# Extreme makeovers: Crossing critical thresholds into desertification
by Shirley Kurc[1]

Introduction

In mathematics, *stationarity* describes time series (i.e., a series of measurements showing the magnitude of a given variable at regular points in time). Generally speaking, stationarity is achieved when a variable of interest fluctuates about some stable average. Additionally, for a stationary time series, the rules that govern the series do not change over time. Nearly everything going on around us can be described with time series, e.g. arrival times at your work place each day (Figure 1). Say on average you get to work at 8:45 am, give or take 5 minutes. However, sometimes you have to get in early, say at 6:30 am to finish up a memo that has to go out at 9 am. Similarly, sometimes you get in late; perhaps a doctor's appointment delays your arrival until 11 am. Because you have a stable average around 8:45 am, however, your time series of work arrival adheres to stationarity. A non-stationary change in your time might occur if you switched to a new job that asked you to come into work at 12 pm (Figure 1).
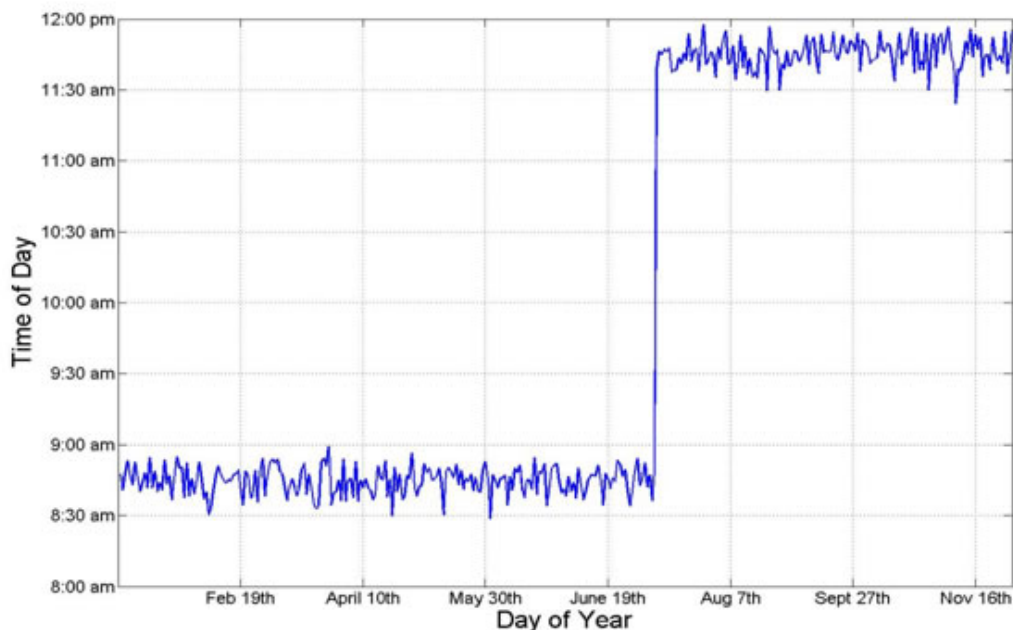

Figure 1. *Stationarity revolving around an 8:45 arrival at work. A non-stationarity when the job time is changed to 12 pm, which results in a new stationary period revolving around a 11:45 arrival at work.*

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Non-stationarities can also occur in more natural systems. In ecology, a moment of *non-stationarity* represents crossing some critical threshold. For any given ecosystem, this threshold is essentially:

1. a boundary in space and time between two different ecosystem states; and
2. a boundary that once crossed cannot be reversed without considerable interference or management inputs (Archer 1989; Friedel 1991; Laycock 1991).

A non-stationarity in the time series of an ecosystem might be represented by an abrupt transition from a humid to a more arid state. As a classic example of this, information extracted from eolian dust (i.e. very fine, eroded dust picked up by wind in one area and deposited in another area) showed that in the past, the Saharan region of Africa stably fluctuated in a humid state, until it was abruptly driven into its current desertified state in less than a century (deMenocal et al. 2000; Scheffer et al. 2001; Scheffer and Carpenter 2003) (Figure 2).
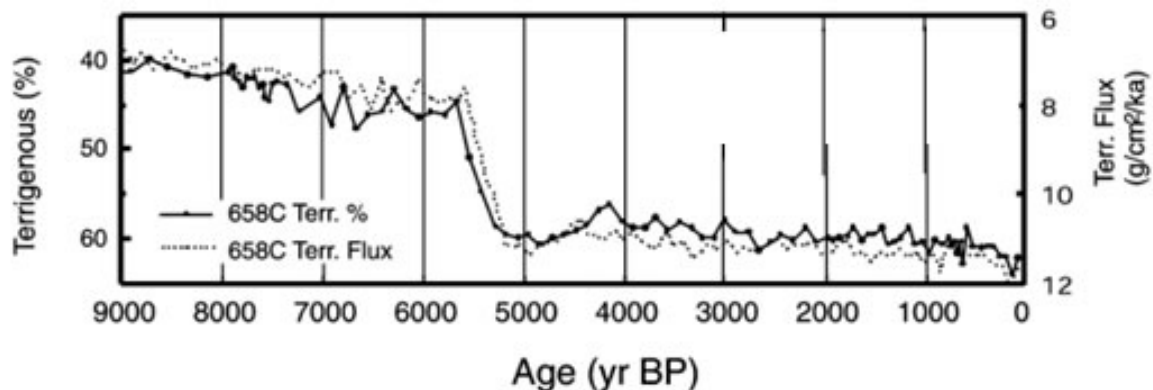
Figure 2. *An abrupt transition (non-stationarity) in the eolion dust supply signifying a shift in the Sahara region from more a more humid climate to a drier climate – thus from more mesic to more xeric ecosystems. Taken from deMenocal et al. (2000).*

Relatively small environmental changes such as short-term drought, overgrazing, changes in fire dynamics, etc., when combined in effect, could cause a total shift from one stable state to another. Resulting from a combination of these types of changes, desertification is often referred to as "*land degradation in arid, semi-arid, and dry-subumid areas resulting from climatic variations and human activities*"; but a stricter definition of desertification might assert that the alteration of the landscape is so severe that even upon return of normal conditions, a full recovery is impossible (Schlesinger et al. 1990). In an ecological context then, desertification of a landscape would imply that some critical threshold must have been crossed and the landscape will not return to its previous state even if climate or land use conditions return to their previous states.

Desertification by means of invasive plants
Native grasslands worldwide have been subject to woody plant encroachment for at least hundreds of years, from species ranging from various types of mesquite (*Prosopis* spp.) to creosotebush (*Larrea tridentata*) (e.g. Archer 1994; Asner et al. 2003; Buffington and Herbel 1965; Gardner 1951; Van Auken, 2000). As 80 % of all rangelands occur in arid and semiarid areas (Snyman 1999), woody encroachment is compromising the sustainability of grazing practices in drylands worldwide (Wilcox and Thurow 2006). Additionally, woody encroachment often leads to desertification by exposing more bare soil, increasing surface connectivity and thus erosion, etc. (Wilcox 2007).

Invasive non-woody species can provoke desertification, as well. A conversion from a native ecosystem such as a grassland or pristine desert into non-native grassland can represent a desertification transition. Often, invasive grasses are introduced into a region with the best of intentions, such as providing improved rangeland forage for livestock. Unfortunately, attributes that make these grasses desirable as forage are usually the same attributes that make them an ecological threat: they can thrive in the presence of fire and drought, they can produce inexhaustible seed, they can completely interfere with the ecosystem functioning, and they can significantly reduce biodiversity (Arriaga et al. 2004; Bock et al. 2007; Franklin et al. 2006).

The transition from plant establishment to species invasion is poorly understood. Invasive species often stay where purposely planted, or persist in obscurity for decades before erupting. Establishment of invasive plants often begins in linear habitats such as roadsides (Mutz and Scifres 1975) where seeds may be readily dispersed via vehicular transport. They may then disperse laterally, away from the roadside. However, available evidence suggests invasive species rarely move as a continuous front from the linear habitats where they first appear (Pysek and Hulme 2005), but instead form nascent foci in the uplands as small isolated patches (Moody and Mack 1988). Although invasive species tend to spread via local dispersal, rare long distance dispersal events are suspected to govern the rate, extent, and spatial pattern of the spread (Richardson and Pysek 2006; With 2004).

The mechanisms by which extensive continuous stands develop from small, isolated patches within established ecosystems are not well understood. Phase transition theory suggests there are critical values of cover at which landscapes switch abruptly from a fragmented collection of small patches to a network of connected canopies (e.g., Milne et al. 1996) or a continuous large patch (Moody and Mack 1988). Progression to widespread invasion is generally correlated with seed dispersal and resource availability, but disturbance or abrupt climatic changes are believed to play a key role (Clarke et al. 2005). Under such conditions, the invasive species may dramatically alter the native ecosystem structure and functions.

The case of exotic buffelgrass (*Pennisetum ciliare*)

Substantive exotic grass invasions with extensive areal coverage have been documented worldwide (D'Antonio and Vitousek 1992; Fairfax and Fensham 2000; Mack 1981; Williams and Baruch 2000). D'Antonio and Vitousek (1992) convey the unequivocal importance of exotic grass invasions with four main points:

(1) exotic grass invasion is aided through deliberate planting as forage (e.g. Daehler and Goergen 2005; Fairfax and Fensham 2000);
(2) exotic grasses out-compete native species in a variety of ecosystems (e.g. Callaway et al. 2004; Jackson 2005);
(3) exotic grasses have the ability to significantly alter ecosystem processes (e.g. Huxman et al. 2004; Wilcox and Thurow 2006; Williams and Baruch 2000); and
(4) exotic grasses are often tolerant of fire and indeed, even promote a positive feedback fire cycle (e.g. Billings 1994; Clarke et al. 2005; Williams and Baruch 2000).

Additionally, exotic grass species pose a significant global threat to the biodiversity of native ecosystems (e.g. Clarke et al. 2005; Jackson 2005; Kolar and Lodge 2001; Richardson et al. 2000; Sala et al. 2000).

Native to southern Asia and eastern Africa, different varieties of exotic buffelgrass can now be found in drylands worldwide (Arriaga et al. 2004). Buffelgrass was accidentally introduced into western Australia by Afghan camel harnesses in the late 1800s; it now now dominates 30–50 million hectares in

Queensland alone (Jackson 2005) and has the potential to cover 60% of mainland Australia (Lawson et al. 2004). In Mexico, native desert vegetation has been purposely converted to buffelgrass pasture in many areas (Martin-R et al. 1999; Williams and Baruch 2000) owing to its perceived value as livestock pasture forage. Now, buffelgrass covers more than 1 million hectares in Sonora, Mexico, and is spreading at an exponential rate; it has been estimated to have the potential to cover 53 % of Sonora alone and up to 12 % of Mexico overall (Arriaga et al. 2004). In 1946, buffelgrass seeds were shipped to the United States from the Turkana Desert collection (Cox et al. 1988) to improve and increase pasture (Arriaga et al. 2004). From the 1940s through the 1980s, buffelgrass plantings were made in the Sonoran Desert of Arizona (Ward et al. 2006). Today in Arizona, there is growing concern that buffelgrass will escape these plantings, as has already occurred in Australia and Mexico, and dramatically alter the ecology of the hot desert ecosystems of the Southwest (e.g. http://wwwpaztcn.wr.usgs.gov/buffelgrass/), much as *Bromus tectorum* has transformed many Great Basin cold desert ecosystems (Mack 1981).

Buffelgrass is a C_4 perennial bunchgrass with asexual seed production (Daehler and Goergen 2005; Shafer et al. 2000) and dense root systems (Arriaga et al. 2004). It has dimorphic seeds with some seeds on a plant having greater mass and higher germinability than others (Lahiri and Kharabanda 1961). The seeds tend to be light and fluffy enabling them to potentially spread by wind, water, and animal fur (Arriaga et al. 2004; Jackson 2005). Sown seeds remain viable in the soil for 2 to 4 years (Bean 2007; Winkworth 1971). This longevity is enhanced by microbial inhibitors associated with the glumes which enclose the seeds and appear to retard their decay in semiarid systems (Parihar and Patil 1984).

A single wetting event, of at least 6 mm (Ward et al., 2006), immediately followed by a drydown period, provides conditions for about 50 % of viable buffelgrass seeds to germinate (Emmerich and Hardegree 1996).   A series of wetting events can significantly reduce seed germinability (Emmerich and Hardegree 1996), so there does appear to be a fine line between too much rainfall for germination and enough rainfall for seedling establishment (Sharif-Zadeh and Murdoch 2001). However, whereas native grasses are relatively susceptible to drought (Clarke et al. 2005), established buffelgrass plants are highly drought tolerant (Jackson 2005).

Though broadly identified as a highly invasive exotic species, buffelgrass does have clear habitat preferences. Available data suggests peak performance on soils with sand loam or loamy sand texture (Daehler and Goergen 2005; Fairfax and Fensham 2000; Ibarra-F et al. 1995; Martin-R et al. 1999; Mutz and Scifres 1975). However, buffelgrass has also been found on deep cracking clays (Butler and Fairfax, 2003). Regardless of soil texture, buffelgrass requires some seed cover for optimum emergence (Mutz and Scifres 1975). It is generally found at elevations ranging between about 20 and 700 m (Ibarra-F et al. 1995; Martin-R et al. 1999), although it currently resides on Tumamoc Hill in Tucson, Arizona, at elevations over 900 m (Bean 2007).

Climatologically, buffelgrass has the most success in a bimodal precipitation regime totaling about 200 – 550 mm annually (Clarke et al. 2005; Daehler and Goergen 2005; Ibarra-F et al. 1995; Martin-R et al. 1999). The critical minimum temperature for buffelgrass germination has been reported as 12.5^{o} C (Ivory and Whiteman 1978; Ward et al. 2006), a minimum temperature for significant growth is 10° C (Ivory and Whiteman 1978), and the optimum growth temperature is about 38° C (Ivory and Whiteman 1978). With a warm climate and bimodal precipitation pattern, Tucson is an ideal location for continued buffelgrass invasion.

As temperatures continue to climb in line with predicted climatic changes, uncontrollable spread of buffelgrass both northward and into higher elevations may be inevitable in the southwestern United

States. Additionally, buffelgrass is highly tolerant of fire; following summer fire, plants have been observed to regenerate vegetatively within 5 to 10 days (Martin-R et al. 1999). Establishment in hot deserts of the southwestern United States has the potential to convert normally fireproof and picturesque desert scrub communities into monospecific grassland—especially since the native plants in these communities are not adapted to fire and can easily be killed by it (Figure 3).

Figure 3. *Native Sonoran Desert in Tucson Mountains, AZ (left); buffelgrass invasion of a saguaro stand in the Tucson Mountains (center); fire-damaged saguaro (right). See http://wwwpaztcn.wr.usgs.gov/buffelgrass/ for additional details (Photo left by Mark Dimmitt; photo center by Travis Bean, photo right by Ben Wilder).*

Implications for Management

Currently, two primary management strategies are being used to eradicate buffelgrass in southeastern Arizona: herbicides and manual removal. Herbicides are only used when a majority of the plant is green and actively growing, whereas manual removal of the plants can be effective year round. Both of these management strategies rely heavily on knowing the locations of existing buffelgrass patches, and often the good will of citizen volunteers, thus limiting their impact. While the development of remote sensing techniques (e.g. Franklin et al. 2006) may be able to help locate current stands, making these management strategies more effective, they still focus more on removal than on prevention.

A better understanding of pre-threshold states can benefit these management efforts (Friedel 1991; Kolar and Lodge 2001). Preventing new invasions is the most successful and economical management strategy, whether in the Sonoran Desert or elswhere around the globe (Dale et al. 2005; Vitelli and Pitt 2006). Insights into mechanisms and patterns of invasive plant establishment are fundamental to forecasting their spread (Pysek and Hulme 2005). Spatially explicit predictive models based on ecological theory are needed to direct limited local and regional resources to monitor and manage those portions of the landscapes most likely to be invaded (Hastings et al. 2005; King and With 2002; Marone et al. 1998). Given the potential for rising global temperatures to turn exotic grasses and other invasive species into agents of irreversible desertification around the globe, the development of such tools and insights is of the utmost importance.

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