Ecological Indicators 30 (2013) 35–44

Contents lists available at SciVerse ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Phenology-based, remote sensing of post-burn disturbance windows in rangelands

Joel B. Sankey[a,*,1], Cynthia S.A. Wallace[b], Sujith Ravi[c]

[a] U.S. Geological Survey, Western Geographic Science Center and USA-National Phenology Network, 2255 N Gemini Drive, Flagstaff, AZ 86001, USA
[b] U.S. Geological Survey, Western Geographic Science Center, 520 N Park Avenue, Tucson, AZ 85719, USA
[c] Stanford University, Environmental Earth System Science, 473 Via Ortega, Stanford, CA 94305, USA

ARTICLE INFO

Article history:
Received 31 October 2012
Received in revised form 29 January 2013
Accepted 1 February 2013

Keywords:
Great Basin
Seeding
Sagebrush steppe
MODIS
AVHRR
Fourier
Harmonics
Erosion
Artemisia

ABSTRACT

Wildland fire activity has increased in many parts of the world in recent decades. Ecological disturbance by fire can accelerate ecosystem degradation processes such as erosion due to combustion of vegetation that otherwise provides protective cover to the soil surface. This study employed a novel ecological indicator based on remote sensing of vegetation greenness dynamics (phenology) to estimate variability in the window of time between fire and the reemergence of green vegetation. The indicator was applied as a proxy for short-term, post-fire disturbance windows in rangelands; where a disturbance window is defined as the time required for an ecological or geomorphic process that is altered to return to pre-disturbance levels. We examined variability in the indicator determined for time series of MODIS and AVHRR NDVI remote sensing data for a database of ∼100 historical wildland fires, with associated post-fire reseeding treatments, that burned 1990–2003 in cold desert shrub steppe of the Great Basin and Columbia Plateau of the western USA. The indicator-based estimates of disturbance window length were examined relative to the day of the year that fires burned and seeding treatments to consider effects of contemporary variability in fire regime and management activities in this environment. A key finding was that contemporary changes of increased length of the annual fire season could have indirect effects on ecosystem degradation, as early season fires appeared to result in longer time that soils remained relatively bare of the protective cover of vegetation after fires. Also important was that reemergence of vegetation did not occur more quickly after fire in sites treated with post-fire seeding, which is a strategy commonly employed to accelerate post-fire vegetation recovery and stabilize soil. Future work with the indicator could examine other ecological factors that are dynamic in space and time following disturbance – such as nutrient cycling, carbon storage, microbial community composition, or soil hydrology – as a function of disturbance windows, possibly using simulation modeling and historical wildfire information.

Published by Elsevier Ltd.

1. Introduction

Disturbance by fire is an important component of many ecosystems. Variations in intensity and frequency of fires induced by climatic changes or anthropogenic factors can affect ecosystem structure and functioning (Millennium Ecosystem Assessment, 2005; Westerling et al., 2006). However, in the past few decades, persistence of warm spells and frequent droughts (induced by climatic changes) coupled with anthropogenic activities, have led to more frequent disturbances globally, as evidenced by the increasing number of wildfires worldwide (Millennium Ecosystem Assessment, 2005; Westerling et al., 2006; Zhao and Running, 2010). Further, these factors have in instances delayed the post disturbance vegetation recovery, resulting in detrimental impacts on ecosystems (van der Werf et al., 2008; Zhao and Running, 2010). An especially prevalent environmental effect of increased fire activity is enhanced soil degradation due to accelerated erosion immediately after fire (see reviews by Shakesby, 2011; Shakesby and Doerr, 2006).

Fire combusts vegetation and litter that otherwise provide resistance to erosion, and this is widely considered a dominant factor that can result in environmental degradation (Dieckmann et al., 1992; Inbar et al., 1998; Shakesby and Doerr, 2006; White and Wells, 1979). Further, fires are shown to greatly increase soil erodiblity by altering the soil physical and chemical properties such as surface sealing, inducing soil hydrophobicity, and reducing biological soil crusts (Bowker et al., 2004; Ravi et al., 2009; Shakesby and Doerr, 2006). Accelerated soil erosion by wind and

* Corresponding author. Tel.: +1 928 556 7289; fax: +1 928 556 7100.
E-mail address: jsankey@usgs.gov (J.B. Sankey).
1 Currect address: U.S. Geological Survey, Grand Canyon Monitoring and Research Center, US.

1470-160X/$ – see front matter. Published by Elsevier Ltd.
http://dx.doi.org/10.1016/j.ecolind.2013.02.004

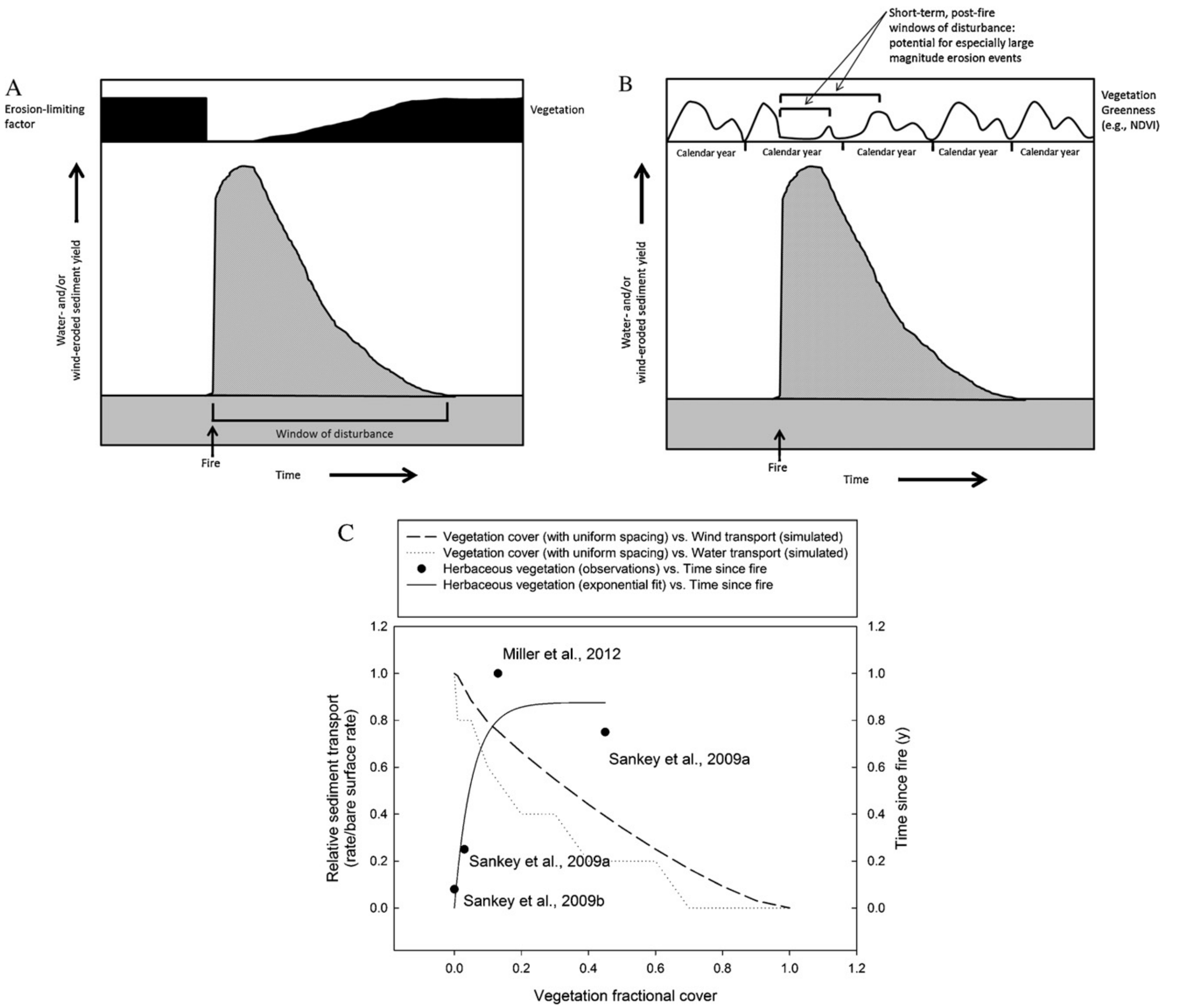

Fig. 1. Disturbance windows relative to the timing of fire, vegetation response, and post-fire erosion: (A) Modified from Shakesby and Doerr (2006) – the hypothetical changes in erosion with fire and post-fire vegetation response; (B) hypothetical changes in erosion with fire and post-fire vegetation greenness dynamics that might be characteristic of a rangeland with continental climate, in which patterns of vegetation green up in the spring, senescence in the summer and a secondary, lesser peak in greenness in the late fall are observed; (C) illustration of relationships of herbaceous vegetation cover with, (i) time since fire for several post-fire erosion studies in the cold desert shrub steppe, and (ii) sediment transport by wind and water simulated using the WEMO (Okin, 2008) and WEPP (WEPP Web Interface, 2012) models.

water has adverse effects on ecosystem function, biodiversity, crop and range productivity, environmental quality and climate (Lal, 1990). Moreover, reduction in vegetation and overall biodiversity via degradation of soil resources can further increase the rate of soil erosion and result in loss of vital ecosystem services including primary production and carbon sequestration, and decrease the ecosystem resilience – the ability of ecosystem to recover from disturbances (Chapin et al., 1997; Elmqvist et al., 2003).

Fires occurring in different seasons and at different intensities may affect ecosystem processes that in turn affect time for recovery. In the ecology and geomorphology literature, a commonly discussed temporal metric of the relationship between fire effects on vegetation and erosion is the disturbance window (Prosser and Williams, 1998 – Fig. 1A). The disturbance window is generally expressed as the time required for eroded sediment yield to return to pre-fire levels (Fig. 1A). The trajectory of erosion potential in response to fire is often described as an immediate (post-fire) increase that peaks in the weeks to months after fire, followed by a steady decline with time since burning. The relative presence of vegetation is a major factor contributing to soil stability (Chaudhary et al., 2009), and the decline in erosion potential with time since burning has been demonstrated to follow a path of non-linear or step decreases in erosion potential that coincide with changes in the relative abundance (reemergence and growth) of vegetation (Fig. 1B and C; Ravi et al., 2012; Sankey et al., 2009a,b; Sass et al., 2012; Shakesby and Doerr, 2006). The longer term decrease in erosion potential relative to vegetation recovery during multiple years post-fire can be punctuated by discrete episodes of increased sediment yield that correspond to rain and/or wind events (Brown, 1972; Miller et al., 2012). However, the largest magnitude erosion events are often observed in the first months to year after a fire when the protective presence of vegetation is at lowest levels (Brown, 1972; Burgess et al., 1981; Prosser, 1990; Prosser and Williams, 1998). Observations of vegetation reemergence and growth after fire therefore provide a useful surrogate for monitoring erosion potential and indicator of overall resistance to erosion.

Satellite remote sensing is a useful tool for observing post-fire vegetation dynamics over large spatial extents and at high temporal frequency that might not be accomplished with traditional

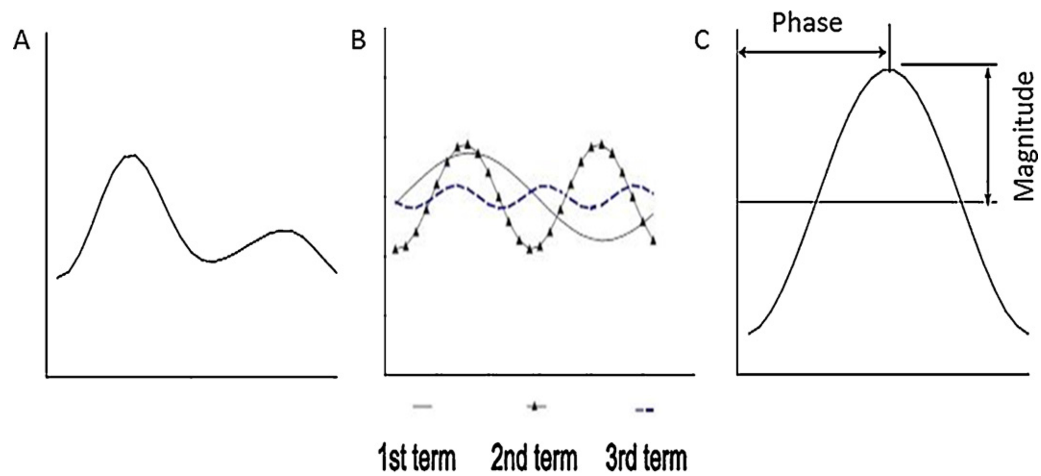

Fig. 2. Modified from Jakubauskas et al. (2001). Examples illustrating components of Fourier analysis in which a complex curve (A) is deconstructed into a set of simple cosine waves (terms) of different frequencies (B); each cosine wave is defined by its phase and magnitude (C).

field-based methods of ecological monitoring (Casady and Marsh, 2010; Ravi et al., 2012; Van Leeuwen et al., 2010; Wittenberg et al., 2007). Satellite remote sensing can provide reasonable estimates of vegetation parameters such as presence, cover, and biomass in a wide variety of land cover types, particularly at higher abundances of vegetation (Wallace et al., 2008; Wylie et al., 2002; Zha et al., 2003). However, the temporal dynamics of vegetation greenness (remote sensing phenology or phenometrics) that can be characterized for relatively short time steps (Bradley and Mustard, 2008; Jakubauskas et al., 2001; Moody and Johnson, 2001; Wallace and Thomas, 2008) might provide a more sensitive indicator of the relatively low levels of vegetation reemergence that can contribute to decreased erosion potential in the short-term, post-fire (Ravi et al., 2012; Sankey et al., 2009a,b; Sass et al., 2012; Shakesby and Doerr, 2006).

In this study, remote sensing observations of vegetation greenness at high temporal frequency are used as a proxy to quantify variability in disturbance windows after historic wildfires in rangelands of the western USA. In the western USA, fire has been documented to increase erosion in a wide variety of biomes and land cover types including forests, shrublands, and grasslands (e.g., Field et al., 2011; Lamb et al., 2011; Pierson et al., 2008; Ravi et al., 2012; Robichaud et al., 2009). Fire activity is known to have varied historically as a function of climate (Heyerdahl et al., 2008; Littell et al., 2009; Marlon et al., 2012), and increases in frequency and size of fires as well as the length of the annual fire season are thought to have occurred in recent decades in the western USA due to contemporary changes in climate (Westerling et al., 2006). We focus on disturbance by fire in the physiographic region of the Great Basin and Columbia Plateau, a vast region of cold desert dominated by shrub-steppe vegetation (Charley and West, 1975; Davies et al., 2011). The cold desert shrub steppe provides critical wildlife habitat for over 350 wildlife species including Greater Sage-Grouse (Centrocercus urophasianus Bonaparte) (Suring et al., 2005). Wildland fire activity has increased in the cold desert shrub steppe due to annual grass invasion and climate change (Keane et al., 2008; Whisenant, 1990). Post-fire management including seeding and soil stabilization are commonly performed to promote the rapid recovery of vegetation for soil stabilization and a desirable vegetation community (Knutson et al., 2009). There is impetus for evaluating effectiveness of such management activities in part due to the perceived increases in fire activity and the potential for vegetation and habitat conversion associated with interactions of fire, invasion, soil disturbance, and anthropogenic activities (Davies et al., 2011; Keeley et al., 2006).

1.1. Study objective

The overall objective of this study was to examine how short-term disturbance windows might be influenced by the time of year that fires burn. This question is particularly relevant in light of the magnitude of historic variability and potential recent increases in fire frequency, size and season length. While increased fire frequency and size might be expected to increase the prevalence of post-fire soil degradation due to reductions in the protective presence of vegetation, effects of an increased length of the fire season and specifically a change in the prevalence of early and/or late season fires is less clear. Therefore, a primary research question examined in this study was: What is the nature of the relationship between fire date and the length of time (disturbance window) between fire date and vegetation reemergence? We considered whether there is rationale to allocate resources and prioritize post-fire treatment efforts (seeding or other site stabilization techniques) for burned areas based on fire date. For example, do early season fires pose a risk of longer disturbance windows due the inherent timing of reemergence of vegetation? Moreover, stabilization and rehabilitation treatments are commonly employed in the cold desert shrub steppe with the objectives to rapidly provide a protective cover of vegetation and stabilize soil immediately after fire, as well as to promote the establishment and recovery of a desirable vegetation community in the longer term post-fire. Therefore we also examined how short-term disturbance windows might be influenced by common seeding treatments performed in the months after fire.

2. Methods

2.1. Calculation and theory – remote sensing phenology disturbance window indicator

Fourier (harmonic) analysis of remote sensing NDVI time series is a method that has been used successfully to describe vegetation dynamics (phenology) in a variety of land cover types (Azzali and Menenti, 2000; Canisius et al., 2007; González Loyarte and Menenti, 2008; Jakubauskas et al., 2001; Wagenseil and Samimi, 2006; Wallace, 2002). A remote sensing time series (Fig. 2A) can be deconstructed using Fourier analysis into a set of simple cosine waves of different frequencies (Fig. 2B) and an "additive term" (horizontal line in Fig. 2C). Multiple frequency terms (e.g., Fig. 2B) sum together to form the original complex curve (Fig. 2A). Each cosine wave is defined by: phase, equal to the offset of the wave from the origin (Fig. 2C); magnitude, equal to the one-half the height of the wave (Fig. 2C); and frequency, equal to the number of complete wave cycles at unit time.

The phase and magnitude of the first term cosine wave is useful for identifying the timing and degree of vegetation green-up for the main growing season in both agricultural and wildland settings (Azzali and Menenti, 2000; Jakubauskas et al., 2001). The phase of the first and second order cosine waves (terms) have been used to identify the timing of the two major periods of vegetation green-up in systems with a bimodal annual pattern such as: cropping systems with winter wheat; rangeland ecosystems with assemblages of annual and perennial vegetation with differing phenology; or deserts that have a bimodal annual distribution of precipitation (Azzali and Menenti, 2000; Canisius et al., 2007; Jakubauskas et al., 2001; Wallace and Thomas, 2008).

The cold desert shrub steppe consists of desert rangeland that has a continental climate consisting of cool winters with snow and spring seasons that can be relatively wet; summers are warm and dry (Fig. 3), and wildfires are often ignited by lightning during the mid to late summer when herbaceous and grass vegetation has senesced. Fall precipitation can be variable but precipitation generally increases later in the fall (Fig. 3). Vegetation greenness patterns observed in satellite remote sensing follow a trend with peak greenness at the height of the growing season in late spring, decreased greenness when vegetation senesces through summer, and often a secondary muted peak or plateau in greenness due in particular to some cool season perennials (e.g., *Poa secunda* J. Presl)

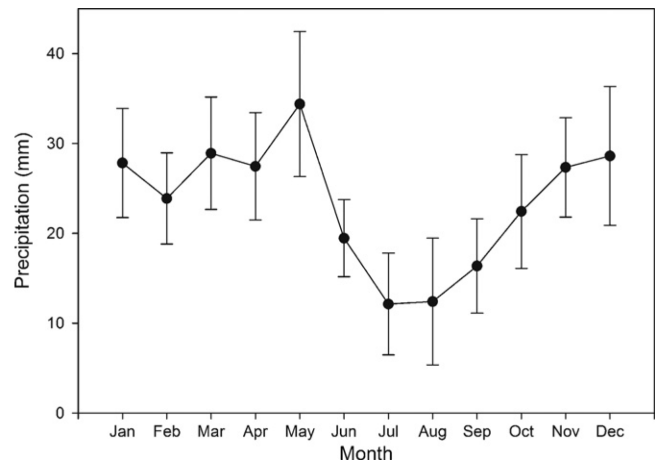

Fig. 3. Average (standard deviation) monthly precipitation during the past ~3 decades among fire project locations analyzed in this study.

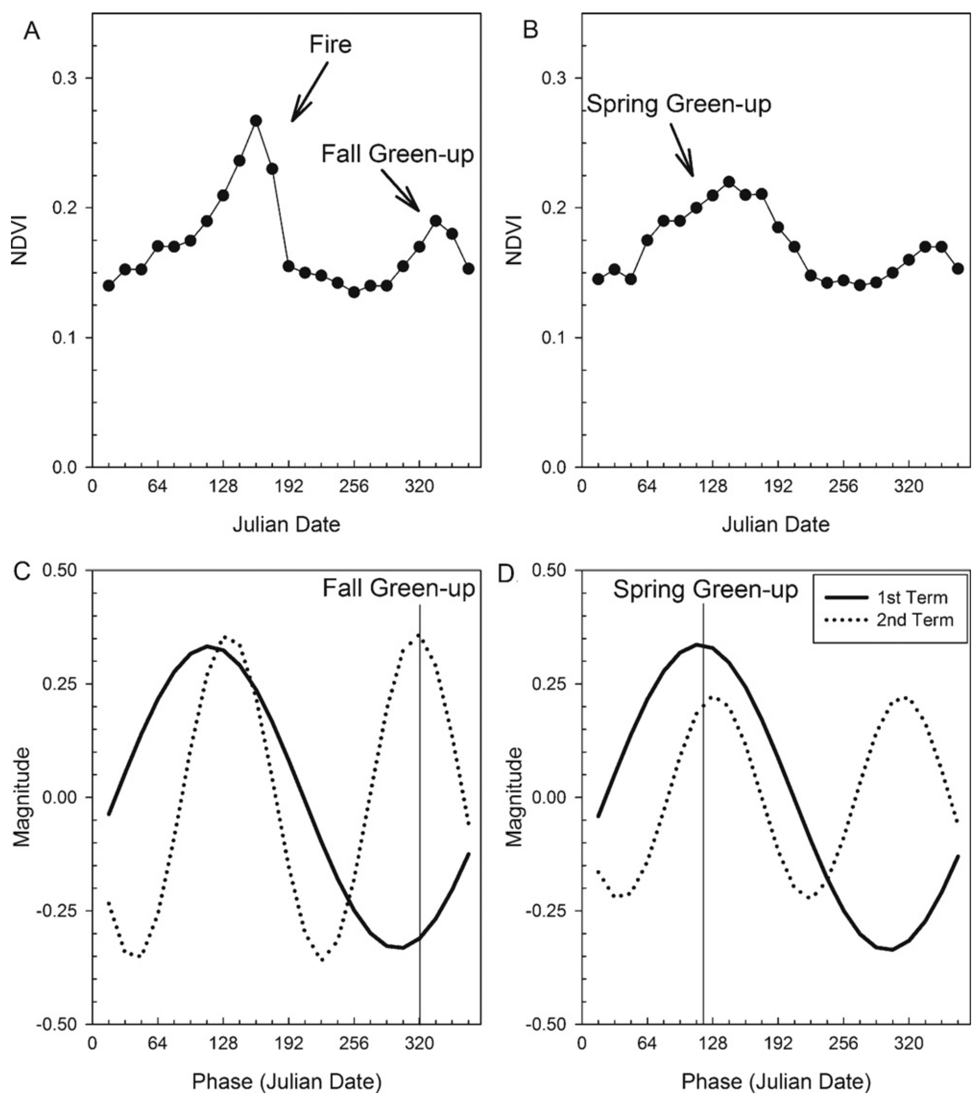

Fig. 4. Illustration of MODIS-NDVI time series (A and B) and 1st and 2nd terms determined from Fourier analysis (C and D).

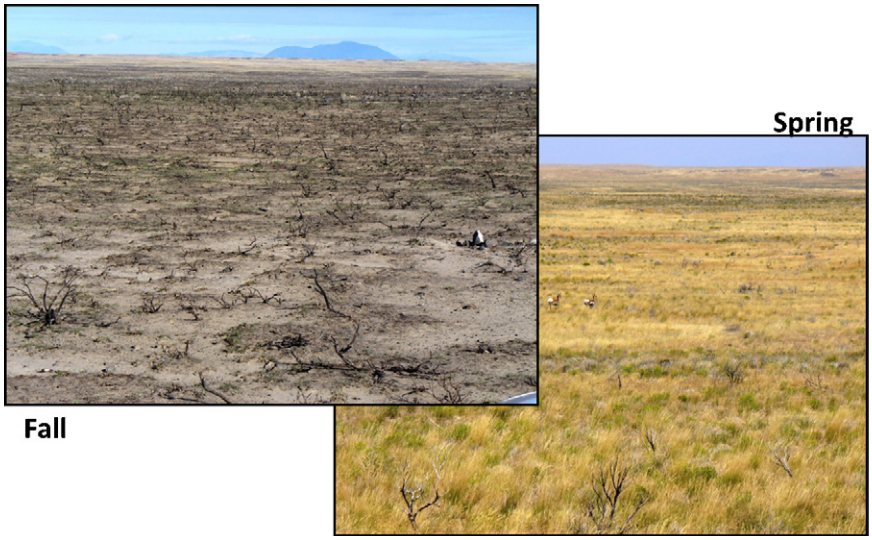

Fig. 5. Examples of short-term post-fire vegetation dynamics for a late-summer wildfire that burned in the study region. Note reemergence (i.e., slight green-up of herbaceous and grass vegetation) in the fall, and substantial regrowth the subsequent spring. Erosion was monitored at this site by a previous study and illustrated substantial step decreases in sediment transport that coincided with vegetation regrowth in these two time periods (Sankey et al., 2009a).

and annuals (e.g., *Bromus tectorum*) that can be green in the mid to late fall and winter (e.g., Fig. 4A; Bradford and Lauenroth, 2006; Bradley and Mustard, 2008). Cold desert shrub steppe consists predominantly of cool season (C_3) plants, and as such, vegetation greenness patterns can be relatively consistent between years with wet relative to dry summers since many plants cannot photosynthesize at high (i.e., mid-late summer) temperatures (Comstock and Ehleringer, 1992).

Vegetation greenness indices observed by satellite remote sensing, and specifically the Normalized Difference Vegetation Index (NDVI), immediately decrease after fire (e.g., Fig. 4A; Van Leeuwen et al., 2010). Because of the timing of natural vegetation dynamics in the cold desert shrub steppe, remote sensing indices of vegetation greenness provide an opportunity to quantify disturbance windows that are important for the short-term erosion potential of burned cold desert shrub steppe. For example, the timing of vegetation green-up in the late-fall after summer fire can provide a measure of the time when the first step decrease in erosion after fire might be expected to occur, and the timing of green-up in the subsequent spring can provide a measure of the time when the next step decrease might occur (Figs. 1A, 4A, B and 5; Sankey et al., 2009b).

In this study, we employed the fast Fourier transform to decompose time series of NDVI (Azzali and Menenti, 2000) from burned areas for the year of the fire as well as the year after the fire, to identify the timing of reemergence of vegetation in the fall and subsequent spring after wildfire (Fig. 4C and D). Specifically, the phase angle of the second term from the NDVI times series of the calendar year of the fire, and the phase angle of the first term from the NDVI time series of the calendar year after the fire, can be converted to a Julian date to identify green-up (timing of maximum greenness) fall and subsequent spring, respectively, after the fire. The number of days between the fire date and fall or spring green-up are estimated to produce two approximations of important short-term disturbance windows in this environment.

2.2. Fire and seeding treatment data sets

Data were acquired from a U.S. Geological Survey fire and seeding inventory study (Knutson et al., 2009) available through the Land Treatment Digital Library (Pilliod, 2009). The complete dataset consisted of 101 projects. Each project was a historical wildland fire that burned on a loamy ecological site in sagebrush (Artemisia tridentata Nutt.) steppe of the Great Basin and Columbia Plateau (Snake River Plain), USA (Fig. 6). The historical fires burned from 1990 to 2003. Each project had unburned and burned treatments replicated 3 times on soils that would support the same ecological site. Burned sites were only known to have burned once during the period of fire records (generally within the last 50 years). Most projects had seeding treatments implemented to revegetate burned areas and stabilize soil [varying combinations of drill seeded, aerial seeded, drill + aerial (mixed), not seeded] replicated 3 times. The total number of burned plots was 540. The plot dimensions as established were 110 m × 110 m, and plot boundaries were located more than 150 m from the boundaries of the fire and seeding treatment perimeters. A majority of the burned plots were evaluated as moderate-high burn severity (http://www.mtbs.gov/, accessed May 2012).

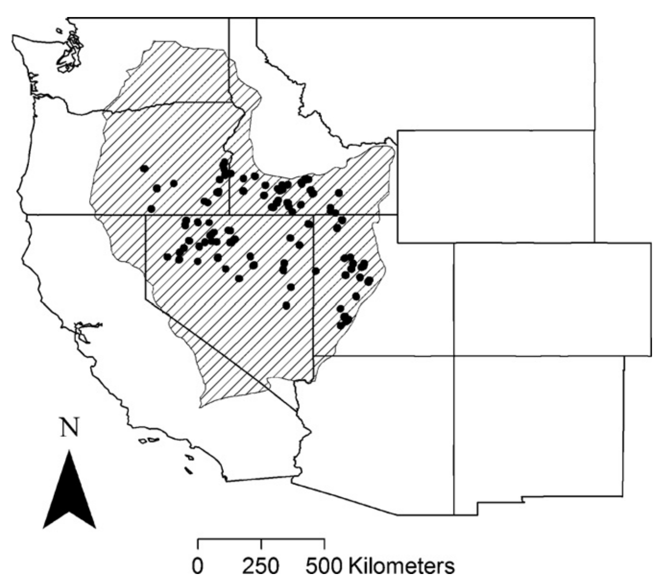

Fig. 6. Map showing locations of historical wildland fires (black points) analyzed in this study in the Great Basin and Columbia Plateau physiographic region (hatched area) and western USA.

2.3. Duration bare – MODIS

Sixteen day composites of MODIS NDVI data (250 m spatial resolution) were acquired from the University of Arizona Remote Sensing Center for 2000–present (Solano et al., 2010). Time series of NDVI values were extracted for all burned plot locations for the 32 projects that burned during 2000–2003 and hence within the period of MODIS availability.

Though the fire plots were located well within fire and seeding treatment perimeters, they were established with dimensions that are smaller than a MODIS pixel (250 m). Therefore, to compare the relative ability for pixel-based measurements to accurately characterize vegetation greenness conditions on the ground, we collected a set of ground based NDVI measurements with a spectroradiometer for plots replicated 3 times in burned and unburned areas at 3, 2011 wildland fires in the Great Basin. For each plot, spectroradiometer measurements were collected at 1 m intervals along 3, 55 m transects that radiated (N, SE, SW) from the plot center. NDVI was determined from the spectroradiometer measurements by aggregating reflectance determined at 10 nm intervals for the full-width half-mast bandwidth of MODIS red and NIR bands. We compared ground-based NDVI collected on two dates in September and November 2011 (September and November) to the respective MODIS–NDVI composite pixels using linear regression analysis.

The primary method used to estimate the date of post-fire vegetation green-up was to apply a Fourier decomposition of NDVI time series (Jakubauskas et al., 2001) to extract the phase of the first harmonic for the calendar year after the fire as an estimate of spring green-up, and the phase of the second harmonic for the calendar year of the fire as an estimate of fall green-up (henceforth "Fourier method"). Phase units were converted from radians to Julian date. To evaluate the Fourier approach, a variation of an additional common method (Moulin et al., 1997) was also used in which the dates of the midpoint of the first instance of three consecutive dates with increasing NDVI after the fire in the fall and in the subsequent spring were estimated for post-fire fall and spring green-up, respectively (henceforth "slope midpoint" method).

The length of time between fire date and the dates of fall and spring green-up were calculated for the Fourier and slope midpoint methods and henceforth will be referred to as "duration bare – fall" and "duration bare – spring". The relationship of fire date and the duration bare variables were examined for each method to consider the effect of fire date on the window of time that soils remained relatively bare of the protective cover of vegetation after fire. Linear regressions of duration bare as a function of fire date were performed for burned plots irrespective of seeding treatments. Additionally, linear regressions of duration bare as a function of fire date and total precipitation by season (i.e., fall or spring after fire) were performed.

The Fourier decomposition of the NDVI time series was also used to extract the magnitude of the first harmonic for the calendar year after the fire (green-up magnitude – spring), and the magnitude of the second harmonic for the calendar year of the fire (green-up magnitude – fall). The effects of seeding on the duration bare and magnitude variables derived from the Fourier method were examined with linear mixed effect models with the duration bare and magnitude variables as response, seeding, fire date, and their interaction as fixed factors, and with random effects of project and plot. Prior to the analysis, the response variables were tested for normality and transformations were applied as necessary; the square of duration bare – fall and the natural log of green-up magnitude – spring were used. Post hoc tests were used to identify significant differences ($p < 0.10$).

2.4. Duration bare – AVHRR

Seven day composites of AVHRR NDVI data, that are available for a longer time frame but at lower spatial resolution (1 km spatial resolution) relative to MODIS, were acquired from the U.S. Geological Survey EROS Data Center for 1989–present (http://earlywarning.usgs.gov/USphenology/; accessed 4/1/2012). Time series of NDVI values were extracted for all burned plot locations for the 101 projects that burned during 1990–2003. AVHRR data were not available for 1990 or 1994 due to satellite failure, meaning that data could not be extracted for (possibly the year-of- *or* year-after) fires that burned in 1990, 1993, or 1994. The resulting dataset consisted of AVHRR-NDVI extracted from plots within 89 distinct historical fires, 57 of which burned 1990–1999 (prior to MODIS availability), and 32 of which burned 2000–2003 (i.e., the same fires examined using MODIS in Section 2.3).

The spatial resolution of the AVHRR data (1 km pixels) was much coarser than the MODIS data and the fire plot dimensions. It was anticipated that this might impact the accuracy of analyses and results. Therefore, duration bare metrics were derived using the Fourier approach from the AVHRR-NDVI data for all plots and values (i.e., fires that burned 1990–2003), and the metrics for the plots that specifically burned 2000–2003 were examined for similarities with the MODIS-based estimates from the same plots using Pearson correlation coefficients (R). It was anticipated that a significant, positive relationship between estimates derived from the remote sensing data sources of differing resolution might provide justification for extending the analysis of duration bare as a function of fire date to the fires in the database that burned prior to MODIS availability. Linear regressions of duration bare (from AVHRR) as a function of fire date were performed for burned plots irrespective of seeding treatments. Effects of seeding treatments were not examined using AVHRR data because of the large mismatch in scale between AVHRR pixels and the plot dimensions.

3. Results

Ground-based NDVI measurements were significantly related with MODIS composite NDVI for the September and November measurement dates (September – Ground NDVI = −0.05 + 1.22 * MODIS NDVI, $R^2 = 0.69$, $p < 0.001$; November – Ground NDVI = −0.03 + 1.11 * MODIS NDVI, $R^2 = 0.77$, $p < 0.001$). This suggests that the MODIS-NDVI measurements were of a suitable spatial resolution to examine variability among the study plots.

3.1. Duration bare and fire date

Fire dates in the data ranged from early June to late September. Among the different methods and datasets used, relationships of duration bare metrics as a function of fire date indicated that fires that burned earlier, relative to later, in the year tended to result in longer windows of time until the subsequent reemergence of vegetation. The duration bare metrics for fall and spring were significantly and negatively related to fire date, irrespective of seeding treatment (Figs. 7 and 8). The relationships of duration bare and fire date were very similar for duration bare derived from MODIS for fires that burned 2000–2003 and duration bare derived from AVHRR for fires that burned 1990–2003 (Figs. 7 and 8). The duration bare estimates derived from AVHRR and MODIS for fires that burned 2000–2003 were significantly related (fall $R = 0.61$, $p < 0.001$; spring $R = 0.65$, $p < 0.001$). The relationship of duration bare and fire date evaluated with the slope midpoint method produced comparable results to the Fourier method, though the strength of the relationships differed (slope

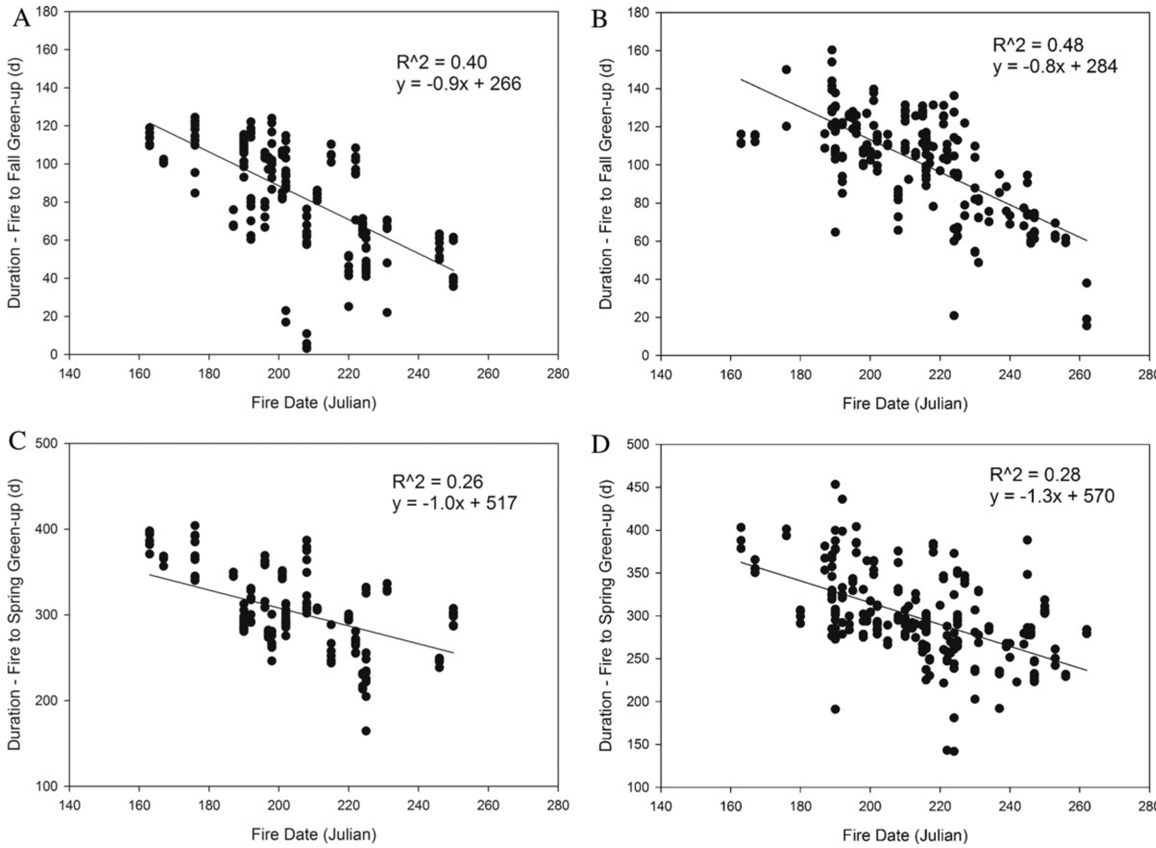

Fig. 7. Relationship of fire date and duration of time from fire date to green-up in the fall and subsequent spring determined with the Fourier transform method for: (A) MODIS-NDVI and plots that burned 2000–2003; and (B) AVHRR-NDVI and plots that burned 1990–2003.

midpoint method: duration bare fall = −0.23 * fire date + 105, $p = 0.02$, $R^2 = 0.03$; duration bare spring = −0.04 * fire date + 21, $p < 0.01$, $R^2 = 0.67$). Incorporation of precipitation received during the first year after fire as an additional predictor variable increased the strength of the correlation only slightly (e.g., Fourier method – duration bare ~ fire date * fall or spring precipitation: $R^2 = 0.40$ for fall MODIS, 0.28 for spring MODIS, 0.51 for fall AVHRR, and 0.29 for spring AVHRR; all $p < 0.05$).

3.2. Seeding treatments

Seeding treatment was significant as a fixed effect explaining variability in the fall and spring duration bare metrics (Tables 1 and 2, Fig. 8A), and regressions of duration bare as a function of fire date were significant when determined separately for seeded and not seeded [e.g. for MODIS-NDVI: duration bare fall (not seeded) = −0.87 * fire date + 263.96, $p < 0.001$, $R^2 = 0.42$; duration bare fall (seeded) = −0.90 * fire date + 267.19, $p < 0.001$, $R^2 = 0.37$; duration bare spring (not seeded) = −1.19 * fire date + 549.17, $p < 0.001$, $R^2 = 0.29$; duration bare spring (seeded) = −0.84 * fire date + 470.25, $p < 0.001$, $R^2 = 0.21$]. However results did not provide substantial evidence that the seeding treatments resulted in shorter windows of time after fire until the subsequent reemergence of vegetation. In the fall, duration bare for the treatments that were not seeded did not differ significantly from the drill and mixed seeded treatments, and duration bare for the aerial seeded treatments was significantly larger. For the spring, the not seeded, drill seeded, and aerial seeded treatments did not differ significantly, and the mean duration bare for the mixed seeding treatment was significantly larger.

Seeding treatment was not significant as a fixed effect explaining variability in the fall magnitude metric, but was for the spring magnitude metric (Tables 3 and 4, Fig. 8B). Results did not provide evidence that the seeding treatments resulted in a larger magnitude of greenness in the short term after fire (i.e., greater NDVI at the peak of the fall and spring green-up after fire). The magnitude of

Table 1
F-Values, significance level (*p*-value), and degrees of freedom from mixed model analysis for response of length of time between fire date and fall green-up ["duration bare (fall)"] derived from the Fourier method on burned plots to the fixed effects of seed (aerial, drill, drill + aerial, not seeded), fire date, and their interaction, and with random effects of project, and plot.

Effect	Square transformed – duration bare (fall)		
	F-Value	*p*-Value	Degrees of freedom
Intercept	118.595	0.000	1
Seed	3.944	0.022	3
Fire date	1.510	0.274	20
Seed * fire date	2.061	0.010	24

Table 2
F-Values, significance level (*p*-value), and degrees of freedom from mixed model analysis for response of length of time between fire date and spring green-up ["duration bare (spring)"] derived from the Fourier method on burned plots to the fixed effects of seed (aerial, drill, drill + aerial, not seeded), fire date, and their interaction, and with random effects of project, and plot.

Effect	Duration bare (spring)		
	F-Value	*p*-Value	Degrees of freedom
Intercept	4612.536	0.000	1
Seed	7.401	0.005	3
Fire date	4.466	0.015	20
Seed * fire date	1.150	0.447	24

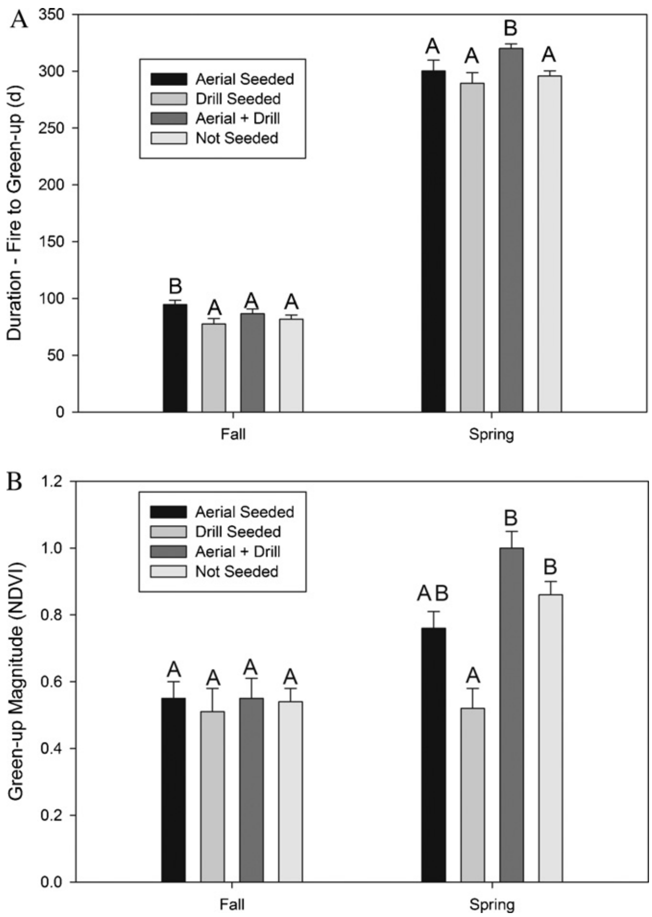

Fig. 8. (A) Duration of time from fire date to green-up in the fall and subsequent spring determined with the Fourier method for seeded (drill and/or aerial) and not seeded locations. (B) Magnitude of green-up in the fall or subsequent spring determined with the Fourier method for seeded (drill and/or aerial) and not seeded locations. Error bars indicate standard errors. Means with different letters indicate statistically significant differences ($p < 0.10$).

Table 3
F-Values, significance level (*p*-value), and degrees of freedom from mixed model analysis for response of magnitude of fall green-up ["magnitude (fall)"] derived from the Fourier method on burned plots to the fixed effects of seed (aerial, drill, drill + aerial, not seeded), fire date, and their interaction, and with random effects of project, and plot.

Effect	Ln magnitude (fall)		
	F-Value	*p*-Value	Degrees of freedom
Intercept	38.699	0.000	1
Seed	0.017	0.997	3
Fire date	0.747	0.718	20
Seed * fire date	0.857	0.642	24

Table 4
F-Values, significance level (*p*-value), and degrees of freedom from mixed model analysis for response of magnitude of spring green-up ["magnitude (spring)"] derived from the Fourier method on burned plots to the fixed effects of seed (aerial, drill, drill + aerial, not seeded), fire date, and their interaction, and with random effects of project, and plot.

Effect	Magnitude (spring)		
	F-Value	*p*-Value	Degrees of freedom
Intercept	4612.536	0.000	1
Seed	7.401	0.005	3
Fire date	4.466	0.015	20
Seed * fire date	1.150	0.447	24

spring green-up for plots that did not receive a seeding treatment was not statistically different relative to aerial and mixed seeded plots, but was significantly greater than the drill seeded plots.

4. Discussion

4.1. Disturbance window indicator

In this study we examined the timing of vegetation greenness patterns during the first year after historic wildfires to approximate variability in disturbance windows when the potential for soil degradation could be expected to be most substantial due to decreased protective presence of vegetation. In general terms, the disturbance window is defined as the time required for an ecological or geomorphic process that is altered to return to pre-disturbance levels (Prosser and Williams, 1998). A common application of the metric in the study of landscape response to wildfire has been to examine (qualitatively or quantitatively) the time required for eroded sediment yield to return to pre-fire levels (Shakesby and Doerr, 2006; Fig. 1). While we did not measure sediment transport after the historical wildfires examined in this study, there is abundant evidence of large magnitude sediment transport events by water and wind in many environments around the world, including rangelands of the Great Basin and Columbia Plateau, and there is specifically strong evidence that the largest magnitude events occur in the short-term (i.e., months–year) after fire, prior to the reemergence of vegetation (e.g., Brown, 1972; Burgess et al., 1981; Miller et al., 2012; Prosser, 1990; Prosser and Williams, 1998; Ravi et al., 2012; Sankey et al., 2009a,b; Sass et al., 2012; Shakesby and Doerr, 2006). The novel application of harmonic analysis with the fast Fourier transform produced computationally efficient, quantitative estimates of the timing of vegetation greenness patterns after fire that were internally consistent within the remote sensing time series. Future work could incorporate knowledge of relationships between vegetation parameters (e.g., cover, structure, biomass), satellite greenness, and time since fire, in modeling efforts that simulated the magnitude of potential sediment transport as a function of disturbance window characteristics measured with this method. The relevance of the disturbance window indicator has been presented with respect to post-fire erosion processes in this manuscript, however, it certainly could be important for other soil and ecological factors, such as nutrient cycling, carbon storage, microbial community composition, or soil hydrology, that can be immediately impacted by fire and the subsequent regrowth of vegetation (Cerda and Doerr, 2005; Doerr and Cerda, 2005; Robichaud, 2000; Sankey et al., 2012a,b; Shakesby and Doerr, 2006).

4.2. Disturbance windows and fire season length

Relationships between estimates of the length of disturbance windows and fire dates indicated that for each 1 day increment earlier in the year that fire burned, there was a slightly less than 1 day increase in the window of time between fire date and vegetation green-up in the fall, and a slightly more than 1 day increase between fire date and spring green-up (Figs. 7 and 8). The timing of vegetation greenness patterns in the short-term after fire therefore appeared to be relatively invariant with respect to the time of year that fires burned, which resulted in longer disturbance windows after fires that burned earlier relative to later in the year. In the western USA in general, there is evidence of a trend of increased wildfire activity in recent decades and the suggestion that the length of the fire season increased during this time period (Running, 2006; Westerling et al., 2006). Longer term study of forests in the western USA indicates that climate, and specifically the extent that moisture is, or is not, abundant in the spring, influences the length of the fire season (Heyerdahl et al., 2008). In grass- and shrublands of the western USA, such as the cold desert shrub steppe, the extent of wildfire activity in any year can additionally be influenced by antecedent climate; specifically precipitation in the previous year (Littell et al., 2009). Therefore, an important implication of our findings is that variability in the length of the annual fire season that can exist at biome or other scales due to antecedent conditions of the spring and/or previous year might indirectly influence post-fire soil degradation through the length of time that soils remain relatively bare of the protective cover of vegetation after fires.

4.3. Seeding treatments

In situations or seasons when it is necessary to prioritize and allocate post-fire efforts in order to maximize the benefits of available rehabilitation resources, the findings of this study indicate there is rationale in considering the time of year that a fire burns as a factor for evaluation. Specifically, large fires that burn earlier in the fire season might be particularly important candidates for rehabilitation and stabilization efforts. Certainly, however, rehabilitation efforts selected should also have proven effectiveness. In this study, we investigated relationships of common seeding treatments to our estimates of post-fire disturbance windows. Results demonstrated that average temporal patterns of vegetation greenness as well as the magnitude of peaks in greenness in the short-term after fires were not notably related to common seeding treatments that are employed after fire. Specifically, seeding treatments did not appear to result in shorter disturbance windows or larger magnitude greenness (e.g., which might be indicative of greater vegetation biomass) relative to untreated burned surfaces.

While differences existed among treatments, and the largest mean values for duration bare and magnitude metrics were observed for different seeding treatments for the two seasons examined, some of the differences might be accounted for by conventions and nuances of the seeding operations. For example, the somewhat counterintuitive finding that aerial seeding was associated with longer disturbance windows in the fall, might be at least partially confounded by the fact that aerial seeding is sometimes not performed until late fall or winter when there is snow on the ground after fire. While this impacts the scope of inference regarding the effects of aerial seeding in the fall, the comparison between duration bare for plots that were drill seeded vs. those that received no treatment in the fall is relevant. The mean time between fire date and fall green-up for these two treatments differed by just 4 days with substantial overlap in the standard error of the means (Fig. 8A). In the spring, seeding treatment effects can be compared with more confidence as each of the common treatments after fires were almost certainly performed prior to the beginning of the next year's growing season. The longer window of time between fire date and green-up observed in the spring for mixed (drill + aerial) seeding is somewhat counterintuitive, and it is not apparent whether there are characteristics of this seeding convention that might have inherently contributed to longer disturbance windows. Importantly, the substantial overlap in standard error bars for the aerial, drill, and not seeded treatments in Fig. 8A illustrate that none of the seeding treatments resulted in shorter windows of time between fire and vegetation reemergence after fire; a finding that was consistent for fall and spring seasons.

While seeding treatments are commonly employed with the intention of promoting a desirable vegetation community in the longer term after fire, another common goal in the short-term is to promote soil stability through the rapid reemergence of herbaceous vegetation (Brown and Amacher, 1999; James and Svejcar, 2010; Monsen and Stevens, 2004). Therefore, it is noteworthy that surfaces seeded after fire did not produce a more rapid reemergence of vegetation (measured by satellite-derived vegetation greenness) compared to surfaces that were not seeded.

5. Conclusion

This study employed a novel ecological indicator based on remote sensing phenology to estimate variability in vegetation greenness dynamics as a proxy for short-term, post-fire disturbance windows in rangelands of the western USA. The indicator estimates were examined relative to the day of the year that fires burned and common seeding treatments to consider effects of contemporary variability in fire regime and management activities in this environment. A key finding was that contemporary changes in the length of the annual fire season could have indirect effects on soil degradation, as early season fires appeared to result in longer time that soils remained relatively bare of the protective cover of vegetation after fires. A more rapid reemergence of vegetation was not evident after post-fire seeding treatments that are commonly employed in the rangelands. Future work with the indicator could employ simulation modeling and historical wildfire information to examine the timing and magnitude of common post-disturbance processes such as erosion, soil hydrologic response, or carbon storage, as function of disturbance windows.

Acknowledgements

This research was supported by a U.S. Geological Survey Mendenhall Fellowship (Joel Sankey). The authors thank Matthew Bowker, David Pilliod, and David Pyke for providing helpful comments on this manuscript. Any use of trade, product, or firm names in this paper is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Azzali, S., Menenti, M., 2000. Mapping vegetation–soil–climate complexes in Southern Africa using temporal Fourier analysis of NOAA–AVHRR NDVI data. Int. J. Remote Sens. 21, 973–996.

Bowker, M.A., Belnap, J., Rosentreter, R., Graham, B., 2004. Wildfire-resistant biological soil crusts and fire-induced loss of soil stability in Palouse prairies, USA. Appl. Soil Ecol. 26, 41–52.

Bradford, J.B., Lauenroth, W.K., 2006. Controls over invasion of *Bromus tectorum*: the importance of climate, soil, disturbance and seed availability. J. Veg. Sci. 17, 693–704.

Bradley, B.A., Mustard, J.F., 2008. Comparison of phenology trends by land cover class: a case study in the Great Basin, USA. Global Change Biol. 14, 334–346.

Brown, J.A.H., 1972. Hydrological effects of a bushfire in a catchment in N.S.W. J. Hydrol. 15, 77–96.

Brown, R.W., Amacher, M.C., 1999. Selecting plant species for ecological restoration: a perspective for land managers. In: Holzworth, L.K., Brown, R.W. (Eds.), Proceedings: Revegetation with Native Species. US Department of Agriculture Forest Service, Rocky Mountain Research Station, Ogden, UT, USA, pp. 1–16.

Burgess, J.S., Reiger, W.A., Olive, L.J., 1981. Sediment yield change following logging and fire effects in dry sclerophyll forest in southern New South Wales. IAHS Publ. 132, 375–385.

Canisius, F., Turral, H., Molden, D., 2007. Fourier analysis of historical NOAA time series data to estimate bimodal agriculture. Int. J. Remote Sens. 28 (24), 5503–5522.

Casady, G.M., Marsh, S.E., 2010. Broad-scale environmental conditions responsible for post-fire vegetation dynamics. Remote Sens. 2, 1–22.

Cerda, A., Doerr, S., 2005. Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. Int. J. Wildland Fire 14, 423–437.

Chapin, F.S., Walker, B.H., Hobbs, R.J., Hooper, D.U., Lawton, J.H., Sala, O.E., Tilman, D., 1997. Biotic control over the functioning of ecosystems. Science 277, 500–504.

Charley, J.L., West, N.E., 1975. Plant-induced soil chemical patterns in some shrub-dominated semi-desert ecosystems of Utah. J. Ecol. 63, 945–963.

Chaudhary, V.B., Bowker, M.A., O'Dell, T.E., Grace, J.B., Redman, A.E., Rillig, M.C., Johnson, N.C., 2009. Untangling the biological contributions to soil stability in semiarid shrublands. Ecol. Appl. 19, 110–122.

Comstock, J.P., Ehleringer, J.R., 1992. Plant adaptation in the Great Basin and Colorado Plateau. Great Basin Naturalist 52, 195–215.

Davies, K.W., Boyd, C.S., Beck, J.L., Bates, J.D., Svejcar, T.J., Gregg, M.J., 2011. Saving the sagebrush sea: an ecosystem conservation plan for big sagebrush plant communities. Biol. Conserv. 144, 2573–2584.

Dieckmann, H.H., Motzer, H., Harres, H.P., Seuffert, O., 1992. Vegetation and erosion, investigations on erosion plots in southern Sardinia. Geo-Ö ko-Plus 3, 139–149.

Doerr, S.H., Cerda, A., 2005. Fire effects on soil system functioning: new insights and future challenges. Int. J. Wildland Fire 14, 339–342.

Elmqvist, T., Folke, C., Nystrom, M., Peterson, G., Bengtsson, J., Walker, B., Norberg, J., 2003. Response diversity, ecosystem change, and resilience. Front. Ecol. Environ. 1, 488–494.

Field, J.P., Breshears, D.D., Whicker, J.J., Zou, C.B., 2011. Interactive effects of grazing and burning on wind- and water-driven sediment fluxes: rangeland management implications. Ecol. Appl. 21, 22–32.

González Loyarte, M.M., Menenti, M., 2008. Impact of rainfall anomalies on Fourier parameters of NDVI time series of northwestern Argentina. Int. J. Remote Sens. 29, 1125–1152.

Heyerdahl, E.K., McKenzie, D., Daniels, L.D., Hessl, A.E., Littell, J.S., Mantua, N.J., 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651–1900). Int. J. Wildland Fire 17, 40–49.

Inbar, M., Tamir, M., Wittenberg, L., 1998. Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 24, 17–33.

Jakubauskas, M.E., Legates, D.R., Kastens, J.H., 2001. Harmonic analysis of time-series AVHRR NDVI data. Photogramm. Eng. Remote Sens. 67, 461–470.

James, J.J., Svejcar, T.J., 2010. Limitations to postfire seedling establishment: the role of seeding technology, water availability, and invasive plant abundance. Range. Ecol. Manage. 63, 491–495.

Keane, R.E., Agee, J.K., Fule, J., Keeley, J.E., Key, C., Kitchen, S.G., Miller, R., Schulte, L.A., 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? Int. J. Wildland Fire 17, 696–712.

Keeley, J.E., Allen, C.D., Betancourt, J., Chong, G.W., Fotheringham, C.J., Safford, H.D., 2006. A 21st century perspective on postfire seeding. J. Forestry 104, 1–2.

Knutson, K.C., Pyke, D.A., Wirth, T.A., Pilliod, D.S., Brooks, M.L., Chambers, J.C., 2009. Chronosequence feasibility assessment of emergency fire rehabilitation records within the Intermountain Western United States—Final Report to the Joint Fire Science Program—Project 08-S-08. U.S. Geological Survey Open-File Report 2009-1099, 20 p.

Lal, R., 1990. Soil erosion and land degradation: the global risks. Adv. Soil Sci. 11, 129–172.

Lamb, M.P., Scheingross, J., Swanson, E., Amidon, W., Limaye, A., 2011. A model for post-fire sediment flux by dry ravel in steep landscapes. J. Geophys. Res. 116, F03006.

Littell, J.S., McKenzie, D., Peterson, D.L., Westerling, A.L., 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecol. Appl. 19, 1003–1021.

Marlon, J.R., Bartlein, P.R., Gavinb, D.G., Long, C.J., Anderson, R.S., Briles, C.E., Brown, K.J., Colombaroli, D., Hallett, D.J., Power, M.J., Scharf, E.A., Walsh, M.K., 2012. Long-term perspective on wildfires in the western USA. Proc. Natl. Acad. Sci. U. S. A., http://dx.doi.org/10.1073/pnas.1112839109.

Millennium Ecosystem Assessment, 2005. http://www.grida.no/graphicslib/detail/number-of-major-wild-fires-by-continent-and-decade-since-1950_11d6 (accessed 04.13.12).

Miller, M.E., Bowker, M.A., Reynolds, R.L., Goldstein, H.L., 2012. Post-fire land treatments and wind erosion – lessons from the Milford Flat Fire, UT, USA. Aeolian Res., http://dx.doi.org/10.1016/j.aeolia.2012.04.001.

Monsen, S.B., Stevens, R., 2004. Seedbed Preparation and Seeding Practices. US Department of Agriculture Forest Service Rocky Mountain Research Station, Fort Collins, CO, USA, 698 p.

Moody, A., Johnson, D., 2001. Land-surface phenologies from AVHRR using the discrete Fourier transform. Remote Sens. Environ. 75, 305–323.

Moulin, S., Kergoat, L., Viovy, N., Dedieu, G., 1997. Global-scale assessment of vegetation phenology using NOAA/AVHRR satellite measurements. J. Climate 10, 1154–1170.

Okin, G.S., 2008. A new model of wind erosion in the presence of vegetation. J. Geophys. Res. 113, F02S10.

Pierson, F.B., Robichaud, P.R., Moffet, C.A., Spaeth, K.E., Hardegree, S.P., Clark, P.E., Williams, C.J., 2008. Fire effects on rangeland hydrology and erosion in a steep sagebrush-dominated landscape. Hydrol. Process. 22, 2916–2929.

Pilliod, D., 2009. Land treatment digital library—a dynamic system to enter, store, retrieve, and analyze federal land treatment data. U.S. Geological Survey Fact Sheet 2009-3095, 2 p.

Prosser, I.P., 1990. Fire, humans and denudation at Wangrah Creek, Southern Tablelands N. S. W. Aust. Geogr. Stud. 28, 77–95.

Prosser, I.P., Williams, L., 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrol. Process. 12, 251–265.

Ravi, S., D'Odorico, P., Wang, L., White, C.S., Okin, G.S., Macko, S.A., Collins, S.L., 2009. Post-fire resource redistribution in desert grasslands: a possible negative feedback on land degradation. Ecosystems 12, 434–444, http://dx.doi.org/10.1007/s10021-009-9233-9.

Ravi, S., Baddock, M.C., Zobeck, T.M., Hartman, J., 2012. Field evidence for differences in post-fire aeolian transport related to vegetation type in semi-arid grasslands. Aeolian Res., http://dx.doi.org/10.1016/j.aeolia.2011.12.002.

Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. J. Hydrol. 231, 220–229.

Robichaud, P.R., Wagenbrenner, J.W., Brown, R.E., Spigel, K.M., 2009. Three years of hillslope sediment yields following the Valley Complex Fires, western Montana. Research Paper, RMRS-RP-77, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Running, S.W., 2006. Is global warming causing more, larger wildfires? Science 313, 927–928.

Sankey, J.B., Germino, M.J., Glenn, N.F., 2009a. Relationships of post-fire Aeolian transport to soil and atmospheric conditions. Aeolian Res. 1, 75–85.

Sankey, J.B., Germino, M.J., Glenn, N.F., 2009b. Aeolian sediment transport following wildfire in sagebrush steppe. J. Arid Environ. 73, 912–919.

Sankey, J.B., Ravi, S., Wallace, C.S.A., Webb, R.H., Huxman, T.E., 2012a. Quantifying soil surface change in degraded drylands: shrub encroachment and effects of fire and vegetation removal in a desert grassland. J. Geophys. Res., http://dx.doi.org/10.1029/2012JG002002.

Sankey, J.B., Germino, M.J., Sankey, T.T., Hoover, A., 2012b. Fire effects on the spatial patterning of soil properties in sagebrush steppe, USA: meta-analysis. Int. J. Wildland Fire, http://dx.doi.org/10.1071/WF11092.

Sass, O., Heel, M., Leistner, I., Stoger, F., Wetzel, K., Friedmann, A., 2012. Disturbance, geomorphic processes and recovery of wildfire slopes in North Tyrol. Earth Surf. Process. Landf., http://dx.doi.org/10.1002/esp.3221.

Shakesby, R.A., 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth-Sci. Rev. 105, 71–100.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth-Sci. Rev. 74, 269–307.

Solano, R., Didan, K., Jacobson, A., Huete, A., 2010. MODIS Vegetation Indices, (MOD13) C5—User's Guide. The University of Arizona, Tucson, 38 p.

Suring, L.H., Rowland, M.M., Wisdom, M.J., 2005. Identifying species of conservation concern. In: Wisdom, M.J., Rowland, M.M., Suring, L.H. (Eds.), Habitat Threats in the Sagebrush Ecosystem: Methods of Regional Assessment and Applications in the Great Basin. Alliance Communications Group, Lawrence, KS, pp. 150–162.

van der Werf, G.R., Randerson, G.T., Giglio, L., Gobron, N., Dolman, A.J., 2008. Climate controls on the variability of fires in the tropics and subtropics. Global Biogeochem. Cycles 22, GB3028, http://dx.doi.org/10.1029/2007GB003122.

Van Leeuwen, W.J.D., Casady, G.M., Neary, D.G., Bautista, S., Alloza, J.A., Carmef, Y., Wittenberg, L., Malkinson, D., Orr, B.J., 2010. Monitoring post-wildfire vegetation response with remotely sensed time-series data in Spain, USA and Israel. Int. J. Wildland Fire 19, 75–93.

Wagenseil, H., Samimi, C., 2006. Assessing spatio-temporal variations in plant phenology using Fourier analysis on NDVI time series: results from a dry savannah environment in Namibia. Int. J. Remote Sens. 27, 3455–3471.

Wallace, C.S.A., 2002. Extracting temporal and spatial information from remotely sensed data for mapping wildlife habitat. The University of Arizona, Tucson, Ph.D. Dissertation, 198 p.

Wallace, C.S.A., Thomas, K.A., 2008. An annual plant growth proxy in the Mojave Desert using MODIS-EVI data. Sensors 8, 7792–7808.

Wallace, C.S.A., Webb, R.H., Thomas, K.A., 2008. Estimation of perennial vegetation cover in the Mojave Desert using MODIS-EVI data and pre-existing field data. GIScience Remote Sens. 45, 167–187.

WEPP Web Interface, 2012. http://milford.nserl.purdue.edu/wepp/weppV1.html (accessed 06.15.12).

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313, 940–943.

Whisenant, S.G., 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In: McArthur, E.D., Romney, E.M., Smith, S.D., Tueller, P.T. (Eds.), Proceedings – Symposium on Cheatgrass Invasion, Shrub Die-Off, and other Aspects of Shrub Biology and Management. 5–7 April 1989, Las Vegas, NV. USDA Forest Service, Intermountain Research Station, General Technical Report INNT-276, Ogden, UT, pp. 4–10.

White, W.D., Wells, S.G., 1979. Forest fire devegetation and drainage basin adjustments in mountainous terrain. In: Rhodes, R.D., Williams, G.P. (Eds.), Adjustments to Fluvial Systems. Kendall/Hunt Publishers, pp. 199–223.

Wittenberg, L., Malkinson, D., Beeri, O., Halutzy, A., Tesler, N., 2007. Spatial and temporal patterns of vegetation recovery following sequences of forest fires in a Mediterranean landscape, Mt. Carmel Israel. Catena 71, 76–83.

Wylie, B.K., Meyer, R., Tieszen, D.J., Mannel, L.L.S., 2002. Satellite mapping of surface biophysical parameters at the biome scale over the North American grasslands: a case study. Remote Sens. Environ. 79, 266–278.

Zha, Y., Gao, J., Ni, S., Liu, Y., Jiang, J., Wei, Y., 2003. A spectral reflectance-based approach to quantification of grassland cover from Landsat TM imagery. Remote Sens. Environ. 87, 371–375.

Zhao, M., Running, S.W., 2010. Drought-induced reduction in global terrestrial net primary production from 2000 to 2009. Science 329, 940–943.