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The two faces of Janus: Processes can be both exogenous forcings and endogenous feedbacks with wind as a case study

Sarah A. R. Payne[1] | **Gregory S. Okin**[1] | **Abinash Bhattachan**[1,2] | **Michael R. Fischella**[1]

[1]Department of Geography, University of California Los Angeles, Los Angeles, California, USA

[2]Department of Geosciences, Texas Tech University, Lubbock, Texas, USA

Correspondence
Gregory S. Okin
Email: okin@ucla.edu

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Abstract

Janus is the Roman god of transitions. In many environments, state transitions are an important part of our understanding of ecological change. These transitions are controlled by the interactions between exogenous forcing factors and stabilizing endogenous feedbacks. Forcing factors and feedbacks are typically considered to consist of different processes. We argue that during extreme events, a process that usually forms part of a stabilizing feedback can behave as a forcing factor. And thus, like Janus, a single process can have two faces. The case explored here pertains to state change in drylands where interactions between wind erosion and vegetation form an important feedback that encourages grass-to-shrub state transitions. Wind concentrates soil resources in shrub-centered fertile islands, removes resources through loss of fines to favor deep-rooted shrubs, and abrades grasses' photosynthetic tissue, thus further favoring the shrub state that, in turn, experiences greater aeolian transport. This feedback is well documented but the potential of wind to act also as a forcing has yet to be examined. Extreme wind events have the potential to act like other drivers of state change, such as drought and grazing, to directly reduce grass cover. This study examines the responses of a grass-shrub community after two extreme wind events in 2019 caused severe deflation. We measured grass cover and root exposure due to deflation, in addition to shrub height, grass patch size, and grass greenness along 50-m transects across a wide range of grass cover. Root exposure was concentrated in the direction of erosive winds during the storms and sites with low grass cover were associated with increased root exposure and reduced greenness. We argue that differences between extreme, rare wind events and frequent, small wind events are significant enough to be differences in kind rather than differences in degree allowing extreme winds to behave as endogenous forcings and common winds to participate in an endogenous stabilizing feedback. Several types of

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state change in other ecological systems in are contextualized within this framework.

KEYWORDS
aeolian processes, cusp-catastrophe model, feedbacks, shrub encroachment, state change, wind erosion

INTRODUCTION

Ecological state changes are transitions that occur when an ecosystem transitions from one state and reorganizes around a new set of self-reinforcing feedbacks defining another state, typically as a result of exogenous driving forces (Ratajczak et al., 2017). Multi-state dynamics have been shown to be present in many terrestrial ecosystems such as: systems that experience insect outbreaks (e.g., Ludwig et al., 1978), boreal forest tree line (e.g., Bonan et al., 1992), mangroves at their poleward extent (e.g., Cavanaugh et al., 2014), coastal scrub (e.g., Huang et al., 2020), tropical forest-savanna mosaics (e.g., Staver et al., 2011), landslide-prone forests (e.g., Runyan & D'Odorico, 2014), barrier islands (e.g., Stallins, 2005), and salt-affected soils (e.g., Lambers, 2003).

The cusp-catastrophe model is often used to describe the conditions that give rise to state change (Figure 1, Loehle, 1989; Rose & Harmsen, 1981; Scheffer et al., 2001; Scheffer, 2009). Such a model posits that an exogenous forcing (or multiple exogenous forcings) influences a state variable. In the absence of endogenous feedbacks, the state variable changes smoothly and reversibly (without hysteresis) in response to gradual changes in the exogenous forcing (Figure 1A,B, Slice 1). Alternative stable states can be induced by positive endogenous feedbacks that fold the state variable response to the exogenous forcing(s) (Figure 1B), causing a bifurcation and two stable states separated by a critical transition (Figure 1C, Slice 2). As the state variable responds to the exogenous forcing(s), movement past the edge of the fold causes abrupt changes in the value of the state variable (Figure 1C, gray arrows) that cannot be reversed along the same path (i.e., hysteresis). Different regions of the state variable values (i.e., the top solid curve vs. the bottom solid curve in Figure 1C) are often interpreted as different states and as such a system might be called 'bistable' where two separate regions of stable state variable values exist. Transitions from one state to another often occur relatively rapidly and appear effectively irreversible due to the feedback-induced hysteresis (Ratajczak et al., 2017).

The transition from grasslands to shrublands occurring in drylands around the world (Archer et al., 2017; Browning et al., 2008; Buffington & Herbel, 1965; D'Odorico et al., 2012) is an important, almost iconic, instance of state change. Shrub encroachment is often observed to occur as a discontinuous and mostly irreversible transition from grass to shrub dominance, suggesting that the underlying dynamics might exhibit alternative stable states and hysteresis, consistent with the cusp-catastrophe model of state change (e.g., D'Odorico et al., 2012; Moreno-de las Heras et al., 2016; Westoby et al., 1989). Thus, while a mixed grass-shrub system can be in "equilibrium" when grass- or shrub-dominance is controlled by exogenous forcing factors and only weakly affected by positive feedbacks (Figure 1A, Slice 1), multiple equilibria exist in the presence of strong positive feedbacks (Figure 1A, Slice 2), often called "non-equilibrium" in the range and dryland ecology literature (e.g., Bestelmeyer et al., 2004). Exogenous forcings of shrub encroachment include climate change, drought, trends in fire management, and grazing pressure (Archer et al., 2017; Brown & Archer, 1989; D'Odorico et al., 2012; Schlesinger et al., 1990). These drivers may interact with internal feedback processes to reinforce one state or another causing the system to move toward one of multiple possible "attractor" states (Figure 1C). Processes related to fire (Archer et al., 1995), sediment transport (Okin et al., 2006), herbivory by small mammals (Bestelmeyer et al., 2007), ecohydrological relations (Turnbull et al., 2008), and microclimate (e.g., D'Odorico et al., 2010) have been invoked as internal feedbacks that reinforce the shrub state.

Transport by wind has been implicated as an endogenous feedback that reinforces the shrub state because wind erosion increases with decreasing grass cover and differentially affects grasses compared to shrubs (Alvarez et al., 2012; Li et al., 2007; Okin et al., 2006; Okin et al., 2009; Yizhaq et al., 2007). Aeolian (i.e., wind) transport is typically considered part of a shrub state-enforcing feedback in areas with wind-erodible soils because semi-regular small transport events move small amounts of sediment that can change the surface height, remove grass seeds from the soil, and abrade plants. The effects of any individual event may be small but the repeated exposure of soils to small amounts of transport can, over time, result in large changes to community composition due to the interaction of biotic (i.e., plant) and abiotic (i.e., transport) processes.

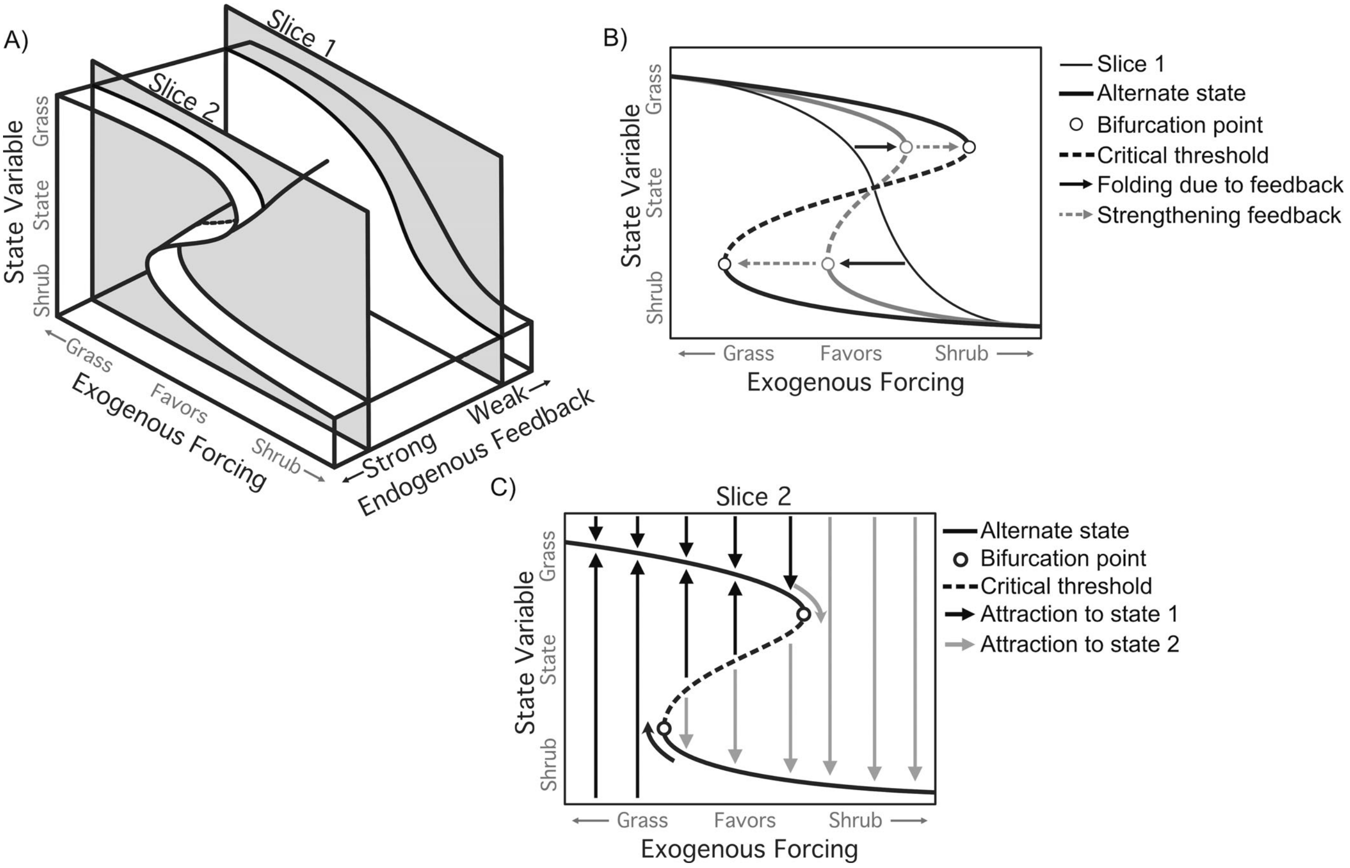

FIGURE 1 (A) The cusp-catastrophe model of state change where a mixed grass-shrub system can be in either uni-stable (equilibrium, Slice 1) or bistable (non-equilibrium, Slice 2). (B) The response surface is folded by the presence of feedbacks to give rise to two, increasingly overlapping states (Slice 1 → Slice 2). (C) Exogeous forcing interacts with endogenous feedbacks to reinforce one state or another causing the system to move toward one of multiple possible attractor equilibrium states.

Whereas semi-regular small transport events are the norm, extreme aeolian transport events occur, but rarely. One relatively early scientific report of such an event, which occurred in the Central Valley of California in December 19–20, 1977 (the so-called "Great Bakersfield Dust Storm"), had sustained winds of over 20 m/s, with decimeters of soil erosion observed from this single event (Sakamoto-Arnold, 1981). The rarity and unpredictability of these extreme windstorms make the consequences of these events in arid landscapes extremely difficult to study. Experiments with pre- and post-event measurements are nearly impossible due to the unpredictable nature of these events. But, when extreme events do occur, their consequence for the landscape demands attention because, compared to the more frequent small events, these events are perhaps so different in degree as to be different in kind. In other words, wind may function as both a component of endogenous feedbacks, which is already well-established, as well as an exogenous forcing in its own right, an idea that has not appeared in the literature.

D'Odorico et al. (2006) have developed a simplistic model for random, extreme events that can be used to conceptualize how wind might behave as an exogenous forcing. Although the model was developed for fires in savannas, it can be easily reinterpreted for wind-induced grass mortality (see Appendix S1) and mapped onto a cusp-catastrophe diagram (Figure 1A). Specifically, in the model, the parameter describing the likelihood of an extreme wind event, λ_o, is the primary measure of the strength of the exogenous forcing, which is to say, the likelihood of a grass mortality event (i.e., extreme wind event = exogenous forcing). By the same token, b describes how the frequency of erosional events changes with grass cover where events become less likely with increasing grass cover because the threshold for initiation of an event increases with vegetation cover (Raupach et al., 1993). As such, b is a measure of the strength of the endogenous feedback that folds the response surface to induce bistability.

An important criterion for the realistic interpretation of the D'Odorico et al. (2006) model for extreme wind events is the ability of these events to directly result in dramatic losses in grass. In Spring of 2019, two rare, high wind events were recorded in south-central New Mexico. Serendipitous observations after the fact showed considerable erosion and, importantly for the grass-shrub state change, considerable excavation of grass roots by wind erosion in an area where shrub encroachment has been researched for several decades (Figure 2A,B, Buffington & Herbel, 1965). Here, we report on the quantitative observations of root exposure and its consequences, and place these observations into the context of the cusp-catastrophe model to elucidate the dual role that wind plays in state change. Later, we situate this argument in terms of other ecosystems with process-dependent, dynamic interplays between exogenous extreme events and endogenous feedbacks. Janus is the Roman god of transitions and is depicted with two faces. He is an apt symbol here as we discuss whether one fundamental process can have two manifestations that control the transitions between (or stability of) ecosystem states.

METHODS

Study area, meteorological data, and sediment transport data

Located within the Chihuahuan Desert, the USDA-ARS Jornada Experimental Range (JER) in south-central New Mexico is composed of semiarid shrublands and grasslands with plant communities having shifted from abundant grasslands to shrublands in the past 150 years (Buffington & Herbal, 1965). On March 13th and April 10th of 2019, strong wind speeds were recorded at the nearby San Augustin Pass in New Mexico (Alba-Soular, 2019a, 2019b). Here, daily average windspeed measurements from the JER headquarters (Anderson, 2022) from 1983 to 2021 were used to contextualize the two 2019 windstorms by calculating the return interval (in years) of days with average winds in 0.25-m/s windspeed bins (above 3 m/s). Aeolian sediment transport measurements from a Big Springs Number Eight

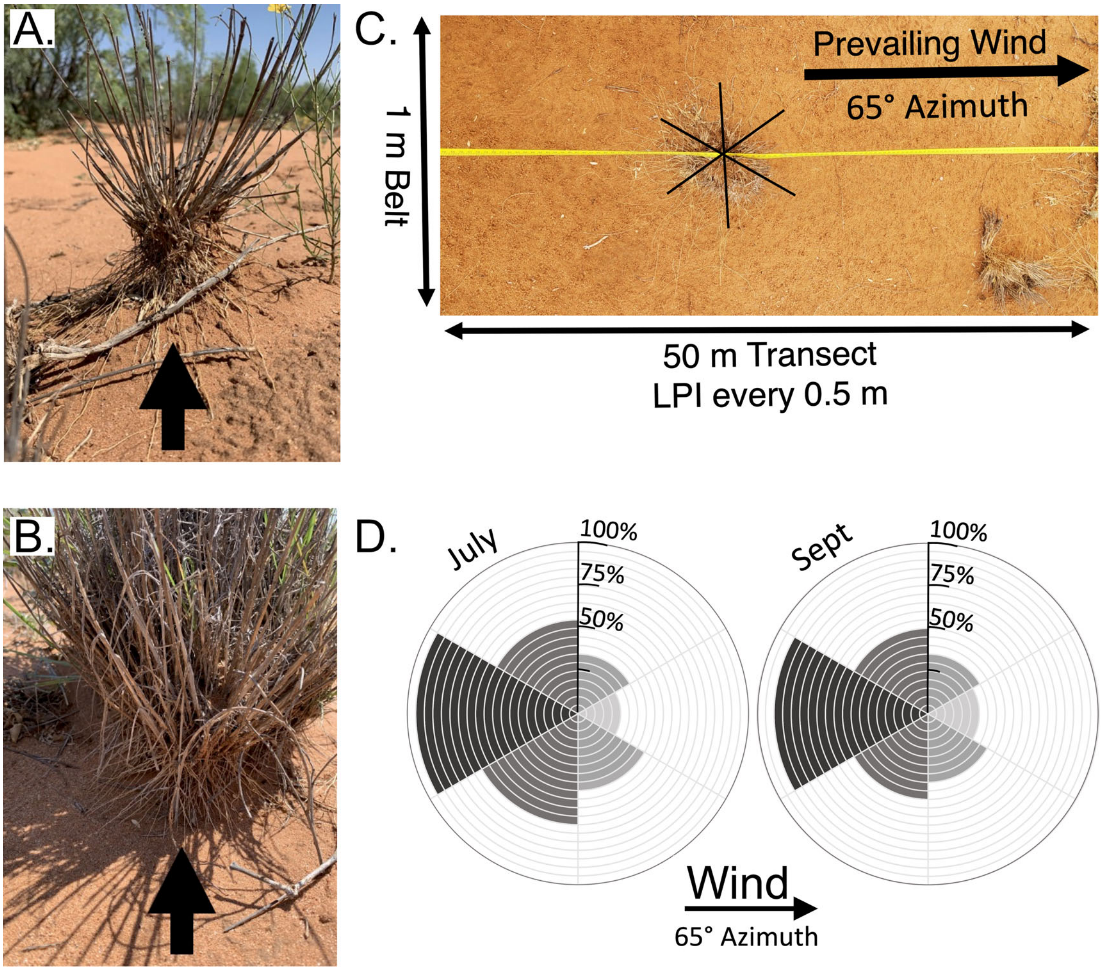

FIGURE 2 (A) and (B) Photographs of grasses with root exposure due to erosion. The black arrow represents the direction of the wind. (C) Nadir view of a portion of a transect and collocated one-meter belt transect. Direction of root exposure was quantified using six sectors. (D) Rose diagrams showing fraction of plants that had exposed roots in each of the sectors in July and September. Photo credits for (A)–(C): S. Payne. LPI, line point intercept.

(BSNE, Fryrear, 1986) samplers within ~1 km of the field sampling sites were used to verify that aeolian transport had been extreme during the spring of 2019 (Okin & Gillette, 2022). Sample collection at these sites started in spring 2007 and we analyzed data through winter 2021. Using standard methods, weights from quarterly collections were divided by the size of the opening and the number of days of collection to estimate aeolian flux in gram per square centimeter per days at each of four BSNE heights. These were then fit to an exponential function of height, which was then integrated from 0 to 1 m to estimate total horizontal aeolian flux in gram per centimeter per days (Gillette et al., 1997).

There are several 10-m meteorological tower with 5-min measurement intervals located within the JER. The closest of these to the site of field data collection (M-Nort, Okin, 2022b) also recorded the strong storms on March 13 and April 10, 2019. From the 10-m anemometer data from the closest of these to the site of field data collection (M-Nort), we calculated Drift Potential (DP, Fryberger & Dean, 1979):

$$DP = \sum \begin{cases} U^2(U - U_t)/T & U > U_t \\ 0 & U \le U_t \end{cases},$$

where U is the windspeed at 10-m and U_t is the threshold wind speed for initiation of transport (6 m/s was used here), and T is the time of each record, 5 min. DP provides an index of the wind's ability to move sand and is not intended as a quantitative measure of flux. DP during March and April of 2019 was primarily toward the northwest (average weighted DP direction was toward 41° azimuth). For March and April from 2016 to 2020, the average weighted DP direction was toward 80° azimuth. These calculations caused us to align our vegetation transects at 65° azimuth, between the 2019 and whole-record average DP directions (Li et al., 2007).

Field data collection

Field data are published as Okin (2022a). In July 2019, we selected six locations based on low, medium, and high cover of *Sporobolus spp*. and *Bouteloua eriopoda* (two dominant grass species at JER). All sites were on the "sand sheet," a relatively homogenous fine sandy loam/loamy fine sand surface of aeolian sediments derived from the Rio Grande River (Monger et al., 2006). As such, they have very similar threshold wind speeds for entrainment and essentially unlimited sediment supply (Webb et al., 2016). There is no dominant slope in this area of the sandsheet, with the surface being generally very flat with very shallow, wide depressions. Furthermore, the infiltration rate for the soils at the site is very high, meaning local runoff is rarely generated and when it is, has no preferred direction of flow. *Prosopis glandulosa* (mesquite) is the dominant shrub at all sites and across the sand sheet. At each site, we established a 100-m by 200-m plot and randomly placed three 50-m transects within the plot oriented at 65° azimuth. Initial measurements were made in July and again in September 2019, which is typically near peak annual biomass in the region (Huenneke et al., 2002). We recorded vegetation composition using line point intercept (LPI) measurements along the transects at every half meter. The height of shrubs that intersected the transects were measured using the disk-drop method (Li et al., 2013). Grass species were dormant during the July 2019 measurements but could be identified to the genus level. We identified shrubs to the species level, though mesquite was the dominant woody species at the sites.

A belt transect was centered along the same transect tape as LPI to characterize patterns of root exposure recording a grass if it was within 50-cm from the transect. Grasses with overlapping canopies were considered a single patch because they function as a single roughness element for the wind. For each patch, we noted if a grass had any green shoots and whether roots were exposed. The orientation of the exposed root was captured by designating six sectors radiating from the center of the grass, each subtending 60° (Figure 2C).

Between September 5th and 12th, 2019, we captured high resolution aerial imagery of the transects to determine the relationship between percent grass cover and relative greenness of grass patches. We flew a Phantom 4 quadcopter (DJI, Shenzen, China), over the established 18 transects at a height of 10-m with 80% overlap between each flight path. Flights were between 10 AM and 2 PM to establish similar lighting conditions and limit canopy shadow. Approximately 120 images per transect were collected to generate orthomosaics of 3.5 mm/pixel resolution using PhotoScan (Agisoft Metashape Pro 1.6). In ENVI 5.5 (L3Harris Geospatial, Boulder, CO), for nine of the 18 transects, we placed region of interest (ROI) polygons covering every grass patch within one meter of the transect tape, which was visible in the orthomosaic. These ROIs were used to estimate grass cover. For each ROI, we calculated the Green Chromatic Coordinate (GCC):

$$\mathrm{GCC} = \frac{\mathrm{DN}_{\mathrm{green}}}{\mathrm{DN}_{\mathrm{red}} + \mathrm{DN}_{\mathrm{green}} + \mathrm{DN}_{\mathrm{blue}}}$$

where DN is the digital number for each of the three color channels (red, green, and blue) (Richardson et al., 2007).

The GCC is a greenness index that is a measure of the proportion of the reflected light that is in the green channel, irrespective of total amount of visible light reflected (i.e., irrespective of total brightness).

RESULTS

The 1983–2021 wind data indicate that the March 13, 2019 event had the highest daily average wind speed (9.4 m/s) in the entire record (Figure 3A). The April 10, 2019 event was equivalent to a 7-year return interval event (7.4 m/s). In general, the return interval was well fit by an exponential function with $R^2 = 0.73$. The measurements of total horizontal aeolian flux at a BSNE stem near the location of field data collection showed that from 2007 to 2021, spring 2019 had the highest flux (~0.8 g cm^{-1} days^{-1}), nearly four times the next highest flux (Figure 3B).

Our belt transects recorded a total of 2577 patches of grass across 18 transects, of which 29% had root exposure. These statistics do not include individual grass patches that were apparently completely removed during the wind events, with only a few roots remaining to mark where they had been. These patches, though clearly a component of the total grass loss, were nonetheless difficult to quantify, and the degree to which they could have been adequately sampled is unclear. If these lost patches had been included, the total amount of root exposure would have been higher than reported here.

Root exposure was concentrated in the upwind direction (65° azimuth, Figure 2D). The rose diagram in September remained essentially the same as in July. We observed that green shoots were not concentrated in the downwind direction, but were evenly distributed (not shown). The fraction of grass patches that exhibited exposed roots decreased strongly with grass cover (Figure 4A power law fits through curves have $R^2 = 0.68$ [July] and 0.74 [September]) for both July and September measurements, though the number of plants exhibiting exposed roots increased in September compared to July. Above 20% grass cover, between 80% and 100% of the grass patches exhibited green shoots, whereas this fraction was highly variable below this threshold (Figure 4B). This pattern was similar in both July and September. The percent of grasses with green shoots decreased strongly with the fraction of plants exhibiting root exposure (Figure 4C), though overall the number of plants exhibiting green shoots increased from July to September, which is consistent with the latter being near peak greenness in typical years. There was no clear relationship between the height of mesquite (*P. glandulosa*) shrubs, as a general indicator of plant size, and the fraction of grasses in the transects that were green, particularly when mesquite were <60 cm in height (Figure 4D). Multiple regression of the fraction of grasses exhibiting green shoots against grass cover, shrub (mesquite) cover, and mesquite height show significant positive relationships with grass cover ($p = 0.014$) but not shrub (mesquite) cover ($p = 0.283$) or mesquite height ($p = 0.130$) with the overall model having adjusted $R^2 = 0.568$ ($p = 0.002$). Likewise, multiple regression of the fraction of grasses with exposed roots against grass cover, shrub (mesquite) cover, and mesquite height show significant negative relationships with grass cover ($p = 0.005$) but not shrub (mesquite) cover ($p = 0.539$) or mesquite height ($p = 0.140$) with the overall model having adjusted $R^2 = 0.586$ ($p = 0.001$). Though there is a significant negative relationship between shrub (mesquite) cover and grass cover (not shown, $R^2 = 0.46$, $p = 0.002$), variance inflation factors (VIFs) for mesquite height, mesquite cover, and grass cover are between 1.3 and 2.3, well below the minimum threshold of 10 often used as a rule of thumb to indicate the need for correction of collinearity (O'Brien, 2007).

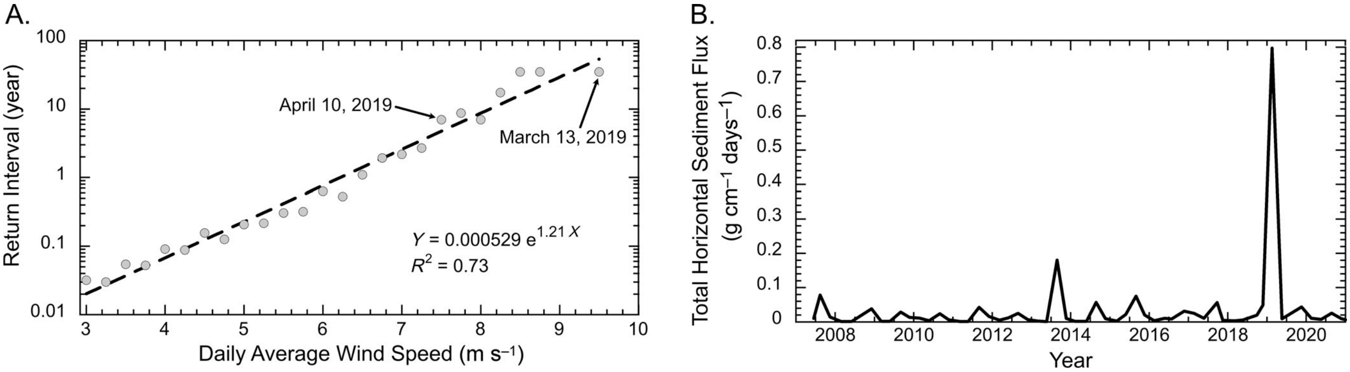

FIGURE 3 (A) Return interval for average daily windspeed from 1983 to 2021 for the Jornada Experimental Range headquarters meteorological tower. (B) Seasonal total horizontal aeolian flux from 2007 to 2021 from a site near the locations of field data collection.

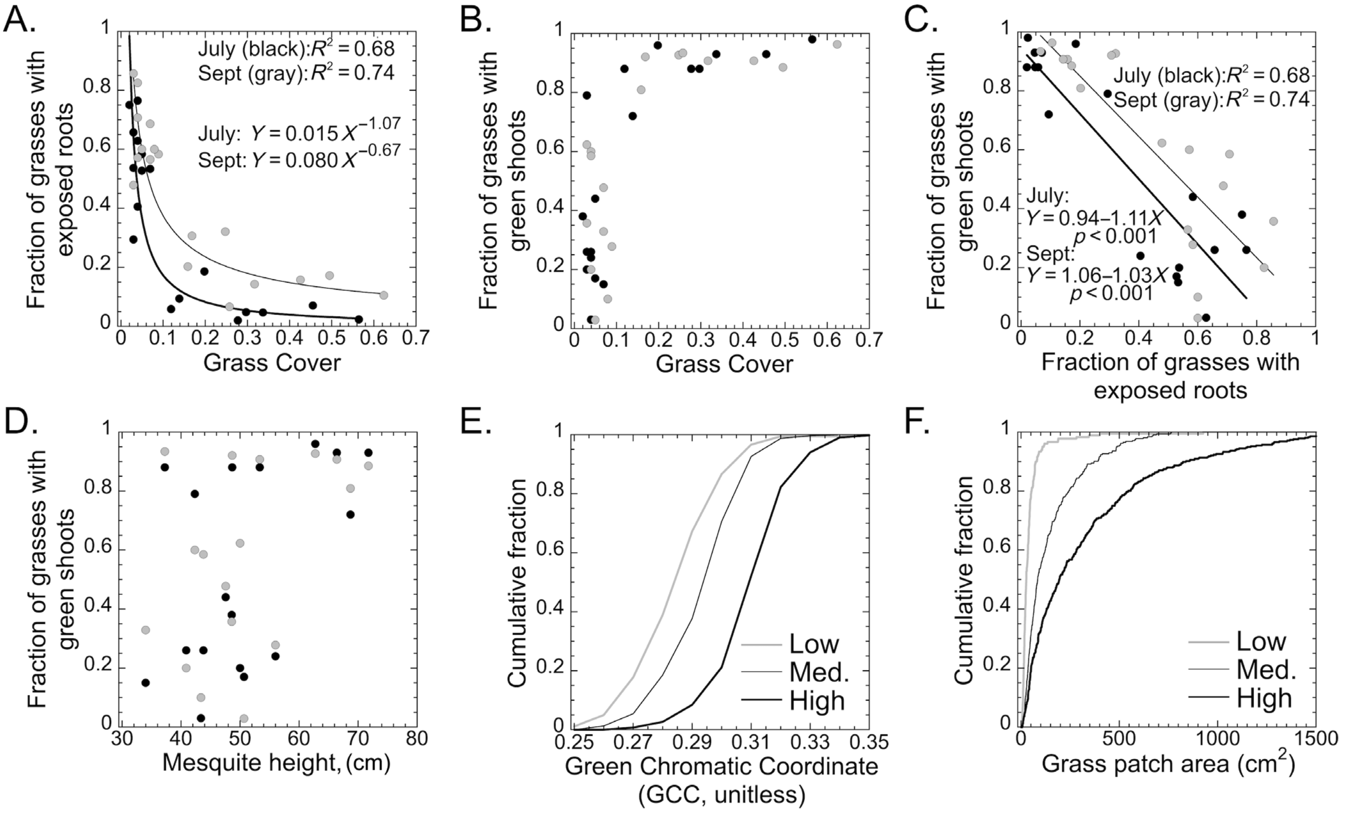

FIGURE 4 (A)–(D) Results of belt and line point intercept (LPI) transect data collection in July (black symbols) and September (gray) symbols for all 18 transects showing how the fraction of grasses that have exposed roots or green shoots vary with grass cover and mesquite cover. (E) and (F) Results of orthomosaic analysis showing cumulative distribution functions (CDFs) of green chromatic coordinate (GCC) and grass patch area (cm^2), respectively, for site with low, medium (med.), and high cover.

UAV-derived orthomosaics show that the low grass cover sites had between 3% and 4% grass cover, the medium cover sites had between 16% and 25% grass cover, and the three high cover sites ranged between 43% and 62% grass cover. The high grass cover sites exhibited generally greener grass patches (as indicated by GCC) and patch size than the low grass cover sites, with the medium grass cover sites in between (Figure 4E,F). Two-tailed heteroscedastic Students' *t* tests indicate that the mean GCC and grass patch sizes values for the three populations (low, medium, and high) are significantly different ($p < 0.001$ in all cases). A significant positive ($p < 0.001$) log-linear relationship exists between individual grass patch area and GCC (not shown).

DISCUSSION

Our analysis of a nearly 40-year (1983–2021) wind record from the Jornada, as well as a ~14 year record of aeolian flux near the site of our field data collection indicate that spring 2019 was, indeed, extraordinary both in terms of wind speed and measured total horizontal aeolian flux (Figure 3). In both records, the highest daily average wind speed and the highest seasonal flux in both records were observed during this season. Thus, the observations we made in summer 2019 can shed light on the impact of extreme wind events on grass cover.

Though we did not have the benefit of collocated pre-windstorm measurements, our field observations nonetheless present a compelling story. Specifically, the orientation of root exposure is consistent with the direction of the erosive winds (drift potential) during the storms (Figure 2D). The fraction of grasses with exposed roots was a strongly decreasing function of grass cover (Figure 4A), which is consistent with higher-cover sites experiencing less erosion due to the protective role of grass against wind transport (e.g., Okin, 2008) and sites with initially lower cover experiencing higher erosion because the soil was more exposed. The erosive/abrasive effects of the wind did not just expose roots (Figure 4A–C), but also appears to have inhibited the emergence (in July) and growth (by September) of green shoots (Figure 4C,D). Sites with less than 20% grass cover appeared to experience some suppression of greening, while sites with greater than 20% grass cover appeared unaffected (Figure 4B). The difference in greenness could be observed both through transect measurements and UAV-derived orthomosaics which show that the low grass sites had the smallest remaining grass patches as well as the least green grass patches as measured by GCC (Figure 4E,F). Our data do not suggest that edaphic effects are responsible for suppressed greening. If belowground interactions or competition for water were responsible for suppressed greening, we would expect a negative relationship between the fraction of grasses exhibiting green shoots and the size of shrubs or shrub cover. Grass cover appears in our data to be the only factor that significantly (and with a positive slope) explains grass greenness, thus suggesting that higher aeolian fluxes associated with lower grass cover are alone responsible for suppressing grass greening.

To be sure, the initial conditions of our low, medium, and high grass cover plots were different before the storms, with patterns in (unmeasured) initial grass cover likely mimicking patterns in grass cover from our data, measured after the storms. Differences in soil texture, crusting, or sediment availability could, in theory, give rise to differences in wind erosion (e.g., Gillette & Chen, 2001), but in the relatively small area over which our data were collected (sites within 3 km of on another) there are no appreciable differences in any of these soil parameters (Soil Conservation Service, 1980), largely as a result of their common geomorphic origin (Monger et al., 2006). The sites with higher grass cover have lower shrub cover, consistent with this area experiencing encroachment of shrubs into grasslands. But in the absence of meaningful differences in soil properties or shrub cover, it is logical to conclude that pre-storm grass cover drove differences in aeolian transport, leading to significant differences in plant response.

We were unable to characterize the extent to which grass patches were completely removed along our transects, though this obviously occurred, as evidenced by patches of exposed grass roots without any attached shoots. The process for complete removal of the aboveground portion of grasses is not completely clear, but in many cases, we observed the complete or near-complete removal of soil from beneath the above-ground portion of the grass patch, leaving it suspended several centimeters above the soil surface, propped up by roots (e.g., Figure 2A). Cases where the propping roots had broken, causing the suspended grass to lay on the soil surface held in place by only a few roots, were also frequent. It is not hard to imagine that during the storm, or just after, full detachment of grasses that were previously anchored is the likely mechanism for complete loss of a grass patch.

The reduced greenness during peak biomass in September in the low cover plots and observed for the smaller grass patches in all plots (Figure 4E,F) suggests that the erosion/abrasion of the grass roots influenced the plant's ability to respond to moisture. The complete loss of grass patches, though unquantifiable because of the inability to quantify an absence, clearly contributes to the reduction in potential grass growth that we observed in the remaining grass patches.

As we argued earlier, direct grass loss is a criterion for realistic reinterpretation of the D'Odorico et al. (2006) extreme event model in terms of wind's effect on grass cover. Our field observations show that extreme wind events can directly reduce grass cover. We argue that extreme wind events, like those the effects of which are presented here, are so different from small relatively frequent wind events as to be considered a different dimension of grass-shrub dynamics altogether. Rather than folding the response curve, which is what wind-as-a-feedback does, extreme wind events have effects that are not modulated by vegetation dynamics and act as exogenous drivers causing more direct and abrupt state changes as the system responds after the event (Figure 1). The ecologically instantaneous fatal wounding and removal of grasses over large areas that occurs during extreme wind events is a phenomenon that has more in common with drivers like overgrazing or extreme drought, which directly remove or kill grasses, than regular wind events.

We have argued here that one basic process (here, wind) can participate as both exogenous driver during extreme events and endogenous feedback during non-extreme events. Many ecological/ecogeomorphic systems exhibit bistability. There are systems besides the grass-shrub/wind system that also appear to display dynamic interplay between extreme and non-extreme events of a single episodic process. For example,

1. Feedback between storms and barrier island erosion (even under mild storm conditions) maintains these systems in bistable states: (1) high, vegetated, and minimally susceptible to storm erosion or (2) low, sparsely vegetated, and maximally susceptible to storm erosion. However, extreme storm surge events can entirely destabilize the barrier island system, causing immediate transition from the vegetated to unvegetated states, and possibly leading to barrier island disintegration (Durán Vinent & Moore, 2015).
2. Many mixed grass-woody systems exhibit multiple stable states and regular fires interact with vegetation to stabilize a mosaic of closed-canopy forest, open savanna, and grasslands (Beckage et al., 2009; Minnich, 2007; van Wagtendonk et al., 2018). However, extreme fires in closed-canopy forests (especially if encouraged by

previous fire suppression) can cause stand replacement and precipitate a transition to open, grassy savanna (Beckett et al., 2022; Goforth & Minnich, 2008).

3. Vegetated hillslopes can exhibit bistability in soil depth due to feedbacks between hillslope erosion and rock weathering (D'Odorico, 2000). Extreme events, such as landslides induced by heavy rains, however, can completely remove the soil, moving areas in the with-soil state to the without-soil state (Runyan & D'Odorico, 2014).
4. Abandoned agricultural terraces can be maintained by feedbacks between vegetation growth, terrace stabilization, and increased infiltration of runoff precipitation. However, extreme events can induce terrace-cascading avalanches that erase terracing (Moreno-de-las-Heras et al., 2019).
5. Herbivory by small rodents constitutes part of a feedback process whereby shrub invasion limits grass establishment via seedling herbivory (e.g., Bestelmeyer et al., 2007). However, large-scale herbivory by domestic livestock constitutes and extreme herbivory that functions as a forcing factor, directly causing state change (e.g., Archer et al., 1995).
6. Uneven infiltration caused by patchy vegetation in shrub-encroached grasslands contributes to unequal distribution of infiltration that enhances heterogeneity which favors further shrub encroachment (e.g., Turnbull et al., 2012). Thus regular rainfall events participate in a state-stabilizing feedback. But extreme drought acts broadly as a forcing factor across the landscape to directly kill grasses and support the shrub state (e.g., Gherardi & Sala, 2015; Peters et al., 2012).

In all of these cases, as with the shrub-grass/wind system, the regular, episodic process (in the examples above, coastal erosion, fire, hillslope erosion, runoff/infiltration, herbivory, and precipitation, respectively) behaves as a component of the endogenous feedbacks that strengthen bistability during regular non-extreme events but as an exogenous forcing during rare extreme events. When this happens, the process causes rapid, nearly instantaneous change over large areas, irrespective of any state-stabilizing feedbacks.

As a final observation, there may be a connection between spatial scale and process-as-feedback versus process-as-forcing. We note that when a process primarily acts at finer spatial scales, such as local redistribution by wind (or, referring to examples above, (1) regular sediment movement on a beach, (2) small, cool fires in a forest, (3) hillslope erosion, (4) terrace-scale runoff and infiltration, (5) herbivory by small mammals, (6) patch-scale distribution of runoff), they appear to behave as endogenous feedbacks. Even if these fine-scale, endogenous interactions are repeated many times (i.e., for the small territories of many small herbivores or many plant-interspace pairs) over a broad area, the interactions are essentially local. In contrast, when a factor acts at broad scales, such as large windstorms that cause heavy erosion over large areas (or, referring to examples above, (1) large waves causing beach erosion, (2) major crown fires, (3) and (4) heavy rains that saturate soil over areas leading to slope failure, (5) livestock grazing over large areas, (6) regional drought), the same basic factor is essentially exogenous and behaves as a forcer.

CONCLUSION

In wind's role as a feedback, it has normally been understood that the regular action of the wind on soils and plants causes incremental changes that can push, over time, a grassland into a shrub-dominated state from which recovery is difficult or impossible under present climatic conditions, as evidenced by the failure of several bush control or removal programs (D'Odorico et al., 2012). Our field observations, in contrast, clearly show that extreme wind events can directly reduce grass cover, acting as an endogenous forcer of grass cover. Indeed, there are many ecological systems with key processes that display this dual behavior whereby the process can, when at low intensity, function as an endogenous feedback to stabilize the existence of individual separate states and, in doing so, hinder transitions between them. At the same time, during extreme events, the same process can function as an exogenous forcer, leading directly to state change.

Extreme climate events are becoming more frequent under, and attributable to, human-induced climate change (NASEM, 2016). Climate change thus impacts the Earth's ecosystems not just through gradual change, but by increasingly frequent extremes. Processes which are part of endogenous feedbacks may, in the extreme, cause direct and rapid change to ecosystems and thus participate, Janus-like, in both endogenous feedbacks and exogenous forcers. Ecosystems that are maintained in stable states due to the interaction of exogenous and endogenous factors can thus be expected to respond to the increasing frequency of extreme events, even in the absence of changes to processes traditionally considered to be exogenous. As a result, the changing balance between feedbacks and forcers caused by increasingly frequent extremes may have large impacts on presently-stable ecosystem state and may thus cause considerable change to ecosystems worldwide.

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CONFLICT OF INTEREST STATEMENT

The authors declare no financial or other conflicts of interest in the publication of this research.

DATA AVAILABILITY STATEMENT

Aaeolian flux data (Okin & Gillette, 2022) are available in the Environmental Data Initiative (EDI) Data Portal at https://doi.org/10.6073/pasta/6574d6825e7167d7bec234c97973c46d. Meteorological data are available in Anderson (2022) and Okin (2022b) in the EDI Data Portal at https://doi.org/10.6073/pasta/f1389876889b4390f01b904760a54604 and https://doi.org/10.6073/pasta/c3d84bfdf21f2ef432faed9f6f857eb7, respectively. Other field data (Okin, 2022a) are available in the EDI Data Portal at https://doi.org/10.6073/pasta/d6c8a362bc012028a13a4683b7e67634.

ORCID

Gregory S. Okin https://orcid.org/0000-0002-0484-3537
Abinash Bhattachan https://orcid.org/0000-0002-7102-8233

REFERENCES

Alba-Soular, D. 2019a. *102 mph Wind Gust Reported at San Augustin Pass*. Las Cruces, NM: Las Cruces Sun-News.

Alba-Soular, D. 2019b. *Weather: Wind Gust Hits 104 mph at San Augustin Pass; Train Derails*. Las Cruces, NM: Las Cruces Sun-News.

Alvarez, L., H. E. Epstein, J. Li, and G. S. Okin. 2012. "Aaeolian Process Effects on Vegetation Communities in an Arid Grassland Ecosystem." *Ecology and Evolution* 2: 809–21.

Anderson, J. A. 2022. "Jornada Basin LTER Weather Station Daily Summary Climate Data ver 121. Environmental Data Initiative." https://doi.org/10.6073/pasta/f1389876889b4390f01b904760a54604

Archer, S., D. S. Schimel, and E. A. Holland. 1995. "Mechanisms of Shrubland Expansion: Land Use, Climate or CO_2." *Climatic Change* 29: 91–9.

Archer, S. R., E. M. Andersen, K. I. Predick, S. Schwinning, R. J. Steidl, and S. R. Woods. 2017. "Woody Plant Encroachment: Causes and Consequences." In *Rangeland Systems: Processes, Management and Challenges*, edited by D. D. Briske, 25–84. Cham, Switzerland: Springer.

Beckage, B., W. J. Platt, and L.J. Gross. 2009. "Vegetation, Fire, and Feedbacks: A Disturbance-Mediated Model of Savannas." *The American Naturalist* 174: 805–18.

Beckett, H., A. C. Staver, T. Charles-Dominique, and W. J. Bond. 2022. "Pathways of Savannization in a Mesic African Savanna–Forest Mosaic Following an Extreme Fire." *Journal of Ecology* 110: 902–15.

Bestelmeyer, B. T., J. E. Herrick, J. R. Brown, D. A. Trujillo, and K. M. Havstad. 2004. "Land Management in the American Southwest: A State-and-Transition Approach to Ecosystem Complexity." *Environmental Management* 34: 38–51.

Bestelmeyer, B. T., N. I. Kalil, and D. P. C. Peters. 2007. "Does Shrub Invasion Indirectly Limit Grass Establishment Via Seedling Herbivory? A Test at Grassland-Shrubland Ecotones." *Journal of Vegetation Science* 18: 363–70.

Bonan, G. B., D. Pollard, and S. L. Thompson. 1992. "Effects of Boreal Forest Vegetation on Global Climate." *Nature* 359: 716–8.

Brown, J. R., and S. Archer. 1989. "Woody Plant Invasion of Grasslands - Establishment of Honey Mesquite (Prosopis-Glandulosa Var Glandulosa) on Sites Differing in Herbaceous Biomass and Grazing History." *Oecologia* 80: 19–26.

Browning, D. M., S. R. Archer, G. P. Asner, M. P. McClaran, and C. A. Wessman. 2008. "Woody Plants in Grasslands: Post-Encroachment Stand Dynamics." *Ecological Applications* 18: 928–44.

Buffington, L. C., and C. H. Herbel. 1965. "Vegetational Changes on a Semidesert Grassland Range from 1858 to 1963." *Ecological Monographs* 35: 139–64.

Cavanaugh, K. C., J. R. Kellner, A. J. Forde, D. S. Gruner, J. D. Parker, W. Rodriguez, and I. C. Feller. 2014. "Poleward Expansion of Mangroves is a Threshold Response to Decreased Frequency of Extreme Cold Events." *Proceedings of the National Academy of Sciences of the United States of America* 111: 723–7.

D'Odorico, P. 2000. "A Possible Bistable Evolution of Soil Thickness." *Journal of Geophysical Research: Solid Earth* 105: 25927–35.

D'Odorico, P., J. D. Fuentes, W. T. Pockman, S. L. Collins, Y. He, J. S. Medeiros, S. DeWekker, and M. E. Litvak. 2010. "Positive Feedback between Microclimate and Shrub Encroachment in the Northern Chihuahuan Desert." *Ecosphere* 1: 17.

D'Odorico, P., F. Laio, and L. Ridolfi. 2006. "A Probabilistic Analysis of Fire-Induced Tree-Grass Coexistence in Savannas." *American Naturalist* 167: E79–87.

D'Odorico, P., G. S. Okin, and B. T. Bestelmeyer. 2012. "A Synthetic Review of Feedbacks and Drivers of Shrub Encroachment in Arid Grasslands." *Ecohydrology* 5: 520–30.

Durán Vinent, O., and L. J. Moore. 2015. "Barrier Island Bistability Induced by Biophysical Interactions." *Nature Climate Change* 5: 158–62.

Fryberger, S. G., and G. Dean. 1979. "Dune Forms and Wind Regime." In *A Study of Global Sand Seas*, Vol. 1052, 137–69. Washington, DC: US Government Printing Office.

Fryrear, D. W. 1986. "A Field Dust Sampler." *Journal of Soil and Water Conservation* 41: 117–20.

Gherardi, L. A., and O. E. Sala. 2015. "Enhanced Precipitation Variability Decreases Grass- and Increases Shrub-Productivity." *Proceedings of the National Academy of Sciences* 112: 12735–40.

Gillette, D. A., and W. A. Chen. 2001. "Particle Production and Aaeolian Transport from a "Supply- Limited" Source Area in the Chihuahuan Desert, New Mexico, United States." *Journal of Geophysical Research* 106: 5267–78.

Gillette, D. A., D. W. Fryrear, T. E. Gill, T. Ley, T. A. Cahill, and E. A. Gearhart. 1997. "Relation of Vertical Flux of Particles Smaller than 10 mm to Aaeolian Horizontal Mass Flux at Owens Lake." *Journal of Geophysical Research* 102: 26009–15.

Goforth, B. R., and R. A. Minnich. 2008. "Densification, Stand-Replacement Wildfire, and Extirpation of Mixed Conifer Forest in Cuyamaca Rancho State Park, Southern California." *Forest Ecology and Management* 256: 36–45.

Huang, H., L. D. L. Anderegg, T. E. Dawson, S. Mote, and P. D'Odorico. 2020. "Critical Transition to Woody Plant Dominance through Microclimate Feedbacks in North American Coastal Ecosystems." *Ecology* 101: e03107.

Huenneke, L. F., J. P. Anderson, M. Remmenga, and W. H. Schlesinger. 2002. "Desertification Alters Patterns of Aboveground Net Primary Production in Chihuahuan Ecosystems." *Global Change Biology* 8: 247–64.

Lambers, H. 2003. "Introduction: Dryland Salinity: A Key Environmental Issue in Southern Australia." *Plant and Soil* 257: v–vii.

Li, J., G. S. Okin, L. J. Hartman, and H. E. Epstein. 2007. "Quantitative Assessment of Wind Erosion and Soil Nutrient Loss in Desert Grasslands of Southern New Mexico, USA." *Biogeochemistry* 85: 317–32.

Li, J., G. S. Okin, J. E. Herrick, J. Belnap, M. E. Miller, K. Vest, and A. E. Draut. 2013. "Evaluation of a New Model of Aaeolian Transport in the Presence of Vegetation." *Journal of Geophysical Research: Earth Surface* 118: 288–306.

Loehle, C. 1989. "Catastrophe Theory in Ecology: A Critical Review and an Example of the Butterfly Catastrophe." *Ecological Modelling* 49: 125–52.

Ludwig, D., D. D. Jones, and C. S. Holling. 1978. "Qualitative Analysis of Insect Outbreak Systems: The Spruce Budworm and Forest." *Journal of Animal Ecology* 47: 315–32.

Minnich, R. A. 2007. "Southern California Conifer Forests." In *Terrestrial Vegetation of California*, edited by M. Barbour, T. Keeler-Wolf, and A. A. Schoenherr, 502–38. Oakland, CA: University of California Press.

Monger, H. C., G. H. Mack, B. A. Nolen, and L. H. Gile. 2006. "Regional Setting of the Jornada Basin." In *Structure and Function of a Chihuahuan Desert Ecosystem: The Jornada Basin Long Term Ecological Research Site*, edited by K. Havstad, L. F. Huenneke, and W. H. Schlesinger, 15–43. Oxford: Oxford University Press.

Moreno-de las Heras, M., L. Turnbull, and J. Wainwright. 2016. "Seed-Bank Structure and Plant-Recruitment Conditions Regulate the Dynamics of a Grassland-Shrubland Chihuahuan Ecotone." *Ecology* 97: 2303–18.

Moreno-de-las-Heras, M., F. Lindenberger, J. Latron, N. Lana-Renault, P. Llorens, J. Arnáez, A. Romero-Díaz, and F. Gallart. 2019. "Hydro-Geomorphological Consequences of the Abandonment of Agricultural Terraces in the Mediterranean Region: Key Controlling Factors and Landscape Stability Patterns." *Geomorphology* 333: 73–91.

NASEM. 2016. *Attribution of Extreme Weather Events in the Context of Climate Change*. Washington, D.C.: National Academies Press.

O'Brien, R. M. 2007. "A Caution Regarding Rules of Thumb for Variance Inflation Factors." *Quality & Quantity* 41: 673–90.

Okin, G. 2022a. "Grass Exposure, Mesquite Height, and Plant Greenness at a Jornada Basin LTER Sand-Sheet Site, July and September 2019." Environmental Data Initiative https://doi.org/10.6073/pasta/d6c8a362bc012028a13a4683b7e67634.

Okin, G. 2022b. "Jornada Basin LTER Wireless Meteorological Station at MNORT Wind Tower Site: 5-Minute Summary Data, 2006 - Ongoing (Provisional) ver 1." Environmental Data Initiative https://doi.org/10.6073/pasta/c3d84bfdf21f2ef432faed9f6f857eb7.

Okin, G., and D. Gillette. 2022. "Aaeolian Dust Weights Sampled by BSNE Collectors in 18 Locations at the Jornada Basin LTER Site, 1998-Ongoing ver 103." Environmental Data Initiative https://doi.org/10.6073/pasta/6574d6825e7167d7bec234c97973c46d.

Okin, G. S. 2008. "A New Model for Wind Erosion in the Presence of Vegetation." *Journal Of Geophysical Research-Earth Surface* 113: F02S10.

Okin, G. S., P. D'Odorico, and S. R. Archer. 2009. "Impacts of Feedbacks on Chihuahuan Desert Grasslands: Transience and Metastability Driven by Grass Recruitment." *Journal of Geophysical Research* 114: G01004.

Okin, G. S., J. E. Herrick, and D. A. Gillette. 2006. "Multiscale Controls on and Consequences of Aaeolian Processes in Landscape Change in Arid and Semiarid Environments." *Journal of Arid Environments* 65: 253–75.

Peters, D. P. C., J. Yao, O. E. Sala, and J. P. Anderson. 2012. "Directional Climate Change and Potential Reversal of Desertification in Arid and Semiarid Ecosystems." *Global Change Biology* 18: 151–63.

Ratajczak, Z., P. D'Odorico, S. L. Collins, B. T. Bestelmeyer, F. I. Isbell, and J. B. Nippert. 2017. "The Interactive Effects of Press/Pulse Intensity and Duration on Regime Shifts at Multiple Scales." *Ecological Monographs* 87: 198–218.

Raupach, M. R., D. A. Gillette, and J. F. Leys. 1993. "The Effect of Roughness Elements on Wind Erosion Threshold." *Journal of Geophysical Research* 98: 3023–9.

Richardson, A. D., J. P. Jenkins, B. H. Braswell, D. Y. Hollinger, S. V. Ollinger, and M. L. Smith. 2007. "Use of Digital Webcam Images to Track Spring Green-Up in a Deciduous Broadleaf Forest." *Oecologia* 152: 323–34.

Rose, M. R., and R. Harmsen. 1981. "Ecological Outbreak Dynamics and the Cusp Catastrophe." *Acta Biotheoretica* 30: 229–53.

Runyan, C. W., and P. D'Odorico. 2014. "Bistable Dynamics between Forest Removal and Landslide Occurrence." *Water Resources Research* 50: 1112–30.

Sakamoto-Arnold, C. M. 1981. "Eolian Features Produced by the December 1977 Windstorm, Southern San Joaquin Valley, California." *The Journal of Geology* 89: 129–37.

Scheffer, M. 2009. *Critical Transitions in Nature and Society*. Princeton, NJ: Princeton University Press.

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke, and B. Walker. 2001. "Catastrophic Shifts in Ecosystems." *Nature* 413: 591–6.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. "Biological Feedbacks in Global Desertification." *Science* 247: 1043–8.

Soil Conservation Service. 1980. *Soil Survey of Doña Ana County, New Mexico*. Washington, DC: United States Department of Agriculture, Soil Conservation Service.

Stallins, J. A. 2005. "Stability Domains in Barrier Island Dune Systems." *Ecological Complexity* 2: 410–30.

Staver, A. C., S. Archibald, and S. A. Levin. 2011. “The Global Extent and Determinants of Savanna and Forest as Alternative Biome States.” *Science* 334: 230–2.

Turnbull, L., J. Wainwright, and R. E. Brazier. 2008. “A Conceptual Framework for Understanding Semi-Arid Land Degradation: Ecohydrological Interactions across Multiple-Space and Time Scales.” *Ecohydrology* 1: 23–34.

Turnbull, L., B. P. Wilcox, J. Belnap, S. Ravi, P. D’Odorico, D. Childers, W. Gwenzi, et al. 2012. “Understanding the Role of Ecohydrological Feedbacks in Ecosystem State Change in Drylands.” *Ecohydrology* 5: 174–83.

van Wagtendonk, J. W., J. A. Fites-Kaufman, H. D. Safford, M. P. North, and B. Collins. 2018. “Sierra Nevada bioregion.” In *Fire in California’s Ecosystems*, edited by J. W. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Schaffer, and J. A. Fites-Kaufman, 249–79. Oakland, CA: University of California Press.

Webb, N. P., M. S. Galloza, T. M. Zobeck, and J. E. Herrick. 2016. “Threshold Wind Velocity Dynamics as a Driver of Aaeolian Sediment Mass Flux.” *Aaeolian Research* 20: 45–58.

Westoby, M., B. Walker, and I. Noy-Meir. 1989. “Range Management on the Basis of a Model which Does Not Seek to Establish Equilibrium.” *Journal of Arid Environments* 17: 235–9.

Yizhaq, H., Y. Ashkenazy, and H. Tsoar. 2007. “Why Do Active and Stabilized Dunes Coexist under the Same Climatic Conditions?” *Physical Review Letters* 98: 188001.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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