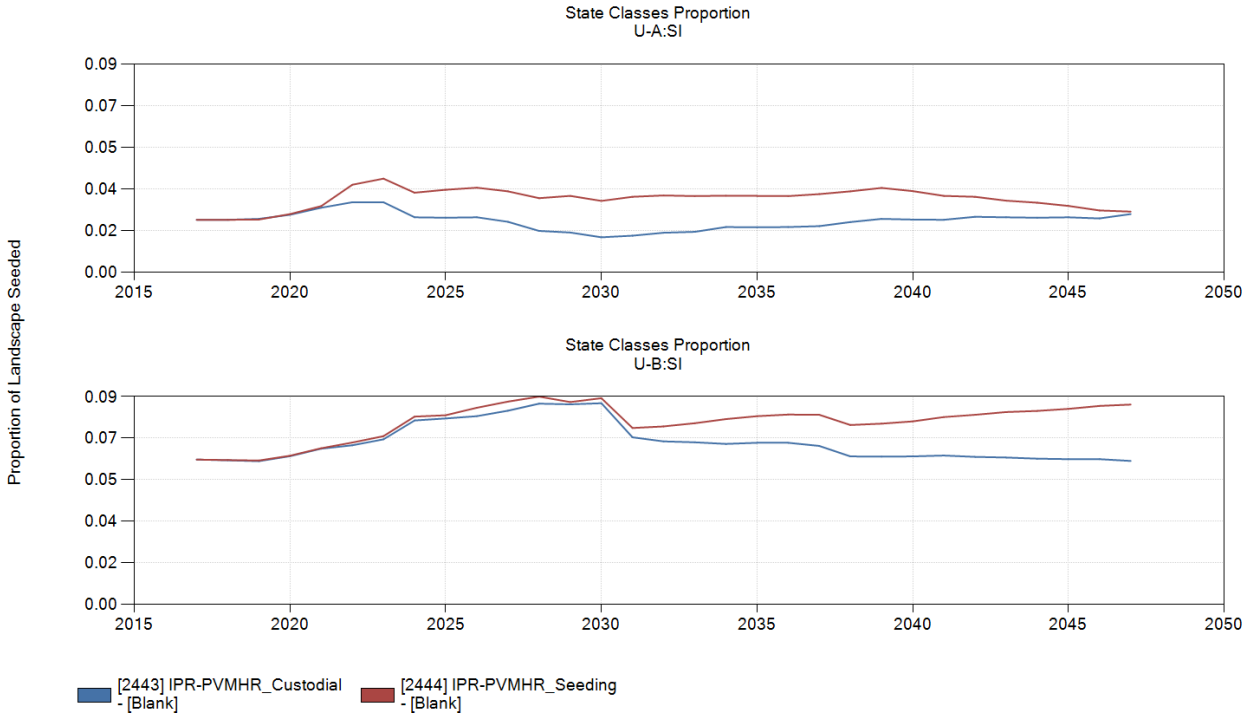


Figure 15. Proportion of landscape seeded in U:SI-A class (e.g., early successional class; top graph) and U:SI-B (e.g., mid-successional class; bottom graph for the Custodial and Seeding management scenarios in the PVMH landscape. The real increase of the U:SI-B class due to woody succession from the U:SI-A class is about in 2037 in the Seeding management scenario because succession in pre-existing seedings is confusing the signal.

The PVMH landscape, IL Ranch, and TS-Horseshoe Ranch with historic fire were sinks of carbon (i.e., positive NBP) without seeding showing 84, 9, and 11 g $C \cdot m^{-2} \cdot yr^{-1}$, respectively, whereas the TS-Horseshoe Ranch with frequent fire was a source of carbon to the atmosphere at-35 g $C \cdot m^{-2} \cdot yr^{-1}$ (Table 3). All landscapes showed that seeding resulted in treated systems becoming sinks of carbon as the NBP difference between scenarios were positive, except for the positive difference (still a sink) found for TS-Horseshoe Ranch's with historic fire as the 95% CI did overlap with zero (Table 3). However, the IL Ranch, which was considered a landscape closer to reference conditions (Provencher et al. 2015) requiring only small level of seeding, was a very weak sink of C at 0.71 ± 0.65 g $C \cdot m^{-2} \cdot yr^{-1}$, whereas values for the PVMH landscape and TS-Horseshoe Ranch with frequent fire, respectively, were two orders of magnitude higher at 19.9 ± 10.6 g $C \cdot m^{-2} \cdot yr^{-1}$ and 19.7 ± 5.0 g $C \cdot m^{-2} \cdot yr^{-1}$ (Table 3). Both the PVMH landscape and TS-Horseshoe Ranch have vegetation more departed from reference conditions (Provencher et al. 2015, Provencher et al. 2020), of which the TS-Horseshoe being by far the most departed from reference conditions. This difference indicated than the ecological condition of the systems in which seedings were implemented will affect how much more sink uplift can be achieved by restoration. The more extensively degraded the ecological systems in a specific landscape, including with too frequent fire relative to normal sagebrush fire regimes, the greater the potential sink on average per square meter; however, each individual seeded pixel in isolation might still contribute similarly to carbon sequestration among geographies.

Table 3. Net Biome Productivity (NBP), Net Biome Productivity difference (NBP Δ) between the Seeding and Custodial management scenarios, and cost of seeding in three different climatic regions of the Basin and Range geologic province and southern Columbia Plateau ecoregion estimated at the pixel, climatic region, and generic ranch levels. 95% C.I. is the 95% percent confidence interval. N = 10.

Area	**NBP (± 95% C.I.)**	**NBP Δ (± 95% C.I.)**		**Cost of seeding (± 95% C.I.)**
MONSOONAL BASIN AND RANGE (UT)				
PVMH ($g \cdot C \cdot m^{-2} \cdot yr^{-1}$)	83.6 ± 18.5	19.9 ± 10.6	$	$2.81·10^{+6} ± $857,842
			$·m^{-2}	$0.0421 ± <$0.01
Utah AOI 6,830 km^2 (Metric Ton of $C \cdot yr^{-1}$)		136,132 ± 72,378	$	$287.5·10^{+6}
Ranch Level 4,047 ha of NNAGF (Metric Ton of $C \cdot yr^{-1}$)		806.6 ± 428.8	$	$1,703,204
SOUTHERN AND CENTRAL BASIN AND RANGE – **FREQUENT FIRE** (NV & CA)				
TS-Horseshoe Ranch	-35.3 ± 9.8	19.7 ± 5.0	$	$15.5·10^{+6} ± $450,350

($g\ C \cdot m^{-2} \cdot yr^{-1}$)				
			$\$ \cdot m^{-2}$	\$0.0424 ± <\$0.01
Nevada & Utah AOI 5,177 km^2 (Metric Ton of $C \cdot yr^{-1}$)		102,156 ± 25,972	\$	$\$219.5 \cdot 10^{+6}$
Ranch Level 4,047 ha of NNAGF (Metric Ton of $C \cdot yr^{-1}$)		799 ± 203	\$	\$1,716,433
SOUTHERN AND CENTRAL BASIN AND RANGE – **HISTORIC FIRE** (NV & CA)				
TS-Horseshoe Ranch ($g\ C \cdot m^{-2} \cdot yr^{-1}$)	10.6 ± 5.6	0.61 ± 0.88	\$	$\$5.5 \cdot 10^{+6}$ ± \$342,443
			$\$ \cdot m^{-2}$	\$0.0447 ± <\$0.01
Nevada & Utah AOI 5,177 km^2 (Metric Ton of $C \cdot yr^{-1}$)		3,196 ± 4,541	\$	$\$231.5 \cdot 10^{6}$
Ranch Level 4,047 ha of NNAGF (Metric Ton of $C \cdot yr^{-1}$)		25 ± 35	\$	\$1,809,907
SOUTHERN COLUMBIA PLATEAU (NV, ID, OR, & CA)				
IL Ranch ($g\ C \cdot m^{-2} \cdot yr^{-1}$)	8.9 ± 5.2	0.71 ± 0.65	\$	$\$8.8 \cdot 10^{+6}$ ± \$92,618
			$\$ \cdot m^{-2}$	\$0.0426 ± <\$0.01
Northern Basin and Range 18,047 km^2 (Metric Ton of $C \cdot yr^{-1}$)		12,743 ± 11,751	\$	$\$769 \cdot 10^{+6}$
Ranch Level 4,047 ha of NNAGF (Metric Ton of $C \cdot yr^{-1}$)		28 ± 26	\$	\$1,726,709

Most of the carbon in the total ecosystem carbon was found in the soil, considered a stable form of carbon. For the IL Ranch, the proportion of soil carbon ranged from 83% to 86% (Fig. 16). This same proportion ranged between 90% and 95% for the TS-Horseshoe Ranch with frequent fire (Fig. 17A), whereas with historic fire the proportion of soil carbon progressively decreased to 87% as the proportion of growing living biomass increasingly accumulated carbon (Fig. 17B). In other words, frequent fire volatilized the carbon in living biomass. The proportion of soil C progressively decreased from about 78% to 58% in the PVMH landscape (Fig. 18). Differences between scenarios were non-existent (Figs. 16-17) to small (Fig. 18).

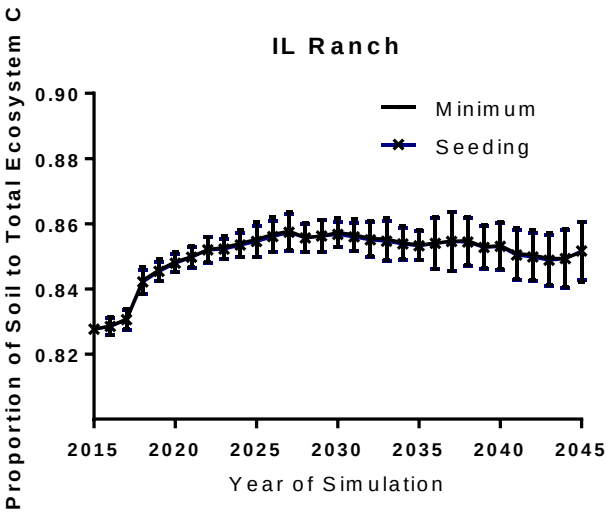


Figure 16. Average proportion of soil to total ecosystem C for the Custodial and Seeding management scenarios in the IL Ranch.

B – Frequent Fire

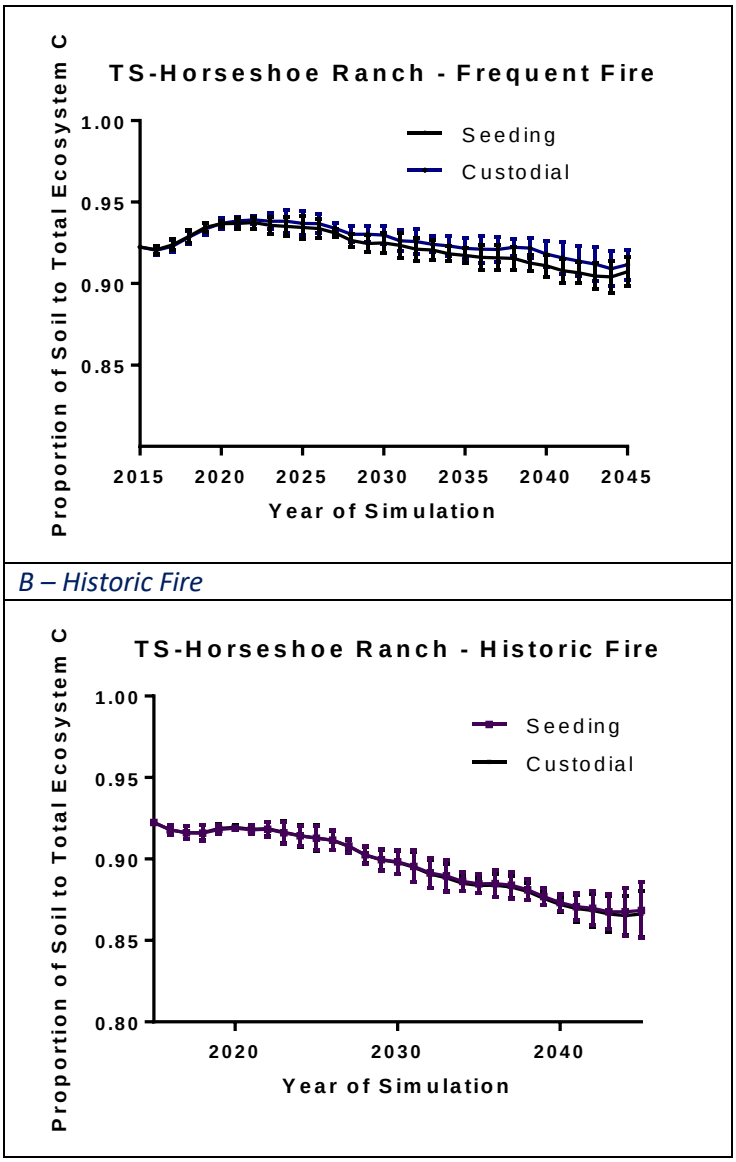


Figure 17 A&B. Average proportion of soil to total ecosystem C for the Custodial and Seeding management scenarios in the TS-Horseshoe Ranch with frequent fire.

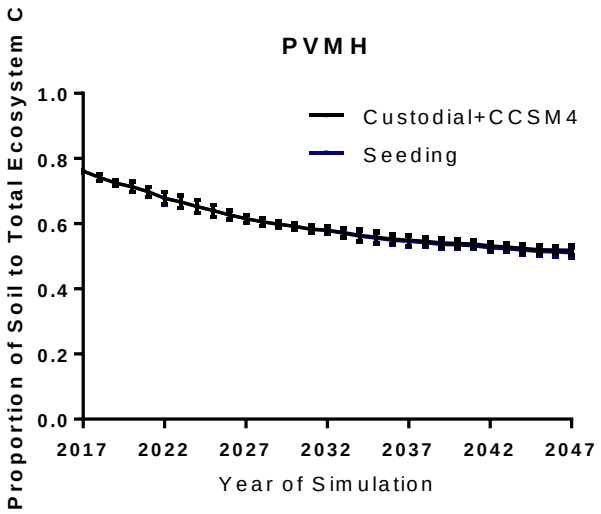


Figure 18. Average proportion of soil to total ecosystem C for the Custodial and Seeding management scenarios in the PVMH landscape.

3.6 Estimating Cost of Restoring Rangelands

The cost per square meter and the generic ranch level assuming 4,047 ha of NNAGF, respectively, was about $\$0.04 \cdot m^{-2}$ in all landscapes and ranged from $1.64 million for the PVMH landscape at the low end to about $1.71 million for the TS-Horseshoe Ranch at the high end (Table 3). These small differences reflected decimal point differences at the m^2 scale among landscapes, which was smaller in Utah. The lower cost in Utah was simply due to the lower cost per unit area of seeding because cost effective chaining was used to imprint seed into the soil at slopes up to 30% (and not felling trees or shrubs). Land managers rarely, if ever anymore, use chaining, to imprint seed in Nevada, Idaho, or California. Not only was the operation cheaper in Utah, the access to steeper slopes by equipment allowed a greater prevention of future fires and reduction of areas dominated by NNAGF.

The quantity of carbon stored at the ranch level, primarily in the soil, ranged from 806.6 metric $Ton \cdot yr^{-1}$ in Utah's PVMH landscape, 799.0 metric $Ton \cdot yr^{-1}$ on the TS-Horseshoe Ranch with frequent fire, 28 metric $Ton \cdot yr^{-1}$ on the IL Ranch to 25 metric $Ton \cdot yr^{-1}$ on the TS-Horseshoe Ranch with historic fire (Table 3). These numbers are less than numbers reported in Bradley et al. (2006); however, our analysis was constrained to sites where current practices could be used to convert NNAGF to shrublands (e.g., exclusion of steep slopes and isolated small patches of NNAGF). Considering that 1 metric Ton is 1,000 Kg, 800 metric Tons is 800,000 Kg of C. Therefore, the cheapest seeding that produced the most carbon sequestered in soils was in Utah's PVMH landscape.

3.7 Estimating Total Carbon Stored and Total Cost in the AOI

The AOI was partitioned in broad climatic regions corresponding to the monsoonal influence, which was basically Utah's Great Basin ecoregion, the southern Great Basin ecoregion in Nevada and California, and the northern basin and range geologic province that occupies the Snake River volcanic batholith and maps to the southern Columbia Plateau ecoregion of TNC in Nevada, Oregon, Idaho, and California. The total area of NNAGF per climatic zones was presented in Table 3 for extrapolation of total C stored and cost. Numbers were very large with the most stored carbon obtained in Utah at 136,132 metric $Ton \cdot yr^{-1}$ at a cost of $287 million (Table 3). The smallest amount of carbon was obtained from the TS-Horseshoe

Ranch with historic fire in the southern and central basin and range at 3,196 metric Ton·yr^{-1} at the highest cost of $23 million (Table 3).

4. CONCLUSIONS

Net biome productivity reported in this study ranged from -35 to 84 g C·m^{-2}·yr^{-1} without seeding (Table 3). These values were smaller than the maximum values reported for other more productive systems. Values from other systems were from -599 (source to atmosphere) to 847 g C·m^{-2}·yr^{-1} (sink from atmosphere) in different wet meadows of the northern Sierra Nevada (Reed et al. 2020), -0.47 (source) to 153 g C·m^{-2}·yr^{-1} (sink) in northern Sierra Nevada forests (Campbell et al. 2009; Potter 2010; Hudiburg et al. 2011), 403 g C·m^{-2}·yr^{-1} (sink) in evergreen tropical forests (Luyssaert et al. 2007), and 215 g C·m^{-2}·yr^{-1} (sink) in tropical wetlands (Sjögersten et al. 2014). As with our study, other studies such as Reed et al. (2020) and the meta-analysis by Nagy et al. (2020) found that C was mostly stored in soils. Our soil stock included all belowground stocks of carbon (e.g., organic soil carbon and belowground biomass) which may help explain why our proportion of soil carbon relative to total ecosystem carbon was generally higher than rates for rangelands (Meyer 2012; Nagy et al. 2020), but similar to generalized estimates of desert carbon stocks (Janzen 2004).

In the Piedmont region of the USA, Liu et al. (2016) estimates from modeled land use and cover changes that forest ecosystems sequestered 25 g C·m^{-2}·yr^{-1} as measured by NBP. This value is within the range reported here for arid rangelands, indicating that rangelands compared favorably to Appalachian forests for carbon sequestration. The TS-Horseshoe Ranch with frequent fire, a somewhat unrealistic case with ten times the normal fire return interval, was the exception, which was highly dominated by shrublands dominated by NNAGF understory.

Few studies of carbon stocks and flows are reported for sagebrush systems. Svejcar et al (2008) synthesized rangelands studies where Net Ecosystem Exchange (NEE) values were reported. NEE is the net flux of carbon between photosynthesis and respiration (NEP, net ecosystem productivity, is similar, but longer term albeit reported per year) and does not include loss of carbon to disturbances (e.g., fire volatilizing woody and herbaceous sources of carbon). NBP does include change in carbon flux due to natural disturbances over the long term. Two NEE sagebrush values from Oregon and Idaho were reported at 73 and 83 g C·m^{-2}·yr^{-1} that are of the same order of magnitude, albeit slightly higher, than NBP values modeled in our study, which considered change of flux due to disturbances. Fellows et al. (2018) at a site in Idaho found 103-157 g C·m^{-2}·yr^{-1} of NEP pre-fire and 122-182 g C·m^{-2}·yr^{-1} of NEP after the fire, which implies a change of 20-30 g C·m^{-2}·yr-1 in NEP due to fire. However, the NEP values of Fellows et al. (2018) were twice the NBP values found here with simulations (Table 3) and difficult to compare.

The purpose of this study was to estimate the added amount of NBP that could be sequestered in sagebrush systems due to seeding perennial grass and woody species (i.e., sagebrush and antelope bitterbrush [*Purshia tridentata*]). This was in the spirit of the Natural Climate Solutions approach (Griscom et al. 2017; Fargione et al. 2018; Graves et al. 2020). The modeled carbon sink difference estimated in this study ranged from 0.74 g C·m^{-2}·yr^{-1} to 19.9 g C·m^{-2}·yr^{-1}. In the case of the PVMH, seeding could result in 104 g C·m^{-2}·yr^{-1} (104 = 84 from NBP + 20 from NBP difference due to seeding) total sequestered (Table 3), mostly as a stable form in the soil. Seeding would lessen the carbon source

to the atmosphere of the TS-Horseshoe Ranch to -15 g $C \cdot m^{-2} \cdot yr^{-1}$ (compared to -35 g $C \cdot m^{-2} \cdot yr^{-1}$), which is a significant offset (Table 3).

In Oregon sagebrush systems, Graves et al. (2020) used published literature and static Monte-Carlo methods to estimate the loss and uncertainty of CO_2 equivalents to the atmosphere from standing sagebrush communities burning and converting to NNAGF and the gain in CO_2 equivalents to the system from restoration of NNAGF areas to standing sagebrush communities by seeding. Graves et al. (2020) only considered living biomass CO_2 equivalents, thus did not include soil carbon as they considered soil carbon stocks and flows estimates too uncertain in sagebrush systems. They estimated that CO_2 equivalents gains and losses (i.e., the difference due to fire or seeding) of only living biomass were equivalent at 0.81 ± 0.44 metric T of CO_2 equivalents$\cdot ha^{-1} \cdot yr^{-1}$. Given that we estimated about 800 metric T of $C \cdot yr^{-1}$ for 4,047 ha in two of our climatic zones or 0.19 metric T of $C \cdot ha^{-1} \cdot yr^{-1}$ caused by seeding (Table 3), we believe that Graves et al.'s (2020) estimate should be closer to 0.06 metric T of CO_2 equivalents$\cdot ha^{-1} \cdot yr^{-1}$ if soil carbon accounts for 60% to 80% of total biome carbon. However, it was not possible to confirm if Graves et al. (2020) used NBP, NEP, or other flux measures.

An interesting result of models reported here was that the extent of NNAGF-dominated areas appeared to positively correlate with the magnitude of the carbon sink difference per unit area (Table 3). In other words, greater NNAGF area offered the greatest opportunity for uplift from restoration as it would be difficult to improve the ecological condition of a system already close to the reference condition. Being closer to the reference condition usually means the system has a higher representation of more mature vegetation classes that, presumably, have grown a more substantial root system whose decay contributes to relatively stable soil carbon. Preventing the loss of established shrublands, therefore, to too frequent fire and NNAGF should be a conservation strategy. We believe that the IL Ranch fits that description. Reed et al. (2020) reported that greater C sinks were generally associated with the greater ecological integrity (i.e., less departed from vegetation reference conditions) of Sierra Nevada wet meadows, whereas degraded wet meadows were sources of C to the atmosphere; however, they discussed that restoration of degraded wet meadows offered the greatest chance to offset C emissions by turning these meadows into sinks. Additionally, Bradley et al. (2006) estimated a total of 8,000,000 metric T of C had been transported to the atmosphere due to historic conversion of shrublands to cheatgrass stands within the Great Basin.

A critical aspect of the concept of carbon sequestration uplift was, and will be, the cost per m^2 of seeding (Table 3). The cheapest average cost of seeding was in Utah that translated to about \$66$\cdot ha^{-1}$ (\$163$\cdot ac^{-1}$), whereas the highest cost was \$69$\cdot ha^{-1}$ (\$170$\cdot ac^{-1}$), at the TS-Horseshoe Ranch. These small differences in cost resulted in large financial differences when extrapolated to large areas of the AOI (Table 3). Lowering the cost of seeding is critical to achieve large-scale seedings and sizable carbon sequestration. Carbon sequestration in rangelands appeared to be a viable strategy because, for example, spending \$1.6 million to seed 4,047 ha (10,000 acres) of perennial species to replace undesirable NNAGF and sequester an additional 800 metric T of $C \cdot yr^{-1}$ was in line with current range improvement practices and cost (Monsen et al. 2004). Seedings to increase carbon sequestration is likely to have co-benefits with other natural resource goals (e.g., improved wildlife habitat, erosion control, etc.).

Further lowering the cost per unit area of seeding operations is achievable. In our models, the cost per unit area was fixed; however, this cost would decrease if contractors could bid for seeding increasingly

larger areas through a state-level public-private entity, as performed by the Watershed Restoration Initiative in Utah (https://wri.utah.gov/wri/). The cost per unit area might increase; however, if several contractors seeding very large areas depleted commercial seed reserves with appropriate local genetic sourcing and drive the price of seed up. Therefore, coordinating introduced and native species seeding supply and sourcing might also be part of lowering the cost per unit area of seeding.

Increasing the success of seed germination and establishment would also lower costs because the success rates of introduced and native species decreases appreciably with decreasing elevation; as a result, taxpayers pay for 30-50% of seeds of introduced species that fail to germinate and/or establish at lower elevations (as assumed in our sagebrush models), and sometimes repay to have failed seedings to be reseeded. The success rate of native species seedings is even lower than those of introduced species at lower and middle elevations (Monsen et al. 2004). Germination and establishment success could be improved by mechanical methods, such as using rangeland drills of non-rocky soils and various seed imprinters and chaining (Monsen et al. 2004), or by improving seed technology (Madsen et al. 2016; Baughman et al. 2022).

Climate change bet hedging may offer another option to lower the cost per unit area of seedings. Bet hedging is an evolutionary strategy by which an organism sacrifices a fraction of its fitness during favorable conditions to better survive during stressful conditions (Olofsson et al. 2009). Under current climate, a land manager may seed a certain variety of crested wheatgrass (*Agropyron cristatum*) and/or bluebunch wheatgrass (*Pseudoroegneria spicata*) at the 25 to 30 cm (10-12 inch) precipitation zone in upland soils of Wyoming big sagebrush communities as these species should perform well in loamy soils and moisture conditions. Instead, droughtier species such as Siberian wheatgrass (*Agropyron fragile*) and/or Thurber's needlegrass (*Stipa thurberiana*) could be seeded to survive the periodic droughts associated with Pacific El Niño/La Niña cycles in Nevada and future climate warming. Moreover, a land manager may include all these species in one seed mix and let soils and future climate sort out winners and losers.

We recommend continued investment in understanding carbon dynamics in rangelands, particularly at belowground stocks which hold most of the ecosystem's carbon yet are the least studied. Despite the lower level of attention rangelands have gathered in carbon sequestration discussion, we believe this study highlights the ability of rangeland restoration to contribute to natural solutions to reduce carbon emissions.

5. Acknowledgments

We are grateful for funding from the US Climate Alliance Grant Program for NWL Research, which was made possible by a grant from the Doris Duke Charitable Foundation, transferred to the Nature Conservancy through the not-for-profit American Forests and Nevada Division of Natural Heritage (NDNH). Kristin Szabo from NDNH is thanked for reaching out to TNC. We thank sub-contractors Apex Resource Management Solutions Ltd for populating the stock-and-flow sub-model of the ST-Sim software and writing the carbon methodology, and Spatial Solutions Inc for remote sensing advice to TNC's Sentinel imagery analysis. The following reviewed drafts of the report: Michael Clifford, Kristin Szabo, Nathan Welch, and Elaine York.

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Appendix A

Increasing carbon storage in invaded cold deserts: A case study from northern Nevada

Kevin Badik, Louis Provencher, Leonardo Frid

Introduction

Deserts and semideserts hold roughly 8% of the global terrestrial carbon pool, a relatively small number given these biomes cover nearly 30% of the land area (Janzen 2004). Within western North America, cold deserts, such as the Great Basin and portions of the Columbia Plateau, are estimated to contain 5% of the total terrestrial carbon (Zhu et al. 2012). As such, deserts are often ignored in discussions regarding sequestration of carbon to mitigate against anthropogenic caused climate change (Svejcar et al. 2008, Meyers 2012). Despite the low relative ecosystem carbon compared to other North American biomes, Meyer (2012) has outlined several reasons why cold deserts might be a good system to invest in carbon sequestration. Of the total carbon stock within deserts, the vast majority tend to be in the soil organic carbon (SOC) compared to the standing biomass stock. SOC is considered a more stable pool of carbon than standing biomass, which more readily decomposes. The North American cold deserts (NACD) are dominated by woody species, which tend to have greater root:shoot ratios and deeper root systems compared to other ecosystems. The greater belowground biomass coupled with low decomposition rates means that relatively little carbon is released into the atmosphere. Several studies have confirmed that shrub dominated cold deserts act as sinks for carbon (Hunt et al. 2004, Svejcar et al. 2008). In addition to low natural carbon emissions, Meyer (2012) argues that economic factors in cold deserts means there is less conflict in managing lands for carbon sequestration than other ecosystems, such as forests or prairies where timber or agricultural demands are higher.

Despite the cold desert currently acting as carbon sinks, conversion of dominant vegetation may shift the carbon cycle. Across much of the NACD, the invasive annual species cheatgrass (*Bromus tectorum*) now dominates much of the landscape. Cheatgrass can alter the carbon cycle in multiple ways. Sites dominated by cheatgrass have more rapid carbon cycling than intact native vegetation (Meyer 2012) and store less aboveground carbon than native communities (Bradley et al. 2006). As an annual, cheatgrass produces greater amounts of litter which more readily moves carbon from the terrestrial to atmospheric pools. Cheatgrass also tend to have shallower root systems, preventing the storage of carbon in deeper, more stable SOC stock (Meyer 2012). Lastly, cheatgrass has altered the native fire regime, resulting in more frequent fires which promote the establishment of itself over the more slowly recovering native species (Whisant 1990, D'Antonio and Vitousek 1992). Bradley et al. (2006) have suggested that the conversion from a native shrubland to cheatgrass dominated landscapes is likely to shift the Great Basin from a carbon sink to carbon source. However, management actions within the Great Basin may be able to reverse the conversion to invasive species monocultures and thus increase carbon storage in these impacted landscapes.

The Nature Conservancy's Nevada Field Office (NVFO) has pioneered a landscape scale land use planning tool called "Landscape Conservation Forecasting™" (LCF). LCF couples high resolution satellite imagery, state-and-transition simulation models (STSM, see Daniel and Frid 2012, Provencher et al. 2015 for discussion of STSM), and stakeholder engagement in order to simulate how management actions impact vegetation dynamics (Low et al. 2010). LCF allows land managers to understand how management strategies impact the landscape in the future and compare these strategies under different scenarios (e.g., budgets, climate, etc.). Recent advances in the STSM software ST-Sim (ApexRMS 2016) have allowed LCF to incorporate vegetation derived metrics, such as single species habitat suitability (Provencher et al. 2017) and ecosystem carbon cycling (Sleeter et al. 2015).

NVFO used LCF to model carbon fluxes to ask the question, "Can management actions reduce fire frequency and cheatgrass dominance to increase carbon storage in a cold desert shrubland?" Two

management scenarios were modeled: 1) using fuel breaks to limit fire spread (hereafter Fuel Break Only scenario) and 2) combination of fuel breaks and cheatgrass treatment (hereafter Fuel Break+Seeding scenario). These scenarios were compared against a scenario where no active management (Minimum scenario was done on the landscape. While carbon dynamics have been coupled with STSM previously (Sleeter et al. 2015), those analyses have generally been at the regional to continental scale and few have focused on carbon fluxes in the NACD.

Methods

For a previous project, NVFO mapped 485,732 acres (1,965 km^2) in northern central Nevada. The goal of this previous project was to model management strategies to improve habitat for Greater-sage grouse (Provencher et al. 2016). Simulated management in this project included a wide suite of actions to restore ecosystem function, including: cheatgrass treatment, fuel breaks, riparian restoration, stand age heterogeneity, and exotic forb control. For the current analysis, a subset of the previously collected satellite imagery was used, totaling 376,434 acres (1,523 km^2). Located in northern Nevada bordering Idaho, the Owyhee Study Area (OSA) is located in the southern Columbia Plateau. Generally, the OSA has flat topography and volcanic geology. Vegetation is mostly Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*), with infrequent pockets of low sagebrush (*Artemisia arbuscula*), basin wildrye (*Leymus cinereus*), and wet meadows. A portion of the southern and eastern OSA has experienced fire within 15 years, which resulted in a matrix of cheatgrass and native plants. Unlike other portions of the NACD ecosystem, conifers are absent. Within the study area both livestock graze and wild horse grazing occurs, with most of the project area coinciding with the Owyhee Herd Management Area.

Satellite imagery was captured in 2013 using 5-m resolution RapidEye. Two vegetation layers were generated describing the ecological system and the vegetation structural class. Ecological systems are similar to the "Biophysical Setting" used by the LANDFIRE project (Rollins 2009) and described the dominant potential vegetation based on vegetation, climatic, and other physical characteristics. The structural class is defined by the current vegetation composition and incorporates successional stage (e.g., early seral), canopy structure (i.e., open or closed), and anthropogenic influences (e.g., native or exotic species).

Each ecological system has a state-and-transition model which includes all likely classes. The classes include the "reference" classes, which are the classes expected to be present in the system pre-European settlement. Additionally, "uncharacteristic" classes are represented. These are classes caused by post-European settlement actions, such as agriculture, roads, invasive species. Classes can change to another classes via "transitions". These can be deterministic (e.g., time causes transition from an early seral to mid-seral class) or probabilistic. Age of the class is an important component for carbon dynamics as vegetation tends to accumulate carbon as it ages. Probabilistic transitions can be natural disturbance, such as fire or drought, or anthropogenic actions like restoration treatments. Each probabilistic transition has a rate associated with it derived from published literature or expert opinion when data are scarce. These rates vary with the characteristics of the vegetation class (e.g., cheatgrass pixels are more likely to burn than other less flammable vegetation). The same transition type may have multiple outcomes. For example, fire in a mature sagebrush class may cause shift to an early seral state or conversion to cheatgrass. The vegetation rasters and the state-and-transition models used in the STSM software ST-Sim. ST-Sim allows for spatial Monte Carlo replications, where external factors like climate or atmospheric CO_2 concentrations can also be inputted to vary the inherent rates of transitions (see Low et al. 2010 and Provencher et al. 2016 for a detailed description of STSM methodology). Each scenario had 10 replicates. External time series of climate, CO_2, and temporal patterns of transitions (e.g., occurrence of large fire years) was the same across scenarios. This means high fire activity

occurred in the same simulated years across all scenarios but location of the fire starts was randomly positioned.

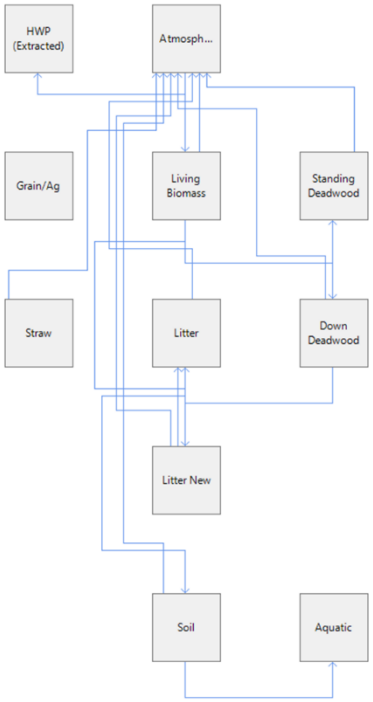

Figure A1. Carbon stock-and-flow model used in this analysis. Model parameters were adapted from Sleeter et al. (2015). Note, no flow was modeled to the "Aquatic", "GrainAg", HWP (Extracted)", and "Straw" stocks.

Modeling the carbon dynamics followed the methodology in Sleeter et al. (2015). This method utilizes a stock-and-flow model, where the total carbon storage of a system is divided into several stocks, including: atmosphere, living biomass, litter, and soil organic carbon (Figure). Moreover, storage or release of carbon is synchronized with the natural processes in the non-carbon STSMs. Flows represent the processes that move carbon from one stock to another and similar to "transitions" in the STSM. Flows can either be automatic, such as plant growth, decomposition, or soil emissions, or event-based, such as land use change or fire. As flows occur, the model tracks the proportion of carbon that is moved from one stock to others. The amount released was determined by assigning a severity rank to the removal (complete release to the atmosphere, partial, dead biomass standing, slow decomposition, etc.) and a quantity of carbon per acre of each broad vegetation type. During a fire of the carbon lost from the living biomass stock, a portion will be emitted into the atmosphere and a portion will be converted to standing dead biomass. As with the STSM, external factors such as climate can be inputted into the simulation to model climate impacts on carbon storage. We used CO_2 concentration and plant growth rates from 1970 to 2000 as this time series was previously calibrated in prior work by the US Geological Survey (Sleeter et al. 2015). This time frame means that carbon flux did not interact with increasing temperatures and changing precipitation that would be assumed in models projecting future climate. The first 5 yrs. of the simulation were used as spin-up times to allow the carbon model to reach equilibrium before any transitions were modeled.

Two management actions were modeled in this study: conversion of cheatgrass monocultures to perennial grass/shrubland and fuel breaks. To convert cheatgrass dominated sites to perennial dominated sites, we modeled a combination of herbicide application and perennial grass seeding. The goal of this treatment was two-fold: reduce likelihood of fire and increase carbon storage through establishment of perennial species. In seeded areas, the likelihood of fire is reduced due to the inherent lower flammability of the seeded vegetation compared to cheatgrass monocultures. While native species seeding is often the desired state due to multiple benefits (e.g., wildlife use, fire resistance, soil stability, etc.), we assumed the seeding was dominated by crested wheatgrass (*Agropyron cristatum*), an introduced grass species. Crested wheatgrass has long been used in the NACD because of its increased establishment success in semi-arid environments, fire resistance, and ability to outcompete cheatgrass (Maestas et al. 2016). In addition to the benefits of crested wheatgrass, sufficient supplies of native seed are often not available to seed in large areas. The goal was to model likely management outcomes rather than the ideal ones. The second action was creation of fuel breaks. Fuel breaks may be bare soil (brownstrips) or vegetated with highly fire resistant plants (greenstrips). The modeled fuel breaks were assumed to be composed of the introduced forage kochia (*Kochia prostrata*) along either side of an existing road. This species is often used in fuel breaks as it has a low stature, thus reducing flame lengths, resist cheatgrass invasion, and maintains high levels of moisture during the summer. While fuel

breaks are known to be effective in slowing down fire spread (Maestas et al. 2016), they are not absolute treatment as extreme fire conditions (e.g. high winds, low humidity, favorable pre-fire weather) can cause fires to "jump" over even wide fuel breaks. We assigned a fire probability of 0.0001 (1 pixel out of a 10,000 per year of simulation will be burn or be permeable to fire spread) to model the permeability in real-life fuel breaks. Modeled fuel breaks were placed along existed roads to reduce habitat fragmentation.

STSM classes were translated into broader land use categories: barren, forest, grassland, shrubland, water, and wetland. While previous use of these categories was used at the system level, we modified the approach so that land use category was designated based on our vegetation class characteristics, such that pixels within the "Big Sagebrush" system would be classified as grassland or shrubland depending on the mapped vegetation structure and composition. Three management scenarios were simulated: Minimum, where no additional management beyond grazing was modeled, Fuel Break Only, where the only additional management modeled where fuel breaks, and Fuel Break+Seeding, modeled actions included fuel breaks, cheatgrass removal, and perennial grass seeding. Grazing followed a rotation plan that alternated the timing of grazing across pastures on the ranch. Modeled grazing was a simplification of grazing strategies currently employed. Fuel breaks were along existing roads, and fuel breaks were sited in the same location for the two scenarios (**Error! Reference source not found.**). For the Fuel Break+Seeding scenario, the yearly planned implementation rate of the cheatgrass removal and seed was fixed at 1,000 acres per year. The treatment was only implemented in pixels of cheatgrass monocultures. The planned implementation rate was rarely achieved as other factors inherent to the model limited the acres treated in the simulation (average yearly implemented was 225 acres). For example, treatment was excluded in areas with greater than 15% slope and treatment was not applied to smaller patches. These limitations were simulated to mimic real life limitations of machinery on steep slopes and unlikely case of treating small patches of cheatgrass that are widely dispersed.

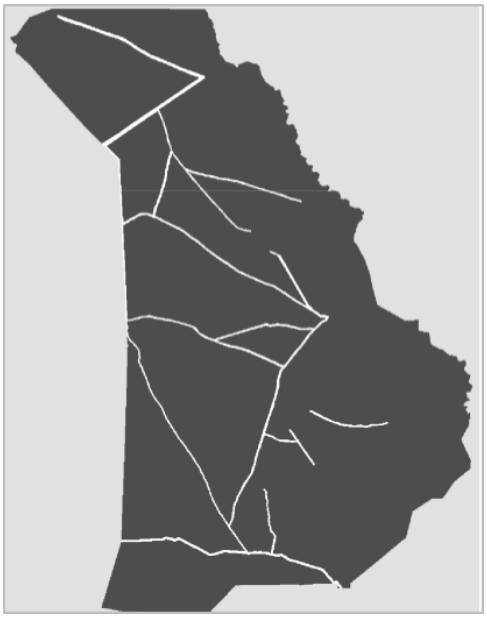

Figure A2. Location of fuel breaks implemented in Owyhee Study Area are shown in white

Table A1. Average carbon storage for the terrestrial stock ands and total ecosystem carbon for the three scenarios from the last 10 years of the simulations.

Scenario	Litter (lbs.)	Living Biomass (lbs.)	Soil (lbs.)	Standing Deadwood (lbs.)	Total Ecosystem Carbon (lbs.)
Minimum	287,184,382	925,297,748	12,334,864,286	765,519,443	14,808,361,705
Fuel Break Only	287,984,539	927,006,384	12,343,890,973	763,717,767	14,818,799,728
Fuel Break+Seeding	288,095,808	930,776,747	12,351,113,074	765,778,691	14,831,194,814

Results/Discussion

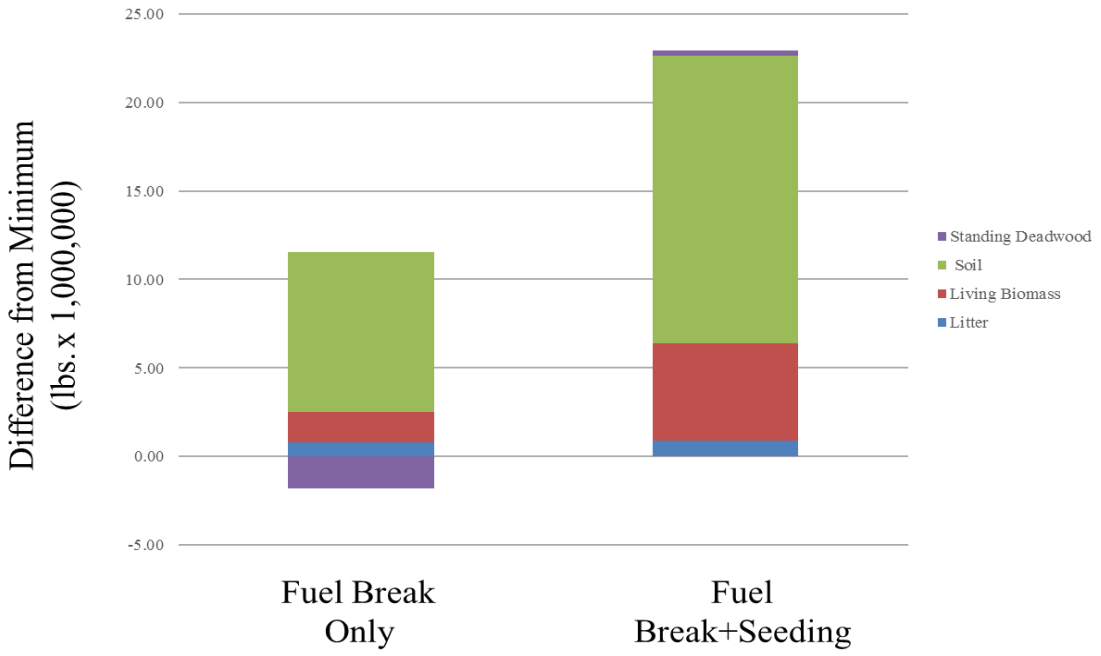

Figure A4. The difference between the two management scenarios and the Minimum scenario for the four terrestrial carbon stocks.

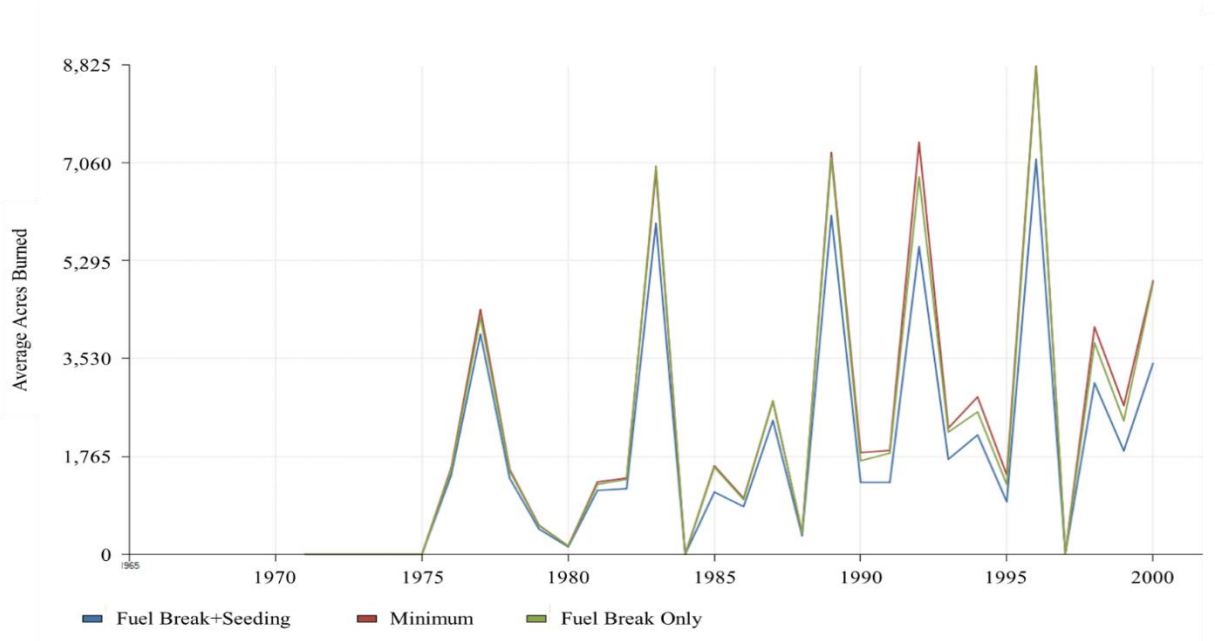

Figure A3. Average acres burned per year for the three scenarios: Minimum, Fuel Break Only, and Fuel Break+Seeding. Note, the first 5 years were a calibration period for the stock-and-flow model and no transitions, including fire, were allowed.

Differences among scenarios were more noticeable in the last 10 yrs. of the simulation as the legacy of past fire activity and seeding treatment began to show. On average, Fuel Break+Seeding stored the most carbon followed by Fuel Break Only and Minimum, respectively (**Error! Reference source not found.** A1, Figure). By the last year of the simulation, Fuel Break+Seeding had 15,058,524,650 lbs. of carbon compared to 5,007,002,315 lbs. for Minimum and 15,030,092,731 for Fuel Break Only. These differences were mostly due to increased stored carbon in the soil carbon, though differences were also observed in the

living biomass and litter stocks as well. Overall, soil carbon represented over 80% of the total ecosystem carbon with living biomass the next most important stock.

Table A2. Average acres burned across all years and across all replicates for each of the 3 scenarios.

Replicate	Minimum	Fuel Break Only	Fuel Break+Seeding
1	68,264	64,270	54,745
2	34,146	33,486	29,900
3	49,179	44,940	39,479
4	75,745	70,507	57,744
5	82,152	80,092	60,505
6	33,361	32,375	28,810
7	86,877	90,076	73,188
8	148,155	140,016	112,315
9	66,252	67,071	56,057
10	45,866	44,053	37,082
Average	**69,000**	**66,689**	**54,983**

The carbon differences among scenarios were most likely caused by difference in fire activity. Although the total acres burned was not significantly different among the scenarios, generally the Minimum scenario experienced more fire than the two active management scenarios with Fuel Break+Seeding having lower fire activity (**Error! Reference source not found.**). The difference between Minimum and Fuel Break+Seeding was most pronounced when comparing large fire years (**Error! Reference source not found.**). This implies that the combination of fuel breaks and establishment of fire resistant perennial leads to lower fire activity than fuel breaks alone. Fuel beaks more visibly created sharper edges to the fires, where they stopped (Fig. A 3A *vs*. B). In addition to total acres burned, the likelihood of pixels reburning also differed among the three scenarios (**Error! Reference source not found.**). Across all scenarios, repeated fires were more likely to occur along the eastern portion of OSA. However, in the Fuel Break+Seeding reburning was much less common. Given the spatial patterns of reburning in all scenarios, this difference is likely more attributable to seeding than the fuel breaks. Increasing the length of time between fire is an important component of NACD management as recovery from fire among many native woody species and herbaceous perennials is a slow process and native community resilience to fire tends to increase with age.

At the beginning of the simulation, roughly 1,140 acres were classified as cheatgrass monoculture. By the final year of the simulation, cheatgrass monocultures accounted for 7,480 acres in the Minimum management. In the Fuel Break Only scenario, cheatgrass monocultures were slightly reduced (average acres =6,925). The Fuel Break+Seeding scenario dramatically reduced cheatgrass levels compared to both the Minimum and Fuel Break Only scenario (average acres =270). The cheatgrass reduction corresponds with large increases in the early and mid-seral seeded classes in the Fuel Break+Seeding. Early seral seeded classes are characterized by perennial grasses, but as the class matures to mid- and late seral classes more woody species are incorporated in the vegetation structure.

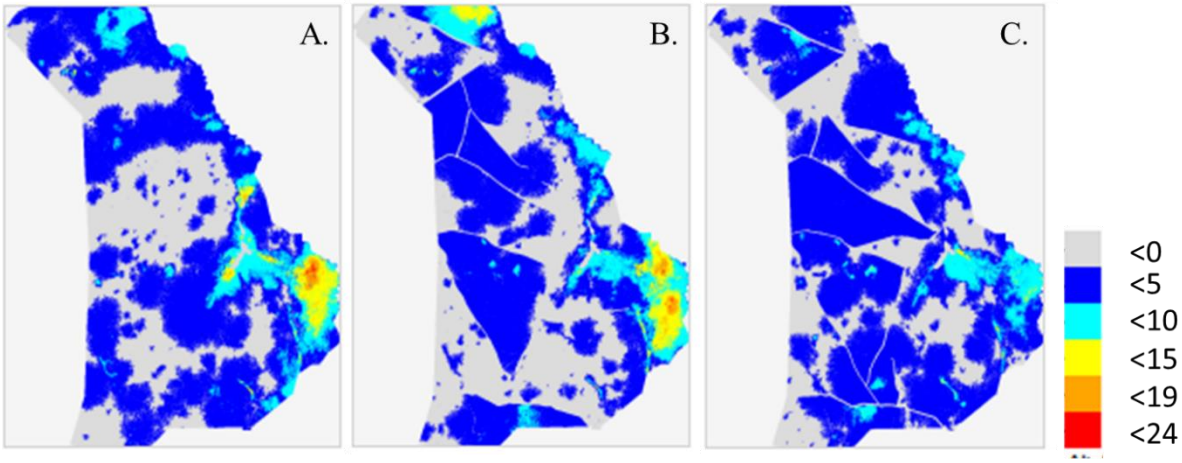

Figure A5. Fire frequency maps for the 3 simulated scenarios: A) Minimum, B) Fuel Break Only, and C) Fuel Break+Seeding. Colors indicate the number of times a pixel burned across all time steps and replicates.

Among the scenarios, net primary production (NPP) and net ecosystem production (NEP) did not differ. NPP represents the net carbon uptake minus carbon lost due to plant respiration. NEP equals the NPP of the system minus the loss of carbon due to decomposition. As there was not major conversion in land use (e.g. change from agricultural use to urban development), these results were expected. Both NPP and NEP showed increased carbon flow over the course of the simulation indicating an increasing ability of the system to store carbon. There were observed differences among the scenarios for net biome production (NBP). NBP measures the net carbon flux after losses due to disturbances are accounted. Increases in NBP are thought to increase long-term storage of carbon. The results mirrored the fire activity results, with Fuel Break+Seeding having the highest carbon uptake rate in the last ten years of the simulation (average of 43,320,423 lbs.), followed by Fuel Break Only (41,553,911 lbs. of carbon) and Minimum scenarios (39,664,630 lbs. of carbon). As expected, the majority of the carbon transferred from the terrestrial stocks to the atmosphere was due to fire in the landscape. Other emissions sources were livestock grazing and grazing by wild horses. Grazing emissions did not vary among the scenarios, and the carbon lost due to grazing was at least an order of magnitude lower than carbon emitted from fire. In low to moderate fire years, carbon uptake is expected to occur for NBP. However, models indicated that large fire years may cause the system to shift from a carbon sink to a carbon source. Across all years and iterations, the Minimum had 40 events where NBP was a carbon source (i.e. more carbon transferred to the atmosphere than stored in the terrestrial stocks), while Fuel Break Only had 39 and Fuel Break+Seeding had 34 events.

Our results suggest that reducing fire and establishing perennial species in areas previously dominated by an invasive annual species may be viable strategy to increase carbon storage in NACD. In fact, some of the benefit of the treatment is underrepresented as the carbon flow model designated both annual species monocultures and perennial grass sites as the “grassland” land use category. The

limited classification system was necessary has data are not available to finer model carbon fluxes for the mapped area. This coarse scale model ignores the inherent carbon dynamic differences between those two vegetation communities, such as: quicker carbon cycling and greater litter output of annual species and deeper, more extensive root system in the perennial species (Meyer 2012, Bradley et al. 2006). In addition to increased carbon sequestration, the management actions we modeled have other ecosystem benefits, including improved wildlife habitat, forage for livestock, and soil stabilization. These congruent benefits are unique among other areas of North America, where management for economic interests and carbon storage opportunities tend to differ.

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