Effects of Increased *Heteropogon contortus* (Tanglehead) on Rangelands: The Tangled Issue of Native Invasive Species☆

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ARTICLE INFO

ABSTRACT

Heteropogon contortus recently and rapidly increased in dominance in grasslands where it once had been a minor component. Ecological effects of this increase are unknown, but land managers are concerned about the potential negative economic and ecological impacts. We examined compositional and structural characteristics of the vegetation community along a gradient of dominance of *H. contortus* to quantify changes, compare the effects to invasions by nonnative grasses, and provide insights about management. As *H. contortus* increased, grass richness decreased across the gradient by 6 species·m^{-2} (95% CI: $2-10$) in summer and 10 species·m^{-2} ($6-15$) in winter. Cover of other native grasses decreased $8-10\%$ in both seasons for every 10% increase in *H. contortus*. Presence of seven individual plant species and cover of five species decreased, but presence of five species and cover of one species increased with *H. contortus*. Canopy cover increased and soil nutrients were higher in dense *H. contortus,* potentially facilitating further ecological changes. We suggest that managing *H. contortus* and other species that become invasive within the ecosystem where they were once native likely requires reducing rather than wholly eliminating the species, which may differ from management strategies for nonnative species.

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Introduction

Species invasions have altered the structure and function of ecosystems worldwide (D'Antonio and Vitousek, 1992). Although invasions by nonnative species have been examined most often, population outbreaks can also occur in native organisms (Valéry et al., 2008). Alpert et al. (2000) suggested that an invasive species is any species that "both spreads in space and has negative effects on species already in the space that it enters." Essentially, a species does not need to move from one geographic region to another to become invasive. Likewise, Valéry et al. (2008) stated that invasion could result from a "change OF the environment," such as nonnative invasion from one geographic location to another or a "change IN the environment," meaning anthropogenic changes in native communities could allow a native species to become invasive (e.g., de la Cretaz and Kelty, 1999; Nielsen et al., 2011; Carey et al., 2012). Despite this, many native species may not be recognized as invasive because increases in dominance generally are assumed to be natural shifts in the community (Simberloff and Vitule, 2014).

Heteropogon contortus (tanglehead) is a perennial bunchgrass found in semitropical grasslands throughout the world, including northern Mexico and the southwestern United States (Heuzé et al., 2013). *Heteropogon contortus* forms small stands and typically has been a minor component of the vegetation community (Johnston, 1963; Tjelmeland, 2011). Within the past $15-20$ years, however, this grass has become the most dominant species in rangelands of southern Texas (Tjelmeland, 2011), resulting in visible changes to rangelands and perceived negative impacts on economic resources. This rapid shift in dominance seems to have originated from a single point and has now spread to > 150 km^2. Although we do not have direct evidence supporting a "change of" or "change in" the environment, the sudden shift may be related to anthropogenic changes, namely a reduction in grazing, given that land management has shifted from cattle production to recreational hunting (Smith, 2009; Tjelmeland, 2011). The change in land management has the potential to help *H. contortus* become invasive because reduced grazing pressure increases seedling recruitment, seed production, and plant survival (Grice and McIntyre, 1995; Orr et al., 2004; Heuzé et al., 2013) and fire, used to improve wildlife habitat, can increase recruitment (Tjelmeland, 2011). Alternatively, a nonnative variety could be responsible for the increased dominance as two phenotypic varieties have been proposed, one that is much more abundant, but we lack supporting genetic evidence (Tjelmeland, 2011).

We sought to determine if increased dominance of *H. contortus* has negative effects on vegetation composition and structure in grasslands and, therefore, should be managed as a native invasive species or if this grass is simply a native species undergoing population changes. If the increased dominance of *H. contortus* has resulted in ecological effects similar to those documented with other invasive grasses, we predicted that richness and cover of other native plants would be lower and vegetation structure in invaded areas should differ from more diverse communities (D'Antonio and Vitousek, 1992; Sands et al., 2009). We also predicted soil nutrients would differ with *H. contortus*, as has been observed with several other species of nonnative invasive plants (Ehrenfeld, 2003; Dassonville et al., 2007). If the increase in *H. contortus* can be considered an invasion, this grass would add to the scant literature on native invasive species that invade their original range and ecosystem.

Methods

Study Area

Our study area lies in semiarid grasslands within the Tamaulipan Biotic Province, at the convergence of Gulf Coastal Grassland and Tamaulipan Thornscrub (Johnston, 1963; Jahrsdoerfer and Leslie, 1988; Fulbright et al., 1990). We chose several pastures on the Borregos and Alta Vista ranches (~ 19 km apart) in Jim Hogg County, Texas because natural or mechanical disturbance was minimal, grazing had been maintained at a stocking rate of 0.1 animal unit·ha^{-1} (~3× lower than 30 years ago), and the increase in *H. contortus* was first reported here. A mosaic of grasses and brush characterizes these grasslands, but woody vegetation primarily occurs in 0.02- to 0.24-ha stands of *Prosopis glandulosa* (honey mesquite). Sandy ridges throughout the site are composed of bare ground, semiwoody *Monarda punctata* (spotted beebalm), and short grasses. The climate is both semiarid and subtropical punctuated by rainfall extremes, such as high rainfall in 2009 − 2010 and droughts in 2008 − 2009 and 2010 − 2013 (South Central Climate Science Center, 2013).

Sampling

We randomly selected 70 study plots (125-m radius) in relatively open grasslands with *H. contortus* ranging from 0% to 80% relative cover (0 − 60% absolute cover). Plots were > 325 m apart, > 150 m from large woody thickets, > 125 m from two-tracks/roads, and had minimal woody vegetation and nonnative grasses. We sampled vegetation four times. Summer (June) sampling occurred during a wet (2010) and dry (2011) growing season, and winter (January/February)

Table 1

Estimated linear change in compositional and structural characteristics of the vegetation for every 10% increase in of *Heteropogon contortus*, after accounting for covariates ($n = 70$ plots) along with 95% confidence intervals and *P* values, Jim Hogg County, Texas, 2010 − 2011. To reflect the magnitude of the effect differing between years, estimates are presented in separate rows, with 2010 first and 2011 second. Numbers in bold indicate a significant difference.

Variable	SUMMER		WINTER	
	Estimate (LCI, UCI)	*P*	Estimate (LCI, UCI)	*P*
Richness (number of species)				
Native grasses	**-5.6** **(-10.4, -1.6)**	**0.009**[1]	**-10.4** **(-14.8, -6.0)**	**< 0.0001**[a]
Forbs	**-8** **(-14.4, -2.1)** 0.6 (-5.1, 6.6)	**0.03**		
Absolute cover (percent)				
Native grasses	**-7.5** **(-8.8, -6.2)** **-10.4** **(-11.5, -9.2)**	**0.0004**	**-10.1** **(-10.9, -9.3)**	**< 0.0001**
Forbs	**-2.1** **(-3.2, -1.1)** **0.3** **(-0.6, 1.2)**	**0.0003**		
Non-native	0.3 (-0.3, 0.9)	0.32	0.4 (-0.4, 1.1)	0.31
Bare ground	-2.4 (-3.5, -1.3) -1.1 (-2.1, -1.7)	0.07	**-1.1** **(-1.9, 0.2)**	**0.02**
Leaf litter	-0.04 (-0.8, 0.7)	0.9	1.4 (0.3, 2.4) 1.0 (-0.7, 1.4)	0.08
Total cover	**1.0** **(0.4, 1.5)**	**0.001**[a]	**2.4** **(1.8, 3.0)**	**0.02**
canopy-level			**1.6** **(1.0, 2.2)**	
Vertical structure (number of contacts)				
Density	**11.0** **(5.9, 16.1)** 1.1 (-3.2, 5.5)	**0.0009**	2.1 (-1.8, 6.0)	0.29[a]
Density at	**3.6** **(2.5, 4.6)**	**< 0.0001**	**2.3** **(0.1, 3.5)**	**0.0006**[a]
canopy-level	**0.0** **(-0.9, 0.9)**			

[a] $df = 68$. For analyses of all other variables, $df = 67$.

sampling in 2010 and 2011 occurred after a dry and wet growing season (summer 2009 and 2010).

We characterized vegetation on plots within 17 quadrats (0.5 × 0.5 m, elevated 1-m high) spaced 30 m apart, along two perpendicular 250-m transects. We quantified cover of bare ground, litter, and each grass and forb species in 5% increments. We estimated horizontal cover in two vertical strata, ground level (≤ 0.4-m high) and canopy level (> 0.4 − 1.1 m), because the canopy could change independently of vegetation cover near the ground. In addition, we quantified vertical structure of the vegetation within the two strata using a Wiens pole in each quadrat (Wiens and Rotenberry, 1981) to record the number of times each plant species contacted the pole in 0.1-m increments.

During sampling, we observed subtle differences among the plots in the amount of interstitial space, which was not captured in preliminary analyses. Therefore, we classified plots into four categories on the basis of the interstitial space between large bunchgrasses: 1) sparse native, 2) thick native, 3) sparse *H. contortus*, and 4) thick *H. contortus*. We selected a subset of four plots per category (out of the 70 total plots), all on Borregos Ranch, for further sampling in 2011. On each of these 16 plots, we sampled soils within three, randomly placed quadrats. We collected soil samples at 5- to 50-cm depths ($n = 48$), which were analyzed by the Texas AgriLife Extension Service, Soil, Water, and Forage Testing Laboratory.

Analysis

We used relative cover of *H. contortus* to characterize the dominance gradient and analyzed data for summer and winter separately to explore seasonal differences. We used generalized linear mixed models, selected the appropriate distribution and link function for each response variable, and accounted for repeated sampling on the same plots over time by treating plots as subjects and incorporating a compound symmetric covariance structure (Littell et al., 2006). In addition, we included an interaction term to determine if the effects related to *H. contortus* changed over time. We log-transformed cover for individual plant species and soil characteristics to meet model assumptions. We also compared differences among the four categories of plots using a discriminant function analysis and soil characteristics based on analysis of variance (ANOVA). Additional details are presented in Bielfelt (2013).

Results

Richness and cover of forbs and native grasses and bare ground all decreased with increased *H. contortus* (Table 1). In summer, cover and density of vegetation at both ground and canopy levels also increased with *H. contortus* but depended on rainfall because we only observed

Table 2

Magnitude and direction of multiplicative changes in presence and cover of plants for every 10% increase in of *Heteropogon contortus*, after accounting for covariates ($n = 70$ plots) along with 95% confidence intervals and *P* values, County, Texas, 2010 − 2011. Species where the magnitude of the effect differed between years are presented in separate rows, with 2010 first and 2011 second.[1] Numbers in bold indicate a significant difference.

| Species | SUMMER | | | | | WINTER | | | | |
	Presence (LCI, UCI)	P	Cover (LCI, UCI)	df	P	Presence (LCI, UCI)	P	Cover (LCI, UCI)	df	P
Decreased										
Aristida oligantha/purpurea	−2 (−23, 24)	0.85	**−14 (−22, −5)**	**47**	**0.004**	1 (−20, 26)	0.96	**−10 (−17, −3)**	**42**	**0.01**
Cenchrus spinifex[2]	−6 (−28, 22)	0.62[3]	−4 (−12, 5)	47	0.41	−3 (−25, 28)	0.82[3]	**−13 (−21, −3)**	**43**	**0.02**
Elionurus tripsacoides	**−24 (−40, −4)**	**0.02**	−4 (−16, 9)	39	0.53	**−17 (−33, 4)**	**0.10**	−8 (−17, 2)	41	0.12
Galactia/rhynchosia	**−30 (−44, −14)**	**0.001**								
Monarda punctata	**−52 (−69, −26)**	**0.04**[3]				**−56 (−76, −17)**	**0.01**			
	−18 (−37, 6)									
Paspalum plicatulum	−21 (−43, 7)	0.13	−19 (−36, −2)	15	0.07	**−37 (−56, −11)**	**0.009**	**−30 (−47, −7)**	**18**	**0.02**
Schizachyrium littorale	**−29 (−42, −13)**	**0.001**	**−22 (−30, −13)**	**47**	**< 0.0001**	**−51 (−61, −39)**	**< 0.0001**	**−8 (−5, −2)**	**45**	**0.04**
								−2 (−4, −1)		
Setaria/Urochloa	**39 (−24, 155)**	**0.04**[3]	−5 (−14, 5)	37	0.29	6 (−21, 41)	0.71	−1 (−9, 8)	42	0.80
	−27 (−43, −7)									
Trachypogon spicatus	**−54 (−68, −33)**	**0.04**[3]	−11 (−28, 10)	30	0.26	**−56 (−69, −37)**	**< 0.0001**	0 (−15, 17)	27	0.99
	−39 (−54, −20)									
Increased										
Commelina erecta (2010 only)	**43 (4, 97)**	**0.03**[3]								
Digitaria/Panicum	**59 (24, 104)**	**0.0004**				**36 (11, 65)**	**0.003**			
Eragrostis lehmanniana	−28 (−12, 236)	0.11	1 (−9, 11)	44	0.88	−7 (−12, 29)	0.50	−2 (−12, 9)	**43**	**0.02**
								14 (3, 27)		
Eragrostis secundiflora[4]	**33 (5, 68)**	**0.02**				14 (−13, 50)	0.32			
Eragrostis sessilispica	**25 (1, 55)**	**0.04**								
Pennisetum ciliare						**57 (17, 111)**	**0.003**			
Did not change										
Acalypha radians (2010 only)	21 (−12, 65)	0.23[3]								
Ambrosia psilostachya	7 (−31, 65)	0.76								
Asteraceae (2010 only)			−6 (−20, 3)	63	0.13					
Bouteloua hirsute	7 (−14, 31)	0.55				−1 (−20, 22)	0.92	−10 (−21, 2)	30	0.10
Croton spp. (2010 only)	−9 (−29, 17)	0.47[3]								
Evolvulus spp. (2011 only)	−17 (−36, 11)	0.21[3]								
Fabaceae	14 (−6, 39)	0.18								
Paspalum setaceum	−3 (−21, 19)	0.75	−10 (−24, 6)	8	0.17					
Phlox spp. (2010 only)	−26 (−52, 15)	0.18[3]								
Physalis spp.	10 (−13, 39)	0.43								

[1] Slope estimates are back-transformed. Details regarding other model terms are in Bielfelt (2013).
[2] Presence analyzed for summer 2011 and winter 2010 only.
[3] $df = 67$. For all other presence estimates, $df = 68$.
[4] Presence analyzed for summer 2010 and 2011, but for winter only in 2011.

these differences in summer 2010 (see Table 1). However, there was an increase in both cover and density at the grass canopy level in winter. The data on the grass canopy also indicated that the canopy tended to be taller than other native areas.

Overall, we observed 84 species of herbaceous plants during the study (Bielfelt, 2013). Presence and cover of *Schizachyrium littorale* (seacoast bluestem) and *Paspalum plicatulum* (brown-seed paspalum) decreased with *H. contortus* in one or more seasons (Table 2). Presence also decreased for *Elionurus tripsacoides* (Pan-American balsam scale), *Setaria* spp./*Urochloa* spp. (signalgrass), *Trachypogon spicatus* (spiked crinkleawn), *Galactia canescens/Rhynchosia americana* (hoary milkpea/ American snoutbean), and *M. punctata* in at least one season. Cover decreased for *Aristida oligantha/A. purpurea* (threeawn species) in both seasons (see Table 2). Presence of *Digitaria cognata/Panicum capillare* (witchgrass), *Eragrostis sessilispica* (tumble lovegrass), *E. secundiflora* (red lovegrass), nonnative *Pennisetum ciliare* (buffelgrass), and *Commelina erecta* (dayflower) increased with *H. contortus* in at least one season (see Table 2). Cover of nonnative *E. lehmanniana* (Lehmann lovegrass) increased with *H. contortus* in winter 2011.

Based on the discriminant analysis, the four categories of plots were distinctly different. Plots with *H. contortus* generally had higher concentrations of three cations (magnesium, calcium, and potassium), compared with plots with native vegetation. Plots with sparse *H. contortus* or thick native vegetation had concentrations of potassium and sodium that were intermediate between plots with sparse, native vegetation and thick *H. contortus* (Fig. 1). Soil pH was highest in plots with sparse native vegetation and lowest in areas with thick native vegetation, with *H. contortus* plots having intermediate pH values (see Fig. 1).

Sand content was not significantly different, but it was slightly less sandy in plots with *H. contortus*.

Discussion

Much like nonnative invasive grasses, increased *H. contortus* was associated with fewer species of forbs and other native grasses, less bare ground, and a taller, denser, and more contiguous canopy (D'Antonio and Vitousek, 1992; Sands et al., 2009). The differences we observed in soil nutrients are consistent with patterns observed with other non-native invasive plants (e.g., Ehrenfeld, 2003; Dassonville et al., 2007). *Heteropogon contortus* was more abundant along roads and rapidly regrows after fires (Tjelmeland, 2011), additional characteristics frequently exhibited by nonnative invasive species (D'Antonio and Vitousek, 1992; Hobbs and Huenneke, 1992). On the basis of these results, we suggest *H. contortus* be characterized and managed as a native invasive species.

Plants that affect light absorption and nutrient uptake often have a competitive advantage over other plants (D'Antonio and Vitousek, 1992; de la Cretaz and Kelty, 1999; MacDougall and Turkington, 2005). Plots dominated by *H. contortus* had a denser, taller, and more contiguous canopy that could alter light availability and absorption. Decreases in bare ground associated with increased *H. contortus* could reduce available space for other native plants, and differences in cations and phosphorus could provide *H. contortus* with a competitive advantage over other native species. Furthermore, *H. contortus* is deficient in magnesium, calcium, sodium, and phosphorus (Paliwal and Manoharan, 1997; Heuzé et al.,
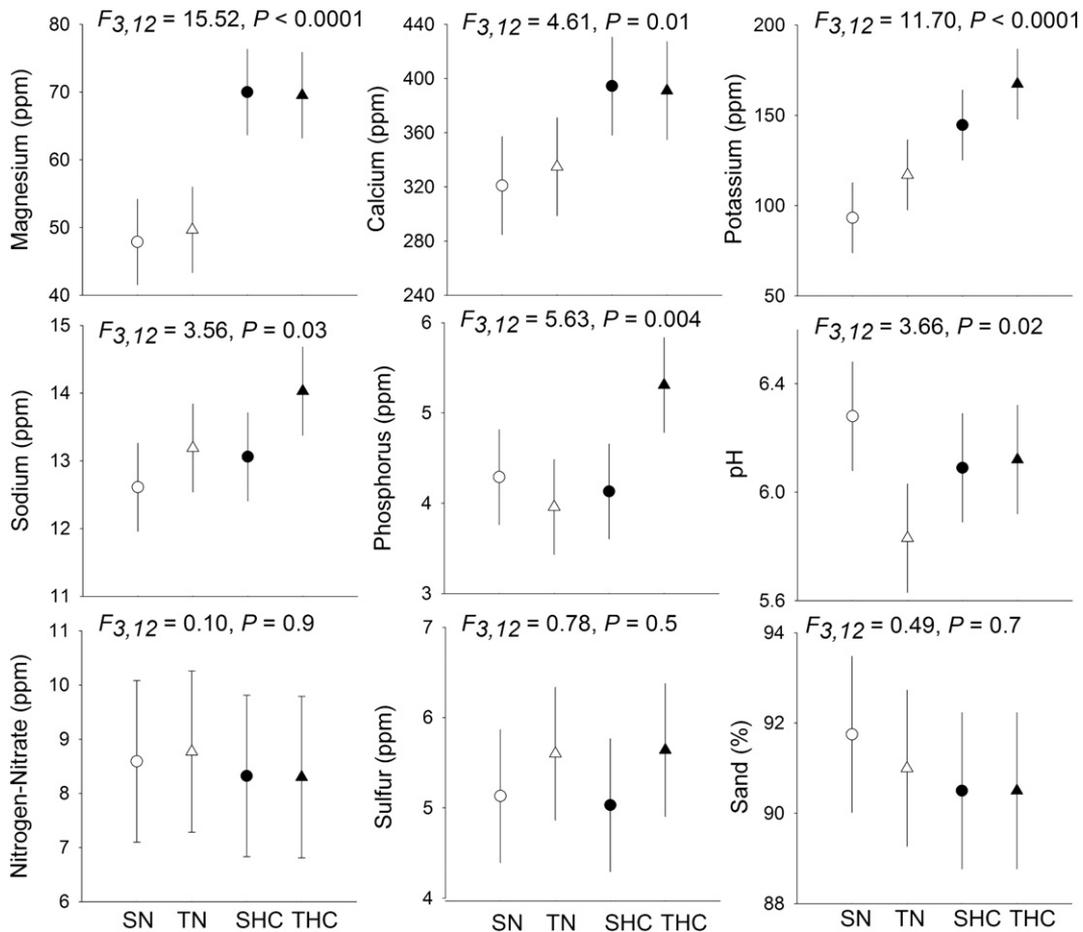

Figure 1. Means and 95% confidence intervals for soil chemistry and texture for each subcommunity (*n* = 16), June 2011, Jim Hogg County, Texas. SHC indicates sparsely-vegetated *Heteropogon contortus;* SN, sparsely-vegetated native; TCH, thickly-vegetated *H. contortus;* TN, thickly-vegetated native.

2013); low uptake leads to higher soil nutrients and could create a positive feedback loop.

Alternatively, *H. contortus* could be a "passenger" that is associated with existing differences among sites rather than driving changes (MacDougall and Turkington, 2005). Indeed, four of the species that increased in presence or cover with *H. contortus* favor disturbance, including *P. ciliare* and *E. lehmanniana* (despite selecting plots with limited nonnatives). If disturbance caused a change in the ecosystem, then perhaps *H. contortus* is taking advantage of this change. Pre-existing soil conditions also may dictate where *H. contortus* becomes dominant. The slight differences we observed in sand content may represent a biological difference that could limit *H. contortus* growth, given that this species tends to favor soils that better retain moisture (Heuzé et al., 2013). The current increase in dominance by *H. contortus* has been slower within the coastal sand sheet to the east (Tjelmeland, 2011). Therefore, we suspect old dunes and sand ridges may be a barrier to dense *H. contortus,* providing refuges for other native plants.

Implications

Although native species can have effects that are similar to nonnative invasive species, understanding the origin, genetics, and details of the changes is important to inform appropriate management. Most native invasive species invade adjacent ecosystems (Nielsen et al., 2011; Carey et al., 2012), unlike *H. contortus* and a few others (e.g., de la Cretaz and Kelty, 1999). Native species that become invasive within the ecosystem where they were once noninvasive require different management strategies than nonnative species, where the goal is complete elimination. Indeed, *H. contortus* plays a role for wildlife when at its historic, and assumed natural, abundance levels (Bielfelt, 2013), such that management should aim to reduce abundance without wholly eliminating this grass. Even if the increased dominance of *H. contortus* is the result of a nonnative variety, further genetic study and careful management are required to retain the native variety. The recent and rapid increase in dominance of *H. contortus* warrants proper management to contain the negative consequences on other native plants and the overall community.

Acknowledgments

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