9-1-2022

Patch-Burn Grazing Impacts Forage Resources in Subtropical Humid Grazing Lands

Elizabeth H. Boughton
Archbold Biological Station

Nuria Gomez-Casanovas
University of Illinois Urbana-Champaign

Hilary Swain
Archbold Biological Station

Carl Bernacchi
USDA Agricultural Research Service

Raoul K. Boughton
Archbold Biological Station

See next page for additional authors

Boughton, Elizabeth H.; Gomez-Casanovas, Nuria; Swain, Hilary; Bernacchi, Carl; Boughton, Raoul K.; Brinsko, Keith; Li, Haoyu; Rivero, Alan; DeLucia, Evan H.; and Sparks, Jed, "Patch-Burn Grazing Impacts Forage Resources in Subtropical Humid Grazing Lands" (2022). *Publications from USDA-ARS / UNL Faculty*. 2593.
https://digitalcommons.unl.edu/usdaarsfacpub/2593

Authors

Elizabeth H. Boughton, Nuria Gomez-Casanovas, Hilary Swain, Carl Bernacchi, Raoul K. Boughton, Keith Brinsko, Haoyu Li, Alan Rivero, Evan H. DeLucia, and Jed Sparks

PAtch-Burn Grazing Impacts Forage Resources in Subtropical Humid Grazing Lands☆

Authors: Boughton, Elizabeth H., Gomez-Casanovas, Nuria, Swain, Hilary, Bernacchi, Carl, Boughton, Raoul K., et al.

Source: Rangeland Ecology and Management, 84(1) : 10-21

Published By: Society for Range Management

URL: https://doi.org/10.1016/j.rama.2022.05.004

Rangeland Ecology & Management 84 (2022) 10–21

Contents lists available at ScienceDirect

Rangeland Ecology & Management

journal homepage: www.elsevier.com/locate/rama

Original Research

Patch-Burn Grazing Impacts Forage Resources in Subtropical Humid Grazing Lands☆

Elizabeth H. Boughton[1,*], Nuria Gomez-Casanovas[2,3], Hilary Swain[4], Carl Bernacchi[5], Raoul K. Boughton[1], Keith Brinsko[1], Haoyu Li[1], Alan Rivero[1], Evan H. DeLucia[6], Jed Sparks[7]

[1] *Archbold Biological Station, Buck Island Ranch, 300 Buck Island Ranch Road, Lake Placid, FL 33852*
[2] *Institute for Sustainability, Energy, and Environment, University of Illinois at Urbana-Champaign, Urbana, Illinois, USA*
[3] *Center for Advance Bioenergy and Bioproduct Innovation, University of Illinois at Urbana-Champaign, Urbana, Illinois, USA*
[4] *Archbold Biological Station, 123 Main Drive, Venus, FL 33960*
[5] *Global Change Photosynthesis Research Unit, USDA/ARS, Urbana, IL, 61801, United States of America*
[6] *nstitute for Sustainability, Energy, and Environment, University of Illinois at Urbana-Champaign, Urbana, IL, USA*
[7] *Cornell University, Department of Ecology & Evolutionary Biology, E409 Corson Hall, Ithaca, NY 14853*

ARTICLE INFO

Article history:
Received 6 December 2021
Revised 15 April 2022
Accepted 16 May 2022

Key Words:
cattle
fire
forage nutritive value
forage quantity
LTAR
pyric herbivory

ABSTRACT

Subtropical humid grazing lands represent a large global land use and are important for livestock production, as well as supplying multiple ecosystem services. Patch-burn grazing (PBG) management is applied in temperate grazing lands to enhance environmental and economic sustainability; however, this management system has not been widely tested in subtropical humid grazing lands. The objective of this study was to determine how PBG affected forage resources, in comparison with the business-as-usual full-burn (FB) management in both intensively managed pastures (IMP) and seminative (SN) pastures in subtropical humid grazinglands. We hypothesized that PBG management would create patch contrasts in forage quantity and nutritive value in both IMP and SN pastures, with a greater effect in SN pastures. A randomized block design experiment was established in 2017 with 16 pastures (16 ha each), 8 each in IMP and SN at Archbold Biological Station's Buck Island Ranch in Florida.

PBG management employed on IMP and SN resulted in creation of patch contrast in forage nutritive value and biomass metrics, and recent fire increased forage nutritive value. Residual standing biomass was significantly lower in burned patches of each year, creating heterogeneity within both pasture types under PBG. PBG increased digestible forage production in SN but not IMP pastures. These results suggest that PBG may be a useful management tool for enhancing forage nutritive value and creating patch contrast in both SN and IMP, but PBG does not necessarily increase production relative to FB management. The annual increase in tissue quality and digestible forage production in a PBG system as opposed to once every 3 yr in an FB system is an important consideration for ranchers. Economic impacts of PBG and FB management in the two different pasture types are discussed, and we compare and contrast results from subtropical humid grazing lands with continental temperate grazing lands.

Introduction

Globally, grass-dominated ecosystems, including grasslands, savannas, and grassy woodlands, cover approximately 3.5 billion ha, about 30–40% of the Earth's land surface (White et al. 2000; Gibson 2009; Dixon et al. 2014) and approximately 20% of the tropics (Parr et al. 2014; Bond 2016), including both old-growth grasslands (Veldan et al. 2015) and human-modified grasslands (Veldman 2016). A large portion of these natural and managed grasslands is used for livestock grazing (Bengtsson et al. 2019); in the United States, 11 million ha of subtropical humid grasslands support about 30% of the US beef cow herd (Sigua 2010).

Grass-based forage is the nutritional foundation for all cattle-grazing enterprises, both temperate and tropical (Peters et al. 2013; Scasta et al. 2016). Forage availability and nutritive value drive cattle weight change and reproductive performance and are there-

---

☆ This study was funded by NIFA (2016-67019-24988) and a US Dept of Agriculture (USDA) cooperative agreement with Archbold (58-0202-7-001). This research is a contribution of the Long-Term Agroecosystem Research network, supported by the USDA.

* Correspondence: Dr Elizabeth H. Boughton. Archbold Biological Station, 300 Buck Island Ranch Rd, Lake Placid, FL 33852.

E-mail address: eboughton@archbold-station.org (E.H. Boughton).

https://doi.org/10.1016/j.rama.2022.05.004

fore critical to economic returns (Arthington et al. 2007; Thornton 2010). Optimizing forage resources is a critically important management objective, especially in subtropical humid pastures because they are dominated by C4 grasses naturally lower in forage nutritive value (Archimède et al. 2011). Productive grass forages also provide environmental benefits such as soil and water conservation, nutrient retention, carbon uptake, and habitat for grassland birds (Liebman et al. 2013; Bengtsson et al. 2019). Given the agronomic and ecological importance of grassland agroecosystems, combined with their large global land cover, it is essential to assess innovative strategies for grassland management.

Prescribed fire has been used globally as a management strategy to improve forage availability and nutritive value in fire-prone subtropical humid grasslands in Australia, Brazil, and southeastern United States (Overbeck et al. 2005; Noss, 2013; Swain et al. 2013; Cowley et al. 2014). Often mimicking natural fire processes, prescribed fire removes dead biomass, allows for new regrowth, reduces woody encroachment, and manages cattle distribution (Tothill 1971; McGranahan et al. 2016). Historically, lightning-ignited fires occurred frequently in subtropical humid grasslands of the United States and there is no evidence for anthropogenic fire ignitions (Noss, 2013; Boughton et al. 2018). Cattle managers continued to maintain fire on the landscape even when federal government fire-suppression policies were promulgated, continuing to use prescribed fire to burn entire pastures, typically every 3 yr.

One management innovation is patch-burn grazing (PBG), burning a fraction of a pasture annually versus burning whole pastures (full burns [FBs]) every second or third yr, the business-as-usual prescribed fire strategy (Fuhlendorf and Engle 2001, 2004). PBG might enhance both forage quality and availability while creating spatially heterogeneous vegetation structure (Fuhlendorf and Engle 2001, 2004; Allred et al. 2011, 2014; McGranahan et al. 2012; Augustine and Derner 2014; Scasta et al. 2016; Fulgoni et al. 2020; Spiess et al. 2020; Wang et al. 2020). PBG includes targeted spatial and temporal manipulations of fire to manage specifically for pyric herbivory, the fire-grazing interaction where grazing is driven by fire (Fuhlendorf and Engle 2001, 2004; Scasta et al. 2016). PBG includes rotational application of fire to spatially discrete portions of a pasture that attracts cattle to graze primarily on recently burned areas, where responding growth increases in digestibility (i.e., the proportion of organic matter in the forage that is digested) and has greater amounts of crude protein and other important nutrients, such as phosphorus (needed for rumen microbes Satter et al. 2005), while avoiding unburned areas (Allred et al. 2011; McGranahan et al. 2013; Scasta et al. 2016). In contrast, typical FB management promoted homogeneous spatial distribution of cattle grazing (McGranahan et al. 2016). Although most studies on PBG have been conducted in temperate grasslands, a few studies suggest PBG will improve forage resources in fire-prone subtropical humid grasslands (Duvall and Whitaker 1964; McGranahan et al. 2013, 2014; Scasta et al. 2016).

PBG can improve or maintain production outcomes in cattle-grazing operations compared with traditional management, but effects may vary depending on grassland type, productivity, and stocking density (Duvall and Whitaker 1964; Vermeire et al. 2004; Limb et al. 2011; Augustine and Derner 2014; McGranahan et al. 2014; Spiess et al. 2020). Within patch-burned systems, forage in burned patches typically has greater crude protein than forage in unburned patches (Allred et al. 2011; McGranahan et al. 2014; Scasta et al. 2016). However, low productivity or high stocking can limit the benefits of PBG by reducing fuel loads and, in turn, burn completeness, leading to reduced patch contrast in crude protein and other attributes between burned and unburned patches (McGranahan et al. 2013; Augustine and Derner 2014; Scasta et al. 2016; Spiess et al. 2020). Differences in stocking density or productivity could drive different responses in seminatural grasslands versus intensively managed pastures. However, studies in grasslands across management intensites have shown relatively consistent results. For example, McGranahan et al. (2014) found that in an old field pasture located in the southeast United States dominated by *Andropogon virginicus,* a weedy native bunchgrass, PBG increased crude protein content and created spatially heterogeneous vegetation. Similarly, in North Dakota, in the northern Great Plains of the United States, a patch-burning study conducted on grasslands dominated by introduced grasses and legumes showed that PBG maintained or improved livestock performance during drought compared with conventional management and created patch contrast in vegetation attributes (Spiess et al. 2020).

In contrast to the extensively studied temperate grasslands, there are significant knowledge gaps on the impacts of PBG on forage resources in subtropical humid grazing lands, as well as the usefulness of PBG across subtropical grasslands with variable management intensities and abundance of introduced grasses. Like other global grasslands, subtropical humid grasslands contain variation in species composition, stocking densities, and management intensities. Throughout the humid C4 grasslands of the southeastern United States, the highest-intensity, improved pastures are monocultures of introduced grasses with relatively high stocking density while lower-intensity, seminative pastures (mix of native and introduced grasses) have moderate stocking density and native rangeland (only native grasses) has low stocking densities.

To test the impacts of PBG on subtropical humid grasslands we took advantage of a large-scale experimental pasture array installed at Buck Island Ranch (BIR) in Florida, part of the Archbold−University of Florida US Department of Agriculture Long-Term Agroecosystem Research (LTAR) site. This is one of 18 sites forming a nationwide network designed to assess strategies for sustainable intensification of agriculture at the national scale (Kleinman et al. 2018; Spiegal et al. 2018). LTAR sites are all contributing toward the network's continental scale "Common Experiment," each site comparing production, environmental, and social outcomes of an aspirational treatment and a business-as-usual treatment. We present here forage quantity and quality results from the Archbold−University of Florida LTAR site's Common Experiment, with PBG as the aspirational treatment and FB as the business-as-usual treatment. We asked "How did PBG affect annual aboveground net primary productivity (ANPP), annual digestible forage production, residual biomass, vegetation growth rate, and forage nutritive value (crude protein, in vitro organic matter digestibility [IVOMD], and total phosphorus) compared with traditional FB pasture management in high-intensity improved pastures and low-intensity seminative pastures?" We hypothesized that PBG management would create patch contrasts in forage quantity and forage nutritive value in both improved and seminative pastures. However, we expected the magnitude of forage responses to PBG would be greater in native grass−dominated seminative pastures with lower stocking densities compared with improved pastures with higher stocking densities.

Methods

Study site

This study took place at the Archbold Biological Station's BIR, located in south-central Florida (27°09′N, 81°11′W), a 4 254-ha commercial cattle ranch (Fig. 1). The climate is subtropical with a mean annual temperature of 22°C (1998−2008). Mean annual precipitation is 1 218 mm (1992−2008), of which 69% falls during the wet season (June−October) (Swain et al. 2007).

BIR is a typical cow-calf operation with ~2 500 cows, 200 heifers, and 200 bulls. The land has been used for cattle production since the 1920s. BIR contains 1 953 ha of intensively man-

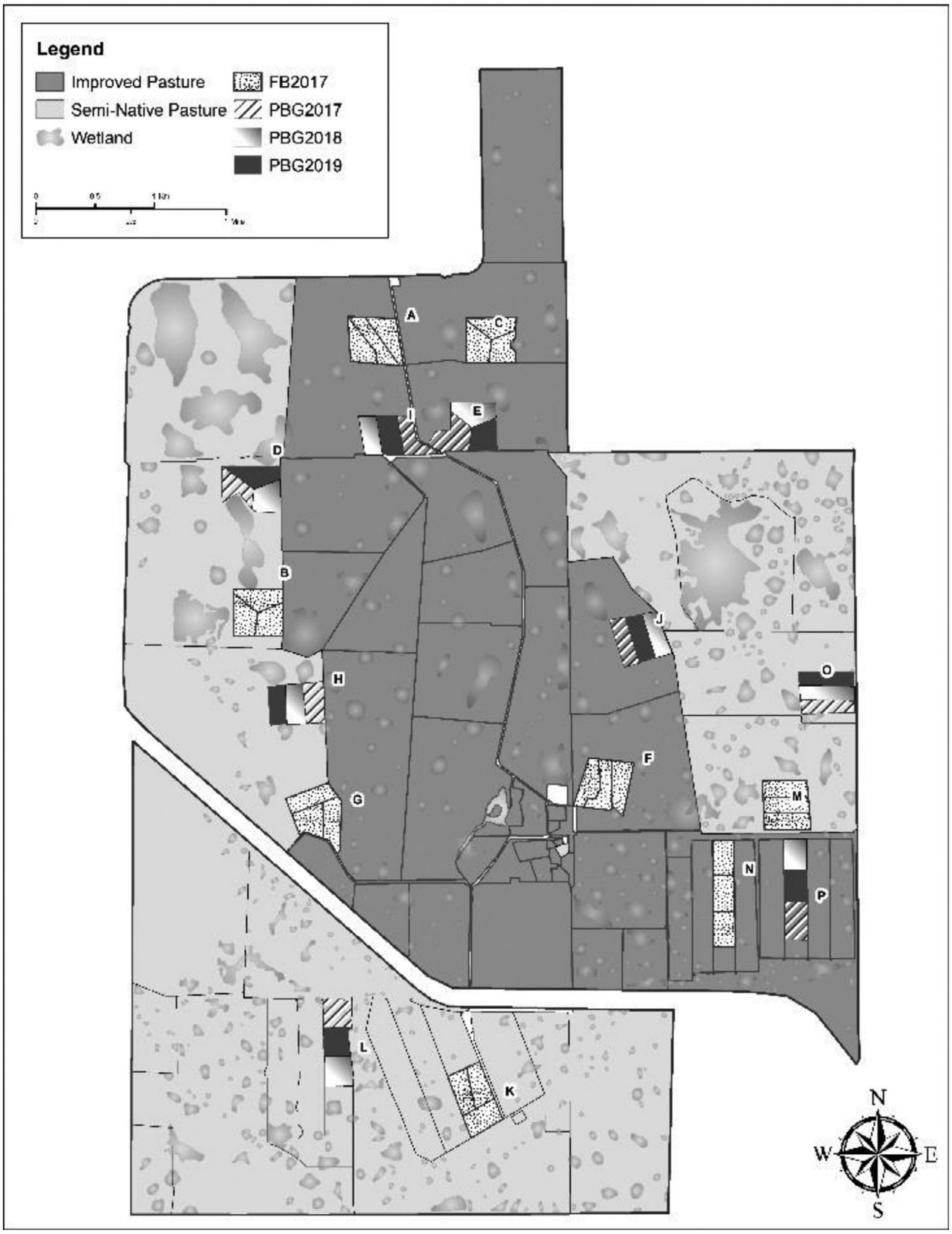

Figure 1. Archbold's Buck Island Ranch (BIR), a 4 254-ha working cattle ranch located in south-central Florida. Experimental pastures are denoted with letters. This experiment is BIR's Long-Term Agroecosystem Research Common Experiment. FB indicates full burn; PBG, patch-burn grazing; 2017-2019, yr of prescribed fire treatments. Lines within pastures represent the pasture patches used to delineate fire in PBG treatments and for sampling stratification. Pasture exteriors are fenced, but there are no fences around patches. See Table 1 for pasture attributes.

aged, improved pasture (IMP) dominated by Argentine Bahiagrass *(Paspalum notatum)* and 2 290 ha of seminative (SN) pastures dominated by a mixture of bahiagrass and native grasses such as purple bluestem (*Andropogon glomeratus* var. *glaucopsis*), redtop Panicum (Coleataenia longifolia), carpetgrasses (*Axonopus* spp)., and creeping bluestem *(Schizachyrium stoloniferum)* (see Fig. 1). Both pasture types contain embedded seasonal wetlands that make up about 12% of the total ranch area (Boughton et al. 2010, 2016; Gomez-Casanovas et al. 2020). The typical ranch operation at BIR uses rotational grazing on IMP pastures in the summer months (May–September) and rotational grazing on seminative pastures in the winter months (October–April). Different IMP pastures (based on soil tests) are selected to be fertilized every 2 yr with N, most likely NH_4SO_4 or NH_4NO_3 (~26 kg ha^{-1}), and IMPs were historically fertilized annually also with P until 1987 (1960s–1987, 40 kg P_2O_5 ha^{-1}), resulting in P accumulation in soils. None of the experimental pastures were fertilized during the experiment. Paudel et al. (in review) showed that IMP pastures had higher nutritive quality than SN pastures at BIR. Arthington et al. (2007) estimated that cattle removed, on average, 32% of the forage production at BIR.

Experimental design

To assess the effect of fire regime, FB and PBG, on multiple ecosystem services in two different pasture-types, we used a randomized block design with four blocks to account for known environmental differences in soil type and elevation at BIR (Table 1). The experiment was designed so that within each block, pastures were similar in elevation, soil type, and soil moisture (see Table 1; Appendix 1, Fig. S1). Within each of the four blocks, we assigned

Table 1
Attributes of 16 experimental pastures at Archbold's Buck Island Ranch. Stocking density in hectares per animal unit mo (AUM^{-1}).

Pasture	Block	Fire treatment	Pasture type	Soil type	Elevation (masl)	Average annual soil moisture ± SD (%)	Prescribed fire dates 2017-2019	Area (ha)	2017 ha AUM^{-1}	2018 ha AUM^{-1}	2019 ha AUM^{-1}
A	2	FB	IMP	Basinger fine sand, Immokalee sand	10.63	22.98 ± 11.4	31-Jan-17	16.04	0.09	0.09	0.09
C	1	FB	IMP	Immokalee sand	10.82	26.06 ± 8.5	1-Feb-17	16.1	0.09	0.08	0.09
F	4	FB	IMP	Immokalee sand, Basinger fine sand	8.93	22.72 ± 14.2	3-Feb-17	16.1	0.09	0.08	0.09
N	3	FB	IMP	Felda fine sand	7.99	27.43 ± 12.9	8-Feb-17	16.1	0.09	0.09	0.09
E	1	PBG	IMP	Valkaria fine sand, Immokalee sand, Olsdmar fine sand	10.25	23.98 ± 12.4	E1: 31-Jan-17; E2: 22-Jan-18; E3: 29-Jan-19	16.1	0.09	0.09	0.09
I	2	PBG	IMP	Basinger fine sand, Immokalee sand	10.53	25.48 ± 10.1	I3: 31-Jan-17; I1: 22-Jan-18; I2: 29-Jan-19	16	0.09	0.09	0.09
J	4	PBG	IMP	Immokalee sand	9.19	23.36 ± 13.3	J1: 3-Feb-17; J3: 22-Jan-18; J2: 29-Jan-19	16	0.09	0.09	0.09
P	3	PBG	IMP	Felda fine sand; Tequesta muck	7.99	25.43 ± 12.6	P1: 6-Feb-17; P3: 23-Jan-18; P2: 29-Jan-19	16.3	0.09	0.09	0.09
B	1	FB	SN	Malabar fine sand	9.99	23.14 ± 13.8	2-Feb-17	16	0.37	0.49	0.41
G	2	FB	SN	Placid fine sand, depressional, Oldsmar fine sand	9.49	11.44 ± 10.1	3-Feb-17	16.1	0.36	0.71	0.37
K	3	FB	SN	Pineda sand, Malabar fine sand, Felda fine sand, Pineda sand	8.23	35.13 ± 10.5	8-Feb-17	16.2	0.34	0.71	0.19
M	4	FB	SN	Felda fine sand, Hicoria mucky sand, depressional	8.04	27.34±13.1	30-Jan-17	16.1	0.40	0.96	0.19
D	1	PBG	SN	Malabar fine sand, Valkaria fine sand, Immokalee sand	10.16	21.82±10.3	D1: 2-Feb-17; D3: 31-Jan-18; D2: 11-Feb-19	16	0.36	0.56	0.41
H	2	PBG	SN	Immokalee sand	9.43	20.80±12.6	H3: 3-Feb-17; H2: 1-Feb-18; H1: 6-Feb-19	16	0.35	0.90	0.40
L	3	PBG	SN	Felda fine sand	8.25	25.06±13.1	L1: 6-Feb-17; L3: 8-Feb-18; L2/L1: 17-Feb-19	16	0.38	0.81	0.19
O	4	PBG	SN	Felda fine sand	8.12	26.71±13.7	O3: 30-Jan-17; O2: 24-Jan-18; O1: 29-Jan-19	16.1	0.40	0.84	0.20

four treatments (FB/IMP, PBG/ IMP, FB/SN, PBG/SN), resulting in a total of 16 fenced pastures within the experiment.

For fire application and sampling, each pasture was divided into thirds, or patches (~5.3 ha). All pastures had exterior fencing with no fences between patches. In 2017, we implemented eight FB treatments (4 IMP and 4 SN) with all three patches burned (see Table 1). PBG pastures were burned one-third annually (~33%), with the first third burned in 2017, the second in 2018, and the third in 2019 (see Table 1). Therefore, all patches were assigned one of four burn treatments: FB2017, PBG2017, PBG2018, and PBG2019. All prescribed fires were conducted in January and February, the typical fire season for the region (Swain et al. 2013) (see Table 1). Backfires and ploughed firebreaks around pasture borders were used to control fires. Mowed strips and water lines were used to conduct PBs within PBG treatments. Any unburned area within the patches were relit until 95% of the patch was burned. Following current management practices, cattle grazing was deferred approximately 30 d after FBs (Putnam 2008) and 1–2 wk after PBG (Fuhlendorf and Engle 2004).

Every effort was made to graze pastures equally within the same pasture type each year (see Table 1). Hurricane Irma, a category 4 storm, occurred on September 11, 2017, and all gates had to be opened for a period of 2 mo so that cattle could seek high ground. During this time we assumed cattle were not in experimental pastures. Typically, average stocking density (2014–2020) at the ranch scale for BIR are 0.18-ha animal unit months $(\text{AUM})^{-1}$ (5.7 AUM ha^{-1}) in SN pastures to 0.08 ha AUM^{-1} (13.3 AUM ha^{-1}) in IMP pastures.

On average for the 3-yr study, experimental IMP pastures were grazed at 0.08–0.09 ha AUM^{-1} (0.09 ha AUM^{-1}, 0.084 ha AUM^{-1}, 0.09 ha AUM^{-1}, in 2017, 2018, 2019, respectively; see Table 1). In SN experimental pastures, we had lower stocking density compared with the ranch scale average (0.18 ha AUM^{-1}) and stocking density in experimental pastures varied annually (but was similar across SN pastures within a year), at 0.36 ha AUM^{-1}, 0.77 ha AUM^{-1}, and 0.30 ha AUM^{-1}, in 2017, 2018, 2019 respectively (see Table 1). Grazing in SN pastures varied considerably by year in response to differences in annual precipitation and more variable available forage.

Environmental conditions

Throughout the course of this study, rainfall and temperature were measured at a central weather station on BIR (rainfall: TE525WS-L25-PT, temperature: HC2S3). Within pastures, soil moisture was measured periodically (~3–4 × /yr) at random locations with a handheld soil moisture probe (CS658 HydroSense II Water Content Sensor with 20-cm rods) (see Table 1).

Pretreatment biomass data

Before implementation of the experimental treatments, we collected pretreatment standing biomass in November 2016 in all 16 pastures. Nine random points per pasture, stratified by patch (3 per patch), were identified using ArcGIS. The Collector application was used to navigate to points in the field. A 0.25-m^2 circular plot was placed at the location, and all standing biomass was clipped after litter was removed. Biomass was dried to constant mass at 50–60°C for ~2–3 d in a drying oven, and dry weight was measured using a precision balance. These data were collected to assess if there were pretreatment differences in the amount of standing biomass at both patch and pasture scales, as well as to provide a reference condition for biomass in unburned patches for use in ANPP calculations. We found no pretreatment differences in standing biomass among pasture-scale fire treatments or blocks (Appendix 2, Table S1). At the patch scale, there was a significant interaction between patch × type, with FB2017 patches having the highest standing biomass in IMP and the lowest in SN (Appendix 1, Fig. S2, Appendix 2, Table S3). A Tukey post-hoc test showed there were no significant differences in standing biomass among any of the combinations of patch treatments and type in standing biomass before the experiment (Appendix 2, Table S4).

Forage productivity

We measured aboveground net primary productivity (ANPP, i.e., the quantity of biomass produced over the 12 calendar mo of a year; see equation later) using the moveable exclosure (ME) method (McNaughton 1985; Knapp et al. 2012) with ~clipping every 3 mo of 0.25-m^2 plots within wire cage exclosures(ME; > 0.5 m^2) that were randomly relocated after clipping within each patch to prevent cattle grazing and of paired 0.25-m^2-grazed plots (paired plots [PPs]) located next to the exclosures. For woody species, only new growth was harvested. For FB pastures, we setup three exclosures in one of the patches ($N_{FB} = 3$). For PBG pastures, we set up three exclosures in each patch ($N_{PB} = 9$) to account for each patch being burned in a different year of the study. This sampling design allowed for similar sampling intensity among patches (PBG and FB patches) and allowed us to reduce excessive sampling in FB pastures where all patches had the same fire regime and where preliminary analysis showed that standing biomass did not differ among patches within a pasture (analysis not shown). Biomass was dried to constant mass at 50°C–60°C for approximately 2–3 d in a drying oven, and dry weight was measured using a precision balance. Twelve of the 16 pastures were selected for ANPP measurements due to logistical constraints (Block 3 was excluded [see Table 1]). Sampling occurred in late 2016 before the start of the experiment and when cattle were rotated out of a pasture (≈3–4 × /yr) (Arthington et al. 2007).

On the basis of these measurements, we calculated the following:

- ANPP (Knapp et al. 2012). $\text{ANPP} = (\sum_{i=1}^{n} \text{ME(T2)} - -\text{PP (T1)}) +$ residual biomass in ME at end of growing season, where ME = exclosure and PP = grazed paired plot. T_1 = time 1 and T_2 = time 2. ANPP is an indicator of annual forage production and an indicator of the amount of forage available for cattle.
- Residual biomass. Residual biomass is the amount of standing biomass in a pasture not removed by grazing. Residual biomass was the amount of biomass measured in the pastures (PP plots) after each grazing event; thus, values represent an average of the residual biomass over time and are not representative of a cumulative annual value.
- Forage biomass accumulation rate. Forage biomass accumulation rate = $\text{ME}(T_2)\text{-PP}(T_1)$/number of days between sampling events.
- Digestible forage production. Digestible forage production = ANPP × IVOMD% (mean annual in vitro organic matter digestibility on a dry matter basis). Mean annual IVOMD per patch was obtained from forage nutritive value sampling described later.

Forage nutritive value

In 2017, forage nutritive value samples were collected ~every 3 mo (upon cattle rotation out of pastures) at the same time as productivity measurements of biomass (see earlier) were made. Starting in January 2018, we sampled plant biomass within each patch once a month (all 16 pastures). The sampling regime was switched to monthly samples to better assess seasonal dynamics. In FB pastures, we sampled biomass at four random locations within one patch using a 0.1-m^2 quadrat and these four samples were composited into one sample. In PBG pastures, we did the same as for FB, except that we sampled in each of the three patches.

Samples were dried, weighed, and ground biomass to fit through a 1-mm screen (Wiley Mill, Thomas Scientific, model 4). All samples were sent to the University of Florida Forage Evaluation Support Laboratory for analysis of the three measures of forage nutritive value: total Phosphorus (Total P [%]), total Nitrogen (Total N [%]), and in vitro IVOMD (%). The method used for total P and total N was a modification of the standard Kjeldahl procedure. Samples were digested using a modification of the aluminum block digestion procedure of Gallaher et al. (1975). Sample weight was 0.25 g, catalyst used was 1.5 g of 9:1 K_2SO_4:$CuSO_4$, and digestion was conducted for at least 4 h at 375°C using 6 mL of H_2SO_4 and 2 mL H_2O_2. P or N in the digestate was determined by semiautomated colorimetry (Hambleton 1977). The procedure used for IVOMD was the "two-stage" (Moore and Mott 1974). Samples were incubated with rumen microorganisms for 48 h followed by incubation with acid-pepsin. IVOMD results are in percentage units, the percentage of organic matter that was "digested" (https://agronomy.ifas.ufl.edu/service-labs-and-facilities/forage-evaluation-support-laboratory/). Crude protein was calculated by multiplying total N (%) by 6.25 (Adegbola T. Adesogan 2017).

Statistical analyses

We assessed average wet and dry season soil moisture in relation to the average elevation of each of the 16 experimental pastures using a linear mixed effects model with elevation as a fixed effect and block as a random effect. An analysis of variance was used to assess the impact of block on average wet season and dry season soil moisture.

To assess pretreatment differences in standing biomass, we used linear mixed effects models. For pasture scale (entire pastures [16 ha] FB vs. PBG), fixed effects were treatment (FB/PBG), pasture type (IMP/SN), their interaction, block, and block × treatment interaction, and block/pasture/patch was as a random effect. For patch scale (each of the three patches in PBG pastures and one patch in FB [5.3 ha per patch]), fixed effects were patch treatment (FB2017, PBG2017, PBG2018, and PBG2019), pasture type (IMP/SN), their interaction, block, and block × treatment. The models included a random effect of block/pasture/patch to account for the randomized block design and nested patches within pastures.

We conducted linear mixed effects models at the patch scale (each of the three patches in PBG pastures and one patch in FB [5.3 ha per patch]) and at the pasture scale (comparisons at the treatment level (entire pastures [16 ha] FB vs. PB) for each dependent variable. For patch-scale analyses, fixed factors were yr (2017, 2018, 2019); patch treatment (FB2017, PBG2017, PBG2018, and PBG2019); and pasture type (IMP or SN) and their interactions. For pasture-scale analyses, fixed factors were year, treatment (PBG or FB), and pasture type and their interactions. All models also included a main effect of block and a block × treatment interaction to assess whether fire treatments responded differently among blocks. Block was included as a fixed effect due to its known relationship to elevation gradient and as a random effect to account for any other environmental variables that may vary among blocks. Year was included as a fixed effect due to variables such as total precipitation, pattern of precipitation, and any differences in grazing rotation patterns that may have occurred among years. All models included a random effect of block/pasture/patch to account for the randomized block design and nested patches within pastures.

Forage quality models also included an additional fixed effect of time since fire in days. The time-since-fire variable was calculated in relation to the experimental fires implemented each year with the intent of understanding forage nutrient dynamics within a year, with zero representing the day when the experimental fires occurred within each year.

We considered alpha = 0.05 to indicate significance. Interactions without clear effect ($P > 0.10$) were omitted from final statistical models (Appendix 2, Tables S1−S18). Residual biomass, biomass accumulation rate, and Total P were log transformed before analysis to meet the assumption of normality. Residual plots were assessed to check assumptions of normality and homogeneity of variance; no violations were detected. All analyses were conducted in R version 4.0.2 (R Core Team 2018) and used *lme4* (Bates et al. 2015), *effects* (Fox and Weisberg 2019), and *sjPlot* (Lüdecke 2021) packages in R. In the next section, we present significant estimates and 95% confidendence intervals in brackets for all responses with full tables of analyses in Appendix 2.

Results

Environmental conditions

Total precipitation for each yr of the study was 1 361 mm, 1 023 mm, and 1 208 mm, in 2017, 2018, 2019, respectively (Fig. 2). Wet -season average soil moisture (%) varied across BIR's elevation gradient, with higher-elevation pastures having lower wet season average soil moisture (%) (estimate = −2.86, $CI_{95\%}$ [−5.28 to −0.45], $P = 0.02$, $R^2 = 0.35$). Therefore, experimental blocks, which were arrayed across BIR's elevation gradient, varied in wet season average soil moisture (block; $F_{[3,12]} =$ 6.67, $P = 0.007$) (see Table 1, Appendix 1, Fig. S1), with the lower-elevation blocks (blocks 3 and 4) having the highest wet season soil moisture. Dry season average soil moisture did not vary across the elevation gradient (−1.65 [−4.82 to 1.52], $P = 0.31$, $R^2 = 0.07$) or by experimental block (block; $F_{[3,12]} =$ 1.28, $P = 0.33$).

Aboveground net primary productivity and digestible forage production

Over the 3-yr period, ANPP was on average 5 474 ± 3 534 kg ha^{-1} yr^{-1}(mean ± standard deviation). ANPP tended to be higher in improved pastures, with on average 5 875 ± 3 250 kg ha^{-1} yr^{-1} in IMP and 5 073 ± 4 240 kg ha^{-1} yr^{-1} in SN pastures (Appendix 1, Fig. S3).

At the pasture scale (FB vs. PBG; 16 ha), ANPP did not differ between FB and PBG treatments and the only clear difference found in ANPP was among years, with the highest ANPP in 2018 (258.54 [150.13–366.95], $P < 0.001$) (Appendix 2, Table S5). ANPP did not differ among pasture types (pasture type [SN]: −89.02 [−247.78 to 69.74], $P = 0.27$), treatments, or blocks (Appendix 2, Table S5).

At the patch scale (~5.3 ha), the model of ANPP included a significant interaction of patch × year (Appendix 2, Table S6). ANPP varied by patch treatment and year (Fig. 3A). In the first year of the study, the FB2017 and PBG2017 were both treated with prescribed fire, and these areas had greater ANPP compared with unburned patches (PBG2018 [−278.89 {−521.16 to −36.63}], $P = 0.02$) and PBG2019 [−243.92 {−486.18 to −1.65}], $P = 0.05$). In the second yr (2018), the highest ANPP occurred in the PBG2017 patch, the patch that was burned the previous yr (282.75 [−16.37 to 581.87], $P = 0.06$), and in 2019, the highest ANPP occurred in the PBG2018 patch (344.56 [45.44–643.68], $P = 0.02$), the patch that was burned the previous yr.

At the pasture scale, annual digestible forage production was impacted by an interaction of fire treatment × year (Appendix 2, Table S7). Digestible forage production was greater in FB pastures compared with PBG in 2017, but in 2018 and 2019, annual digestible forage production in PBG treatments was greater than FB (2018: 113.44 [28.19–198.69], $P = 0.009$; 2019: 99.18 [13.94 –

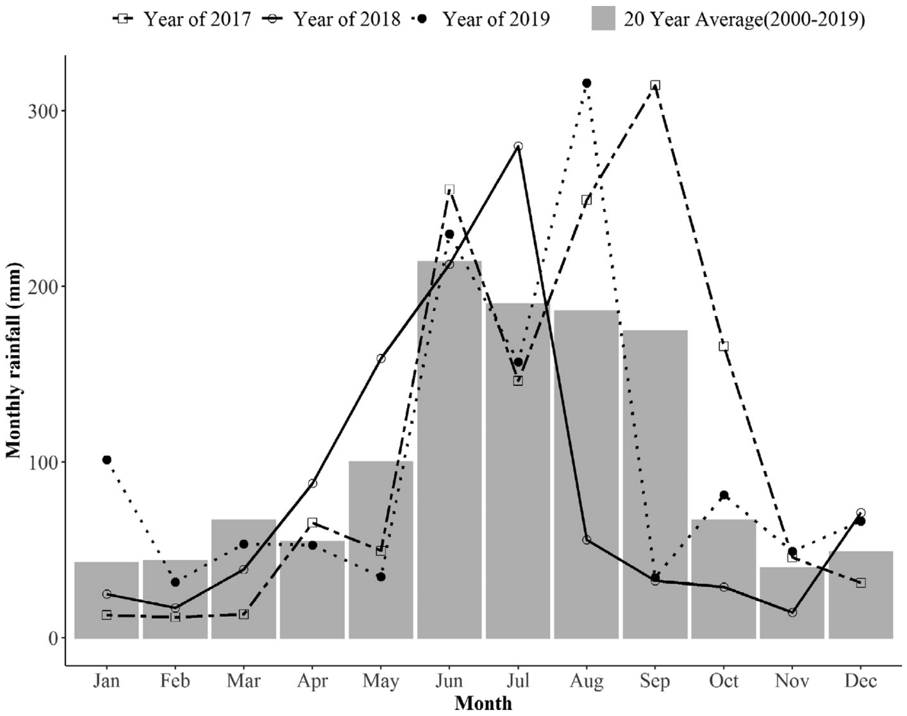

Figure 2. Rainfall conditions during the study period from January 2017 to December 2019 compared with the 20-yr average.

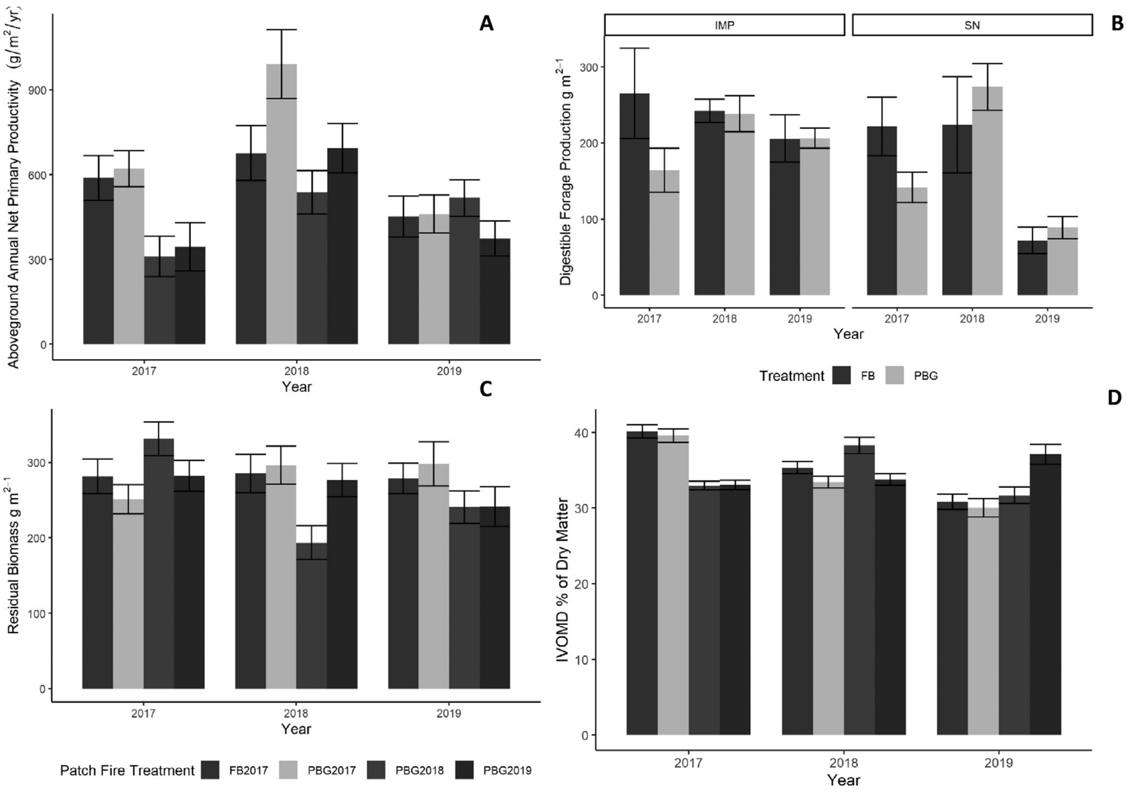

Figure 3. A, Annual aboveground net primary production (ANPP) for each patch-burn treatment and year. **B,** Annual digestible forage production per year and pasture-scale fire treatment (**C**). Log transformed residual biomass for each patch fire treatment and year. **D,** In vitro organic matter digestibility for each patch-burn treatment and year, IMP indicates improved pastures; SN, seminative pastures; FB, full burn; PBG, patch-burn grazing; FB2017, full-burn 2017; PBG2017, patch-burn 2017; PBG2018, patch-burn 2018; PBG2019, patch-burn 2019.

Table 2
Average annual digestible forage produced among pasture types and treatments. All data in $gm/m^2/yr$.

IMP digestible forage production, $g/m^2/yr$ (mean ± standard deviation)	2017	2018	2019
FB	265.08 ± 178	242.11 ± 46	205.80 ± 93
PBG2017	281.74 ± 138	313.54 ± 162	186.78 ± 85
PBG2018	65.44 ± 99	171.67 ± 74	217.74 ± 75
PBG2019	144.78 ± 135	229.82 ± 81	214.61 ± 35
PBG—overall	163.99 ± 124	238.34 ± 105	206.38 ± 65
Percent difference (PBG—FB/FB)	−0.38	−0.02	0.00
SN Digestible Forage Production, g/m^2/year (mean ± sd)			
FB	221.81 ± 116	223.92 ± 190	72.13 ± 52
PBG2017	219.58 ± 92	342.91 ± 159	93.73 ± 73
PBG2018	130.36 ± 82	242.40 ± 153	113.31 ± 78
PBG2019	75.87 ± 91	235.52 ± 161	59.81 ± 75
PBG—overall average	141.94 ± 88	273.61 ± 157	88.95 ± 75
Percent Difference (PBG—FB/FB)	−0.36	0.22	0.23

184.43], $P = 0.02$) (see Fig. 3B). There was not a significant treatment × type interaction; however, trends in the two pasture types were different. In the year of the FB, both pasture types showed greater digestible forage production in FB versus PBG. However, in the second and third yr, pasture-type responses diverged where digestible forage production was relatively similar between fire treatments in IMP while in SN, PBG treatments had greater digestible forage production than FB (Table 2).

Patch scale dynamics showed clear patch × yr interactions for annual digestible forage production (Appendix 2, Table S8). In 2018 and 2019, the PBG 2018 and PBG 2019 patches had greater digestible forage production than those same patches in 2017 (Appendix 2, Table S8).

Average residual biomass and biomass growth rate

At the pasture scale, average residual biomass was greater in IMP versus SN in 2017, while in 2018 and 2019, residual biomass was greater in SN (2018: 0.59 [0.30–0.87], $P < 0.001$; 2019: 0.48 [0.17–0.79], $P = 0.002$) (Appendix 2, Table S9). Fire treatment also interacted with yr, with PBG having lower residual biomass than FB in 2018 and 2019 (2018: −0.33 [−0.65 to −0.00], $P = 0.05$; 2019: −0.45 [−0.81 to −0.09], $P = 0.01$).

At the patch scale, there was less residual biomass in the burned patch of the yr (PBG2018 × Yr2018, −0.74 [−1.26 to −0.23], $P = 0.005$; PBG2019 × Yr2018, −0.92 [1.49 to −0.34], $P = 0.002$) and higher residual biomass in the unburned patches (Appendix 2, Table S10, see Fig. 3C).

Average biomass growth rate varied by year and was on average 10.2 ± 42 kg ha^{-1} d^{-1}. The slowest growth rate was measured in 2019 (−0.60 [−1.0 to −0.16], $P = 0.007$; Appendix 2, Table S11). A pasture-scale fire treatment × yr interaction showed that in 2 of 3 yr, growth rates were higher in FB, except in 2018, when growth rates were higher in PBG (0.54 [0.10–0.99], $P = 0.02$; Appendix 2, Table S11).

At the patch scale, patch treatment interacted with year, with greater growth rates in PBG 2018 (0.87 [0.32−1.42], $P = 0.002$) and PBG 2019 (0.55 [0.00−1.09], $P = 0.05$) in the yr 2018 compared with growth rates in those patches in 2017 (Appendix 2, Table S12).

For models of biomass variables, the main effect of block was never significant, indicating that elevation explained little variation in forage production, residual biomass, or growth rates (Appendix 2, Tables S5−12).

Forage nutritive value

Throughout the 3-yr experiment, average crude protein (% on dry matter basis) was 6.34% ± 1.86% in IMP (mean ± stdev) and 6.36% ± 2.25% in SN, average IVOMD was 36.32% ± 8.77% in IMP and 33.31% ± 9.53% in SN, and Total P was 0.13% ± 0.06% in IMP and 0.12% ± 0.08% in SN.

For crude protein, at the pasture scale, PBG treatments had significantly greater crude protein than FB treatment in 2018 (2.12 [1.24–2.99], $P < 0.001$) and 2019 (2.13 [1.17–3.09], $P < 0.001$) (Appendix 2, Table S13). Patch scale analysis showed greater crude protein in the recently burned patches of each year (Appendix 2, Table S14). There was also a three-way pasture type × patch treatment × year interaction that indicated SN pastures patches had more variable crude protein responses than IMP (Appendix 2, Table S14; Fig. S4).

A treatment × year × pasture type interaction showed that in 2017, Total P was the higher in IMP versus SN in FB treatments but higher in SN versus IMP in PBG. In 2018 and 2019, Total P was greater in IMP forage regardless of fire treatment (Appendix 2, Table S15). Patch scale analysis showed greater Total P in forage of the recently burned patches of each year (Appendix 2, Table S16). At the patch scale, for Total P, a pasture type × patch treatment × year interaction indicated that the pasture types had similar responses to fire, with elevated Total P in forage in response to recent burns, but IMP had higher Total P than SN for the majority of the combinations of pasture type, patch treatment, and year (Appendix 2, Table S16).

In general, IMP forage had greater IVOMD than SN (−2.75 [−4.31 to −1.18], $P = 0.001$). (Appendix 2, Fig. S5 and Table S17). For forage IVOMD, an interaction of treatment × year indicated that in the first year of the study, FB treatments had greater IVOMD than PBG (Appendix 2, Table S17; see Fig. 3D). However, in yr 2 and 3, the PBG treatment had greater IVOMD (2018: 5.66 [2.71−8.61], $P < 0.001$; 2019: 8.27 [5.05−11.05], $P < 0.001$). At patch scales, there was a strong effect of fire on forage IVOMD (see Fig. 3D). In the first year of the study, both the FB2017 and PBG2017 patches had high and similar levels of IVOMD (Appendix 2, Table S18). In the second yr, the PBG2018 patch that was burned had clearly higher IVOMD (11.04 [7.66−14.43], $P < 0.001$) than other patches, and in the third yr, the PBG2019 recently burned patch had the highest IVOMD (14.21 [10.54−17.89], $P < 0.001$).

At patch scales, crude protein, total P, and IVOMD were higher in recently burned patches compared with unburned patches and forage nutritive value metrics peaked at ∼120−150 d post fire. The magnitude of the response of these variables to fire varied among years (Fig. 4A−C). There was no clear effect of block on any of the forage nutritive value metrics.

Discussion

As hypothesized, PBG management created patch contrast (i.e., differences between burned and unburned patches within PBG pastures) in biomass and forage nutritive value metrics. Similar to

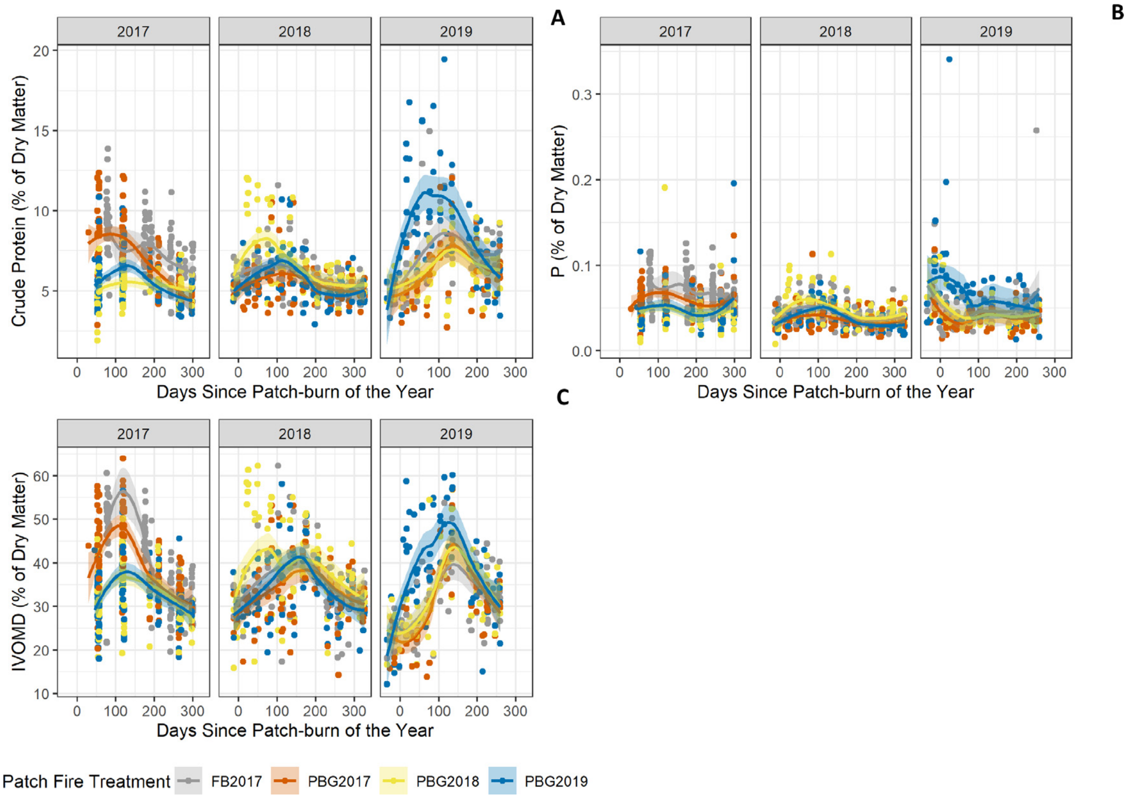

Figure 4. **A,** Crude protein. **B,** Total phosphorus (P). **C,** In vitro organic matter digestibility response to patch-burn treatment, days since the patch-burn of the yr, and yr. FB2017, full-burn 2017; PBG2017, patch-burn 2017; PBG2018, patch-burn 2018; PBG2019, patch-burn 2019.

other studies, recent fire increased all forage nutritive value metrics and reduced residual standing biomass (Duvall and Whitaker 1964; Coppedge et al. 1998; Griebel et al. 1998; Allred et al. 2011; McGranahan et al. 2013). Many studies of PBG do not evaluate ANPP responses, and our study showed variable effects of PBG on forage production and growth rate, with potential lag effects as in 2 of the 3 yr, the highest ANPP occurred in patches that were 1 yr since fire. At the pasture scale, PBG had greater crude protein and IVOMD than FB in 2 of 3 yr, but the effect was not as marked (only an increase of 0.7%–3.1%) as found in other studies (Scasta et al. 2016). When combining forage nutritive value and production metrics to assess digestible forage production, results suggested that the impact of fire management was highly variable across years and differs by pasture type. In IMP, digestible forage production responses ranged from 0% to 38% less in PBG versus FB, while in SN digestible forage production ranged from 23% more production to 36% less in PBG versus FB. This large range was explained by a larger production response to fire in the first year of the experiment in FB versus PBG in both pasture types. Contrary to our hypothesis, but in line with the literature (McGranahan et al. 2014; Spiess et al. 2020), responses of the two pasture types were relatively consistent to fire treatments at both patch and pasture scales. These results suggest that PBG may be a useful management tool for enhancing forage nutritive value and creating patch contrast in both seminative and improved subtropical humid pastures, but PBG does not necessarily increase productivity relative to FB management.

At the patch scale, we found that ANPP was increased after fire in 2017, while at the pasture scale there was no detectable difference between FB and PBG. A positive effect of fire on plant production is due to a combination of three factors: 1) release from light limitation, 2) increased soil moisture, and 3) increased N availability (Blair 1997). In subtropical humid grasslands, it is likely all three of these factors drive ANPP. In terms of light, accumulations of dead biomass in long unburned plant communities can result in depression of ANPP and fire removes dead biomass and increases ANPP (Boughton et al. 2018). A similar effect may be present in the Great Plains in the humid prairie peninsula and tallgrass prairie, where plant production is often higher on burned areas than areas not burned where high standing dead and litter accumulation may depress production (Kucera and Ehrenreich 1962; Old 1969; Bidwell and Engle 1992). Soil moisture has been shown to increase following fire as evapotranspiration is decreased from removal of aboveground biomass. This increase in soil moisture may be especially important for ANPP in the early growing season (Weekley et al. 2007). Lastly, PBG has been shown to increase soil N availability, which can increase ANPP in these subtropical humid grasslands (Boughton et al. 2018). Anderson et al. 2006 found that in a patch-burn system, the fire-grazing interaction alters soil N availability, with greater net N mineralization following recent fire and focal grazing disturbance. Increased soil N availability in burned patches was due to increased grazing pressure in the recently burned patches that was approximately 3 × that of the uniform grazing treatment (Anderson et al. 2006).

In contrast to the first year of the study, ANPP was lowest in recently burned areas in 2018 and 2019, suggesting that ANPP may have been negatively impacted by the combination of recent burning and heavier grazing in burned patches during those years, possibly due to differences in rainfall, grazing patterns, or fire intensity among years. Although many species are able to recover after grazing, negative impacts of defoliation on grasses have been shown as intensity or frequency of defoliation increases (McNaughton 1983), so despite increased N, soil moisture, and light in burned and grazed patches, high grazing pressure may have limited recovery of some species. This was not the case in 2017. However, in both 2018 and 2019, there was a trend for the previously burned patch (1 yr time-since-fire) compared with other patches to have the greatest ANPP. This may also be explained by an intermediate amount of standing dead in these patches that may ameliorate low soil moisture in the dry season and does not yet suppress production. Compared with other rangeland types such as semiarid shortgrass steppe, which showed no impact of prescribed fire on aboveground biomass production (Augustine et al. 2010), our results suggest subtropical humid grasslands are not as resistant to fire and grazing interactions, but they are resilient with potentially important lag effects of the grazing and fire interaction to plant production.

PBG created heterogeneity in standing biomass in subtropical humid grasslands. As expected, residual standing biomass was significantly lower in the burned patches of each year, creating heterogeneity in standing biomass within both pasture types under PBG. It was the combined impact of fire and grazing that created this heterogeneity consistent with our observation of higher grazing intensity in burned areas than unburned patches from cattle GPS collars (RK Boughton and Smith, unpublished). Heterogeneity in a grassland context refers to variability in vegetation stature, composition, density, and biomass and influences species diversity, wildlife habitat, and ecosystem function (Fuhlendorf and Engle 2001). Since PBG results in higher heterogeneity in subtropical humid grasslands, this may have similar positive benefits on biodiversity as those found in the Great Plains, but impacts on higher trophic levels have yet to be studied in subtropical grasslands (Fuhlendorf and Engle 2004; Fuhlendorf et al. 2006).

PBG increased forage nutritive value at both patch and pasture scales as found in other studies. At the patch scale, all forage nutritive value metrics were higher in the burned patch of each year. The annual increase in tissue quality and digestible forage production in a PBG system, as opposed to once every 3 yr in a FB system, is an important consideration for ranchers. Our results show that subtropical humid grassland, like other global rangelands, exhibits increases in forage nutritive value after fire (Scasta et al. 2016); however, crude protein responses to fire were not as great as in other grassland types. There was only a small average annual increase in crude protein between burned and unburned areas in our study at 1.7% in IMP pastures (range 0.69%–3.1% [peak May/June]) and 3.6% in SN pasture (range 2.0%–6.2% [peak May/June]). This is much lower than an average increase of 7.2%–12.8% between burned and unburned areas in shortgrass, mixed grass, and tall grass prairie (Scasta et al. 2016). This difference in the relative increase in crude protein after fire between subtropical versus temperate grasslands may be due to lower overall absolute values of crude protein % in vegetation within subtropical humid grasslands versus temperate graslands. On average, vegetation has 7.12% and 7.23% crude protein in improved and seminative subtropical humid pastures at peak season, respectively. In comparison, vegetation in temperate tallgrass prairie had ~10% crude protein at the peak of the growing season (Spiess et al. 2020). Alternatively, the muted difference between unburned and burned patches in subtropical humid grasslands may be due to the timing of the burn in relation to the growing season. Future research should test the impact of different season of burns on forage nutritive value patch contrast.

There were a few pasture-type differences likely driven by species composition, fertilization practices, and grazing pressure. Improved pastures had greater IVOMD and Total P in vegetation. Compared with *P. notatum*, which dominates IMP, native bunchgrasses that dominate SN pastures have relatively lower nutritional quality (Kalmbacher et al. 1981, 1985). IMP did not receive fertilizer during the study period and only intermittently received N fertilizer before the start of this experiment. In the past, P fertilization was a common practice but ceased in 1986 at BIR (Swain et al. 2007). Higher stocking rates in IMP pastures may have also maintained higher forage nutritive value, as studies have shown that IVOMD generally increases with increased stocking rate (Garay et al. 2004).

The creation of heterogeneity in biomass with PBG that we observed may result in cascading benefits to other ecosystem functions and services in subtropical humid grasslands, and our work is under way to examine cattle behavior, greenhouse gas regulation, water use efficiency, and plant diversity responses to PBG.

Management Implications

Our study resulted in three main conclusions: 1) PBG management employed on improved and seminative subtropical humid pastures created heterogeneity through patch contrast in forage nutritive value (total P, crude protein, and IVOMD); ANPP; digestible forage production, and residual biomass; 2) PBG benefited forage nutritive value in both IMP and SN; and 3) PBG benefited digestible forage production in seminative pastures.

Even though the pasture types responded relatively consistently and we did not detect a clear interaction of pasture type × fire treatments, it is important for a ranchers making management decisions to break down results by pasture type. For the rancher, the cumulative impact of PBG and FB is important. What does this mean from an economic standpoint? Using a simple economic model (Bankovich et al. 2016; Ferrell et al. 2006) that combines productivity and forage digestibility and assuming that digestible forage production was directly related to the ability of a unit area to produced beef and using minimum, median, and maximum calf prices from 2012 to 2020, we estimated the economic impact of using PBG management in IMP and SN (Appendix 3). In IMP, digestible forage production responses ranged from 0% to 38% less in PBG versus FB, while in SN digestible forage production ranged from 23% more production to 36% less in PBG versus FB (see Table 2). On the basis of this difference in forage production, modeled calf production showed a $246–$563 (median $312) loss per ha in IMP pastures when using a 3-yr PBG cycle compared with an FB once every 3 yr. In SN pastures, modeled calf production showed a gain of $19 to $43 (median $24) per ha when using a 3-yr patch-burn grazing cycle compared with an FB once every 3 yr.

The median projected gain for SN pastures was $24/ha and, while modest, is not insignificant when considering the area of seminative pastures. For example, at BIR, with 2 290 ha of SN pastures, this would result in ~$55 000 gain over the 3-yr PBG cycle. In the five-county region surrounding BIR, the economic gain of using PBG in seminative/native range (23 000 ha) is estimated at ~$552 000 over a 3-yr cycle if similar stocking rates to BIR are used. In SN pastures, the real value for implementing PBG is that it is both economically viable and has ecological benefits. Additional economic benefit could come from the integration of PBG into cost-share programs such as the Natural Resources Conservation Service Environmental Quality Incentives Program and Conservation Stewardship Program, which would offset the cost of management for ranchers.

Our economic analysis did not include actual calf production, supplemental feed or fertilizer inputs, and labor costs because of scale and experimental design. In subtropical humid grasslands of Florida, labor costs of PBG and FB may not be very different because PBG could entail organization of burn units using natural fire breaks such as ditches and roads rather than increased plowed fire breaks. One additional benefit of PBG is that pastures always contain forage in unburned areas, potentially reducing risk or increasing resiliency in relation to extreme annual events such as droughts or freezes. Further investigation of these varied socioeconomic impacts could allow us to assess whether PBG allows ranchers access to the best of both worlds, the opportunity for enhanced forage nutritive value in each of their pastures and also a grass bank in case of drought or freeze (Allred et al. 2014; McGranahan et al. 2014; Spiess et al. 2020).

This study is contributing toward the LTAR continental scale cross-site analysis to compare aspirational and business-as-usual management. Aspirational management systems, as defined by the USDA LTAR network, should increase or maintain agricultural production while maintaining or improving environmental quality (Spiegal et al. 2018). Our results suggest that PBG is an aspirational management regime that is both economically and environmentally viable in SN pastures, but while some benefits of PBG occur in IMP, there may be an economic risk to using PBG in IMP. At Archbold-UF LTAR, our next iteration of aspirational management in IMP is under consideration and will be defined with stakeholder input.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

We thank Laurent Lollis and Gene Lollis for implementing prescribed fires and managing cattle for this project. Steffan Pierre provided Global Information System and logistical support. Numerous BIR scientists (Gregory Sonnier, Amartya Saha, Nic McMillon, Hansol Lee, Mandy West, Shefali Azad) and interns assisted with data collection, management, and field work (interns: Ruth Whittington, Sarah Garvey, Erik Iverson, Matthias Gaffney, Jaide Allenbrande, Phoebe Judge, Kate Bazany, Avarna Jain, Sophie Fruchter). We thank two anonymous reviewers for comments that improved the manuscript. This study was funded by NIFA (2016-67019-24988) and USDA cooperative agreement to Archbold (58-0202-7-001). This research is a contribution of the Long-term Agroecosystem Research Network. The LTAR is supported by the United States Department of Agriculture.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rama.2022.05.004.

References

Adegbola T., and Adesogan, J. V. 2017. Defining forage quality. Available at: https://edis.ifas.ufl.edu/ag332. Accessed 13 April 2021.

Allred, B.W., Fuhlendorf, S.D., Engle, D.M., Elmore, R.D., 2011. Ungulate preference for burned patches reveals strength of fire–grazing interaction. Ecology and Evolution 1, 32–144.

Allred, B.W., Scasta, J.D., Hovick, T.J., Fuhlendorf, S.D., Hamilton, R.G., 2014. Spatial heterogeneity stabilizes livestock productivity in a changing climate. Agriculture. Ecosystems & Environment 193, 37–41.

Anderson, R.H., Fuhlendorf, S.D., Engle, D.M., 2006. Soil nitrogen availability in tallgrass prairie under the fire–grazing interaction. Rangeland Ecology & Management 59, 625–631.

Archimède, H., Eugène, M., Marie Magdeleine, C., Boval, M., Martin, C., Morgavi, D.P., Lecomte, P., Doreau, M., 2011. Comparison of methane production between C3 and C4 grasses and legumes. Animal Feed Science and Technology, Special Issue: Greenhouse Gases in Animal Agriculture—Finding a Balance Between Food and Emissions 166–167, 59–64.

Arthington, J.D., Roka, F.M., Mullahey, J.J., Coleman, S.W., Muchovej, R.M., Lollis, L.O., Hitchcock, D., 2007. Integrating ranch forage production, cattle performance, and economics in ranch management systems for southern Florida. Rangeland Ecology & Management 60, 12–18.

Augustine, D.J., Derner, J.D., 2014. Controls over the strength and timing of fire—grazer interactions in a semi-arid rangeland. Journal of Applied Ecology 51, 242–250.

Augustine, D.J., Derner, J.D., Milchunas, D.G., 2010. Prescribed fire, grazing, and herbaceous plant production in shortgrass steppe. Rangeland Ecology & Management 63, 317–323.

Bankovich, B., Boughton, E., Boughton, R., Avery, M.L., Wisely, S.M., 2016. Plant community shifts caused by feral swine rooting devalue Florida rangeland. Agriculture, Ecosystems & Environment 220, 45–54.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software 67 (1), 1–48. doi:10.18637/jss.v067.i01.

Bengtsson, J., Bullock, J.M., Egoh, B., Everson, C., Everson, T., O'Connor, T., O'Farrell, P.J., Smith, H.G., Lindborg, R., 2019. Grasslands—more important for ecosystem services than you might think. Ecosphere 10, e02582.

Bidwell, T.G., Engle, D.M., 1992. Relationship of fire behavior to tallgrass prairie herbage production. Rangeland Ecology & Management/Journal of Range Management Archives 45, 579–584.

Blair, J.M., 1997. Fire, N availability, and plant response in grasslands: a test of the transient maxima hypothesis. Ecology 78 (8), 2359–2368.

Bond, W.J., 2016. Ancient grasslands at risk. Science 351, 120–122.

Boughton, E.H., Bohlen, P.J., Maki, J.H., 2018. Effects of experimental season of prescribed fire and nutrient addition on structure and function of previously grazed grassland. Journal of Plant Ecology 11, 576–584.

Boughton, E.H., Bohlen, P.J., Orzell, S.L., Bridges, E.L., Noss, R.F., 2018. In: Conservation and management of subtropical grasslands in south-central Florida. University of Alabama Press, Tuscaloosa, AL, USA.

Boughton, E.H., Quintana-Ascencio, P.F., Bohlen, P.J., Fauth, J.E., Jenkins, D.G., 2016. Interactive effects of pasture management intensity, release from grazing and prescribed fire on forty subtropical wetland plant assemblages. Journal of Applied Ecology 53, 159–170.

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. Available at. doi:10.1111/j.1600-0587.2009.06010.x.

Coppedge, B.R., Engle, D.M., Toepfer, C.S., Shaw, J.H., 1998. Effects of seasonal fire, bison grazing and climatic variation on tallgrass prairie vegetation. Plant Ecology 139, 235–246.

Cowley, R.A., Hearnden, M.N., Joyce, K.E., Tovar-Valencia, M., Cowley, T.M., Pettit, C.L., Dyer, R.M., Cowley, R.A., Hearnden, M.N., Joyce, K.E., Tovar-Valencia, M., Cowley, T.M., Pettit, C.L., Dyer, R.M., 2014. How hot? How often? Getting the fire frequency and timing right for optimal management of woody cover and pasture composition in northern Australian grazed tropical savannas. Kidman Springs Fire Experiment 1993–2013. Rangeland Journal 36, 323–345.

Dixon, A.P., Faber-Langendoen, D., Josse, C., Morrison, J., Loucks, C.J., 2014. Distribution mapping of world grassland types. Journal of Biogeography 41, 2003–2019.

Duvall, V.L., Whitaker, L.B., 1964. Rotation burning: a forage management system for longleaf pine-bluestem ranges. Rangeland Ecology & Management/Journal of Range Management Archives 17, 322–326.

Ferrell, J.A., Mullahey, J.J., Dusky, J.A., Roka, F.M., 2006. Competition of giant smutgrass (Sporobolus indicus) in a bahiagrass pasture. Weed Science 54, 100–105.

Fox, J., … Weisberg, S., 2019. An R companion to applied regression, 3rd ed. Sage, Thousand Oaks, CA, USA Available at.

Fuhlendorf, S.D., Engle, D.M., 2004. Application of the fire–grazing interaction to restore a shifting mosaic on tallgrass prairie. Journal of Applied Ecology 41, 604–614.

Fuhlendorf, S.D., Engle, D.M., 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. We propose a paradigm that enhances heterogeneity instead of homogeneity to promote biological diversity and wildlife habitat on rangelands grazed by livestock. BioScience 51, 625–632.

Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A., Leslie Jr, D.M., 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological Applications 16, 1706–1716.

Fulgoni, J.N., Whiles, M.R., Dodds, W.K., Larson, D.M., Jackson, K.E., Grudzinski, B.P., 2020. Responses and resilience of tallgrass prairie streams to patch-burn grazing. Journal of Applied Ecology 57, 1303–1313.

Gallaher, R.N., Weldon, C.O., Futral, J.G., 1975. An aluminum block digester for plant and soil analysis 1. Soil Science Society of America Journal 39, 803–806.

Garay, A.H., Sollenberger, L.E., McDonald, D.C., Ruegsegger, G.J., Kalmbacher, R.S., Mislevy, P., 2004. Nitrogen fertilization and stocking rate affect stargrass pasture and cattle performance. Crop Science 44, 1348–1354.

Gibson, D.J., 2009. Grasses and grassland ecology. Oxford University Press, Oxford, England.

Gomez-Casanovas, N., DeLucia, N.J., DeLucia, E.H., Blanc-Betes, E., Boughton, E.H., Sparks, J., Bernacchi, C.J., 2020. Seasonal controls of CO2 and CH4 dynamics in a temporarily flooded subtropical wetland. Journal of Geophysical Research: Biogeosciences 125 e2019JG005257.

Griebel, R.L., Winter, S.L., Steuter, A.A., 1998. Grassland birds and habitat structure in sandhills prairie managed using cattle or bison plus fire. Great Plains Research 255–268.

Hambleton, L.G., 1977. Semiautomated method for simultaneous determination of phosphorus, calcium, and crude protein in animal feeds. Journal of the Association of Official Analytical Chemists 60, 845–852.

Kalmbacher, R.S., Martin, F.G., Andrade, J.M.S., 1981. Yield and quality of creeping bluestem as affected by time of cutting. Rangeland Ecology & Management/Journal of Range Management Archives 34, 471–474.

Kalmbacher, R.S., Martin, F.G., Terry, W.S., Hunter, D.H., White, L.D., 1985. Effects of clipping on burned and unburned creeping bluestem. Journal of Range Management 531–535.

Kleinman, P.J.A., Spiegal, S., Rigby, J.R., Goslee, S.C., Baker, J.M., Bestelmeyer, B.T., Boughton, R.K., Bryant, R.B., Cavigelli, M.A., Derner, J.D., Duncan, E.W., Goodrich, D.C., Huggins, D.R., King, K.W., Liebig, M.A., Locke, M.A., Mirsky, S.B., Moglen, G.E., Moorman, T.B., Pierson, F.B., Robertson, G.P., Sadler, E.J., Shortle, J.S., Steiner, J.L., Strickland, T.C., Swain, H.M., Tsegaye, T., Williams, M.R., Walthall, C.L., 2018. Advancing the sustainability of US agriculture through long-term research. Journal of Environmental Quality 47, 1412–1425.

Knapp, A.K., Hoover, D.L., Blair, J.M., Buis, G., Burkepile, D.E., Chamberlain, A., Collins, S.L., Fynn, R.W.S., Kirkman, K.P., Smith, M.D., 2012. A test of two mechanisms proposed to optimize grassland aboveground primary productivity in response to grazing. Journal of Plant Ecology 5, 357–365.

Kucera, C.L., Ehrenreich, J.H., 1962. Some effects on annual burning on central Missouri prairie. Ecology 43, 334–336.

Liebman, M., Helmers, M.J., Schulte, L.A., Chase, C.A., 2013. Using biodiversity to link agricultural productivity with environmental quality: results from three field experiments in Iowa. Renewable Agriculture and Food Systems 28, 115–128.

Limb, R.F., Fuhlendorf, S.D., Engle, D.M., Weir, J.R., Elmore, R.D., Bidwell, T.G., 2011. Pyric–herbivory and cattle performance in grassland ecosystems. Rangeland Ecology & Management 64, 659–663.

Lüdecke, D., 2021. sjPlot: data visualization for statistics in social science. R package version 2.8.10. Available at https://CRAN.R-project.org/package=sjPlot .

McGranahan, D.A., Engle, D.M., Fuhlendorf, S.D., Winter, S.J., Miller, J.R., Debinski, D.M., 2012. Spatial heterogeneity across five rangelands managed with pyric-herbivory. Journal of Applied Ecology 49, 903–910.

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. Southeastern Naturalist 13, 200–207.

McGranahan, D.A., Hovick, T.J., Dwayne Elmore, R., Engle, D.M., Fuhlendorf, S.D., Winter, S.L., Miller, J.R., Debinski, D.M., 2016. Temporal variability in aboveground plant biomass decreases as spatial variability increases. Ecology 97, 555–560.

McGranahan, D.A., Raicovich, G.M., Wilson, W.N., Smith, C.K., 2013. Preliminary evidence that patch burn-grazing creates spatially heterogeneous habitat structure in old-field grassland. Sena 12, 655–660.

McNaughton, S.J., 1985. Ecology of a grazing ecosystem: the Serengeti. Ecological Monographs 55, 259–294.

McNaughton, S.J., 1983. Compensatory plant growth as a response to herbivory. Oikos 40, 329–336.

Moore, J.E., Mott, G.O., 1974. Recovery of residual organic matter from in vitro digestion of forages. Journal of Dairy Science 57, 1258–1259.

Noss, R., 2013. Forgotten grasslands of the South: natural history and conservation. Island Press, Washington, D.C..

Old, S.M., 1969. Microclimate, fire, and plant production in an Illinois prairie. Ecological Monographs 39, 355–384.

Overbeck, G.E., Müller, S.C., Pillar, V.D., Pfadenhauer, J., 2005. Fine-scale post-fire dynamics in southern Brazilian subtropical grassland. Journal of Vegetation Science 16, 655–664.

Parr, C.L., Lehmann, C.E., Bond, W.J., Hoffmann, W.A., Andersen, A.N., 2014. Tropical grassy biomes: misunderstood, neglected, and under threat. Trends in Ecology & Evolution 29, 205–213.

Peters, M., Herrero, M., Fisher, M., Erb, K.-H., Rao, I., Subbarao, G.V., Castro, A., Arango, J., Chará, J., Murgueitio, E., Hoek, R.van der, Läderach, P., Hyman, G., Tapasco, J., Strassburg, B., Paul, B., Rincón, A., Schultze-Kraft, R., Fonte, S., Searchinger, T., 2013. Challenges and opportunities for improving eco-efficiency of tropical forage-based systems to mitigate greenhouse gas emissions. Tropical Grasslands-Forrajes Tropicales 1, 156–167.

Putnam, A., 2008. Water quality best management practices for Florida cow-calf operations, DACS-P-01280. Florida Department of Agriculture and Consumer Services.

Satter, L.D., Klopfenstein, T.J., Erickson, G.E., … Powell, J.M., 2005. Phosphorus and dairy/beef nutrition. Faculty Papers and Publications in Animal Science 549. Available at https://digitalcommons.unl.edu/animalscifacpub/549 .

Scasta, J.D., Thacker, E.T., Hovick, T.J., Engle, D.M., Allred, B.W., Fuhlendorf, S.D., Weir, J.R., 2016. Patch-burn grazing (PBG) as a livestock management alternative for fire-prone ecosystems of North America. Renewable Agriculture and Food Systems 31, 550–567.

Sigua, G.C., 2010. Sustainable cow-calf operations and water quality: a review. Agronomy for Sustainable Development 30, 631–648.

Spiegal, S., Bestelmeyer, B.T., Archer, D.W., Augustine, D.J., Boughton, E.H., Boughton, R.K., Cavigelli, M.A., Clark, P.E., Derner, J.D., Duncan, E.W., Hapeman, C.J., Harmel, R.D., Heilman, P., Holly, M.A., Huggins, D.R., King, K., Kleinman, P.J.A., Liebig, M.A., Locke, M.A., McCarty, G.W., Millar, N., Mirsky, S.B., Moorman, T.B., Pierson, F.B., Rigby, J.R., Robertson, G.P., Steiner, J.L., Strickland, T.C., Swain, H.M., Wienhold, B.J., Wulfhorst, J.D., Yost, M.A., Walthall, C.L., 2018. Evaluating strategies for sustainable intensification of US agriculture through the Long-Term Agroecosystem Research network. Environmental Research Letters 13, 034031.

Spiess, J.W., McGranahan, D.A., Geaumont, B., Sedivec, K., Lakey, M., Berti, M., Hovick, T.J., Limb, R.F., 2020. Patch-burning buffers forage resources and livestock performance to mitigate drought in the northern Great Plains. Rangeland Ecology & Management 73, 473–481.

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. Rangeland Ecology & Management 60, 1–11.

Swain, H.M., Boughton, E.H., Bohlen, P.J., Lollis, L.O., 2013. Trade-offs among ecosystem services and disservices on a Florida ranch. Rangelands 35, 75–87.

Thornton, P.K., 2010. Livestock production: recent trends, future prospects. Philosophical Transactions of the Royal Society B: Biological Sciences 365, 2853–2867.

Tothill, J.C., 1971. A review of fire in the management of native pasture with particular reference to north-eastern Australia. Tropical Grasslands 5, 1.

Veldman, J.W., 2016. Clarifying the confusion: old-growth savannahs and tropical ecosystem degradation. Philosophical Transactions of the Royal Society B: Biological Sciences 371, 20150306.

Veldman, J.W., Buisson, E., Durigan, G., Fernandes, G.W., Le Stradic, S., Mahy, G., Negreiros, D., Overbeck, G.E., Veldman, R.G., Zaloumis, N.P., 2015. Toward an old-growth concept for grasslands, savannas, and woodlands. Frontiers in Ecology and the Environment 13, 154–162.

Vermeire, L.T., Mitchell, R.B., Fuhlendore, S.D., Gillen, R.L., 2004. Patch burning effects on grazing distribution. Journal of Range Management 57, 248–252.

Wang, T., Olson, K., Harty, A., Johnson, P., Chowdhury, I.U.A., 2020. Cattle producers' willingness to adopt winter patch grazing vs. patch burn grazing in the Northern Great Plains.

Weekley, C.W., Gagnon, D., Menges, E.S., Quintana-Ascencio, P.F., Saha, S., 2007. Variation in soil moisture in relation to rainfall, vegetation, gaps, and time-since-fire in Florida scrub. Ecoscience 14, 377–386.

White, R.P., Murray, S., Rohweder, M., Prince, S.D., Thompson, K.M., 2000. Grassland ecosystems. World Resources Institute, Washington, DC, USA.