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# Vegetative response to water availability on the San Carlos Apache Reservation

Roy Petrakis *, Zhuoting Wu, Jason McVay, Barry Middleton, Dennis Dye, John Vogel

*U.S. Geological Survey, Western Geographic Science Center, 2255 North Gemini Drive, Flagstaff, AZ 86001, USA*

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ABSTRACT

On the San Carlos Apache Reservation in east-central Arizona, U.S.A., vegetation types such as ponderosa pine forests, pinyon-juniper woodlands, and grasslands have significant ecological, cultural, and economic value for the Tribe. This value extends beyond the tribal lands and across the Western United States. Vegetation across the Southwestern United States is susceptible to drought conditions and fluctuating water availability. Remotely sensed vegetation indices can be used to measure and monitor spatial and temporal vegetative response to fluctuating water availability conditions. We used the Moderate Resolution Imaging Spectroradiometer (MODIS)-derived Modified Soil Adjusted Vegetation Index II ($MSAVI_2$) to measure the condition of three dominant vegetation types (ponderosa pine forest, woodland, and grassland) in response to two fluctuating environmental variables: precipitation and the Standardized Precipitation Evapotranspiration Index (SPEI). The study period covered 2002 through 2014 and focused on a region within the San Carlos Apache Reservation. We determined that grassland and woodland had a similar moderate to strong, year-round, positive relationship with precipitation as well as with summer SPEI. This suggests that these vegetation types respond negatively to drought conditions and are more susceptible to initial precipitation deficits. Ponderosa pine forest had a comparatively weaker relationship with monthly precipitation and summer SPEI, indicating that it is more buffered against short-term drought conditions. This research highlights the response of multiple, dominant vegetation types to seasonal and inter-annual water availability. This research demonstrates that multi-temporal remote sensing imagery can be an effective tool for the large scale detection of vegetation response to adverse impacts from climate change and support potential management practices such as increased monitoring and management of drought-affected areas. Different vegetation types displayed various responses to water availability, further highlighting the need for individual management plans for forest and woodland, especially considering the projected drier conditions in the Southwest U.S. and other arid or semi-arid regions around the world.



## 1. Introduction

Arid or semi-arid lands, which experience more surface moisture output through evaporation than input through precipitation, cover approximately 30% of the world's land area (Hochella et al., 2012; Huang et al., 2008; Thornbrugh, 2007). In the Western United States (U.S.), these lands extend from the lower valleys to the higher elevation forests. Over the past several decades, much research has focused on the complex interactions between semi-arid forests and climate variability in the Southwestern U.S. (Allen et al., 2010; Dickinson et al., 1989; Hurteau et al., 2014; Savage et al., 1996; Westerling et al., 2006). Climate and its effects on vegetation across various temporal scales and climate gradients is particularly well documented in the Southwest U.S., through various methods such as dendrochronology, physiological measurements, and image assessment (Allen and Breshears, 1998; Fritts et al., 1965; Ogle et al., 2000).

Forests of the Southwestern U.S. provide immense ecological, social, and economic value to the region including diverse habitat for wildlife (Huffman et al., 2009; Kalies et al., 2010), delayed headwater snowmelt for downstream population centers (Sankey et al., 2015), and an available timber supply (Sánchez Meador et al., 2015). On the San Carlos Apache Reservation in Arizona, forest types range from mixed-conifer and ponderosa pine ecosystems at higher elevations to mixed pinyon-juniper woodlands at the lower elevations (Wu et al., 2016). Timber is a key economic resource in an otherwise limited natural resource economy on the San Carlos Apache Reservation (Tuttle, 2008).

* Corresponding author.
*E-mail addresses:* rpetrakis@usgs.gov (R. Petrakis), zwu@usgs.gov (Z. Wu), jlm683@nau.edu (J. McVay), bmiddleton@usgs.gov (B. Middleton), ddye@usgs.gov (D. Dye), jvogel@usgs.gov (J. Vogel).

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Forests of the Southwest are resilient ecosystems that can function within a range of environmental variables (Buma and Wessman, 2013), including shifting temperature and precipitation regimes. However, these forests can be negatively impacted by extreme temperatures and drought, particularly over extended periods of time (Breshears et al., 2005; Williams et al., 2013). Tree ring records have proven drought to be a major limiting factor for tree growth (Adams and Kolb, 2004; Swetnam and Betancourt, 1998) and a driver of mortality (Breshears et al., 2008; McDowell et al., 2008) in the Southwestern U.S. The response by different vegetation types to large-scale drought can vary. Understanding how each vegetation type responds to changing moisture availability is essential for effective forest management on the San Carlos Apache Reservation and across the West.

Ponderosa pine (*Pinus ponderosa*) forests in this region, and across the Western U.S., have been negatively impacted by the practice of fire suppression beginning in the 20th century (Covington and Moore, 1994). The suppression of low intensity fires, a naturally occurring agent of change, has led to dense, overcrowded, and potentially vulnerable ponderosa pine forests (Pollet and Omi, 2002). This dense condition makes forests particularly susceptible to a variety of threats including large-scale forest fires (Dore et al., 2010; Haire and McGarigal, 2010; Wu et al., 2015), insect outbreaks (Negron et al., 2009; Wallin et al., 2008), and drought (Hoffmann et al., 2011) – all of which are further complicated by the effects of climate change (Bonan, 2008; Ganey and Vojta, 2011). Climate change will bring increased temperatures and disrupt precipitation patterns across the region, leading to longer and more frequent droughts (IPCC, 2014).

Pinyon-juniper woodlands are composed of co-dominant pinyon pine (*Pinus edulis*) and juniper (*Juniperus* sp.) trees and typically occupy the space between higher elevation ponderosa pine forests and lowland shrubs and grasslands. Unlike ponderosa pine forests, which have a clear economic timber value, pinyon-juniper woodlands have less direct economic value. Major uses include cutting for firewood, traditional tribal uses, and the collection of pine nuts from pinyon pines in the fall (Tuttle, 2008). Otherwise, these woodlands go largely unmanaged. Following a severe drought in New Mexico, Breshears et al. (2008) documented extensive mortality of pinyon pine while juniper (*Juniperus monosperma*) survived. Due to fire suppression, grazing, and other factors, pinyon-juniper woodlands have expanded into former shrub and grassland territory (Baker and Shinneman, 2004), and though they may serve as a valuable carbon sink (Huang et al., 2009), less is known about how to effectively manage woodlands during extended drought conditions.

Within the Southwestern U.S., grasslands are generally found within the valleys and lower elevations. They are dominated primarily by C4 grass species, which form 4 carbon compounds during photosynthesis and grow during the warm season (Edwards et al., 2010). However, some C3 grass species, which form 3 carbon compounds and prefer cooler and wetter environments (Edwards et al., 2010), as well as other annual herbaceous plant types can be found at the higher elevations and coexist with pinyon-juniper woodlands and ponderosa pines. The C4 grasses and annual herbaceous plants also extend into the pinyon-juniper woodlands and provide extensive ground cover throughout the woodlands. Tuttle (2008) found that grasslands can provide economic value to the San Carlos Apache Reservation through ranching and grazing purposes. Additionally, they are shown to be an indicator of early drought conditions in the region (Wu et al., 2016). Due to their ecological and economic value, grasslands are essential for consideration within overall land management decisions and are not commonly compared to forest vegetation types.

Remote sensing provides a unique opportunity to measure the response of different forest types to environmental variables including drought. In western North America, satellite imagery has been used to quantify the scale of forest insect outbreaks (Dennison et al., 2010; Meddens and Hicke, 2014; Walter and Platt, 2013), and die-off events (Allen et al., 2010; Anderegg et al., 2013;) associated with persistent drought across the region. The Normalized Difference Vegetation Index (NDVI) is one of the most common vegetation indices used to measure vegetation growth (Anyamba and Tucker, 2005; Tucker, 1979; Yengoh et al., 2015). NDVI is less reliable in arid and semi-arid environments because the background reflectance of exposed rock and soil can distort the vegetation signal (Lu et al., 2015). In these regions, the Modified Soil Adjusted Vegetation Index II ($MSAVI_2$) is a more reliable measure of vegetation conditions because it accounts for background soil brightness through the soil brightness correction factor based on spectral reflectance and the soil line slope (Qi et al., 1994). This correction factor is included within the equation. $MSAVI_2$ has been successfully used to measure vegetation conditions (Gonsamo and Chen, 2014; Heiskanen, 2006; Mariotto and Gutschick, 2010), and has been paired with precipitation data to measure the temporal response of vegetation to drought in Arizona (Wu et al., 2016).

Detecting drought at a landscape scale can be difficult due to the subtle nature and lagged effects of drought onset; therefore, multiple drought indices exist. The Standardized Precipitation Evapotranspiration Index (SPEI) is a multi-scalar drought index that incorporates both sides of the climate water balance equation, precipitation (input) and potential evapotranspiration (output), and is based on a water balance model (Vicente-Serrano et al., 2010). SPEI is highly regarded as an indicator of drought conditions and has been used in multiple studies to suggest or correlate with vegetative response across various periods of time (Dorman et al., 2013; Huang et al., 2015; Vicente-Serrano et al., 2013, 2014). Using SPEI can detect unique aspects of vegetative response; not only to input water availability and precipitation, but rather the full estimated climate water availability. Additionally, SPEI indirectly incorporates temperature, another limiting environmental variable driving vegetation response, through the calculation of potential evapotranspiration.

The objective of this research was to better understand how different vegetation types respond to fluctuating water availability in the arid and semi-arid Southwestern U.S., and help agencies and land owners better manage these ecosystems. We compared remotely sensed climate variables to $MSAVI_2$ for three dominant vegetation types in order to highlight the relationship between vegetative conditions and water availability fluctuations. This analysis also emphasized the dynamic properties of each vegetation type within the larger ecosystem. We then quantified the capacity of each vegetation type to respond to fluctuating seasonal water availability. This analysis provided an increased understanding of the similarities and differences of vegetation responses among vegetation types as well as provided important conclusions on the seasonal response traits for each vegetation type which can help land managers and researchers better monitor bi-modal vegetation response and potentially react accordingly.

## 2. Methods

### 2.1. Study area

#### 2.1.1. Vegetation cover

The study area is located in the northern Forest Management Units (FMUs) region of the San Carlos Apache Reservation in east-central Arizona (Fig. 1) (33°49′5″N, −110°35′53″W to 33°0′45″N, −109°29′45″W). This region covers an area of 3858 $km^2$ with surface elevations ranging from 980 m to 2459 m.

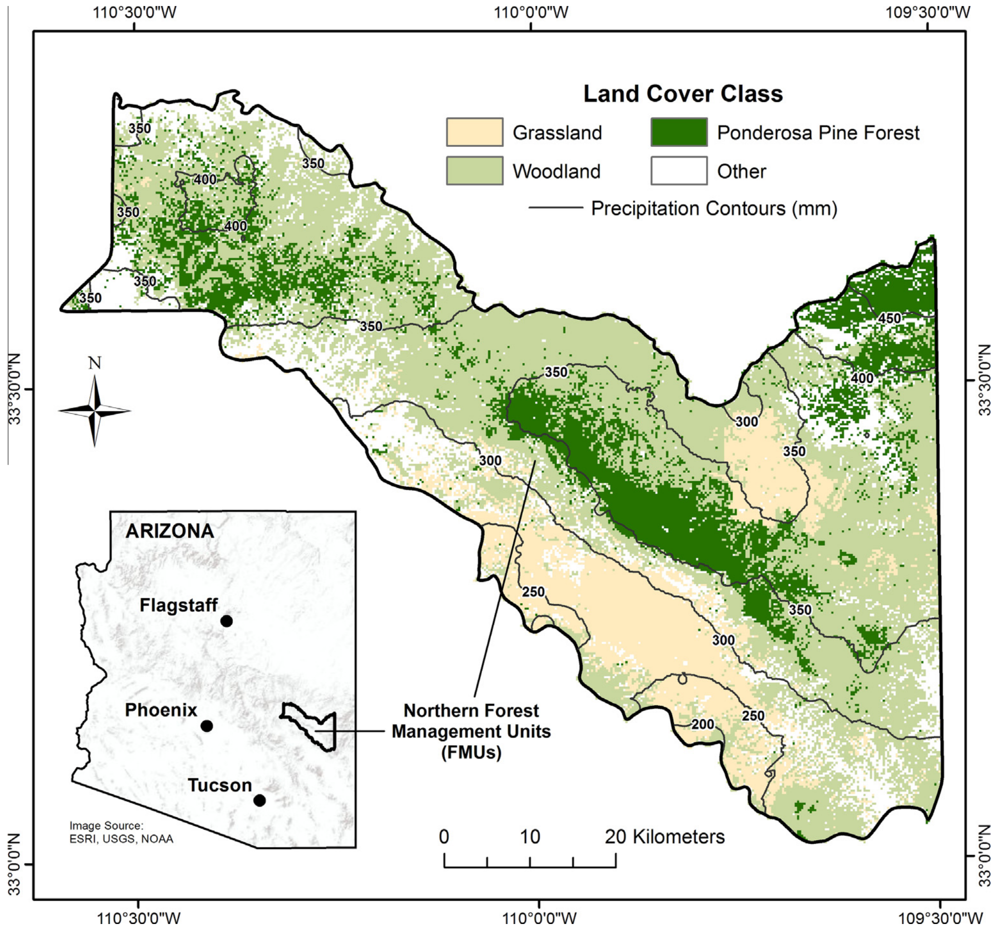


**Fig. 1.** Study area map highlighting the land cover types for the northern portion of the San Carlos Apache Reservation. The ponderosa pine forest land cover type is hereby referred to as forest. Annual total precipitation contours (mm) from 2013 represent average yearly conditions.

Vegetation types vary across the elevation gradient, with grasslands and mixed desert scrub found in the lower elevations and ponderosa pine (*P. ponderosa*) forests mixed with occasional Gambel oak (*Quercus gambelii*) found in the higher elevations. Many of the native C4 grasses found within the study area belong to the grama grass (*Bouteloua*) family and include black grama (*Bouteloua eriopoda*), blue grama (*Bouteloua gracilis*), sideoats grama (*Bouteloua curtipendula*), and slender grama (*Bouteloua repens*) among others. Another dominant native C4 grass is tobosa (*Pleuraphis mutica*). A native C3 species commonly found within the study area is needlegrass (*Stipa* spp.). Woodlands consisting of Arizona white oak (*Quercus arizonica*) and other less common oak species, juniper (*juniperus* sp.), pinyon pine (*P. edulis*), and many mixed grasses are found bordering the ponderosa pine in the mid-elevations (Fig. 1). Areas of desert scrub, ecotones, or indiscriminate vegetation composition were not considered in this study and are mapped as "Other" in Fig. 1.

### 2.1.2. Climate

This region experiences a bi-modal precipitation regime (Fig. 2) (Jacobs et al., 2005). At higher elevations, snow is common during the winter from low pressure weather systems generated from the Pacific Ocean (Sheppard et al., 2002). A warm, dry spring gives way to the North American monsoon season which runs from mid-June through September (Adams and Comrie, 1997). The elevation gradient can also greatly influence the amount and intensity of precipitation, with more typically falling at higher elevations.

## 2.2. Remote sensing data

### 2.2.1. MODIS imagery

We used Moderate Resolution Imaging Spectroradiometer (MODIS) surface reflectance 8-day composite imagery at a 250 m spatial resolution from 2002 through 2014, a period of 13 years. $MSAVI_2$, an index used for vegetation condition or drought studies (Weber and Dunno, 2001; Wu et al., 2016), was calculated from the MODIS imagery. The $MSAVI_2$ better accounts for soil effects, which improves signal-to-noise ratio (Qi et al., 1994). The $MSAVI_2$ was computed as follows:

$$MSAVI_2 = \frac{2 * NIR + 1 - \sqrt{(2 * NIR + 1)^2 - 8 * (NIR - RED)}}{2},$$

where NIR is the Near Infrared band (band 2) and RED is the Red band (band 1) of the MODIS imagery.

### 2.2.2. Southwest Regional GAP land cover

We used the Southwest Regional GAP Analysis Program (SW ReGAP) land cover map (http://gapanalysis.usgs.gov/gaplandcover/data/download/) for Arizona to generate three broad vegetation types. The U.S. Geological Survey SW ReGAP data was published in 2004 and was based on the need for a multi-level, yet consistent classification scheme that covers a five-state region (Arizona, Colorado, Nevada, New Mexico, and Utah) (Lowry et al., 2007). The classification used a combination of 30 m Landsat ETM+ imagery, ground-based field work, supplementary field data,

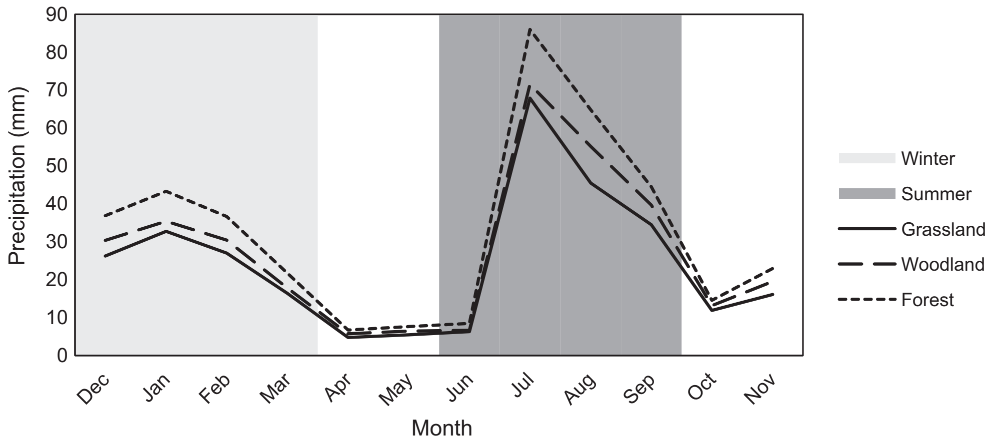


**Fig. 2.** Average monthly precipitation within the study area from 2002 to 2014 for each vegetation type. The two dominant seasons, winter (December through March) and summer (June through September), are highlighted and show the bi-modal precipitation regime present in the study area.

and decision tree modeling (Lowry et al., 2007). With a base spatial resolution of 30 m, we rescaled the classification to 250 m in order to match the spatial resolution of the $MSAVI_2$ imagery.

We used the Ecological Systems Land Use attribute level to define our land cover classes. Initially, 25 unique classes were present. Considering on-the-ground knowledge and vegetation characteristics, we grouped these land cover classes into three dominant vegetation types by combining smaller classes as well as removing mixed or minor classes and ecotones from the classification. The three dominant vegetation types were grassland, woodland, and ponderosa pine forest (hereby referred to as forest) (Fig. 1). Woodland, which is commonly considered a forest type, was assessed separately in this study.

### 2.2.3. PRISM climate data

Long-term and high frequency precipitation data is necessary in order to evaluate the vegetative response to water availability. Because there are very few weather stations located within the study area, we used remotely sensed, gridded precipitation data from the Parameter-elevation Relationships on Independent Slopes Model (PRISM) Climate Group at Oregon State University (http://prism.oregonstate.edu). Within a linear regression function, PRISM uses precipitation and temperature data from surface climate stations, precipitation data acquired from radar, and elevation gradients to interpolate and estimate daily precipitation and temperature (Daly et al., 2002, 2008). PRISM is measured in millimeters and has a 4 km spatial resolution. We downloaded and transformed PRISM daily precipitation data into an 8-day total composite to match the $MSAVI_2$ temporal intervals. Precipitation composites were then summed by month and grouped into summer and winter to match the bi-modal precipitation regime (Fig. 2).

### 2.2.4. Standardized Precipitation Evapotranspiration Index (SPEI)

We used the SPEI to better understand the variability of climate water balance (Beguería et al., 2014; Vicente-Serrano et al., 2010) within the study area, and the resulting impact on the vegetation. Climate water balance is calculated by subtracting potential evapotranspiration (PET) from precipitation (P) for a certain period of time, thus quantifying the amount of water available within the system. Precipitation is the water input and PET is the expected water output under conditions of non-limiting water availability. When quantified, the result shows either water surplus (positive) or deficit (negative) within the system at a specific time (Vicente-Serrano et al., 2010).

SPEI is a multi-scalar index calculated across N (N = 1, 2, 3 ... 72) months, previous to the month at question. The accumulated difference in climate water balance for N months is compared through a probability distribution for that same period, historically. Ultimately, a standardized water availability value for the period is calculated, allowing for comparison to previous periods of time (Vicente-Serrano et al., 2010). A value of 0 indicates average water conditions for the specific month. A positive SPEI value represents an above average period while a negative SPEI value represents a below average period, or drought. We used a six-month time scale for SPEI to align with the bi-modal precipitation regime present in the study area (Fig. 2). The SPEI imagery was downloaded from the West Wide Drought Tracker (http://www.wrcc.dri.edu/wwdt/about.html), which uses PRISM precipitation data for P and temperature data to estimate PET through the Penman-Monteith algorithm.

## 2.3. Analysis approach

### 2.3.1. Random sampled points and time series construction

A total of 30 sample points were randomly selected for each of the vegetation types in order to extract the remotely sensed environmental data. The points were selected using the 250 m land cover classification. The vegetation type at each sample point was validated using the 30 m SW ReGAP data to ensure it represented the focal vegetation type within the 250 m pixel size of $MSAVI_2$. Because vegetation communities undergoing regeneration following disturbance by fire may exhibit responses to climate variability that differ from the undisturbed vegetation type, we excluded fire-disturbance areas from our analysis based on the Monitoring Trends in Burn Severity (http://www.mtbs.gov/) fire perimeter data.

The points for each vegetation type were used to extract values for $MSAVI_2$, PRISM precipitation, and SPEI for their respective vegetation type. The values for all 30 points of each vegetation type were then averaged to produce a mean time series for each of the environmental variables and the $MSAVI_2$ signature. To reduce outliers associated with contamination of the MODIS pixel data by clouds, snow, or other spectral effects (Teillet et al., 1997), we applied temporal smoothing to the 8-day composite time-series of $MSAVI_2$ in a two-step procedure. First, extreme numeric outliers were averaged with the previous and following dates. Second, a seven-date moving average was then applied to smooth the full time-series and highlight the general trend. This was completed only for the 8-day composite time-series, before the data were either averaged or summed by month.

### 2.3.2. Statistical analysis

A two-step analysis approach was applied to determine the magnitude and timing of the response of the three vegetation types to water availability over the 13-year period of MODIS observations. This analysis approach was completed using environmental data that was specific to each vegetation type. First, we determined the relationship between precipitation and the $MSAVI_2$ values derived from the three vegetation types by using a Pearson correlation on a monthly time scale with a lag interval of 0–4 months. Many studies indicate that a lag exists between the timing of precipitation and vegetation indices within the Western U.S. and Great Plains (Ji and Peters, 2003; Wang et al., 2003; Wu et al., 2016). Because of this, we included lag periods extending up to four months for our correlation analysis with the summed monthly precipitation.

The second step was designed to examine the impacts of a bimodal precipitation regime on the various vegetation types. The two environmental variables, precipitation and SPEI, were quantified by season and stratified to show the two dominant precipitation seasons - summer and winter. These relationships were further stratified by vegetation type. For precipitation, the seasonal anomaly percentages of $MSAVI_2$ were compared to the total precipitation for all winter and summer seasons within the study time frame. This comparison was based on the most optimal time lag determined in the correlation analysis. The 2009 winter season was not included for precipitation because of a single extreme precipitation event (8-day total: 135 mm ~40% of yearly average precipitation) which had an irregular impact on $MSAVI_2$ for all three vegetation types. Forest $MSAVI_2$ dropped considerably, likely due to large amounts of snow cover, while woodland and grassland were not significantly impacted due to the short timeframe of the event. For SPEI, the seasonal anomaly percentages of $MSAVI_2$ were also compared to the average SPEI for all winter and summer seasons within the study time frame. SPEI was averaged across the four-month season in order to show average conditions throughout the full season. We used these relationships to determine the multi-seasonal climate pivot point for precipitation as well as SPEI relationships and slope for each vegetation type (Munson, 2013; Scott et al., 2015). In the context of this analysis, the pivot point highlights the value of an environmental variable, specific to the study period, at which shifts in increasing or decreasing plant growth occur, indicating potential ecosystem change (Gremer et al., 2015). Simply, this analysis quantifies the value of each environmental variable at which vegetation growth is sustained. This information can highlight water availability stresses and potential future changes of vegetation in response to climate change (Munson, 2013). The pivot point does not necessarily show long term biophysical significance, but rather a comparative analysis of study period conditions.

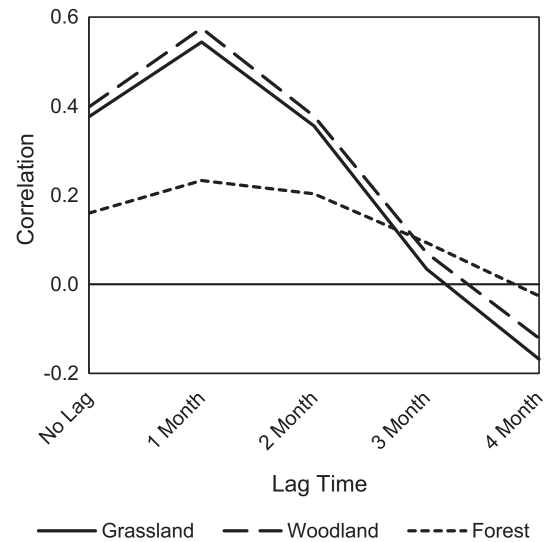


**Fig. 3.** Correlation coefficients between monthly $MSAVI_2$ and precipitation, with multiple temporal lags (no lag to a four-month lag).

## 3. Results

### 3.1. Vegetation response to precipitation variability

The strongest correlation between summed monthly precipitation and average monthly $MSAVI_2$ for all three vegetation types was at a one-month lag (Fig. 3). At a one-month lag, grassland (correlation = 0.54; $p < 0.001$) and woodland (0.57; $p < 0.001$) had the strongest positive correlations, indicating a moderately strong relationship with general precipitation for these vegetation types. Forest $MSAVI_2$ had a weaker positive correlation of 0.23 ($p = 0.004$) at one month with precipitation (Fig. 3). Although statistically significant, this relatively weak correlation suggests a limited response of forest vegetation to precipitation. However, the significance of this correlation does indicate that forests respond to precipitation when available.

Despite the high correlation with a one-month lag, each of the vegetation types experienced statistically significant correlations

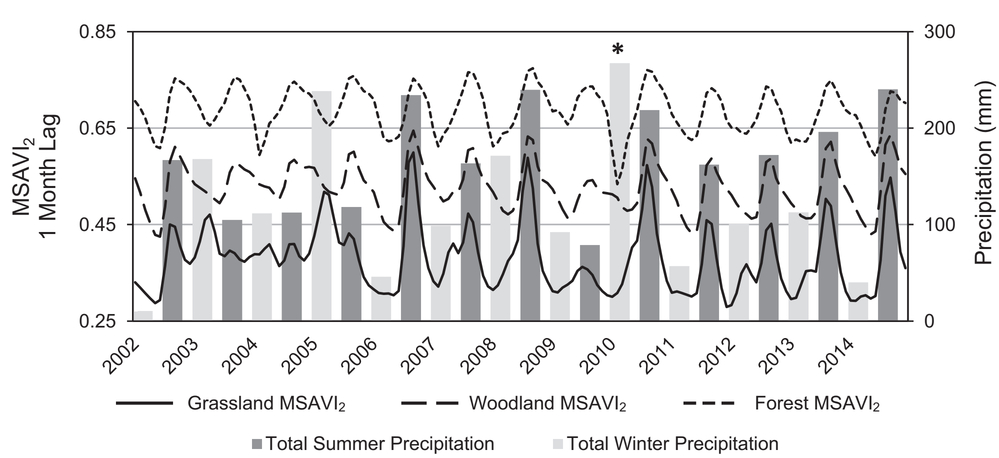


**Fig. 4.** $MSAVI_2$ time-series with one-month lag for each class overlaid above the winter and summer total seasonal precipitation. *The winter of 2009 was not included due to a large single event which occurred in January 2010.

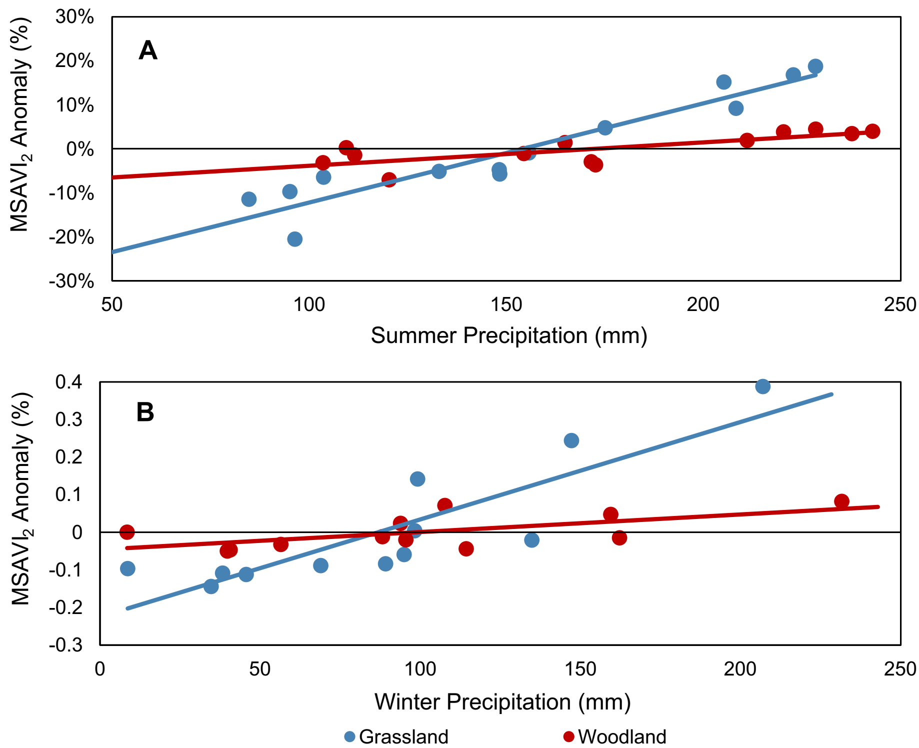


**Fig. 5.** Relationships between $MSAVI_2$ anomaly (%) with a one-month lag and total (A) summer and (B) winter seasonal precipitation for grassland and woodland.

with precipitation for up to two months after precipitation events. By three months, grassland (0.03; $p = 0.68$), woodland (0.07; $p = 0.40$), and forest (0.09; $p = 0.26$) all experienced no relationship with precipitation (Fig. 3). This suggests a longer term response, up to two months, of the full ecosystem to the presence of precipitation.

### 3.2. Seasonal precipitation pivot point

The strongest correlation between precipitation and $MSAVI_2$ for all three vegetation types was with a one-month lag (Fig. 3). Therefore, a one-month lag for $MSAVI_2$ was included in both the seasonal vegetative response time series with precipitation (Fig. 4) as well as the seasonal precipitation pivot point analysis (Fig. 5).

Each of the vegetation types showed distinct responses to precipitation across the two seasons (Fig. 4). Forest reached a more consistent $MSAVI_2$ maximum during the summer despite variability in the total amount of precipitation (Fig. 4). The $MSAVI_2$ signature for grassland corresponded more directly with the total amount of seasonal precipitation (Fig. 4). For grassland, the strongest relationship occurred during the summer (Table 1), signifying a very strong influence of summer monsoonal precipitation on grassland vegetation. This was expected due to the response of C4 grasses to warmer temperatures and summer precipitation. The pivot point for summer was 152.2 mm while the average grassland specific summer precipitation was 154.2 mm. During winter, the relationship was also very strong (Table 1). The strength of this relationship was likely a result of the presence of annual herbaceous plants and limited C3 grasses, which are known to respond to winter precipitation. The pivot point for winter was 84.6 mm while the average winter precipitation for grassland was 88.9 mm. The strong winter and summer relationships imply an overall dependence of grassland vegetation and other annual herbaceous plant types within grasslands on existing bi-modal precipitation patterns.

**Table 1**
Relationship between $MSAVI_2$ and summer and winter precipitation; pivot point and average precipitation for each vegetation type.

| | $R^2$ | *p*-value | Equation | Pivot point (mm) | Average precipitation (mm) |
|---|---|---|---|---|---|
| *Summer* | | | | | |
| Grassland | 0.90 | <0.001 | $y = 0.0023x - 0.35$ | 152.2 | 154.2 |
| Woodland | 0.58 | 0.002 | $y = 0.0005x - 0.09$ | 180 | 172.9 |
| Forest | 0.02 | 0.67 | $y = -6e^{-5}x + 0.01$ | N/A | 203.8 |
| *Winter* | | | | | |
| Grassland | 0.75 | <0.001 | $y = 0.0026x - 0.22$ | 84.6 | 88.9 |
| Woodland | 0.40 | 0.03 | $y = 0.0005x - 0.05$ | 100 | 99.8 |
| Forest | 0.00 | 0.99 | $y = -2e^{-6}x + 0.01$ | N/A | 123.2 |

The strongest relationship for woodland occurred during the summer (Table 1). For winter, the relationship was also moderate and statistically significant (Table 1). Average winter precipitation was nearly the same as the precipitation pivot point value, resulting in near average $MSAVI_2$ (Table 1). The summer and winter precipitation to $MSAVI_2$ relationships had no direct relationship for the forest class and were not assessed or compared (Table 1).

Correlation coefficients indicate that summer monsoon precipitation was more important than winter precipitation to maintain vegetation growth for grassland and woodland (Table 1). The difference between woodland and grassland pivot points for summer (27.8 mm) was greater than that for winter (15.4 mm). This indicates that the summer monsoon was potentially more critical to maintain woodland growth than it was for grassland.

### 3.3. Seasonal SPEI vegetative response

Throughout the study period, SPEI was negative for 111 of the 156 months (71%) (Fig. 6). Because SPEI quantifies climate water balance, this indicates consistent periods of lack of water availability. Predominantly negative periods occurred from 2002–2005 and

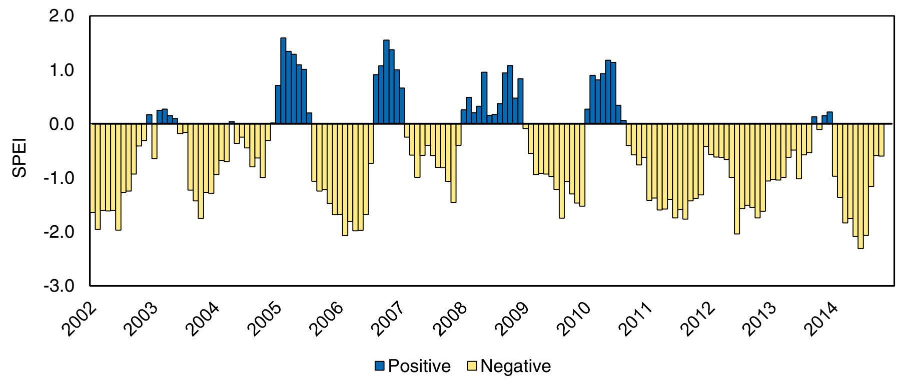


**Fig. 6.** Time-series for SPEI for the study period. Positive SPEI values are shown in blue while negative SPEI values are shown in yellow. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

**Table 2**
Relationships between $MSAVI_2$ anomaly (%) with average summer and winter seasonal SPEI for each vegetation type.

| | $R^2$ | *p*-value | Slope | Pivot point |
|---|---|---|---|---|
| *Summer* | | | | |
| Grassland | 0.60 | 0.002 | 0.11 | −0.76 |
| Woodland | 0.69 | <0.001 | 0.04 | −0.78 |
| Forest | 0.38 | 0.02 | 0.03 | −0.73 |
| *Winter* | | | | |
| Grassland | 0.28 | 0.06 | 0.09 | −0.58 |
| Woodland | 0.29 | 0.06 | 0.02 | −0.56 |
| Forest | 0.02 | 0.63 | N/A | N/A |

2011–2014 (Fig. 6). This resulted in negative SPEI pivot points, which are representative of the study time period and not necessarily long-term trends. Therefore, these results should be considered in comparison by vegetation type as well as across seasons.

For grassland, the summer relationship between $MSAVI_2$ and SPEI was statistically significant (Table 2). For woodland, the relationship between $MSAVI_2$ and SPEI for summer was also statistically significant, and stronger (Table 2). The relationship between $MSAVI_2$ and SPEI for forest was also statistically significant for summer (Table 2). Furthermore, woodland and forest had similar slopes while grassland had a slope nearly 3-times greater (Table 2). This difference in slope highlights the impact of water availability on grassland and its response to both positive and negative conditions. The pivot point for woodland was also slightly more negative, which signifies it may be more resilient to drought conditions (Table 2).

For winter, none of the vegetation types had statistically significant relationships between seasonal $MSAVI_2$ and SPEI (Table 2), indicating an inconsistent response of average winter $MSAVI_2$ to fluctuating winter climate water availability.

## 4. Discussion

### 4.1. Vegetation response summary

We showed that precipitation and short-term water availability, through monthly and seasonal precipitation as well as SPEI, impacted the vegetative responses of each of the three vegetation types in ways that were unique. This includes affecting seasonal patterns as well as influencing the rate of vegetation anomalies in response to levels of water availability. However, it is unlikely that these were the only determining variables driving vegetation response. The timing, intensity, and frequency of precipitation can impact vegetation growth (Weltzin et al., 2003; Yaseef et al., 2010). Temperature can also impact vegetation response (Wang et al., 2011). Increasing temperature is known to be linked to increased risk for disturbances in forests, such as wildfire, bark beetle infestation, or die-offs (Adams et al., 2009; Breshears et al., 2005; Westerling et al., 2006; Williams et al., 2013). Although the timing, intensity, and frequency of precipitation as well as temperature were not directly included in this study, they were indirectly assessed within seasonal precipitation as well as SPEI through potential evapotranspiration.

The response of the grassland vegetation was strong with nearly all of the environmental metrics used in this research, except for winter SPEI. Although this highlights the dependence of grasslands on both bi-modal precipitation and summer climate water availability, the C4 grass species likely only responded strongly during the summer. The strong response to precipitation during the winter was likely a result of other annual plant species found within the grassland region. This was expected due to the characteristics of the grassland vegetation present within the study area. C4 grasses, which make up a majority of the grassland class, respond to sufficient warm-season precipitation and soil moisture (Collatz et al., 1998), conditions present during the summer monsoon (June-September), while the limited C3 grasses and more dominant annuals present within the grasslands at the higher elevations respond to cooler, winter precipitation. The non-significant relationship to winter SPEI indicates that the limited C3 grasses and more dominant annuals did not respond consistently to fluctuating SPEI conditions. Similar results were found in Wu et al. (2016), across the entire San Carlos Apache Reservation, in which grassland/shrubland mixed vegetation was shown to be highly sensitive to precipitation. That study concluded that this vegetation type is an indicator of drought conditions because of its vegetative response (Wu et al., 2016). We found grasslands to be potentially more resistant to drought due to lower precipitation pivot points than both forest and woodland. However, the steeper correlation slopes indicated that drought conditions would have a more immediate and stronger negative impact on grassland vegetative conditions. This is consistent with other studies of Southwestern grasslands (Pennington and Collins, 2007; Wu et al., 2016). Additionally, because grassland receives the least amount of yearly precipitation (Fig. 2), any precipitation deficit may have a stronger influence on its general condition than the same amount for the other vegetation types.

Similarly, woodland had statistically significant relationships for all variables except for winter SPEI, highlighting a strong response to precipitation and climate water availability. Due to the elevation and precipitation characteristics of the woodland class, we expected woodlands to have a more moderate response to the climate variables, compared to both grassland and forest.

However, woodland responded similarly to grassland vegetation. We believe these results suggest a shift is needed in both the assessment and management of woodland vegetation within larger forest ecosystems. Wu et al. (2016) found a stronger relationship between woodland and forest. This is potentially due to the scaling of our study area in which woodlands could be directly analyzed in contrast to forest within the northern FMUs. In addition, our methodological approach treated the relationship with water availability differently, as a response to both precipitation total and climate water availability.

Multiple factors were potentially driving this response of woodlands. First, soil types may have played a role. The soil present in the SW ReGAP class is described as dry and rocky. This soil will not hold as much soil water content, therefore increasing the importance of precipitation, such as from the monsoons. Second, the vegetation composition of woodlands includes a mixture of low shrubs, grasses, and deciduous oaks among pinyon-juniper trees. This vegetation is proven to be susceptible to drought, as is pinyon pine (Breshears et al., 2008). With grassland species and annual herbaceous plant types as a large component of the woodland ground cover composition, the $MSAVI_2$ signature was likely partially influenced by these herbaceous plant types. Our results highlighted this response to both precipitation and SPEI, which can signify drought. However, woodlands demonstrated the ability to respond positively to the presence of bi-modal precipitation, if at or above average for the season. That diverged from forest, which had no direct relationship, and aligned more with grassland vegetation. Again, this was partially a response of the vegetation composition which includes large amounts of grasses and annual herbaceous vegetation types.

Ponderosa pine forest had weaker correlations to both precipitation and SPEI, suggesting a much weaker relationship with estimated water availability. Previous studies have shown that aridity within forest ecosystems can result in die-offs or increased risk for disturbance (Allen et al., 2010; Anderegg et al., 2013; Williams et al., 2010). This may indicate that although forest vegetation is vulnerable to aridity and drought, its direct response to estimated water availability may not be as pronounced on a yearly basis. This is likely due to a deeper root system which allows for access to moisture that grassland and woodland species cannot reach during drought conditions (Wu et al., 2016). Forests are likely more buffered against changes in precipitation because they are less dependent on overall short-term precipitation. Forest vegetation, which is generally found in the higher elevations, also receives more yearly precipitation (Fig. 2) and may have experienced less extreme climate fluctuations due to localized precipitation. Finally, due to the presence of year-round foliage of forest evergreen vegetation types, such as ponderosa pine, seasonal $MSAVI_2$ shifts are reduced and immediate responses to climate fluctuations may be less established. The $MSAVI_2$ signature for pinyon-juniper woodlands is less susceptible to this issue due to reduced tree densities and the greater influence from understory grasses, herbaceous plants, and deciduous oak trees.

### 4.2. Projected climate change

Climate models have identified the Southwestern U.S. as an important climate change hot-spot (Diffenbaugh and Giorgi, 2012). We assessed 13 years of climate and precipitation fluctuations and their impact on three distinct vegetation types. Due to the relatively short study period, these results should be interpreted as a short-term response to climate variability and drought, rather than a response to long-term climate change. Landscape scale changes to vegetation can take place over decades or longer (Swetnam and Betancourt, 1998). This research presents a useful method for detecting early signs of vegetative stress by quantifying the ability of each vegetation type to respond to fluctuating periods of water availability.

Climate change is projected to bring an increase in annual temperatures, disrupt existing precipitation patterns, and bring extended drought conditions to the Southwest (Garfin et al., 2013). The extent of changes in annual precipitation is not yet well known, although decreases in total precipitation, especially winter snow, are expected (Dominguez et al., 2012) as well as an increase in the severity of individual precipitation events (Diffenbaugh and Giorgi, 2012). These changes in climate will have secondary effects of increased forest fire risk (Wu et al., 2015) as well as disease and insect outbreak (Negron et al., 2009; Wallin et al., 2008). Long-term impacts to vegetation types may differ from what our results have shown. Thus, the need for continuous monitoring of these vegetation types is vital.

### 4.3. Management and research implications

Ponderosa pine forests have been impacted by various anthropogenic activities including timber harvesting, thinning, and prescribed burning as well as changing management and restoration views over the past two centuries across the Southwestern U.S. and Arizona (Allen et al., 2002; Schubert, 1974). In an attempt to improve ponderosa pine forest conditions, recent large-scale forest thinning projects such as the Four Forest Restoration Initiative (4FRI) are currently underway on National Forests in Arizona (http://www.fs.usda.gov/4fri). In addition to restoring ponderosa pine forests to more natural conditions as well as reducing fire and disease risk, the timber harvest will have a positive economic impact on local economies. Because forests were shown to be more buffered against decreased water availability conditions, continued management and monitoring may be important simply due to the ecological and economic value of these forests. However, long term drought conditions have been shown to negatively impact forest conditions and heath. Therefore, longer term monitoring of forest condition trends, greater than the length of our study, may indicate a more direct relationship.

Management and restoration of woodlands is much less common. This is especially true on tribal reservation lands (IFMAT, 2013). Coupling our results with projected climate change, woodlands within the San Carlos Apache Reservation and across the Southwestern U.S. are expected to become more water stressed and at higher risk for disturbance. Increased monitoring and management of woodlands within drought affected areas is needed to reduce the risk for future disturbance and to protect the economic and social benefits provided by woodlands such as biofuels, livestock grazing, and traditional uses. Finally, separation of woodlands from the larger forest ecosystem in the assessment of vegetation condition and response can clarify potential threats and suggest beneficial management practices.

Additional potential forest management implications for the full ecosystem were drawn from this research. First, due to the biodiversity present within Southwestern forest ecosystems and the variability of each of the focus vegetation type's responses, it may be necessary to have unique management plans and practices for the various vegetation types present on the landscape. Second, increased forest restoration methods, such as forest thinning, prescribed fires, or other fuel treatments, may reduce the risk for large scale disturbance and help mitigate negative vegetation responses that result from climate fluctuations, although adaption to current climate change conditions must be considered (Agee and Skinner, 2005; Allen et al., 2002; Fulé et al., 1997; North et al., 2012; Spittlehouse and Stewart, 2004). Currently practiced mostly within ponderosa pine forests, expansion of this approach into the pinyon-juniper woodlands throughout the Western U.S. may reduce the impacts of climate change such as drought and vegetation die-offs.

Finally, future research should continue to use remote sensing and other techniques to monitor vegetation response to climate change at the landscape scale. This allows for large spatial scale monitoring as well as the potential for multi-decadal assessment, which can detect vegetation behaviors that are challenging to detect through on-the-ground methods. Additionally, applying remote sensing allows for a methodological approach that can include many types of remotely sensed environmental and vegetation condition data sets. Although limitations exist, such as temporal boundaries, vegetation species mixing within pixels, as well as required computer processing, remote sensing can provide large scale ecological response assessments to benefit the monitoring and management of forests and other vegetation ecosystems. Remote sensing also provides a proven methodological approach to monitor unique ecological cover types which are spatially proximate to each other.

Considered an arid or semi-arid landscape, a land climate designation given to approximately 30% of the world's land area, this study area is especially at risk to shifts in precipitation and temperature (Hochella et al., 2012; Huang et al., 2008; Thornbrugh, 2007). As the climate continues to fluctuate, similar regions around the world may begin to experience similar vegetative responses to the ones shown in this research. Therefore, increased vegetation monitoring and management in various regions around the world may be necessary to fully understand the potential impacts of fluctuating climate variables on arid and semi-arid landscapes.

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