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Ranching practices interactively affect soil nutrients in subtropical wetlands

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ABSTRACT

Growing demand for food from finite agricultural lands requires the optimization of agricultural management, including the potential interactive effects of these practices on ecosystems. We experimentally examined the interactive and temporal effects of three ranching practices (pasture management intensity, livestock grazing, and prescribed fire) on soil nutrients in 40 geographically-isolated seasonal wetlands. Wetlands were embedded in a subtropical ranch, and the long-term experiment used a factorial design with wetlands as experimental units. Soils (0–15 cm) were collected three times over 9 years; at experiment start (2007), one year after prescribed burns started (2009), and seven years later (2016). Samples were analyzed for soil bulk density, organic matter (OM), total nitrogen (N), carbon (C), and phosphorous (P). A lag effect was observed in response to fire; differences were not observed in 2009, but were detected in 2016 after multiple fire cycles had occurred. Rangeland practices showed 2- and 3-way interactive effects, especially for total P and N stocks. Total P increased most in the wetlands embedded within highly managed, grazed, and burned pastures (2.31 ± 0.76 g m^{-2} yr^{-1}), consistent with legacy effects of historical fertilizer application, cattle activity, and ash deposit due to fire. Wetlands in semi-natural and burned pastures had the lowest rates of soil N storage (5.13 ± 7.33 g m^{-2} yr^{-1}) compared to all other treatments (24.5 ± 10.8 g m^{-2} yr^{-1}). Total C stocks were not significantly impacted by ranching practices throughout the study. In summary, ranching practices can additively and interactively alter soil nutrient stocks after a time lag, and legacy effects of P application still impact wetlands decades later. Our study is one of few focused on ranchland wetlands and shows that wetlands in highly managed, grazed, and/or burned pastures can sequester soil P and N, playing an important role in nutrient processing for agricultural landscapes and watersheds.

1. Introduction

The Food and Agriculture Organization (2011) projects that by 2050, food production will need to increase by 70% to feed the growing population and more than 80% of the increase will be from intensification of current agricultural lands. Among agricultural lands, pastures (used for grazing livestock) are globally important because they occupy 25% of all land area that is not permanently frozen (Wright, 1990). Ranching practices, including pasture intensification, managed livestock grazing intensity and timing, and prescribed burns, can alter soil nutrient dynamics and ecosystem health (Swain et al., 2007; McGranahan et al., 2014). Pastures vary in their degree of human alteration (e.g., fertilization, irrigation/drainage, seeding), ranging from natural rangelands to highly managed pastures. Intensive pasture management historically results in land degradation, eutrophication, biodiversity loss, and an increase in greenhouse gas emissions (IPCC, 2014). Livestock grazing influences greenhouse gas emissions soil erosion and nutrient runoff, sometimes resulting in eutrophication of water bodies (Mckergow et al., 2012). Livestock grazing also increases incorporation of surface litter into the soil and root exudation, therefore cattle removal can result in decreased microbial biomass and mineralizable pools of C and N (Bardgett et al., 1999; Wang et al., 2006). Finally, prescribed fire is commonly used to reduce woody plants cover and to increase nutrient concentrations in soil, hence promoting high quality forage grasses (Griffin and Friedel, 1984; Boughton et al., 2015). Prescribed fire will also reduce N in the soil through volatilization (Reinhart et al., 2016). While the short-term effects of individual ranching practices have been studied in detail (e.g., Vermeire et al., 2014; Little et al., 2015; Mérő et al., 2015), few studies have considered the *long-term* and *interactive* effects that these practices have on

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ecosystems (Boughton et al., 2015).

Geographically-isolated seasonal wetlands are common in ranchlands worldwide and play an important role in soil nutrient sequestration (Gathumbi et al., 2005; McCauley et al., 2015; Dunmola et al., 2010; Cohen et al., 2016). Despite being geographically separated from other aquatic bodies, these wetlands are vast in number, act as hydrological conduits between rangelands and downstream water bodies, and serve a vital ecological role as sinks and transformers of carbon (C), nitrogen (N), and phosphorus (P) (McCauley and Jenkins, 2005; Reddy and DeLaune, 2008; McCauley et al., 2015; Cohen et al., 2016). Wetlands in pastures are focal points for ecological management of agricultural lands because wetlands are significant C reservoirs and potential C sinks (depending on local conditions) that can help mitigate global climate change (IPCC, 2014). Furthermore, N and P from fertilizer runoff and animal wastes can either be retained in wetlands and removed (e.g., denitrification), or else released downstream, resulting in eutrophication and algal blooms downstream (Reed-Andersen et al., 2000; Whigham and Jordan, 2003; Gathumbi et al., 2005). Therefore, functional wetlands within agrarian landscapes are essential to sustainable rangeland practices by serving as on-site water treatment systems and nutrient storage reservoirs, in addition to serving as biodiversity hotspots and livestock watering and forage sites (Dunne et al., 2007; Medley et al., 2015). The question then becomes a matter of evaluating the potential benefit of wetlands to agriculture and using that knowledge to optimize wetland ecological functions (McCauley et al., 2015).

This research used a long-term experiment to evaluate main and interactive effects of ranching practices (pasture management intensity, livestock grazing, and prescribed burns) on soil nutrients in wetlands. By analyzing data periodically collected over nine years, we sought to evaluate the lag time between the implementation of practices and observed effects. We hypothesized that there would be a lag time (< 2 years) before interactive ranching practices would alter soils based on previous studies and observations that suggested that subtropical regions generally show quicker soil responses (< 2 years) to management than other systems due to their high mean annual precipitation and primary productivity (Lugo and Brown, 1993; Oesterheld et al., 1999; Brando et al., 2016; Hu et al., 2016). Specifically, we expected: (i) greater organic matter (OM), total C, and N concentrations in soils of highly managed, grazed, and burned wetlands due to fertility, grazing (i.e., cattle "mulching" and trampling during inefficient grazing of emergent wetland vegetation), and vegetation responses to fire; and (ii) greater total P in highly managed wetlands, regardless of other management practices, due to a legacy effect of historical fertilizer application.

2. Materials and methods

2.1. Study area

This study took place in subtropical pastures of Florida's northern Everglades, at the MacArthur Agro-Ecology Research Center (MAERC) at Buck Island Ranch (27°09′N 81°11′W). Buck Island Ranch is a 4170-ha cattle ranch and research laboratory with > 600 wetlands ranging in size from 0.007 to 41.9 ha and inundated up to 10 months out of the year (Boughton et al., 2015). Buck Island Ranch is in the greater Everglades watershed where eutrophication is a major concern and cattle ranching is regionally prevalent (Gathumbi et al., 2005; Smith et al., 2006). Historical pasture management practices led to two distinctive pasture types: semi-natural and highly-managed pastures. Highly-managed pastures (HMPs) were heavily ditched to drain pastures (Fig. 1) and fertilized annually from 1970 to 1987 with 52 kg ha^{-1} of NH_4SO_4 or NH_4NO_3 and 18 kg ha^{-1} of P_2O_5 and K_2O (Boughton et al., 2011). After 1987, P fertilization stopped, but N fertilization continues on a regular basis with 56 kg ha^{-1} of NH_4SO_4 or NH_4NO_3, usually every two years. In contrast, semi-natural pastures (SNPs) have never been fertilized and have lower ditching densities (Fig. 1). Due to modest elevation differences and pasture management practices, the vegetation differs between the two pasture types, and the underlying soil types are a mosaic of Endoaqualfs such as Bradenton, Felda, and Hicoria muck. Higher elevation areas are dominated by Alaquods like Myakka sands (Sperry, 2004). HMPs are dominated by introduced grasses (*Paspalum notatum*) and SNPs contain *P. notatum,* as well as native grasses like *Andropogon* spp., and *Panicum* spp. (USDA, 1989; Boughton et al., 2015).

2.2. Experimental design and soil sampling

A 2 × 2 × 2 complete factorial design was used to understand the interactive effects of three practices: pasture management (HMP or SNP), cattle removal (grazed or fenced), and prescribed burns (burned or unburned) on soils in wetlands embedded within each experimental pasture (Boughton et al., 2015). The eight different treatment combinations were each replicated five times for a total of 40 wetlands in a randomized block design that accounted for spatial differences (Fig. 1). The 40 wetlands were relatively similar in sizes and hydroperiod. All 40 wetlands were sampled during February/March 2007 before fencing and prescribed fire treatments were applied (i.e., representing only pasture effects). Soils were sampled and analyzed for bulk density, organic matter, total P, total C, and total N. Each wetland was split into 4 quadrants bounded by cardinal directions (N, E, S, and W), plus a central zone (5 zones total), and a marker was randomly placed in each zone. Two 0–15 cm soil cores were randomly collected near the marker within each zone and composited to account for heterogeneity of soils.

Differences in pasture management (HMP vs. SNP) had been in effect for almost 4 decades at the study start (2007), but wetland exclosures and experimental prescribed burns had not yet been implemented (however, prior to the experiment, wetlands were exposed to prescribed burns on an irregular basis as dictated by routine management of surrounding pastures).

Barbed-wire fences were installed around 20 randomly selected wetlands in February/March 2007 (10 per pasture type), directly after initial sampling, and left in place for the entire experiment. These fences successfully exclude cattle, though feral pigs could still enter wetlands and disrupt soils there. The grazed pastures were exposed to typical grazing pressure for a cow-calf operation in Florida and based on pasture condition. Throughout the nine years, grazed HMPs had an average stocking density of 2.56 cow ha^{-1} and a stocking rate of 16.7 animal unit month (AUM) ha^{-1}. On the other hand, grazed SNPs had an average stocking density of 1.14 cows ha^{-1} and 6.89 AUM ha^{-1}. Cattle activity was not tracked throughout the experiment other than through tracking stocking rates of individual pastures, but cattle have free access to wetlands within the same pasture and regularly use wetlands to forage and drink.

All 20 wetlands randomly assigned to the burned treatment experienced prescribed burns in February of 2008 and 2011. Twelve of 20 wetlands were burned again in February 2013, and the remaining 8 wetlands were burned in March 2014 (permitted burns are seasonal and weather-dependent). The 40 wetlands were resampled using the above protocol in February 2009 and May 2016.

2.3. Sample analysis

Each bulk soil sample (0–15 cm depth) was weighed and ~100 g from each duplicate sample per quadrant was composited for a total sub-sample weight of 200 g. The composite samples were oven dried in a ThermoFisher Heratherm oven (ThermoFisher Scientific, Waltham, MA) at 70 °C for 72 h and weighed again, where the difference in weights indicated gravimetric moisture content and final weight was used to calculate soil bulk density. Afterwards, the oven dried samples were ground with a SPEX Sample Prep Mixer Mill 8000 M (Metuchen, NJ) for organic matter (OM) content and total C and N analysis on an Elementar Vario Micro Cube (Elementar Americas Inc., Mount Laurel,

Fig. 1. Map of the 40 experimental wetlands at Archbold Biological Station's Buck Island Ranch. Highly managed pastures are shaded dark gray, and semi-natural pastures appear light gray. Cattle exclosure wetlands are surrounded by black diamonds. Burned wetlands are shaded in black and unburned wetlands appear white.

NJ, detection limits = 10 mg/kg). For 2007 and 2009, total C and N (detection limits = 10 mg/kg) were analyzed at the University of Georgia's Stable Isotope Ecology Lab using a Carlo Erba Strumentazione NA 1500C/H/N Analyzer (Milan, Italy). Percent OM was determined on dried, ground subsamples by ashing for four hours at 550 °C in a ThermoFisher Isotemp muffle furnace (ThermoFisher Scientific, Waltham, MA) via the loss-on-ignition method. The ashed subsamples were then digested with 50 mL of 1 M HCl at 150 °C for 30 min after Andersen (1976) and analyzed for total P on an AQ2 Discrete Analyzer (Seal Analytical, Mequon, WI) using EPA method 365.1 Rev. 2 (detection limit = 0.006 mg P/L; U.S. Environmental Protection Agency, 1993). Data were expressed as stocks (i.e., g m^{-2}) by adjusting calculated concentrations (nutrient% soil) for bulk density (g soil cm^{-3}) and height of the soil core (15 cm).

2.4. Statistical analyses

Data were analyzed separately for each time period: 2007 (initial pasture conditions), 2007–2009 (i.e., the change from initial conditions, 1 year after all treatments were applied), and 2007–2016 (i.e., the change from initial conditions, 9 years after all treatments were applied). Two outliers were removed from analyses (one highly managed, fenced, and burned wetland in 2007–2009 analyses, and a semi-natural, fenced, and unburned wetland from the 2007–2016 analyses); outliers were identified using the maximum normed residual test (Grubbs' test) at a significance level of α = 0.05. Data were analyzed using a generalized linear mixed model with the glmmADMB package from R Statistics (R Foundation for Statistical Computing, Vienna, Austria; Bolker et al., 2012), where pasture type (pasture), exclosures (fenced), and fire (burned) were fixed effects and blocks were a random factor (as in Boughton et al., 2015). For all models, treatments were compared to semi-natural, grazed, and unburned wetlands; the combination representing the absence of experimental treatments. Underlying data distributions were evaluated by computing each model using two distributions (Gaussian and Gamma), where the model with the lowest Akaike Information Criterion (AICc) weight was considered the most plausible model (Burnham and Anderson, 2002). Gamma distributions permit flexibility in data distributions, including dispersion, relative to a Gaussian distribution. The AICc weights were computed using the bbmle package in R (Bolker, 2013). Assumptions of normality and homogeneity were evaluated for each most plausible model by residual plots. Planned contrasts and Tukey's Honestly Significant Difference test (HSD) were used for post-hoc analyses to compare means where significant differences were observed, using the glht function in the multcomp package (Hothorn et al., 2016).

3. Results

3.1. Baseline soil properties (2007)

In 2007, pasture management showed no significant differences in measured wetland soil properties despite being in effect for 4 decades (Table A1). Wide variability among wetlands in mean values for soil properties could have potentially obscured any pasture effects. To account for this variation, treatment effects in subsequent years were analyzed as the annual change from these initial conditions.

3.2. Short-term effects (change from 2007 to 2009)

Singular grazing effects and interactive effects of grazing X pasture were observed in 2009. Cattle exclosure (i.e., fenced wetlands) significantly decreased soil compaction by on average 0.09 g cc^{-1} yr^{-1} relative to grazed wetlands in semi-natural pastures (Table 1). A pasture X fence interaction was observed for soil OM and P stocks. Little change in soil OM stocks were observed in wetlands embedded within SNPs (Fig. 2a). In HMPs however, soil OM stocks significantly decreased (Δ OM < 0) in fenced wetlands and increased in grazed wetlands. In contrast to OM stocks, soil P stocks increased in all wetlands, with similar stocks in all fenced wetlands, regardless of pasture type (Fig. 2b). Grazing in HMPs resulted in the greatest increase in soil P concentrations, whereas total P stocks responded less to grazing in SNPs. There was no evidence that prescribed burns one year earlier had changed soil characteristics (Table 1). None of the three experimental ranching practices significantly affected total C or N stocks at this time.

3.3. Long-term effects (change from 2007 to 2016)

Seven years after all treatments were in effect, significant treatment effects were observed in 2016 for every measured soil property except OM and total C (Table 2). Bulk density did not exhibit interactive effects, but significantly decreased in fenced wetlands (-0.019 ± 0.018 g cc^{-1} yr^{-1}) and increased in grazed wetlands (0.007 ± 0.007 g cc^{-1} yr^{-1}). An interactive effect of pasture X prescribed burns was observed in 2016 for total N with net loss in soil N stocks only in wetlands of burned SNPs ($\Delta N < 0$) (Fig. 3). All other wetlands showed notable increases in N (24.5 ± 10.8 g m^{-2} yr^{-1}) in 2016. Only total P stocks showed three-way interactions in 2016 (Table 2). Fencing and grazing did not affect total P stocks in wetlands in SNPs (Fig. 4a), but a strong fence X burn interaction occurred in HMP wetlands. In wetlands embedded within HMPs, we observed the highest increase in soil total P in grazed and burned wetlands (Fig. 4b). Ranching practices did not alter total C stocks significantly in the long term.

In summary, prescribed burns, cattle removal, and long-term pasture management interacted after a time lag to control P sequestration or N removal in wetlands, where greater responses were observed with greater nutrient loading. In general, semi-natural wetlands were more oligotrophic and thus had more muted responses. Maximal P content were observed in highly managed, grazed, and/or burned wetlands, and a significant decrease in soil N storage occurred in semi-natural and burned wetlands.

4. Discussion

4.1. Lag effects and short term effects

This study examined how ranching practices (pasture management intensity, livestock grazing, prescribed fire) affected soil properties after nine years. Here we observed effects of cattle exclosure after two years, and effects of prescribed burning (done in 2008, 2011, and 2013 or 2014) after 2+ years. This delay to prescribed burns is consistent with the literature (Weiss, 1980; Griffin and Friedel, 1984; Laubhan, 1995) and reflects the need for repeated burns to significantly alter soil nutrients through vegetation shifts, aboveground productivity changes and reduction of nutrient input from litterfall (Christy et al., 2014; Sulwiński et al., 2017). This is important but expected, since time is important in ecological studies. Animal-plant-soil relationships are complex and changes in disturbance regimes can take years to manifest, as evidenced by extinction debt, invasion debt, and colonization credit for soil-organism interactions (Jackson and Sax, 2010; Gilbert and Levine, 2013). Determining lag periods for soil-organism interactions, such as those that control nutrient storage and loss, is critical to understanding how land use practices affect ecological structure and function.

4.2. Interactive effects of ranching practices on soil nutrient concentrations

Our results also show the importance of interactive effects of ranching practices on soil nutrient concentrations. Seven years after all treatments began, we observed increased total P stocks in highly managed, grazed, and/or burned wetlands. Furthermore, only semi-natural and burned wetlands showed reduced soil N storage after seven years. These results strongly support the hypothesis that grazing and

Table 1
Model fixed-effect coefficients ± 95% confidence interval for annual changes between 2007 and 2009 in soil properties, where coefficients represent differences relative to wetlands in semi-natural pastures that are not fenced or burned. * indicates p ≤ 0.05, ** indicates p ≤ 0.01; *** indicates p ≤ 0.001.

Factor	Bulk Density (Δ g cc^{-1} yr^{-1})	Organic Matter (Δ kg m^{-2} yr^{-1})	Total C (Δ kg m^{-2} yr^{-1})	Total P (Δ g m^{-2} yr^{-1})	Total N (Δ g m^{-2} yr^{-1})
Pasture (P)	−0.02 ± 0.07	−1.46 ± 1.63	0.18 ± 0.43	−0.35 ± 2.82	35.9 ± 66.5
Fenced (F)	**−0.09 ± 0.07****	−1.6 ± 1.63	0.13 ± 0.43	−2.29 ± 2.82	10.4 ± 66.5
Burned (B)	−0.02 ± 0.07	−0.83 ± 1.63	0.12 ± 0.43	−1.32 ± 2.82	21.9 ± 66.5
P × F	0.05 ± 0.10	**3.74 ± 2.31****	−0.33 ± 0.61	**10.8 ± 7.96****	−66.0 ± 94.0
P × B	0.01 ± 0.10	0.84 ± 2.31	0.01 ± 0.61	1.00 ± 3.98	20.3 ± 94.0
F × B	0.07 ± 0.10	2.22 ± 2.31	−0.25 ± 0.61	2.50 ± 3.98	−46.3 ± 94.0
P × F × B	−0.05 ± 0.15	−4.70 ± 4.78	0.21 ± 0.86	−4.53 ± 7.96	6.75 ± 134

Bold values represent significant coefficients.

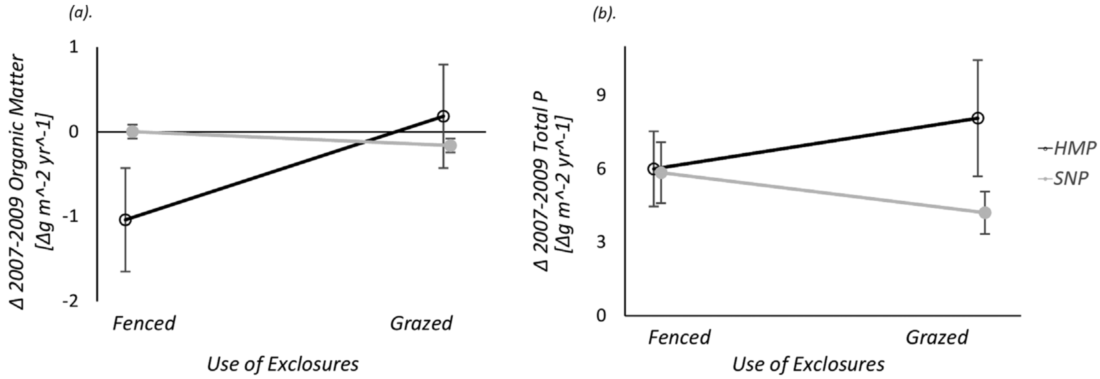

Fig. 2. Significant pasture X cattle removal interaction on OM and total P stocks after 2 years of implementation. Values represent the mean change yr^{-1} ± 95% confidence interval. * indicates values that differed significantly from semi-natural and grazed pastures (absence of experimental treatments).

Table 2
Model coefficients ± 95% confidence interval for annual changes between 2007 and 2016 in soil properties, where coefficients represent differences relative to wetlands in semi-natural pastures that are not fenced or burned. * indicates p ≤ 0.05, ** indicates p ≤ 0.01; *** indicates p ≤ 0.001.

Factor	Bulk Density (Δ g cc^{-1} yr^{-1})	Organic Matter (Δ kg m^{-2} yr^{-1})	Total C (Δ kg m^{-2} yr^{-1})	Total P (Δ g m^{-2} yr^{-1})	Total N (Δ g m^{-2} yr^{-1})
Pasture (P)	−0.01 ± 0.02	−0.22 ± 0.54	0.05 ± 0.10	−0.58 ± 1.08	26.6 ± 17.3**
Fenced (F)	**−0.02 ± 0.02***	−0.34 ± 0.54	0.01 ± 0.10	0.27 ± 1.08	4.08 ± 17.3
Burned (B)	0.00 ± 0.02	0.49 ± 0.54	0.02 ± 0.10	0.06 ± 1.08	13.4 ± 17.3
P × F	−0.01 ± 0.02	0.20 ± 0.77	−0.00 ± 0.15	0.98 ± 1.53	−1.18 ± 24.6
P × B	−0.02 ± 0.02	−0.52 ± 0.77	−0.04 ± 0.15	0.29 ± 1.53	**−27.7 ± 24.6***
F × B	0.01 ± 0.02	−0.56 ± 0.77	0.00 ± 0.15	−0.43 ± 1.53	−0.95 ± 24.6
P × F × B	0.01 ± 0.03	1.05 ± 1.09	−0.03 ± 0.21	**−2.37 ± 2.17***	−8.52 ± 34.9

Bold values represent significant coefficients.

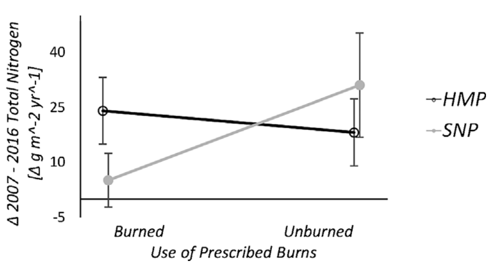

Fig. 3. Significant pasture X burn interaction on total N stocks from 2007 to 2016. Values represent the mean change yr^{-1} ± 95% confidence interval. Positive change in soil properties represent increases in soil property values and negative change values indicate a decrease in the soil property over time. * indicates $p < 0.05$, ** indicates $p < 0.01$; *** indicates $p < 0.001$ for Tukey's HSD test comparison to semi-natural and unburned conditions (absence of experimental treatments).

prescribed burns can additively or interactively enhance nutrient sequestration for wetlands embedded in highly-managed pasture lands. For example, animal-plant-soil interactions played an important role in total P stocks, where highly managed, grazed, and burned wetlands stored P (2.31 g m^{-2} yr^{-1}) at rates comparable to sub-tropical riparian floodplains (3.6 g m^{-2} yr^{-1}; Mitsch et al., 1979). The dominant plant species in grazed and burned wetlands in highly managed pastures was the slow growing and unpalatable *Juncus effusus* which stores less P in aboveground tissues than faster growing dominant species in fenced wetlands (e.g. *Panicum hemitomon)*, resulting in greater soil P concentrations in grazed and burned HMPs.

Soil N stocks was affected by pasture management and prescribed burns. All pasture management X prescribed burns treatments had changes in soil N stocks (24.5 ± 10.8 g m^{-2} yr^{-1}) similar to natural subtropical wetlands (39–46 g m^{-2} yr^{-1}; Mitsch et al., 2005, 2012) except for semi-natural and burned wetlands (5.13 ± 7.33 g m^{-2} yr^{-1}). Most likely, increased nutrient export

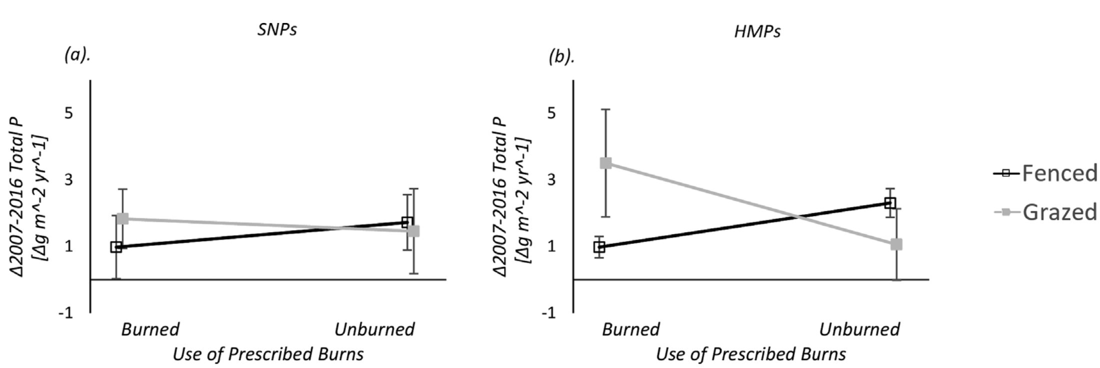

Fig. 4. Significant three-way interactions on total P from 2007 to 2016. Values represent the mean change yr^{-1} ± 95% confidence interval. Positive change in soil properties represent increases in soil property values and negative change values indicate a decrease in the soil property over time. * indicates p ≤ 0.05, ** indicates p ≤ 0.01; *** indicates p ≤ 0.001 for Tukey's HSD test comparison to semi-natural, grazed, and unburned pastures (absence of experimental treatments).

coupled with decreasing nutrient input from litterfall reduced N stocks in semi-natural and burned wetlands. Based on soil nutrient stocks in our study, "isolated" seasonal wetlands can serve substantial roles in watersheds by sequestering nutrients into the soil and thereby limiting downstream eutrophication. Future studies on primary production, greenhouse gas emissions and surface water runoff should be completed to evaluate these wetlands as net nutrient sinks or sources.

4.3. A legacy effect of historical phosphorous application

Lastly, our study demonstrates the importance of a legacy effect of P from fertilizer application over 30 years ago (Swain et al., 2007) and that was made more apparent after other management practices were experimentally implemented. Due to its lack of a gaseous phase and low solubility, P often adsorbs to soil particles (clay, metal oxides, and hydroxides) where it can form insoluble compounds (fixed P) unavailable for plant uptake. Over time, P can build up in the soil until it reaches its saturation point and is removed through microbial metabolism or export from runoff (Reddy et al., 2011). However, due to the long slow nature of P saturation, effects of P addition have generally been documented decades after addition, resulting in a legacy effect of P applications in watersheds (Hart et al., 2004; Gathumbi et al., 2005; Trimble, 2010; Hamilton, 2011; Sharpley et al., 2013).

Pasture intensification (including P application) in the region impacts total P more than other practices, such as cattle stocking rates (Gathumbi et al., 2005; Capece et al., 2007). The detection of a P legacy effect 31 years after application ended aligns with other studies that have shown it can take anywhere from 15 to 100+ years for P application effects to be observed in the top 15 cm of the soil (Reddy et al., 1999). Interestingly, total P was not significantly different in HMPs at the start of the experiment in 2007, suggesting that cattle activity and/or prescribed burns confounded detection of P loading and thus may have played a large role in facilitating P cycling. Our study supports the importance of legacy effects for P in watersheds and the importance of understanding historical land uses applications for current water management practices. Furthermore, results here support the need for more long-term studies on nutrient dynamics and ranching practices.

This experiment was unique in that it studied the interactive effects of different ranching practices for nine years, allowing for long term and interactive effects to be discerned. Significant effects of interactive ranch practices were detected with a sampling regime consisting of new random points every year and despite significant soil heterogeneity. While the results of this study aligned with the literature, results here demonstrate a need for more research on the complex relationships between wetland ecosystem services and management goals. In addition, "isolated" seasonal wetlands in rangelands are often overlooked despite their prevalence and use globally. To better balance ecosystem health and ranching operations, future studies should elucidate the effects of ranching practices on the fate of C, N, and P and include similar experimental designs to compare to results here. Ideally, more frequent sampling intervals and in-depth infiltration rate and runoff studies could be conducted to determine nutrient pathways and examine progressive changes in the soil profile that could influence soil nutrient sequestration.

4.4. Management implications and ecosystem services

Understanding ecosystem services and synergies and tradeoffs among multiple ecosystem services provided by wetlands will improve agroecosystem management (Qiu and Turner, 2013; Butterfield et al., 2016). Land managers who seek to achieve multiple goals associated with soil ecosystem services will need to employ diverse management regimes in wetlands embedded in highly managed pastures (Eastburn et al., 2017). Interestingly, this study showed that wetlands embedded in highly managed pastures were better at soil nutrient sequestration compared to SNPs, but studies on the same wetlands have shown that SNPs contain higher biodiversity and less non-native plant species (Boughton et al., 2010, 2015). Taken together, these studies show evidence of a tradeoff between soil nutrient sequestration and biodiversity. In other words, converting SNP wetlands to HMP wetlands may increase soil nutrient sequestration but decrease biodiversity. Thus, a mosaic of different management practices is likely crucial to maintaining multi-functional ecosystems services in agroecosystems at the landscape level (Power, 2010).

Conflicts of interest

None.

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Appendix A

See Tables A1–A3.

Table A1
Initial mean values ± 95% confidence interval for soil properties in 2007, before wetlands were experimentally burned or fenced to exclude cattle. Note variation among treatments yet to be applied; treatment effects were analyzed as changes after 2007, per wetland.

Pasture	Treatment[a]	Bulk Density (g cc^{-1})	Organic Matter (kg m^{-2})	Total C (kg m^{-2})	Total P (g m^{-2})	Total N (g m^{-2})
Highly Managed Pastures	Fenced × Burned	0.95 ± 0.10	17.6 ± 3.03	8.52 ± 1.14	19.1 ± 5.25	578 ± 73.6
	Grazed × Burned	0.80 ± 0.20	20.7 ± 3.77	9.21 ± 1.43	22.0 ± 9.21	546 ± 77.0
	Fenced × Unburned	0.94 ± 0.06	17.2 ± 2.84	7.94 ± 0.52	15.2 ± 6.17	555 ± 73.0
	Grazed × Unburned	0.97 ± 0.12	15.2 ± 1.98	7.73 ± 1.26	15.2 ± 3.84	534 ± 91.8
Semi-Natural Pastures	Fenced × Burned	0.92 ± 0.07	15.6 ± 2.66	7.74 ± 1.35	17.4 ± 3.86	568 ± 79.6
	Grazed × Burned	0.90 ± 0.13	15.7 ± 5.92	7.90 ± 2.96	13.3 ± 4.63	528 ± 173
	Fenced × Unburned	0.98 ± 0.17	17.1 ± 5.32	8.63 ± 2.94	17.3 ± 5.99	616 ± 179
	Grazed × Unburned	0.97 ± 0.19	15.2 ± 4.59	6.91 ± 2.61	11.0 ± 5.31	469 ± 179

[a] Only pasture treatments (i.e., Highly-managed and Semi-natural pastures) had been applied in 2007; fences to exclude cattle and prescribed fires had not yet been applied.

Table A2
Mean values ± 95% confidence interval for soil properties for the annual change from 2007 to 2009 (i.e., one year after all treatments were applied).

Pasture	Treatment (applied)	Bulk Density (Δ g cc^{-1} yr^{-1})	Organic Matter (Δ kg m^{-2} yr^{-1})	Total C (Δ kg m^{-2} yr^{-1})	Total P (Δ g m^{-2} yr^{-1})	Total N (Δ g m^{-2} yr^{-1})
Highly Managed Pastures	Fenced × Burned	−0.04 ± 0.03	−1.04 ± 1.06	−0.51 ± 0.58	5.85 ± 1.41	−33.1 ± 31.2
	Grazed × Burned	−0.02 ± 0.10	−0.73 ± 2.19	0.75 ± 0.99	6.90 ± 4.05	62.0 ± 54.0
	Fenced × Unburned	0.04 ± 0.08	−1.05 ± 1.54	0.19 ± 0.93	6.15 ± 2.92	9.10 ± 58.5
	Grazed × Unburned	0.00 ± 0.02	1.10 ± 0.82	1.04 ± 0.56	9.25 ± 2.52	65.0 ± 37.1
Semi-Natural Pastures	Fenced × Burned	−0.02 ± 0.03	−0.42 ± 0.65	0.47 ± 0.31	5.20 ± 1.07	23.2 ± 25.7
	Grazed × Burned	0.01 ± 0.05	0.87 ± 1.77	0.07 ± 0.04	38.9 ± 27.6	2.56 ± 1.59
	Fenced × Unburned	−0.07 ± 0.03	0.42 ± 0.65	0.93 ± 0.95	7.50 ± 2.23	45.1 ± 73.0
	Grazed × Unburned	0.00 ± 0.01	−1.18 ± 0.74	0.44 ± 0.99	4.21 ± 1.23	34.7 ± 64.0

Table A3
Mean values ± 95% confidence interval for soil properties for the annual change from 2007 to 2016.

Pasture	Treatment	Bulk Density (Δ g cc^{-1} yr^{-1})	Organic Matter (Δ kg m^{-2} yr^{-1})	Total C (Δ kg m^{-2} yr^{-1})	Total P (Δ g m^{-2} yr^{-1})	Total N (Δ g m^{-2} yr^{-1})
Highly Managed Pastures	Fenced × Burned	−0.02 ± 0.02	0.68 ± 0.37	0.32 ± 0.27	1.96 ± 1.10	20.8 ± 16.4
	Grazed × Burned	−0.01 ± 0.02	1.04 ± 0.44	0.15 ± 0.17	2.31 ± 0.76	6.58 ± 12.2
	Fenced × Unburned	−0.02 ± 0.01	0.72 ± 0.25	0.47 ± 0.22	3.50 ± 2.84	27.4 ± 9.08
	Grazed × Unburned	−0.01 ± 0.02	0.58 ± 0.50	0.08 ± 0.14	1.06 ± 1.07	3.68 ± 9.47
Semi-Natural Pastures	Fenced × Burned	0.00 ± 0.01	1.42 ± 0.83	0.37 ± 0.16	1.67 ± 0.95	19.8 ± 15.8
	Grazed × Burned	0.01 ± 0.01	0.53 ± 0.32	0.52 ± 0.32	1.72 ± 0.83	33.1 ± 20.3
	Fenced × Unburned	−0.00 ± 0.01	0.94 ± 0.39	0.29 ± 0.13	1.82 ± 0.89	16.6 ± 11.1
	Grazed × Unburned	0.01 ± 0.01	0.60 ± 0.68	0.45 ± 0.35	1.46 ± 1.28	29.0 ± 22.1

References

Andersen, J., 1976. An ignition method for determination of total phosphorus in lake sediments. Water Res. 10 (4), 329–331. http://dx.doi.org/10.1016/0043-1354(76)90175-5.

Bardgett, R.D., Mawdsley, J.L., Edwards, S., Hobbs, P.J., Rodwell, J.S., Davies, W.J., 1999. Plant species and nitrogen effects on soil biological properties of temperate upland grasslands. Funct. Ecol. 13 (5), 650–660. http://dx.doi.org/10.1046/j.1365-2435.1999.00362.x.

Bolker, B., Skaug, H., Magnusson, A., Nielsen, A., 2012. Retrieved from: http://glmmadmb.r-forge.r-project.org/glmmADMB.html.

Bolker, B., 2013. Package 'bbmle'. Retrieved from: http://cran.r-project.org/web/packages/bbmle/index.html.

Boughton, E.H., Quintana-Ascencio, P.F., Bohlen, P.J., Jenkins, D.G., Pickert, R., 2010. Land-use and isolation interact to affect wetland plant assemblages. Ecography 33 (3), 461–470. http://dx.doi.org/10.1111/j.1600-0587.2009.06010.x.

Boughton, E.H., Quintana-Ascencio, P.F., Nickerson, D., Bohlen, P.J., 2011. Management intensity affects the relationship between non-native and native species in subtropical wetlands. Appl. Veg. Sci. 14 (2), 210–220. http://dx.doi.org/10.1111/j.1654-109x.2010.01116.x.

Boughton, E.H., Quintana-Ascencio, P.F., Bohlen, P.J., Fauth, J.E., Jenkins, D.G., 2015. Interactive effects of pasture management intensity, release from grazing and prescribed fire on forty subtropical wetland plant assemblages. J. Appl. Ecol. 53 (1), 159–170. http://dx.doi.org/10.1111/1365-2664.12536.

Brando, P.M., Oliveria-Santos, C., Rocha, W., Cury, R., Coe, M.T., 2016. Effects of experimental fuel additions on fire intensity and severity: unexpected carbon resilience of a neotropical forest. Glob. Change Biol. 22 (7), 2516–2525. http://dx.doi.org/10.1111/gcb.13172.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference: a practical information-theoretic approach, 2nd edition. Springer-Verlag, New York.

Butterfield, B.J., Camhi, A.L., Rubin, R.L., Schwalm, C.R., 2016. Tradeoffs and compatibilities among ecosystem services. Ecosyst. Serv. 207–243. http://dx.doi.org/10.1016/bs.aecr.2015.09.002.

Capece, J.C., Campbell, K.L., Bohlen, P.J., Graetz, D.A., Portier, K.M., 2007. Soil phosphorus, cattle stocking rates, and water quality in subtropical pastures in Florida, USA. Rangel. Ecol. Manag. 60 (1), 19–30. http://dx.doi.org/10.2111/05-072r1.1.

Christy, J.A., Mccain, C.N., Greene, S.E., Lippert, J.D., 2014. Fire effects in a Montane wetland, Central Cascade Range, Oregon. Madroño 61 (2), 201–217. http://dx.doi.org/10.3120/0024-9637-61.2.201.

Cohen, M.J., Creed, I.F., Alexander, L., Basu, N.B., Calhoun, A.J., Craft, C., D'Amico, E., DeKeyser, E., Fowler, L., Golden, H.E., Jawitz, J.W., 2016. Do geographically isolated wetlands influence landscape functions? Proc. Natl. Acad. Sci. 113 (8), 1978–1986.

Dunmola, A.S., Tenuta, M., Moulin, A.P., Yapa, P., Lobb, D.A., 2010. Pattern of greenhouse gas emission from a Prairie Pothole agricultural landscape in Manitoba, Canada. Can. J. Soil Sci. 90 (2), 243–256. http://dx.doi.org/10.4141/cjss08053.

Dunne, E., Smith, J., Perkins, D., Clark, M., Jawitz, J., Reddy, K., 2007. Phosphorus storages in historically isolated wetland ecosystems and surrounding pasture uplands. Ecol. Eng. 31 (1), 16–28. http://dx.doi.org/10.1016/j.ecoleng.2007.05.004.

Eastburn, D.J., O'Geen, A.T., Tate, K.W., Roche, L.M., 2017. Multiple ecosystem services in a working landscape. PLoS One 12 (3). http://dx.doi.org/10.1371/journal.pone.0166595.

FAO, 2011. The State of the World's Land and Water Resources for Food and Agriculture (SOLAW) – Managing Systems at Risk. Food and Agriculture Organization of the United Nations, Rome and Earthscan, London.

Gathumbi, S.M., Bohlen, P.J., Graetz, D.A., 2005. Nutrient enrichment of wetland vegetation and sediments in subtropical pastures. Soil Sci. Soc. Am. J. 69 (2), 539. http://dx.doi.org/10.2136/sssaj2005.0539.

Griffin, G.F., Friedel, M.H., 1984. Effects of fire on central Australian rangelands. Fire and fuel characteristics and changes in herbage and nutrients. Aust. Ecol. 9 (4), 381–393. http://dx.doi.org/10.1111/j.1442-9993.1984.tb01375.x.

Gilbert, B., Levine, J.M., 2013. Plant invasions and extinction debts. In: USA. Proceedings of the National Academy of Sciences 110. pp. 1744–1749.

Hamilton, S.K., 2011. Biogeochemical time lags may delay responses of streams to ecological restoration. Freshw. Biol. 57, 43–57. http://dx.doi.org/10.1111/j.1365-2427.2011.02685.x.

Hart, M.R., Quin, B.F., Nguyen, M.L., 2004. Phosphorus runoff from agricultural land and direct fertilizer effects. J. Environ. Qual. 33, 1954–1972. http://dx.doi.org/10.2134/jeq2004.1954.

Hothorn, T., Bretz, F., Westfall, P., 2016. Simultaneous Inference in General Parametric Models. Available from URL: https://cran.r-project.org/web/packages/multcomp/multcomp.html.

Hu, Z., Li, S., Guo, Q., Niu, S., He, N., Li, L., Yu, G., 2016. A synthesis of the effect of grazing exclusion on carbon dynamics in grasslands in China. Glob. Change Biol. 22 (4), 1385–1393. http://dx.doi.org/10.1111/gcb.13133.

IPCC, 2014. In: Core Writing Team, Pachauri, R.K., Meyer, L.A. (Eds.), 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. IPCC, Geneva, Switzerland (151 pp).

Jackson, S.T., Sax, D.F., 2010. Balancing biodiversity in a changing environment: extinction debt, immigration credit and species turnover. Trends Ecol. Evol. 25 (3), 153–160. http://dx.doi.org/10.1016/j.tree.2009.10.001.

Laubhan, M.K., 1995. Effects of prescribed fire on moist-soil vegetation and soil macronutrients. Wetlands 15, 159. http://dx.doi.org/10.1007/BF03160669.

Little, I.T., Hockey, P.A., Jansen, R., 2015. Impacts of fire and grazing management on South Africa's moist highland grasslands: a case study of the Steenkampsberg Plateau, Mpumalanga, South Africa. Bothalia 45 (1). http://dx.doi.org/10.4102/abc.v45i1.1786.

Lugo, A.E., Brown, S., 1993. Management of tropical soils as sinks or sources of atmospheric carbon. Plant Soil 149 (1), 27–41. http://dx.doi.org/10.1007/BF00010760.

McCauley, L.A., Jenkins, D.G., 2005. GIS-based estimates of former and current depressional wetlands in an agricultural landscape. Ecol. Appl. 15, 1199–1208.

McCauley, L.A., Anteau, M.J., Post van der Burg, M., Wiltermuth, M.T., 2015. Land use and wetland drainage affect water levels and dynamics of remaining wetlands. Ecosphere 6 (6), 92. http://dx.doi.org/10.1890/ES14-00494.1.

McGranahan, D.A., Henderson, C.B., Hill, J.S., Raicovich, G.M., Wilson, W.N., Smith, C.K., 2014. Patch burning improves forage quality and creates grass-bank in old-field pasture: results of a demonstration trial. Southeast. Nat. 13 (2), 200–207. http://dx.doi.org/10.1656/058.013.0203.

Mckergow, L.A., Rutherford, J., Timpany, G.C., 2012. Livestock-generated nitrogen exports from a pastoral wetland. J. Environ. Qual. 41 (5), 1681. http://dx.doi.org/10.2134/jeq2010.0435.

Medley, K.A., Boughton, E.H., Jenkins, D.G., Fauth, J.E., Bohlen, P.J., Quintana-Ascencio, P.F., 2015. Intense ranchland management tips the balance of regional and local factors affecting wetland community structure. Agric. Ecosyst. Environ. 212, 207–244. http://dx.doi.org/10.1016/j.agee.2015.06.024.

Mérő, T.O., Lontay, L., Lengyel, S., 2015. Habitat management varying in space and time: the effects of grazing and fire management on marshland birds. J. Ornithol. 156 (3), 579–590. http://dx.doi.org/10.1007/s10336-015-1202-9.

Mitsch, W.J., Dorage, C.L., Wiemhoff, J.R., 1979. Ecosystem dynamics and a phosphorus budget of an alluvial Cypress Swamp in Southern Illinois. Ecology 60 (6), 1116. http://dx.doi.org/10.2307/1936959.

Mitsch, W.J., Day, J.W., Zhang, L., Lane, R.R., 2005. Nitrate-nitrogen retention in wetlands in the Mississippi River Basin. Ecol. Eng. 24 (4), 267–278. http://dx.doi.org/10.1016/j.ecoleng.2005.02.005.

Mitsch, W.J., Bernal, B., Nahlik, A.M., Mander, Ü., Zhang, L., Anderson, C.J., et al., 2012. Wetlands, carbon, and climate change. Landsc. Ecol. 28 (4), 583–597. http://dx.doi.org/10.1007/s10980-012-9758-8.

Oesterheld, M., Loreti, J., Semmartin, M., Paruelo, J.M., 1999. Grazing, Fire, and Climate Effects on Primary Productivity of Grasslands and Savannas. In Ecosystems of Disturbed Ground, 1st ed. Elsevier Science B.V., Amsterdam, the Netherlands, pp. 287–306.

Power, A.G., 2010. Ecosystem services and agriculture: tradeoffs and synergies. Phil. Trans. R. Soc. B 365 (1554), 2959–2971. http://dx.doi.org/10.1098/rstb.2010.0143.

Qiu, J., Turner, M.G., 2013. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proc. Natl. Acad. Sci. 110 (29), 12149–12154. http://dx.doi.org/10.1073/pnas.1310539110.

Reddy, K.R., White, J.R., Wright, A., Chua, T., 1999. Influence of phosphorus loading in soil and water columns of wetlands. Phosphorus Biogeochemistry in Subtropical Ecosystems. Lewis, Boca Raton, FL, pp. 249–274.

Reddy, K.R., DeLaune, R.D., 2008. Biogeochemistry of Wetlands: Science and Applications. CRC Press, Boca Raton.

Reddy, K.R., Newman, S., Osborne, T.Z., White, J.R., Fitz, H.C., 2011. Phosphorus cycling in the Greater Everglades ecosystem: legacy phosphorus implications for management and restoration. Crit. Rev. Environ. Sci. Technol. 41 (6), 149–186. http://dx.doi.org/10.1080/10643389.2010.530932.

Reed-Andersen, T., Carpenter, S.R., Lathrop, R.C., 2000. Phosphorus flow in a watershed-lake ecosystem. Ecosystems 3 (6), 561–573. http://dx.doi.org/10.1007/s100210000049.

Reinhart, K.O., Dangi, S.R., Vermeire, L.T., 2016. The effect of fire intensity, nutrients, soil microbes, and spatial. Plant Soil 409, 203–216. http://dx.doi.org/10.1007/s11104-016-2957-3.

Sharpley, A., Jarvie, H.P., Buda, A., May, L., Spears, B., Kleinman, P., 2013. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. J. Environ. Qual. 42, 1308–1326. http://dx.doi.org/10.2134/jeq2013.03.0098.

Smith, V.H., Joye, S.B., Howarth, R.W., 2006. Eutrophication of freshwater and marine ecosystems. Limnol. Oceanogr. 51 (1 (Part 2)), 351–355. http://dx.doi.org/10.4319/lo.2006.51.1_part_2.0351.

Sperry, C.M., 2004. Soil Phosphorus in Isolated Wetlands of Subtropical Bee Cattle Pastures [Thesis]. University of Florida, Gainesville, FL (65 p).

Sulwiński, M., Mętrak, M., Suska-Malawska, M., 2017. Long-term fire effects of the drained open fen on organic soils. Arch. Environ. Prot. 43 (1). http://dx.doi.org/10.1515/aep-2017-0002.

Swain, H.M., Bohlen, P.J., Campbell, K.L., Lollis, L.O., Steinman, A.D., 2007. Integrated ecological and economic analysis of ranch management systems: an example from South Central Florida. Rangel. Ecol. Manag. 60 (1), 1–11. http://dx.doi.org/10.2111/05-071r1.1.

Trimble, S.W., 2010. Streams, valleys and floodplains in the sediment cascade. Sedim. Cascades 307–343. http://dx.doi.org/10.1002/9780470682876.ch11.

USDA, 1989. Soil Survey of Highland County, Florida. National Soil Conservation Service, Washington, D.C.

U.S. Environmental Protection Agency, 1993. Method 365.1 – Determination of Phosphorus by Semi-Automated Colorimetry, Revision 2.0. (August).

Vermeire, L.T., Crowder, J.L., Wester, D.B., 2014. Semiarid rangeland is resilient to summer fire and postfire grazing utilization. Rangel. Ecol. Manag. 67 (1), 52–60. http://dx.doi.org/10.2111/rem-d-13-00007.1.

Wang, K., Mcsorley, R., Bohlen, P., Gathumbi, S., 2006. Cattle grazing increases microbial biomass and alters soil nematode communities in subtropical pastures. Soil Biol. Biochem. 38 (7), 1956–1965. http://dx.doi.org/10.1016/j.soilbio.2005.12.019.

Weiss Jr., T.E., 1980. The Effects of Fire and Nutrient Availability on the Pitcher Plant Sarracenia flava L., Ph.D Dissertation. University of Georgia, Athens, GA.

Whigham, D.F., Jordan, T.E., 2003. Isolated wetlands and water quality. Wetlands 23 (3), 541–549. http://dx.doi.org/10.1672/0277-5212(2003)023[0541:iwawq]2.0.co;2.

Wright, D., 1990. Human impacts on energy flow through natural ecosystems, and implications for species endangerment. Ambio 19 (4), 189–194 (10.230.7/4313691).