MICROHYDROLOGY OF A PERIODICALLY-PATTERNED ARID
GRASSLAND: SURFACE WATER DYNAMICS, INFILTRATION, AND SOIL WATER
ACCUMULATION IN A SIMULATED RAINFALL EXPERIMENT

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By

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ON MY HONOR, I HAVE NEITHER GIVEN NOR RECEIVED
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TABLE OF CONTENTS

ABSTRACT:

Periodically patterned landscapes are observed in semi-arid and arid ecosystems throughout the world, occurring in several characteristic patterns dependent on factors such as slope and water availability. One such type of self-organized patchy landscape is distinguished by a banded pattern of alternating vegetation strips and bare ground that follow the contours of a gently sloping hillside. The spatial and temporal movement of water and sediment across banded landscapes is not fully understood, and we aimed to address some conflicting conceptions existing in literature regarding water movement into the soil matrix of bands, paying particular attention to how sediment movement might inform water accumulation and infiltration, and vice versa. We wanted to investigate exactly where and how surface water and soil water accumulates across interpatch and patch zones. More generally, we aimed to examine the role of microhydrology and microtopography in pattern maintenance in semi-arid ecosystems. We carried out a simulated rainfall experiment in eight plots at Chico Basin Ranch, which is a shortgrass prairie ecosystem dominated by blue grama grass that receives an average 16 inches of annual rainfall. We watered an even inch (2.54cm) across a plot of interpatch (containing a dying zone of vegetation, and a bare soil zone) and the closest downslope vegetated patch (containing a grass interstitial zone of bare soil, interspersed with grass bunch hummocks of blue grama). While watering, we took qualitative observations on surface water movement and accumulation across the different zones of the interpatch and patch. After fifteen minutes, we stopped rainfall and measured the height of surface water along a transect downslope through the plot, and then extracted soil cores to assess soil water content from each zone after surface water had completely infiltrated or drained from the study area.

During watering, we observed that a consistent film formed within minutes in the interpatch zones of our plot and activated sheetflow downslope into the vegetated patch zones. Water always accumulated in pools in the grass interstitial zones of the vegetated patch and remained long after watering ended. In several plots a pool also formed in the bare zone of the interpatch but did not persist for long after rainfall ended. Surface water was highest in the vegetated patch interstitial zones, where deep pools formed. Pooling in the interstitial zone of the vegetated patch seems to play an especially important role in water use-efficiency of blue grama in our study area. In all plots, we found that percolation depths were greater in vegetated patch zones than interpatch zones, and that total volume of soil water and soil water percentages were also greater in vegetated patch zones than in the interpatch zones. Our results highlight how microtopographic and microhydrologic mechanisms are essentially coupled in the runoff-runon system of patterned landscapes. This coupling facilitates water-use efficiency within the vegetated patch, promoting ecosystem resiliency in the face of increasing aridity.

INTRODUCTION:

Banded vegetation patterns in arid and semi-arid ecosystems have been documented across the world, with studies mostly concentrated in areas of Australia and Africa (Fig. 1). These banded patterns consist of alternating strands of bare soil and dense vegetation, usually following the contour of a slight slope (Fig. 2). These landscape patterns are assumed to form due to limited water availability in arid and semi-arid environments. Banding generally occurs when there is not sufficient rainfall to maintain a homogenous vegetation cover but where gentle, uniform slopes allow for sheet-flow to compensate for this lack of water. In rainfall events, water forms a film on

the bare soil and runs off downslope via sheet-flow into the nearest vegetation patch, where most of the water quickly infiltrates due to the greater surface roughness and higher infiltrability in the vegetation groves (Dunkerley & Brown, 1995, 1999; Thiery et al., 1995). Some water might not soak into the grove area and continues to run downhill, but usually there is insufficient water for the vegetation at the downslope end of the grove to thrive, hence the return to bare soil that allows for rainfall to again form a film on the soil surface and accumulate into sheet flow, running onto and infiltrating into the next band of vegetation.

Positive feedback loops are generally regarded as contributing to the consolidation and maintenance of the vegetation patterns through long distance inhibition and short distance facilitation processes (Lefever & Lejeune, 1997; White, 1971). Short distance facilitation between individual plants on a water-limited slope occurs by neighboring plants increasing soil porosity and root channels, allowing for greater infiltration rates under the plant canopy, which in turn allows for greater water uptake by plants roots, sustaining them in the arid environment.. Additional mechanisms that increase local infiltration rates in the vegetated patches include macropores and tunnels created by soil invertebrates that are more likely found within the vegetation patches than in the bare soil (Ludwig et al., 1999; Valentin et al., 1999; Bonachela et al., 2015). These processes all contribute to the positive infiltration feedback loops in the groves, meaning it is easier for plants to establish and propagate nearby other already established plants, as more water is captured in these areas.

However, higher infiltration rates of the vegetation patches in a water limited environment mean that less water is available downslope for plants competing for the same supply of water to survive. The limited water is captured and infiltrated first by the most upslope plants of the vegetation patch, meaning that plants at the back side of that patch will have the smaller quantity

of water available to sustain them. The plants in the patch consume so much water that the plants downslope of the patch cannot survive, hence the transition back to bare ground interpatch.

The average amount of rainfall over the last 15 year period has been shown to determine the scale on which short-distance facilitation and long-distance inhibition processes interact, including impacting the spacing and widths of the bands. Lower yearly rainfall totals lead to thinner bands of vegetation (described as the infiltration zone) and thicker interbands of bare soil (described as the water shedding zone), as less rainfall can sustain a smaller population of neighboring plants in the infiltration zone (Valentin & d'Herbès, 1999; Dunkerley, 1997; Vega & Montana, 2011).

The sheet-flow necessary for the runoff-runon system characteristic in banded vegetation landscapes is slope dependent, given that a slight slope is needed to impose a flow direction on the water, otherwise the water would move out in all directions. While spatial self-organization into periodic patterns would still be possible, the particular spatial arrangement of the banded vegetation would not naturally occur in a flat landscape, although slope and non-slope dependent bands can be modeled on computers (Dagbovie & Sherratt, 2014). Different non-slope dependent vegetation patterns exist in nature and they can also be modeled. These range from evenly spaced vegetation spots, to labyrinths, to evenly spaced gaps (often referred to as fairy rings), depending on the amount of available precipitation.

To model the runoff-runon mechanism characteristic of slope-dependent banded vegetation patterns and to confirm that the flow of water through the system does create and inform these banded patterns, Rietkerk and others combined three partial differential equations that describe the dynamics of plant density, soil water, and surface water. Using this model, they were able to reproduce general distinct banded patterns broadly consistent with those that have been

observed in arid areas across the globe. This successfully modeled patterning supported their hypothesis that this spatial self-organization of vegetation banding can be caused by the faster infiltration rates of water into vegetated ground than into bare soil, leading to net displacement of surface water into vegetated patches (Rietkerk et al., 2002). This general result has been replicated in multiple models (Dunne et al., 1991; Thiery et al., 1995; Gandhi et al., 2020; Galle et al., 1999).

While simple mathematical models are able to generate banding patterns similar to those observed in nature, these models appear unable to capture important aspects of the dynamics and maintenance of the banded patterns observed in nature. Ritekerk's simple model, along with many others, all simulate banded vegetation patterns migrating upslope over time (Gandhi et al., 2020; Dagbovie & Sherratt, 2014). This is due to the infiltration feedback in the runoff-runon system that supplies most water to the upslope front of the vegetation grove, while the vegetation at the downslope end receives very little of the runon, decaying and dying in contrast to the flourishing upslope plants that slowly propagate uphill. Although some bands have been observed to migrate upwards over time in the field (Ludwig & Tongway, 2001; Leprun 1999; Deblauwe et al., 2012), many vegetation bands have shown no signs of movement, even over 20 years of observation (Dunkerley, 2018). If models only incorporate this infiltration feedback, as Rietkerk does, they cannot be representative of all banded vegetation patterns, given that not all banded patterns migrate upslope. Many of these simple models also assume constant rainfall, but obviously rainfall happens in discrete events of varying intensity. This varying intensity can mean that sometimes there is not enough water to activate sheet-flow and the runoff-runon system, and sometimes there might be so much water that most precipitation simply runs out of the system. These models also abstract away the particular biology of the plants established in the vegetation groves, which overlooks the important roles that specific plants might play in maintaining and facilitating these

banded patterns. Furthermore, focusing only on water flow and infiltration in the system fails to acknowledge the role sediment transport and deposition might play in banded pattern landscapes.

To address these oversights and investigate the mechanisms of sediment movement in banded vegetation patterns, which can perhaps describe both the migrating and stable bands observed around the world, Dunkerley and Saco have examined the interactions of microtopography and microhydrology in pattern formation and resilience. From this perspective, vegetation banding occurs along with a unique pattern of sediment erosion and deposition, creating a stepped microtopography similar to the riffle-pool sequence (Richards, 1976) observed in certain river systems. Small-grain clay and silt sediments are transported from the bare soil interpatch via sheet flow and deposited as the sheet-flow velocity slows at the upslope front of the interpatch, due to the higher surface roughness of the vegetation and the thinner film of water caused by the higher infiltration rates of the vegetation patch. This erosion-deposition process leads to a slight depositional ridge at the downslope end of the bare interzone (Fig. 3), with the vegetated groves located right below this ridge on a generally slightly steeper slope than the bare intergrove (Dunkerley & Brown, 1999). When the runoff-runon system is activated in a substantial enough rainfall event, sheet-flow will run downslope, creating an ephemeral pool at the end of the bare patch until there is enough accumulated water to overflow the ridge and soak into the vegetation zone (Boaler & Hodge, 1964). This pooling and draining at the downslope end of the bare intergrove, coupled with low intergrove infiltration rates, leads to laterally extensive water distribution into the most upslope area of the vegetation grove.

Soil crusts are another mechanism influencing the movement of surface water and infiltrability in the interpatch and vegetated patches of the banded pattern landscape. The distinct makeup and spatial arrangement of soil crusts throughout an intergrove-grove area has been

detailed extensively (Vandervaere et al. 1997; Valentin & d'Herbès, 1999). Soil crusting greatly reduces infiltrability, acting like a layer of armor between the water and the soil beneath, enhancing runoff from the interpatch into the vegetated grove. Sealed surface crusts are generally only observed in the intergrove bare patch, as denser plant cover and leaf litter has been found to prevent most types of soil crusts from forming in the central area of the vegetated patch (Valentin et al., 1999; Dunkerley & Brown, 1995). Soil crusting therefore works to enhance the runoff-runon system providing water from the bare, crusted soil intergrove to the downslope vegetated grove with its higher soil infiltrability.

Little sediment transport is possible across the vegetation patch, increasing the efficiency of the banded landscape not only at water conservation but also at soil conservation (Dunkerley & Brown, 1999). Therefore the banded, concave upward bench system of the steeper grove and concave upward intergrove is very effective at reducing hillslope erosion (Bochet et al. 2000, Saco et al. 2007), acting as an almost fully closed hydrological and topographical system, with very little net outflow or sediment leaking from the landscape (Valentin et al. 1999, Ludwig et al. 1999).

This microhydrological/microtopographical perspective emphasizes the resiliency of banded ecosystems to climactic stressors, but it can call into question the endogenous nature of the banded pattern's formation and maintenance. Could sediment transport and deposition be driving the system, with the vegetation incidentally establishing and propagating due to the increased soil moisture of the ephemeral pools, or could this step-like microtopography only have formed in a pre-established banded landscape? Considered from the Dunkerley microtopographic view, the vegetation band pattern could perhaps be incidental in the landscape, while from the simple modeling perspective detailed earlier, vegetation can form banded patterns irrespective of the

presence of sediment transport processes. Therefore, the role that plants are playing in these banded vegetation patterns is called into question by these two perspectives.

This study explores the interaction of the runoff-runon process through experimental watering of the intergrove-grove area in a semi-arid grassland. In particular, we carefully observed the temporal and spatial dynamics of the sheet-flow and water accumulation in the intergrove and grove areas, and investigated the depth of the surface water sheet-flow and the soil water content throughout the different zones.

The simple model and microtopography perspectives guide the formation of my research framework and hypotheses. From a modeling perspective, we would also expect infiltration to be high and thus soil water content to be high in the zones of vegetation, and we would expect no water to pool anywhere, with sheet flow running off the intergrove soil and continuously soaking into the vegetation grove. We would see a thin film of surface water in vegetation grove areas (as water is soaking in very fast due to roots channels, higher biological activity, etc), and a high thickness of water film in the bare and dying zones of the intergrove areas. This surface water distribution is due to assumptions, made by Rietkerk and others, that do not explicitly incorporate sediment transport into the model framework. In the simple model framework, therefore, no mechanism for pooling exists given that there is no mechanism for sediment transport in these models. Similarly, surface roughness is not accounted for, so we would not expect velocity to change whether water encounters a rougher vegetated patch or the smoother interpatch.

From the microtopography perspective we would expect sheet flow from the bare soil and dying areas, with water running off downslope and running onto the grass zone, where we would see high levels of infiltration, leading to deep wetting horizons, and high soil moisture contents. From the Dunkerley schematic (Fig. 3), we would also expect significant pooling in the bare zone

right before the front of the grove, and the grove would be steeper than the intergrove, with less standing water due to high infiltration rates. We would expect ephemeral ponds in the area just before the front grove grasses, with the sheet flow collecting in these ponds, reaching capacity and overflowing downslope into the groves. This pooling would lead to sediment build-up contributing to the slight ridges, with this process happening in each band, informing erosional and depositional areas much like the riffle-pool sequence in a river. The microhydrology of the system is coupled with the band microtopography, working to increase the water-use efficiency through low hydraulic connectivity, meaning that very little water leaks out of the banded vegetation pattern and goes instead to sustaining the plant population of this arid ecosystem.

METHODS:

Our simulated rainfall experiments were conducted at Chico Basin Ranch, part of the Central Shortgrass Prairie ecoregion in Colorado. Our study sites were located immediately east of the airstrip at Chico Basin Ranch, where periodic patterns have developed since 2011 (Kummel et al., 2021). Chico Basin is best characterized as a high plains ecosystem, and the particular area in which we conducted our study is that of a typical shortgrass prairie landscape. Blue grama, a shallow-rooted perennial bunch-grass (*Chondrosum gracile*), is the dominant grass throughout much of Colorado's eastern plains and also at our site (Benedict, 2008). Due to its dense, fibrous root structure, C_4 metabolism, and high water-use efficiency, blue grama is remarkably drought tolerant, able to survive on only seven inches of rain annually and respond rapidly to small amounts of rainfall (Benedict, 2008). Blue grama is also very resistant to grazing pressures, whilst being one of the most nutritionally dense food-sources for the cows of Chico Basin Ranch (Rondeau et

al., 2016). There are a few other perennial grasses, such as buffalograss (*Bouteloua dactyloides*) and needle and thread (*Hesperostipa comata*) at our site, and some annuals and cacti such as cholla (*Cylindropuntia*), prickly pear cactus (*Opuntia*), and Russian thistle (*Salsola*) (Kelso et al., 2007).

However, blue grama alone forms the periodic pattern that has been emerging over the past five years at our study site. Blue grama grass cover has declined in the shortgrass habitat by 62% below 1999 levels, due to increasing drought (Rondeau et al., 2016). Severe drought years and increased temperatures have been recorded over the past fifteen years in the Chico Basin ecosystem, and while most plants in the ecosystem have suffered extreme losses, the decline in the blue grama population has especially important implications. Its decline is notable not only for its economic implications for ranchers, but also for its seemingly non-random, spatial self-organization emerging out of the increasingly harsh climactic conditions.

Our study areas were selected by first identifying fourteen suitable sites in the Chico Basin landscape close to the air strip, and randomly choosing eight plots from these fourteen (Fig. 13). The experimental areas of the chosen eight plots were 4.5m by 2m, beginning at the most upslope end with a zone of dying vegetation, moving through the bare soil zone and into the vegetated patch most immediately downslope of the interpatch. After demarking the plot area, we placed plastic cups cut to 2.54cm high at different areas (Fig. 12*)* throughout the plot to help us cover the whole plot with an even inch of water. This inch of water simulates a common summer rainstorm, which is a normal occurrence in this area throughout July, August, and September. We also took four 15cm deep soil cores from the dying, bare, grass interstitial, and grass bunch zones outside of the demarcated plot. We took them outside of the 4.5m by 2m area so that we could examine the soil moisture content of the unwatered zones closest to our area of interest without changing the microhydrology of the plot when we began our simulated rainfall experiment.

We watered the plot for fifteen minutes as evenly as possible, moving side to side in such a way that the middle was not watered more than the upslope and downslope ends. While watering, we took qualitative notes on where the water was accumulating and moving through the different zones, recording the process using drone footage taken from a few meters above the plot. Once the fifteen minutes of the rainfall simulation was over, we continued recording qualitative observational notes on surface water movement. Additionally, we measured the height of the surface water at 20cm intervals along a transect through the plot, from the upslope to the downslope end. We remeasured the surface water heights every five minutes until twenty minutes had passed since the rainfall simulation ended.

When there was no standing water left anywhere in our plot, we began taking 15cm deep soil cores within the study area. For each trial, we first took four cores in the dying zone of the interpatch, then four cores in the bare zone of the interpatch, then four respectively from the grass bunch and grass interstitial zones of the vegetated patch. Then we repeated this process so that we had a total of eight soil cores for each of the zones in the study area after the simulated rainfall experiment.

This coring occurred in all eight trials, but two of the trials were conducted a little differently. Plots 1 and 6 were the first rainfall simulation trials we conducted, and we had not yet decided to include the grass bunch zone cores in our measurements. Therefore, for each of these two trials, we only have twenty-four cores in total from after the rainfall simulation - eight from the dying zone, eight from the bare zone, and eight from the grass interstitial zone. The other six trials had thirty-two cores due to the addition of the eight grass bunch cores. All these soil cores were taken back to the lab in sealed bags, weighed, and put in the oven at roughly 65 degrees Celsius for forty-eight hours. After this time, in which theoretically all the water within the soil

should have evaporated, we reweighed the cores, which allowed us to calculate how much of the total soil had been water weight. Subtracting the dry soil core weight from the wet soil core weight then gave us the moisture content present in the soil zones. We also weighed, dried, and reweighed the soil cores taken before the simulated rainfall experiments began in the eight trials. These served as our moisture content control and normalizing values which we used when calculating final values of total volume of soil water and soil water percentage. In particular, we subtracted these control values from our rainfall simulation values to calculate the amount of water gain above the baseline.

To conduct our statistical analyses of percolation depth, total volume of soil water, and soil water percentage, we used one-way ANOVAs on RStudio separately for each experimental run. We first used Levine's Test to assess the homogeneity of variance assumption in the underlying data. If Levine's Test was not significant, we would perform a regular one-way ANOVA, comparing the data by zones, and then perform Bonferroni pairwise comparisons. If Levine's Test was significant, we used the Kruskal-Wallace Test and again performed Bonferroni pairwise comparisons.

RESULTS:

Surface Water Movement:

For the eight plots on which the rainfall simulation was carried out, the average length of time for a consistent film of water on the dying and bare zones of the interpatch to form was 3.25 minutes since beginning watering. Immediately as the interpatch filming became consistent and sheet-flow was activated, the grass interstitial zone in the vegetated patch began to fill with water.

These ephemeral pools continued filling in the grass interstitial zone, and in six of the eight trials, pools also formed in the downslope end of the bare zone. After an average of 9.13 minutes, the grass interstitial zone pools spilled downslope out of our plot area. This downslope spillage happened in all eight trials. In the six trials that had both bare zone and grass interstitial zone pools, the bare zone pool always spilled sideways out of our plot area much sooner than the grass interstitial pools spilled either downslope or sideways out of the plot.

We chose two representative plots to discuss that best illustrate surface water movement in our eight plots: Plot 6 (Fig. 9) and Plot 13 (Fig. 10). In Plot 6, the pool in the bare zone of the interpatch was at its greatest recorded depth of 10mm immediately after the rainfall simulation ended (Fig. 9). Five minutes later, however, the bare zone pool was only 1mm deep and had completely disappeared at the ten-minute timestamp after watering ended (Fig. 9). The most upslope grass interstitial pool along our transect in Plot 6 was 9mm deep immediately after watering ended, while the downslope grass interstitial pool was 20mm deep at the same zero-minute timestamp (Fig. 9). Five minutes later, the upslope pool in the vegetated patch was 6mm deep and the downslope vegetated patch pool was 18mm. At the ten-minute timestamp, the upslope pool was 5mm deep, and the downslope pool was 14mm, meaning neither pool had decreased in height by 50%. Fifteen minutes after watering ended, the upslope pool was 2mm deep and decreased to 1mm deep at the twenty-minute timestamp, after which we stopped taking surface water depth measurements. Meanwhile, the downslope pool was still 10mm deep at the fifteen-minute timestamp and 9mm deep at the twenty-minute timestamp, only just below 50% of its original depth (Fig. 9).

In Plot 13, there was only a very small pool of 1mm in the bare zone that disappeared after five minutes (Fig. 10). The most upslope pool in the vegetated patch was 18mm deep immediately

after the rainfall simulation ended, and decreased to 14mm after five minutes, to 10mm after ten minutes, and then to 5mm after fifteen minutes, at which depth the pool remained twenty minutes after watering ended (Fig. 10). The middle pool in the vegetated patch was 28mm at its deepest point immediately after watering ended, and this depth increased to 29mm at the five-minute timestamp. The greatest recorded depth in the middle pool ten minutes after watering ended was 21mm, then decreased to 15mm at the fifteen-minute timestamp. This depth, however, increased again to 22mm twenty minutes after watering ended. The most downslope pool began with a water surface height of 18mm, which decreased to 16mm after five minutes, then 15mm after ten minutes, then 12mm after fifteen minutes, at which depth the pool remained at the twenty-minute timestamp (Fig. 10). Both the middle and downslope pools, even twenty minutes after the rainfall simulation finished, had surface water heights more than 50% the original height recorded at the zero-minute timestamp immediately after watering ended. There was also a small pool in the dying zone that remained throughout these twenty minutes of recording in Plot 13 between slightly raised clumps of dead plants (Fig. 10).

Percolation Depths:

Considering all the percolation depths in the eight trials, percolation depth was statistically significant in all cases, with F values ranging from 9.44 to 45.95 and p values ranging from 0.003 to less than 0.0001 (Table 1). The grass bunch zone had the greatest percolation depths in all eight plots and were always statistically significantly higher than the dying and bare zone percolation depths (Fig. 7). The grass interstitial zone percolation depths were greater than both dying and bare zones in seven out of the eight plots, and this difference was statistically significant in six of the seven plots (Fig. 7). While the dying zone had the smallest percolation depths in five out of the eight rainfall simulations, in the other three trials the bare zone had smallest percolation depths.

The dying and bare zones of the interpatch were also never significantly different from each other (Fig. 7 & 8). The grass interstitial and grass bunch zones of the vegetated patch were significantly different from each other in four of the six rainfall experiments that contain coring data from both vegetated patch zones (Fig. 7).

Total Volume of Soil Water (g H_2O/cm^2):

As with percolation depths, total volume of soil water was statistically significant in all cases, with F values ranging from 5.71 to 26.18 (Table 2). P values ranged from 0.01 to less than 0.0001 (Table 2). Aside from Plot 12, the grass bunch zones always have the highest total volume of soil water and this difference is statistically significant in four of the six trials that contain grass bunch core data (Fig. 7). The grass interstitial zone has lower or the same total volume of soil water as the grass bunch zones in five out of six plots, as in the Plot 12 the grass interstitial zone has a higher total volume of soil water than the grass bunch (Figure 1). The grass interstitial zone has a greater total volume of soil water than the bare and dying zones in six out of the full eight rainfall experiments (Fig. 7 & 8). In five out the eight trials, the dying zone has the least total volume of soil water, but this is not statistically significantly different from the soil water content in the bare zone, except in Plot 12 (Fig. 7 & 8).

Soil Water Percentage (%):

Again, the soil water percentage was unanimously statistically significant, with F values ranging from 2.99 to 53.69, and p values from 0.04 to less than 0.0001. As with the total volume of soil water, soil water percentages were highest in the grass bunch zones for all trials except Plot 12 (Fig. 7). This difference was only statistically significant in three of the six trials that contained grass bunch coring data. The grass interstitial zone had the same or greater soil water percentages than the dying and bare zones in seven of the eight rainfall experiments (Fig. 7 & 8). As with the

soil water content results, in five of the eight trials, the dying zone had the lowest soil water percentage, but this also was only statistically significantly different from the soil water content in the bare zone in Plot 12 (Fig. 7 & 8).

DISCUSSION:

This study explores the temporal and spatial dynamics of sheet-flow and water accumulation due to the runoff-runon process through experimental watering of the intergrove-grove area in a semi-arid grassland. We carefully observed the movement of water as it formed a film over the interpatch area and began to run downslope into the vegetation patch, collecting in ephemeral pools in between the hummocked bunch grasses that make up our dense vegetation zone. After 2-5 mins the runoff-runon system had been activated as a consistent film developed over the intergrove area and sheet-flow began draining downslope. Initially, after sheet-flow had been activated, the water film was thinner on the vegetated patch area, even though all areas were watered equally and the sheet-flow over the bare interpatch was draining directly into the vegetated patch. The dense vegetation can easily soak most of the water falling directly onto it, likely due to higher infiltration rates that have been observed in our system (Kummel et al., 2021) and in other banded landscapes by modelers parameterizing their models (Thiery et al., 1995, Rietkerk et al., 2002, Dagbovie & Sherratt, 2014) and by microhydrologists (Saco et al., 2007) alike. This difference in surface water film thickness exactly matches the model simulations, where the film is thinner or absent in the vegetation patch and thicker in the bare interpatch (Fig. 4).

However, in a very short amount of time (between 2-5 minutes) pools in our experimental study started to fill in the grass interstitial zones of the vegetated patch, providing ample water to

the infiltration zone. From our observations, the extra input of water into the densely vegetated area due to the sheet-flow from the interpatch caused this pooling between the grass bunches. Infiltration continued, with the pools continuously soaking water while being refilled by the upslope sheet-flow. For the eight trials, it took 7-12 minutes for the vegetated patch pools within our plot to become saturated and spill into the grass downslope of our plot area. Before spillage downslope occured, there were occasional spills sideways outside of our plots, usually from a pool observed in the bare soil zone of the interpatch as the pools extended beyond our study area. The pool that often formed in the bare zone was reminiscent of the ephemeral intergrove pool identified in the microhydrology perspective schematic (Fig. 3).

Given that we did not measure slope change through the interpatch-patch area, we cannot confirm whether our plots contained the depositional ridge associated with the interpatch "ephemeral pond" shown in the Dunkerley schematic, or whether the vegetated grove was steeper than the bare soil intergrove. However, as the film formed in the bare interpatch and moved downslope as sheetflow, we also saw an opacity of the water indicative of small silt and clay sediments suspended in the film of water, being transported and deposited downslope in the runoff-runon process. At least theoretically, the sediment transport could have contributed to a formation of a depositional ridge, however, careful elevational profile measurements are needed to substantiate this conjecture. The observed sediment transport is also congruent with the microhydrology perspective that incorporates the coupling of erosional/depositional mechanisms with vegetation banding (Ludwig et al., 2005; Saco et al., 2007; Dunkerley, 1997; Dunkerley & Brown, 1999; Leprun, 1999). That we see surface water pooling at all in the downslope end of the bare interpatch most closely follows the predictions for surface water movement from the microtopography perspective. The simple modeling view does not predict any pooling occurring,

given that it does not consider sediment transport causing microtopographical differences that could allow for ephemeral ponds (Rietkerk et al., 2002).

To summarize the initial observations and interpretations of the sheet flow and apparent infiltration, for the first 2-5 minutes of watering, the high infiltration rates in the vegetation patch prevent any film from forming and the low infiltrability of the bare interpatch that begins to film and activate the runoff-runon system follow predictions from the simple model view (Rietkerk et al., 2002). Then, the occurrence of pooling after 2-5 minutes is more congruent with the microtopography perspective.

However, both conceptual models (actual models and microtopographical conceptual models) fail to predict the ponding of surface water in the vegetated patches that occurred without fail in all our trails. In the interstitial zone between each hummock of grass in the densely vegetated area, pools formed when the runoff-runon system was activated and began draining more water downslope into the vegetation patch. These pools remained long after the fifteen minutes of our experimental watering was over. For many of the pools in the vegetation patch, even after twenty minutes had passed, they were still 50% full. The pool identified at the downslope end of the bare soil zone, however, had usually drained completely twenty minutes after watering ended.

Not only were the vegetation patch interstitial pools long-lasting, it took 9-12 minutes from initial pond formation in the vegetation patch for these pools to start spilling over downslope out of our plot area. Although the high infiltrability of the vegetated zone was continuously draining the pools, runon from the interpatch sheetflow would offset this infiltration by providing water to refill the pools. The vegetation interstitial pools increased the chances of greater amounts of water infiltrating into the grass bunches by preventing surface water from draining downslope. The runoff-runon process coupled with the microtopography to sustain constant standing water

throughout the grass bunches of the vegetated patch, which neither the model nor microtopography perspectives predict.

We find exceptionally clear differences in surface water accumulation between the interpatch and patch areas throughout our trials, and the spatial distribution of soil water mostly follows the distribution of surface water depth, with a few significant discrepancies. The dying and bare zones of the interpatch area had very few areas of standing water (aside from the bare zone pool already discussed), with generally very low surface water height. The percolation depths in the dying and bare zones were similarly thin, and the lack of high standing water and deep infiltration in most of the interpatch was also reflected in their low total volume of soil water. Although the bare zone pool of the interpatch that occurred in most plots had high standing water, its low percolation depths and low total volume of soil water did not reflect this accumulation of surface water.

While other studies have included the observation that the vegetated patches contain large amounts of bare ground, our study is the first to quantify differences in water gain by the vegetation hummocks/bunches and the bare interstitial space. Our delineation of the vegetated patch into the grass interstitial zone (where the ephemeral pools arise) and the grass bunches (the hummocks of perennial vegetation themselves, raised a centimeter or two above the interstitial zone surface) is an additional parameter that has not been described or included in literature until now. This distinction is important as we identified that the microhydrological pathways of water infiltration differ between those two and the interactions of these pathways seem to enhance the greater infiltrability and total volume of soil water observed in both zones of the vegetated patch.

The grass interstitial zone of the vegetated patch had the highest recorded standing water, and while its percolation depths were greater than those of the interpatch zones, these depths of

wetting were never as profound as those recorded in the grass bunch soil. The grass bunch zones of the vegetated patch, in fact, had the greatest recorded percolation depths across all eight trials, even though there was no surface water accumulation above the grass bunches due to their hummocked nature. These grass bunch percolation depths were always significantly different from the percolation depths of the bare and dying zones. The grass interstitial percolation depths were also usually significantly different from the bare and dying zones, highlighting the greater ability for the water to enter the soil matrix of the vegetated patch.

While the average depth of wetting is greatly dependent on the type of vegetation found in the banded landscape, similar trends in percolation depths of interpatch and patch have been found in other studies. Between our eight trials, we found percolation depths of 25-42mm under the interband surfaces - both dying and bare zones - and 40-85mm deep wetting fronts within the vegetated bands - both grass bunch and grass interstitial zones (Fig. 7 & 8). Dunkerley and Brown found a depth of wetting of only 2-4mm under the interpatch bare, sealed surfaces, while within the vegetated bands depth of wetting values were 10-15mm (Dunkerley & Brown, 1995) in their study site of banded chenopod shrubland. Meanwhile, Ludwig found the wetting front depth in a vegetated band of mulga trees, which have greater root extension than either blue grama or chenopod shrubs, to be up to 700mm, while in the bare interpatch the wetting front depth was around 120mm (Fig. 6). In literature and at our study site, we see that percolation depths are generally much greater in vegetated bands than in the unvegetated interpatch.

The total volume of additional soil water (g H2O/cm^2) in the zones generally reflected the greater percolation depths measured in the vegetated area compared to the interpatch area. Both the grass interstitial and grass bunch zones had greater percolation depths and typically greater total volumes of soil water than the interpatch bare and dying zones. These findings confirm what

previous literature has already described, that higher infiltration rates in the vegetated patches lead to more soil water immediately after rainfall in the vegetated grove than in the intergrove (Thiery et al., 1995; Ludwig et al., 1999, 2005).

The percent water content also followed a similar pattern as the total volume of soil water, but the slight differences observed might be indicative of potential differences in bulk density between zones. To elaborate: we assume bulk density would be higher in depositional areas (such as the bare and interstitial pools) where water is settling and depositing sediments into the soil matrix, reducing pore space and making the soil denser. This denser soil would have a smaller soil water percentage than might be expected given the total volume of water in the soil, because in the calculation for percent soil moisture, the grams of soil water is divided by the soil weight. Higher soil density would lead to a calculated lower soil water percent. This discrepancy is most consistently observed in the bare zone, with a slightly taller bar for bare zone total volume of soil water and a more compressed bar comparatively for bare zone soil water percentage (Fig. 7 & 8). We see these compression differences only minutely, but these differences are generally consistent with our assumptions concerning increased bulk density. Some investigations have found significant differences between patch and interpatch bulk densities (Stavi et al., 2009) but others have not found them clearly (Dunkerley & Brown, 1999). Further studies could identify what role, if any, bulk density might be playing in pattern formation and maintenance.

Bulk density variations are only one small impact of erosion/deposition processes at work in patterned ecosystems. Understanding sediment transport dynamics in the runoff-runon system is a major point of focus for many microtopographists studying banded landscapes. Sediment has been found to be eroded from the bare intergrove, carried by sheetflow downslope, and deposited into pools as crusts (Dunkerley & Brown, 1995, 199, Saco et al., 2007; Boaler & Hodge, 1964;

Valentin & d'Herbès, 1999). Some clay and silt particles likely make their way into the soil matrix via infiltrating water and thereby increase the bulk density of the soil, but many of these particles simply settle on the soil surface, forming soil crusts. Surface soil crusting has been shown by microtopographists and modelers alike to inform many of the differences in infiltration rates in patterned landscapes (Saco et al., 2007; Thiery et al., 1995; Rietkerk et al., 2002).

Soil crusts were clearly identifiable in our study site, not only in the dying and bare zones that make up the interpatch, but also in the grass interstitial zone of the vegetated grove. The soil crusts in the grass interstitial zone were much more cracked and appeared to be thicker than the soil crusts in the bare zone pool, according to qualitative field notes. The recorded differences in percolation depths and soil water content of the ephemeral pool in the bare zone and those in the grass interstitial zone are perhaps due to the zone-specific make-up of the soil crusts. The crusting differences could be caused by variations in slope and associated velocities, which influence where and when particular sediments settle out of the surface water (Thiery et al., 1995; Penny et al., 2013). The importance of the crusts suggested by our qualitative observations underscores the need to take quantitative measurements in the field in our next season.

The greater pressure that higher standing water (associated with the bare and interstitial pools) exerts should lead to an increase in infiltration rates directly below the high pressure area, which would reflect in our results with far greater soil water content and percolation depths wherever pooling occurs. However, soil water content and percolation depths were very low throughout the interpatch, especially in the bare zone, suggesting that some other process or processes, likely including soil crusting, were preventing high infiltration rates from occurring in the interpatch. The crusts in the vegetated patch interstitial zones also may impede some infiltration

from occurring directly into the soil column below the ponds, as percolation depths are still not as great in these pools as they are in the grass bunches.

Even though there is no increased pressure influencing infiltration rates from standing water above the bunches, percolation depths and soil moisture content are typically highest in the grass bunches, precisely because of their protruding topography. The grass bunches are hummocks of looser, more porous, root dense soil raised 1-2 centimeters above the bare, crusted surface of the grass interstitial zones. Below this crusted surface, there are networks of lateral roots growing into the interstitial zone from the grass bunches (Kummel et al., 2021) and when the interstitial pools fill during rainfall, water percolates into the soil in two ways. The first is vertically down, through the soil crusts cracks and in through the laterally extensive root channels beneath the pools. The second is almost horizontal, through the root channels in the porous hummocked grass bunches that make up the sides of the interstitial pools, allowing for high lateral infiltration rates. The lack of these surrounding hummock 'sponges' in the bare interpatch pool likely informs how the bare pond water is able to infiltrate into the soil, with most of this bare zone surface water draining downslope into the vegetated patch pools where more root channels can better facilitate infiltration.

While the simple modeling perspective of water movement through patterned landscapes has identified the influence of soil crusting to some extent, the role of the sediment and spatial specificity of soil crusts - and the potential ephemeral ponds often associated with crusts - has not been adequately considered within their framework. While surface and soil water is always represented in all model equations, it is often represented in simplistic terms and authors rarely discuss the results of the soil water, instead focusing on analyses of the wavelength of the vegetation bands (Rietkerk et al., 2002; Eigentler & Sherratt, 2018). Meanwhile, the

microtopography perspective has incorporated crust types and surface water ponding within their schematics of the runoff-runon system in banded landscapes, but has failed to acknowledge that ponding might occur in multiple areas, both interpatch and patch. This perspective identifies how water and sediment flow might allow for bare patch ponding, but not for ponding in the vegetated patch. The occurrence of ponds within the patch calls into question how the movement of water between and into these ponds might both inform and be informed, not only by erosion/deposition processes and soil crusting, but also by local vegetation mechanisms.

Our qualitative and quantitative results highlight some discrepancies existing within the literature's understanding of the processes integral to pattern formation and evolution in semi-arid ecosystems. Given that water limitations are unanimously considered to be a necessary prerequisite for patterned landscape occurrence, the lack of comprehensive models that accurately predict where and how surface and soil water will move through the system is surprising. Some microtopographists have begun to investigate what other mechanisms might influence water availability and accumulation in banded vegetation landscapes, such as surface roughness and sediment transport, and have incorporated some of these variables into their models (Saco et al., 2007). However, the microtopography perspective, similar to the modeling perspective, still does not adequately address mechanisms present in our patterned landscape field site, such as vegetated patch ephemeral pools or the temporal patterns of water flow and infiltration.

An early 2021 study found that the periodic patterns at our field site are very recent, with significant periodicity being detectable only after 2011 (Kummel et al., 2021). Most literature concerning patterned vegetation dynamics have focused solely on well-established patterns that have been observed in the ecosystem for many decades. Investigating more periodically patterned systems in emergence could help us better understand whether these systems are endogenous

(spatially arranged due to internal plant mechanisms), exogenous (spatially arranged due to environmental heterogeneity), or perhaps a combination of both. Emergent pattern focused studies could also help us not only identify how periodic patterns might evolve in the face of increasing aridity, but also decide how best to manage and anticipate the evolution of these landscapes. Are these patterns enhancing the resiliency of the landscape to more extreme drought conditions, as suggested by (Dunkerley, 1997; Ludwig et al., 1999) or are they one of the first signs of an ecosystem in collapse, as suggested by (Rietkerk et al., 2004; Meron, 2015)?

CONCLUSION:

Understanding whether patterned landscapes are indicators of ecosystems edging towards drought-induced collapse, or resilient self-sustaining responses to increased aridity, is an essential next step in the work of simple modelers and microtopographists alike. Global temperatures are continuing to rise as climate change intensifies, and many drought-prone areas are only seeing precipitation events and intensities decrease further (Pachauri et al., 2014). Monitoring how periodic patterns are forming and evolving in the face of these harsher climactic conditions will enable us to better respond and manage these landscapes as they require.

According to Rietkerk, self-organized patchy landscapes, dependent on the density of the species informing the pattern (in our case, blue grama), are on a continuum of emerging ecosystem collapse (Fig. 5). A fairy ring patterned landscape, where most of the ground is covered in vegetation, is considered the least water-limited and catastrophically-inclined system of the patterned systems (Meron et al., 2004; Rietkerk et al., 2004). Then labyrinth and banded patterns, in which more bare ground is exposed than in a fairy ring system, are slightly closer to catastrophic

collapse. Spotted/stippled periodic patterns, where bare ground is most dominant, indicate the system is closest to ecosystem collapse. Ecosystem collapse, in this case, describes a landscape in which homogenous bare ground is the only steady state that can exist.

Valentin and others, however, argue that despite major rainfall variations no transformation of pattern type has been observed (Valentin & d'Herbès, 1999). The observed lack of a transformative progression of periodicity suggests that spatial self-organization is due more to the topography of the landscape than to lessening degrees of rainfall. Patterns can change or completely disappear, often creating a homogenous bare soil system, due to directional changes in runoff sheetflow caused by roads, rivers, overgrazing, poorly implemented conservation efforts, and numerous other disturbances (Penny et al., 2013). These influences, however, do not indicate that the system is unable to respond or persist in the face of a hotter or drier climate. Mutability between actual pattern types (spots, bands, fairy rings) due to changing aridity has not often been observed in nature, and when pattern transformation is observed it has not necessarily led to landscape desertification (Deblauwe et al., 2011).

However, the wavelength of banded patterns (the width of the bare interpatch compared to the vegetated patch, also known as the IBR: interband-band-ratio) has been shown in nature and in simple models to change with more or less rainfall (Moreno-de las Heras et al., 2012; Valentin & d'Herbès, 1999). Banded patterns can effectively maintain system-level water efficiency through years of greater drought by increasing the IBR, leading to a greater runoff interpatch area and a smaller but more sustainable vegetated patch (Valentin & d'Herbès, 1999). The ability of self-organized patchy landscapes to transform into other patterns or to adapt wavelength in response to rainfall changes is just as much a sign of landscape resiliency and stabilization as of regime collapse. Indeed, many claim that in arid climates, landscape patchiness is vital for

maintaining biodiversity and vice versa, as the two are closely linked (Ludwig et al., 1999; Dunkerley, 1997). Spatially self-organized vegetation such as the blue grama at our study site can play an essential role in soil and water conservation in the landscape, potentially stabilizing the system in the face of increasing aridity and perhaps even preventing ecosystem collapse. Given that blue grama is the dominant high quality, nutritional fodder grass at our study site (Rondeau et al., 2016), the nature of periodic patterns as signals of vegetated landscape resilience or decline likely has severe economic ramifications for ranching operations at Chico Basin Ranch.

While being able to conclusively determine the nature of periodic patterns in enhancing stability and resilience or signaling likely vegetation population collapse is important knowledge to have, management responses may be similar regardless. Overgrazing on patterned landscapes can greatly reduce the system's ability to capture water from sheet-flow and reduce the positive effects of the vegetation's short distance facilitation (Aguiar & Sala, 1999; Kéfi et al., 2007). However, light grazing pressure could enhance the spatial redistribution of water runoff within the landscape and increase the productivity of the patterned system (Moreno de las Heras et al., 2012; Dunkerley, 1997; Stavi et al., 2007). Therefore, intentionally managed ranching can hopefully continue being viable at Chico Basin Ranch.

Over the next few years at our study site, we should take qualitative measurements on the thickness and composition of soil crusts and their spatial distribution across patch and interpatch zones. Expanding our transects so that they run through multiple vegetated bands will allow us to better understand the hydraulic connectivity at our site. Conducting rainfall simulations that allow us to see how far the runoff-runon system extends downslope using these extended transects should also be combined with a detailed slope analysis. Having topographic data at our site will allow us to better compare our patch-interpatch conceptual model with the microhydrology schematic of

Dunkerley and others (Fig. 3). An investigation into the petrocalcic (hardpan) layer, which we found throughout our plots, would also provide greater understanding of how soil composition might be influencing or influenced by landscape patchiness. Hardpans (an impervious layer of soil, typically clay dominated) have been observed to be deeper in vegetated patches compared to interpatches (Boaler & Hodge, 1964; Penny et al., 2013), suggesting that plant roots either aid in reducing the compactness of the soil beneath them, or can proliferate only where the hardpan is deeper. Examining the depth and position of the petrocalcic layer compared to the blue grama patches in Chico Basin would give us more insight into the potential endogenous or exogenous role of vegetation in these patterned ecosystems.

ACKNOWLEDGEMENTS:

FIGURES:

Figure 1: The global distribution of banded vegetation pattern as indicated by available literature compiled by Valentin et al., 1999.

Figure 2: An area of banded vegetation north-west of the town Deming, in New Mexico, USA. Image captured using Google Earth.
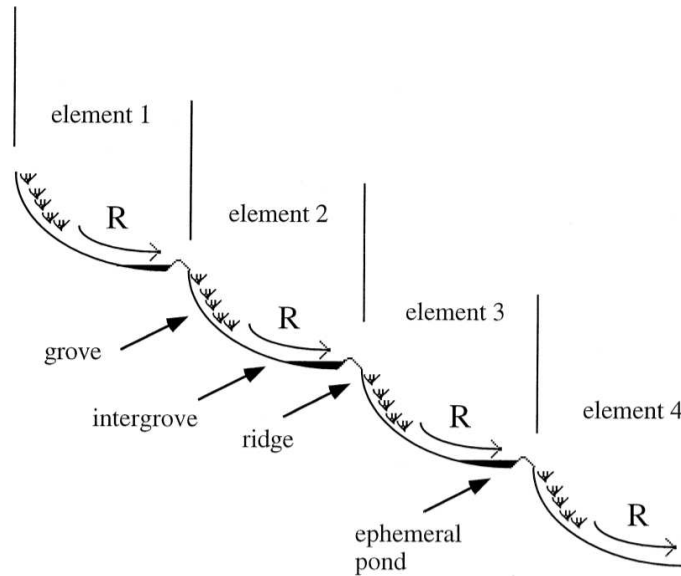
Figure 3: Schematic diagram of the microtopographic framework adopted for the description of banded mosaic characteristics, the conceptual model of the microtopography perspective discussed (Dunkerley & Brown, 1999).
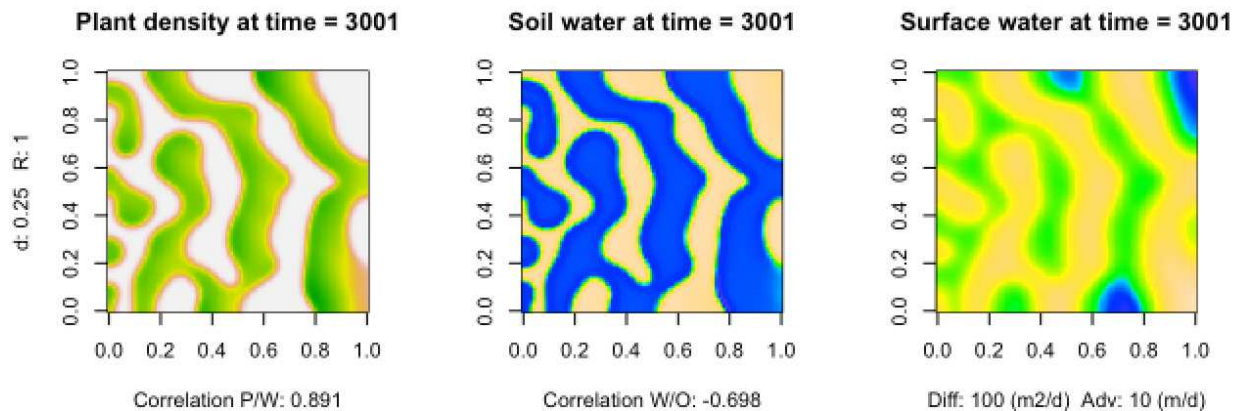
Figure 4: Visualization of vegetation bands with no water in the system (far left figure), soil water accumulation in the banded landscape after rainfall (middle figure), and surface water accumulation in the banded landscape at the same timestamp after rainfall (far right figure). This model was created using the three equations in Rietkerk's simple model (Rietkerk, 2002; Miro & Liu, 2021).
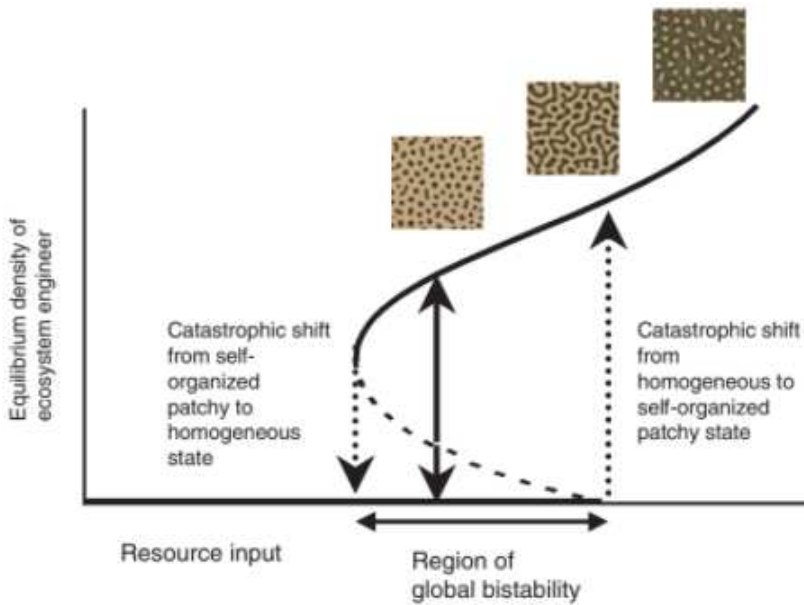
Figure 5: Model showing how ecosystems may undergo a sequence of emerging self-organized patchiness as resource input (water) decreases or increases. Thick solid lines represent mean equilibrium densities of consumers functioning as ecosystem engineers. Dotted arrows represent catastrophic shifts between self-organized patchy and homogeneous states, and vice versa. Dark colors represent high density. The range of resource input for which global bistability and hysteresis exists is between these dotted arrows. Solid arrows represent development of the system toward the coexisting self-organized patchy state or homogeneous state, depending on initial ecosystem engineer densities (Rietkerk et al., 2004).
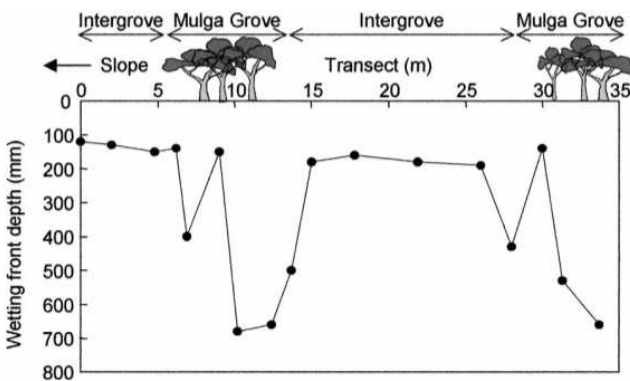
Figure 6: Depth of the wetting front (mm) in soils across a 35m transect cutting downslope (right to left) through small groves of mulga (*Acacia aneura)* trees and across open intergroves (Ludwig et al., 2005).
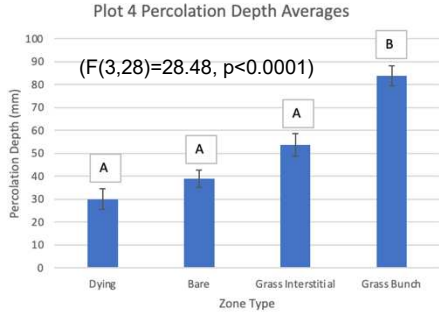
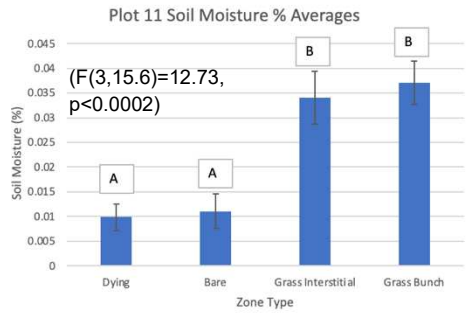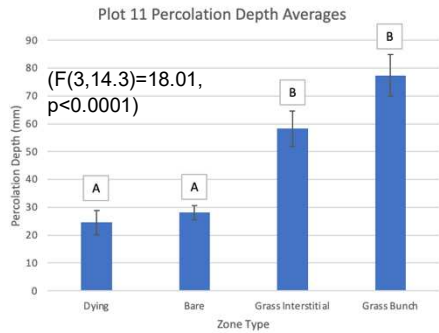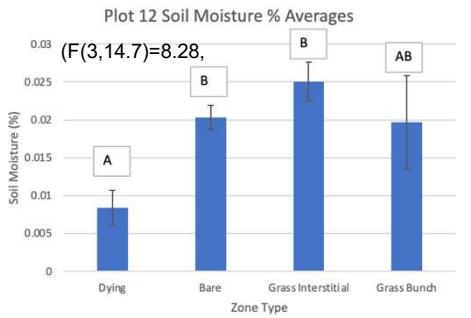| Percolation Depth (mm) | Soil Water Content (g H2O/cm2) | Soil Water Content (%) |

Plot 4

Plot 4 Percolation Depth Averages
(F(3,28)=28.48, p<0.0001)

Plot 4 Soil Moisture Content Averages
(F(3,28)=13.03, p<0.0001)

Plot 4 Soil Moisture % Averages
(F(3,28)=28.29, p<0.0001)

Plot 7

Plot 7 Percolation Depth Averages
(F(3,28)=19.44, p<0.0001)

Plot 7 Soil Moisture Content Averages
(F(3,29)=5.99, p<0.002)

Plot 7 Soil Moisture % Averages
(F(3,29)=2.99, p<0.04)

Plot 11

Plot 11 Percolation Depth Averages
(F(3,14.3)=18.01, p<0.0001)

Plot 11 Soil Moisture Content Averages
(F(3,15.7)=15.80, p<0.0001)

Plot 11 Soil Moisture % Averages
(F(3,15.6)=12.73, p<0.0002)

Plot 12

Plot 12 Percolation Depth Averages
(F(3,14.9)=30.29, p<0.0001)

Plot 12 Soil Moisture Content Averages
(F(3,15.1)=8.55, p<0.0015)

Plot 12 Soil Moisture % Averages
(F(3,14.7)=8.28,

Plot 13

Plot 13 Percolation Depth Averages
(F(3,28)=45.95, p<0.0001)

Plot 13 Soil Moisture Content Averages
(F(3,28)=15.81, p<0.0001)

Plot 13 Soil Moisture % Averages
(F(3,14.1)=32.36, p<0.0001)
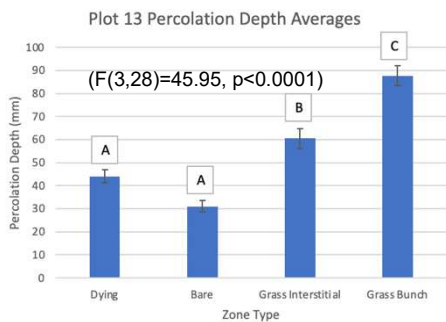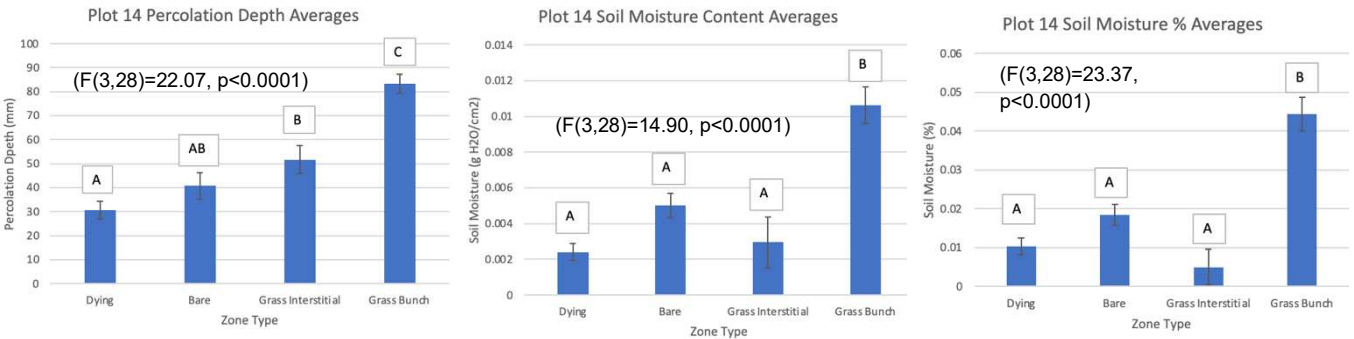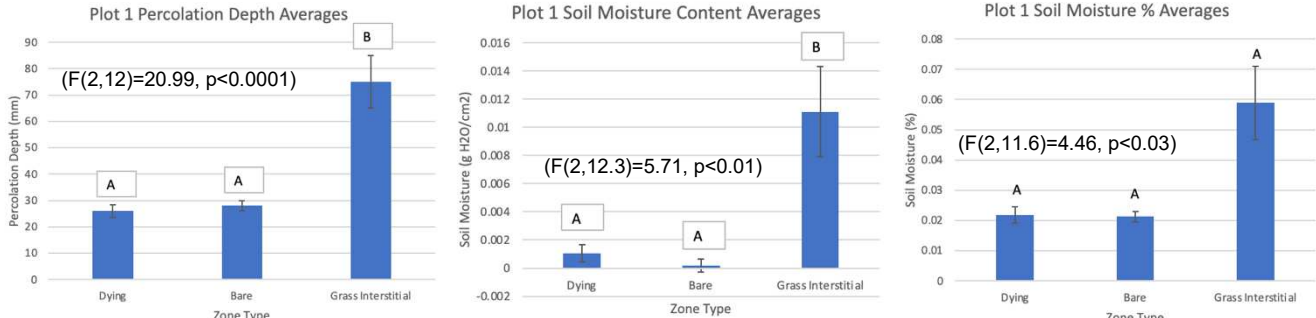
35

Plot 14

Figure 7: Percolation depth (mm), total volume of soil water content (g H_2O/cm^2), and soil water content (percentage) averages calculated from soil cores taken after rainfall simulation conclusion in six of the eight randomly selected plots at Chico Basin Ranch. Letters above each bar indicate statistical relationships between each zone type. The dying and bare zones make up the interpatch, while the grass interstitial and grass bunch zones make up the vegetated patch of each periodic pattern.

Plot 1

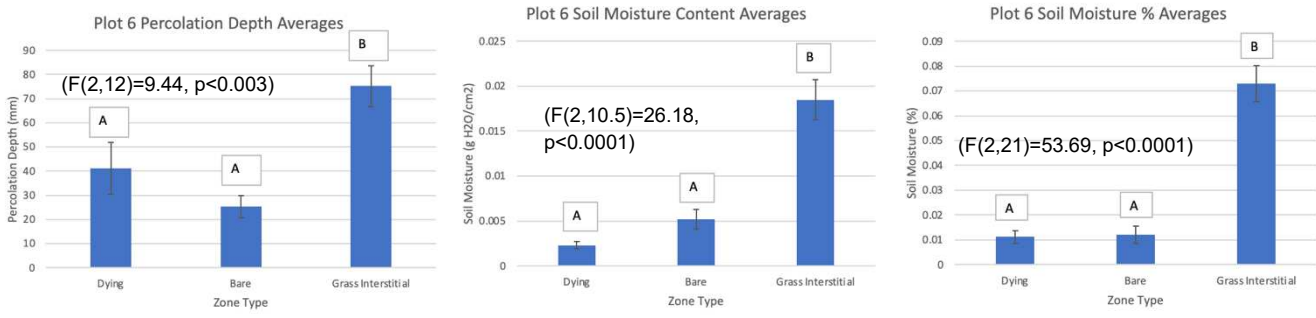**Plot 6**

Figure 8: Percolation depth (mm), total volume of soil water content (g H_2O/cm^2), and soil water content (percentage) averages calculated from soil cores taken after rainfall simulation conclusion in two of the eight randomly selected plots at Chico Basin Ranch. Letters above each bar indicate statistical relationships between each zone type. The dying and bare zones make up the interpatch, while the grass interstitial is only part of the vegetated patch. These two plots do not contain any soil water data from soil cores in the grass bunch, seeing as we had not yet decided to include the grass bunch zone in our investigative framework.
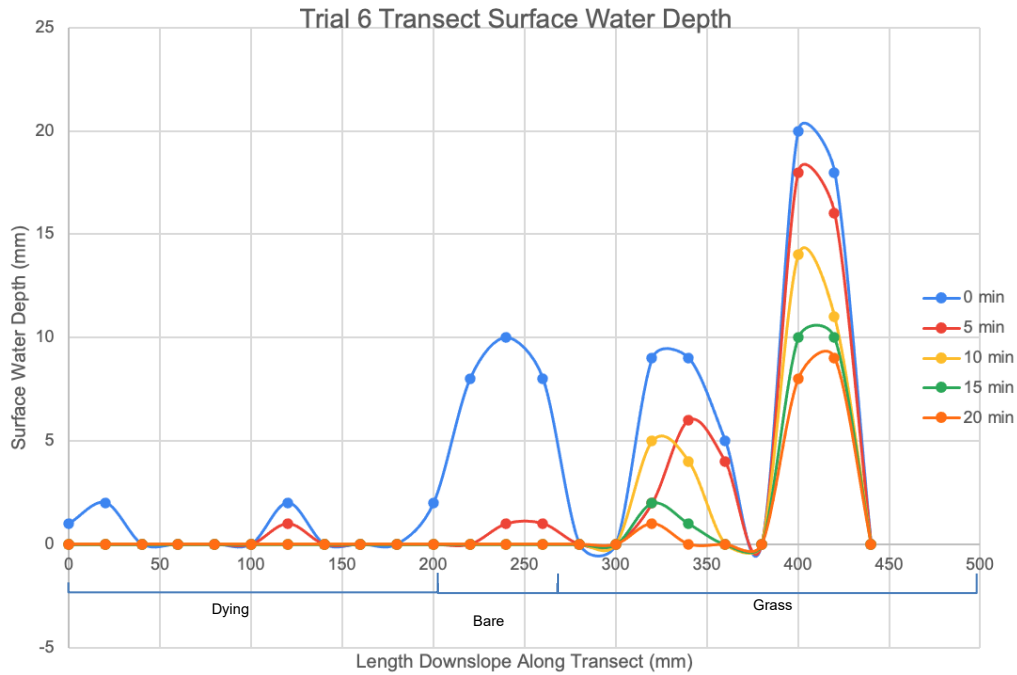
Figure 9: Surface water depth (mm) along a transect through Plot 6 on which the rainfall simulation was conducted. The recorded length of the dying and bare zones of the interpatch are indicated on the graph, while grass indicates the whole vegetated patch as we did not measure every time a grass bunch transitioned to part of the grass interstitial zone and back. Surface water depths at multiple time stamps are shown on the same graph, with 0 min indicating surface water depth immediately after the rainfall simulation ended, and 20 min indicating surface water depth along the transect twenty minutes after the conclusion of the rainfall simulation.
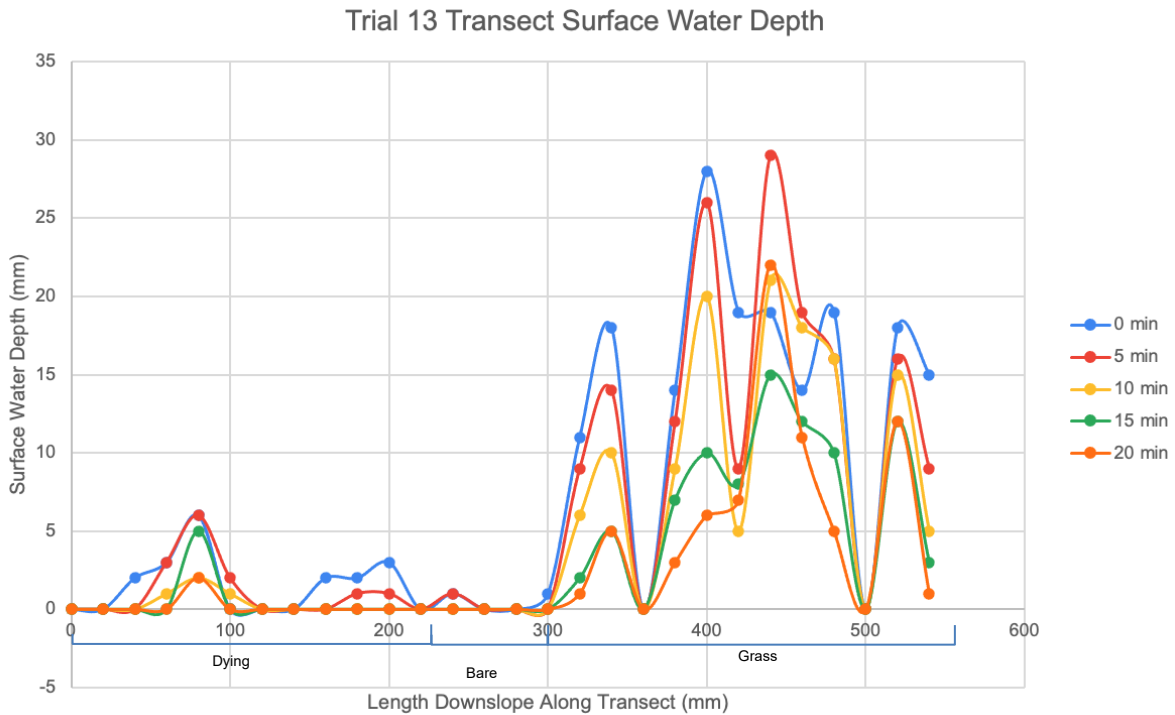
Figure 10: Surface water depth (mm) along a transect through Plot 13 on which the rainfall simulation was conducted. The recorded length of the dying and bare zones of the interpatch are indicated on the graph, while grass indicates the whole vegetated patch as we did not measure every time a grass bunch transitioned to part of the grass interstitial zone and back. Surface water depths at multiple time stamps are shown on the same graph, with 0 min indicating surface water depth immediately after the rainfall simulation ended, and 20 min indicating surface water depth along the transect twenty minutes after the conclusion of the rainfall simulation.
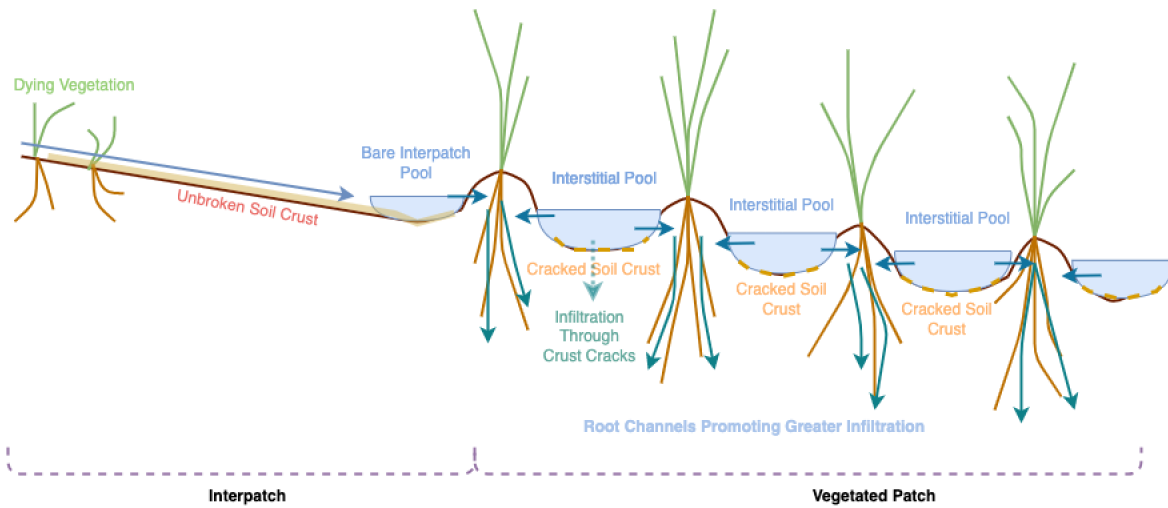

Figure 11: Schematic of microtopographic and microhydrologic mechanisms as observed at our study site at Chico Basin Ranch. Infiltration does occur to some extent in all areas, both interpatch and patch, but arrows emphasize dominant infiltration pathways into the soil matrix.
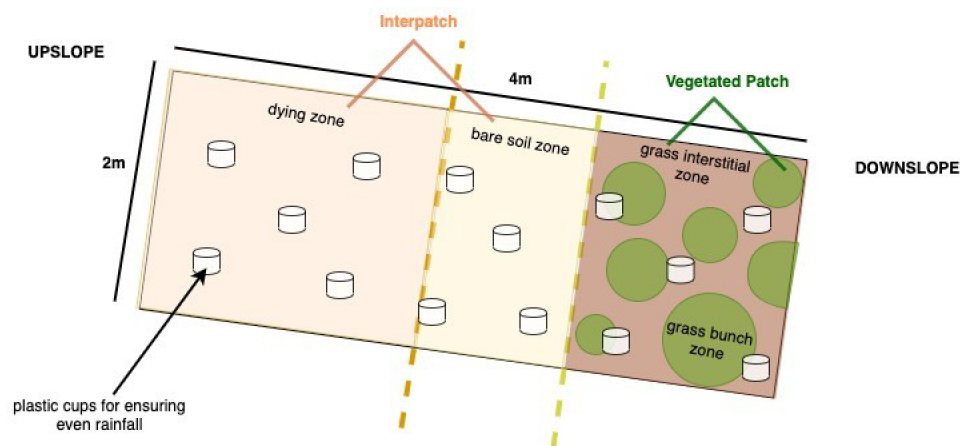

Figure 12: Diagram of plot layout method, describing general zone location, rainfall evenness indicator cup layout, and plot area.

Figure 13: Area of Interest at Chico Basin Ranch Airport, courtesy of Google Earth.

REFERENCES:

1. Aguiar, M. R., & Sala, O. E. (1999). Patch structure, dynamics and implications for the functioning of arid ecosystems. *Trends in Ecology & Evolution*, *14*(7), 273-277.
2. Benedict, A. D. (2008). *The naturalist's guide to the Southern Rockies: Colorado, southern Wyoming, and northern New Mexico*. Cloud Ridge Publishing.
3. Boaler, S. B., & Hodge, C. A. H. (1964). Observations on vegetation arcs in the northern region, Somali Republic. *The Journal of Ecology*, 511-544.
4. Bochet, E., Poesen, J., & Rubio, J. L. (2000). Mound development as an interaction of individual plants with soil, water erosion and sedimentation processes on slopes. *Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group*, *25*(8), 847-867.
5. Bonachela, J. A., Pringle, R. M., Sheffer, E., Coverdale, T. C., Guyton, J. A., Caylor, K. K., … & Tarnita, C. E. (2015). Termite mounds can increase the robustness of dryland ecosystems to climatic change. *Science*, 247(6222), 651-655.
6. Dagbovie, A. S., & Sherratt, J. A. (2014). Pattern selection and hysteresis in the Rietkerk model for banded vegetation in semi-arid environments. *Journal of The Royal Society Interface*, *11*(99), 20140465.
7. Deblauwe, V., Couteron, P., Bogaert, J., & Barbier, N. (2012). Determinants and dynamics of banded vegetation pattern migration in arid climates. *Ecological monographs*, *82*(1), 3-21.

8. Deblauwe, V., Couteron, P., Lejeune, O., Bogaert, J., & Barbier, N. (2011). Environmental modulation of self-organized periodic vegetation patterns in Sudan. *Ecography*, *34*(6), 990-1001.

9. Dunkerley, D. L., & Brown, K. J. (1995). Runoff and runon areas in a patterned chenopod shrubland, arid western New South Wales, Australia: characteristics and origin. *Journal of arid Environments*, *30*(1), 41-55.

10. Dunkerley, D. L. (1997). Banded vegetation: survival under drought and grazing pressure based on a simple cellular automaton model. *Journal of Arid Environments*, *35*(3), 419-428.

11. Dunkerley, D. L., & Brown, K. J. (1999). Banded vegetation near Broken Hill, Australia: significance of surface roughness and soil physical properties. *Catena*, *37*(1-2), 75-88.

12. Dunkerley, D. (2017). An approach to analysing plot scale infiltration and runoff responses to rainfall of fluctuating intensity. *Hydrological Processes*, *31*(1), 191-206.

13. Dunkerley, D. (2018). Banded vegetation in some Australian semi-arid landscapes: 20 years of field observations to support the development and evaluation of numerical models of vegetation pattern evolution. *Desert*, *23*(2), 165-187.

14. Dunne, T., Zhang, W., & Aubry, B. F. (1991). Effects of rainfall, vegetation, and microtopography on infiltration and runoff. *Water Resources Research*, *27*(9), 2271-2285.

15. Eigentler, L., & Sherratt, J. A. (2018). Analysis of a model for banded vegetation patterns in semi-arid environments with nonlocal dispersal. *Journal of mathematical biology*, *77*(3), 739-763.

16. Galle, S., Ehrmann, M., & Peugeot, C. (1999). Water balance in a banded vegetation pattern: A case study of tiger bush in western Niger. *Catena*, *37*(1-2), 197-216.

17. Gandhi, P., Bonetti, S., Iams, S., Porporato, A., & Silber, M. (2020). A fast–slow model of banded vegetation pattern formation in drylands. *Physica D: Nonlinear Phenomena*, *410*, 132534.

18. Google Earth, NW of Deming, NM, USA

19. Kéfi, S., Rietkerk, M., Alados, C. L., Pueyo, Y., Papanastasis, V. P., ElAich, A., & De Ruiter, P. C. (2007). Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. *Nature*, *449*(7159), 213-217.

20. Kelso, T., Bower, N., Halteman, P., Tenney, K., & Weaver, S. (2007). Dune communities of SE Colorado: Patterns of rarity, disjunction and succession. In*: Barlow-Irick, P.; Anderson, J.; McDonald, C., tech eds. Southwestern rare and endangered plants: Proceedings of the Fourth Conference; March 22-26, 2004; Las Cruces, New Mexico. Proceedings. RMRS-P-48CD. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 39-48* (Vol. 48).

21. Kummel, M., McDonald, A., Foster, E., Jennings, A., McMullin, E., Buchband, H. (2021). Desertification in arid Colorado shortgrass steppe? Emerging periodic vegetation pattern formation, possibly driven by climate change. *Ecology Society of America poster.*

22. Lefever, R., & Lejeune, O. (1997). On the origin of tiger bush. *Bulletin of Mathematical biology*, *59*(2), 263-294.

23. Leprun, J. C. (1999). The influences of ecological factors on tiger bush and dotted bush patterns along a gradient from Mali to northern Burkina Faso. *Catena*, *37*(1-2), 25-44.

24. Ludwig, J. A., Tongway, D. J., & Marsden, S. G. (1999). Stripes, strands or stipples: modeling the influence of three landscape banding patterns on resource capture and productivity in semi-arid woodlands, Australia. *Catena*, *37*(1-2), 257-273.

25. Ludwig, J. A., Wilcox, B. P., Breshears, D. D., Tongway, D. J., & Imeson, A. C. (2005). Vegetation patches and runoff–erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology*, *86*(2), 288-297.

26. Meron, E., Gilad, E., von Hardenberg, J., Shachak, M., & Zarmi, Y. (2004). Vegetation patterns along a rainfall gradient. *Chaos, Solitons & Fractals*, *19*(2), 367-376.

27. Meron, Ehud. (2015) Pattern Formation - A missing link in the study of ecosystem response to environmental changes. *Mathematical Biosciences*.

28. Moreno-de las Heras, M., Saco, P. M., Willgoose, G. R., & Tongway, D. J. (2012). Variations in hydrological connectivity of Australian semiarid landscapes indicate abrupt changes in rainfall-use efficiency of vegetation. *Journal of Geophysical Research: Biogeosciences*, *117*(G3).

29. Pachauri, R. K., Allen, M. R., Barros, V. R., Broome, J., Cramer, W., Christ, R., ... & van Ypserle, J. P. (2014). *Climate change 2014: synthesis report. Contribution of Working Groups I, II and III to the fifth assessment report of the Intergovernmental Panel on Climate Change* (p. 151). Ipcc.

30. Penny, G. G., Daniels, K. E., & Thompson, S. E. (2013). Local properties of patterned vegetation: quantifying endogenous and exogenous effects. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *371*(2004), 20120359.

31. Richards, K. S. (1976). The morphology of riffle-pool sequences. *Earth Surface Processes*, *1*(1), 71-88.

32. Rietkerk, M., Boerlijst, M. C., van Langevelde, F., HilleRisLambers, R., de Koppel, J. V., Kumar, L., ... & de Roos, A. M. (2002). Self-organization of vegetation in arid ecosystems. *The American Naturalist*, *160*(4), 524-530.

33. Rietkerk, M., Dekker, S. C., De Ruiter, P. C., & van de Koppel, J. (2004). Self-organized patchiness and catastrophic shifts in ecosystems. *Science*, *305*(5692), 1926-1929.

34. Rondeau, R. J., Doyle, G. A., & Decker, K. (2016). *Vegetation monitoring at Pueblo Chemical Depot: 1999-2015* (Doctoral dissertation, Colorado State University. Libraries).

35. Saco, P. M., Willgoose, G. R., & Hancock, G. R. (2007). Eco-geomorphology of banded vegetation patterns in arid and semi-arid regions. *Hydrology and earth system sciences, 11*(6), 1717-1730.

36. Stavi, I., Lavee, H., Ungar, E. D., & Sarah, P. (2009). Ecogeomorphic feedbacks in semiarid rangelands: A review. *Pedosphere*, *19*(2), 217-229.

37. Thiery, J. M., d'Herbès, J. M., & Valentin, C. (1995). A model simulating the genesis of banded vegetation patterns in Niger. *Journal of Ecology*, 497-507.

38. Tongway, D. J., & Ludwig, J. A. (2001). Theories on the origins, maintenance, dynamics, and functioning of banded landscapes. In *Banded vegetation patterning in arid and semiarid environments* (pp. 20-31). Springer, New York, NY.

39. Valentin, C., & d'Herbès, J. M. (1999). Niger tiger bush as a natural water harvesting system. *Catena*, *37*(1-2), 231-256.

40. Valentin, C., d'Herbès, J. M., & Poesen, J. (1999). Soil and water components of banded vegetation patterns. *Catena*, *37*(1-2), 1-24.

41. Vandervaere, J. P., Peugeot, C., Vauclin, M., Jaramillo, R. A., & Lebel, T. (1997). Estimating hydraulic conductivity of crusted soils using disc infiltrometers and minitensiometers. *Journal of Hydrology*, *188*, 203-223.
42. Vega, E., & Montana, C. (2011). Effects of overgrazing and rainfall variability on the dynamics of semiarid banded vegetation patterns: A simulation study with cellular automata. *Journal of arid environments*, *75*(1), 70-77.
43. White, L. P. (1971). Vegetation stripes on sheet wash surfaces. *The Journal of Ecology*, 615-622.