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Patch-Burn Grazing Effects on the Ecological Integrity of Tallgrass Prairie Streams

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Abstract

Conversion to agriculture, habitat fragmentation, and the loss of native grazers have made tallgrass prairie one of the most endangered ecosystems. One management option for the remaining prairie parcels, patch-burn grazing (PBG), applies a controlled burn to a portion of the prairie to attract cattle, creating a mosaic of more- and less-grazed patches. Although beneficial to cattle and grassland birds, the potential impacts of PBG on streams have not been studied, and a holistic approach is needed to ensure against adverse effects. We used a Before-After-Control-Impact design to assess potential impacts of PBG with and without riparian protection on tallgrass prairie headwater streams. We sampled stream macroinvertebrates and benthic organic matter 2 yr before and 2 yr during PBG treatments on two grazed watersheds with riparian fencing (fenced), two unfenced grazed watersheds (unfenced), and two ungrazed (control) watersheds. Very fine benthic organic matter increased significantly (51%) in unfenced streams compared with controls ($p < 0.007$), and fine particulate organic matter (<1 mm and >250 µm) increased 3-fold in the unfenced streams compared with controls ($p = 0.008$). The contribution of fine inorganic sediments to total substrata increased 28% in unfenced streams during PBG, which was significantly different from controls ($p = 0.03$). Additionally, the abundance of Ephemeroptera, Plecoptera, and Trichoptera taxa decreased from 7635 to 687 individuals m^{-2} in unfenced streams, which was significantly lower than in control streams ($p = 0.008$). Our results indicate that PBG adversely influences prairie streams through sediment inputs and reductions in sensitive invertebrate taxa, but riparian fencing can alleviate these impacts.

Tallgrass prairie is one of the most critically endangered habitats in the world (Noss et al., 1995). Before European settlement, North American prairies covered ~162 million ha of land, but currently prairies occupy <5% of their historic distribution (Samson et al., 2004). Tallgrass prairies support diverse assemblages of plants and animals and provide habitat for some of the rarest species in the midwestern United States (Chapman et al., 1990; Whiles and Charlton, 2006). Because more than 99% of the original tallgrass prairie has been eliminated, tallgrass prairie streams have become endangered and degraded as well. Tallgrass prairie began to decline as European Americans began to cultivate prairie soils, convert the land to pastures of non-native grasses, and suppress natural fires that maintain these systems as grasslands (Knapp et al., 1998; Smith, 2001). Tallgrass prairie streams are particularly imperiled because the remaining prairie fragments are generally too small and degraded to constitute complete functional watersheds (Dodds et al., 2004). These systems experience periodic hydrologic disturbance, resulting in a distinct biota. Specifically, tallgrass prairie streams are subject to drought and flood, and the organisms that inhabit them have adapted to these disturbances (Resh et al., 1988).

Efforts to preserve tallgrass prairies and their streams are typically focused on protecting the few remaining intact systems and establishing proper methods of management (Axelrod, 1985). Patch-burn grazing (PBG), a combination of prescribed burns and grazing, involves burning a different portion of the watershed each spring for consecutive years until the entire watershed has gone through one rotation of burning. Cattle are allowed to graze freely and usually move to the most recently burned area because burned patches are more desirable for foraging (Allred et al., 2011). This disturbance and rest cycle prevents overgrazing and decreases bare ground in the unburned areas, particularly during periods of drought (Teague et al.,

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Abbreviations: AFDM, ash-free dry mass; BACI, Before-After-Control-Impact; CPOM, coarse particulate organic matter; EPT, Ephemeroptera, Plecoptera, and Trichoptera; FPOM, fine particulate organic matter; HBI, Hilsenhoff Biotic Index; NMDS, nonmetric multidimensional scaling; PBG, Patch-burn grazing; VFPOM, very fine particulate organic matter.

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2004). Patch-burn grazing may also help to prevent deleterious effects to riparian zones, which have been so heavily affected by cattle in some cases that they have been described as "sacrifice zones" (Stoddard and Smith, 1955).

Although the effects of fire, independent of grazing, on tallgrass prairie streams have been examined by others (e.g., Larson et al., 2013), few studies have examined impacts of PBG, and there is concern that grazing by cattle may adversely affect tallgrass prairie streams. Various biotic and abiotic factors may be affected by grazing. For example, cattle may influence nutrient levels within stream systems, leading to increased primary production and algal biomass (Schepers et al., 1982; Scrimgeour and Kendall, 2003). Altered channel morphology, bank erosion, and slumping can occur (McInnis and McIver, 2001; Tufekcioglu et al., 2012). Additionally, damage to riparian habitats can increase sediment runoff and fine sediments in the channel (Bengeyfield, 2007). Sedimentation is a leading water quality issue in the United States, and it may be exacerbated with increased compaction of soil by cattle (Simon and Darby, 1999; Connolly and Pearson, 2007; USEPA, 2009). Sedimentation changes the size of substrata, is a primary factor influencing the abundance and distribution of aquatic insects, and can be one of the most significant stressors to aquatic life (Minshall, 1984; Richards and Bacon, 1994; Braccia and Voshell, 2006). Furthermore, dissolved oxygen availability in the substrata and hyporheic zone are reduced because siltation prevents oxygen exchange with the water column (Richards and Bacon, 1994; Kemp et al., 2011). Increases in fine sediments also reduce habitat heterogeneity and refugia from natural disturbances, such as the floods and droughts that characterize prairie streams (Dodds et al., 2004).

The impacts of cattle on the physical template of streams can affect stream communities. For example, excessive sedimentation from cattle activities may hinder filter-feeding macroinvertebrates by clogging their filtering structures, thus reducing their populations (Lemly, 1982; Arruda et al., 1983). Scraping macroinvertebrates may also be affected by sedimentation due to lower quality or complete burial of periphyton (Graham, 1990). Conversely, increased nutrient inputs from cattle could enhance algal resources for scraping macroinvertebrates if light is not limited (Braccia et al., 2014). The abundance of sensitive taxa, such as the Ephemeroptera, Plecoptera, and Trichoptera (EPT), is often positively related to substrate heterogeneity, which is reduced as fine sediments increase (Richards and Host, 1993; Muenz et al., 2006; Burdon et al., 2013). For example, Suren (2005) found that Leptophlebiidae mayfly abundance decreased with increasing sediment deposition on cobble substrata. Ultimately, sedimentation may result in lower taxa diversity and richness and a shift in community composition to more tolerant taxa (Relyea et al., 2000; McIver and McInnis, 2007; Larsen et al., 2011).

Previous studies examining terrestrial responses to PBG have shown positive effects. Patch-burn grazing can benefit the nesting success of species of concern, such as the dickcissel (*Spiza americana*), and can increase grassland bird diversity and richness (Fuhlendorf et al., 2006; Churchwell et al., 2008; Coppedge et al., 2008). Furthermore, livestock in PBG systems have similar weight gains as traditional grazing regimes, while vegetation heterogeneity increases (Fuhlendorf and Engle, 2004; Smart, 2010; Limb et al., 2011). Patch-burn grazing also prevents the growth of trees and shrubs and allows vegetation to accumulate to fuel subsequent years of prescribed burns (NRCS Missouri, 2004). In addition to PBG use in North America, fire, grazing, and various combinations of the two are used worldwide to manage many remaining grasslands (Joy, 1992; Morris and Tainton, 1996)

Although PBG appears effective for managing prairies and some key prairie species, it is unknown the extent to which tallgrass prairie streams may be adversely affected. Fencing could potentially reduce the negative impacts; however, to date, results from livestock exclusion studies have been varied and conflicting (Sarr, 2002). For example, Herbst et al. (2012) reported that small-scale grazing exclosures did not influence the macroinvertebrate community. However, the exclosures in that study did not completely encompass the study reach, allowing for potential upstream impacts from cattle. Riparian fencing can be an appropriate method to prevent cattle from affecting riparian and in-stream habitats (MacLeod and McIvor, 2008; Ranganath et al., 2009; Ash et al., 2011). Excluding cattle from riparian areas may alleviate potential adverse effects from grazing, such as sedimentation, reduced macroinvertebrate diversity, and overall decreased biotic integrity (e.g., reduced EPT presence and increases in taxa with higher pollution tolerance). Additionally, intact riparian vegetation can reduce inputs of sediment and nutrients that adversely affect stream biota and can provide subsidies in the form of allochthonous input and terrestrial invertebrates (e.g., Flory and Milner, 1999; Baxter et al., 2005; Weigel et al., 2011).

Our overall objective was to examine the potential ecological impacts of PBG on tallgrass prairie streams. We quantified stream macroinvertebrate abundance, biomass, and diversity along with benthic organic matter standing stocks in streams draining watersheds subjected to PBG treatments with and without riparian fencing. We hypothesized that study streams without riparian protection would decrease in biotic integrity during PBG. Specifically, we hypothesized that PBG would lead to increased sedimentation and reduce abundance and biomass of sensitive EPT taxa, filter-feeding macroinvertebrates, and overall macroinvertebrate diversity but that riparian fencing would alleviate these effects to some degree.

Materials and Methods

Study Site

The Osage Prairie Natural Area is in the Osage River Basin 8 miles south of Nevada, Missouri (37°75′ N, 94°32′ W) (Fig. 1). Osage is managed by the Missouri Department of Conservation (MDC), Kansas City Region. Nevada receives an annual average rainfall of 49.7 mm. Average annual rainfall was 55.5 mm during the pretreatment years and 44.4 mm during the treatment years. The area encompasses 249 ha and is drained by Landon Branch, a 5-km intermittent stream with several tributaries. The land directly surrounding Osage Prairie is predominantly forest (64%) and cropland (32%). Osage Prairie was historically harvested for hay and grazed from the early 1900s until 1987, but this was deemed an improper method for prairie management and was discontinued in 2009 (Missouri Department of Conservation, 2011). Riparian trees with diameters >10 cm have

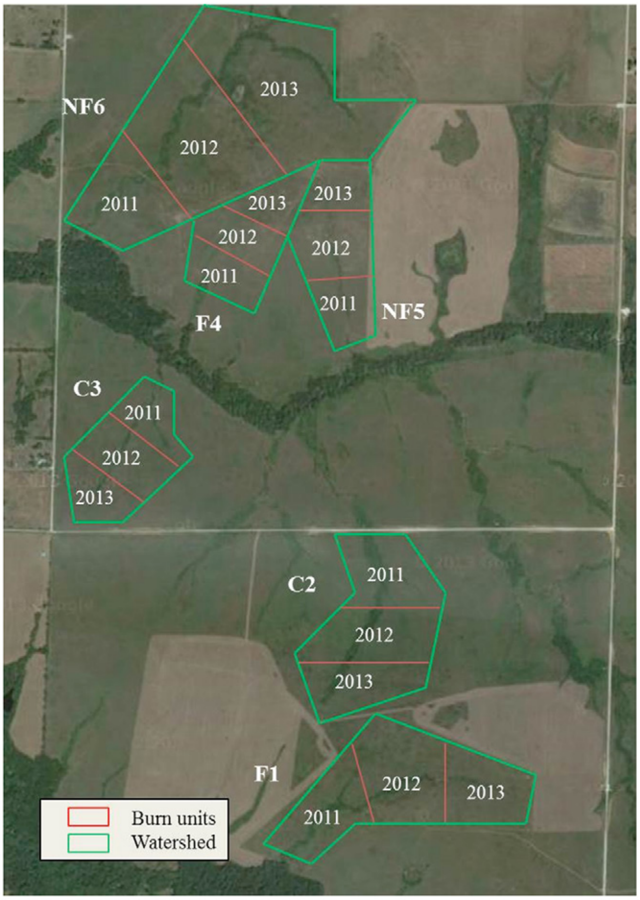

Fig. 1. Burn units for Osage Prairie. Prescribed burns began in May 2011 and concluded in May 2013. C2 and C3, control watersheds; F1 and F4, fenced watersheds; NF5 and NF6, unfenced watersheds.

been periodically removed to prevent establishment of woody vegetation. Osage Prairie is occasionally managed by controlled burns to suppress establishment of woody vegetation and to maintain grass and forb diversity.

Six subwatersheds, each drained by a first-order stream, were used in this study (Fig. 1). As a result of varied channel morphology, stream channel widths ranged from 1.0 to 2.2 m and annual average temperatures ranged from 5 to 14.3°C during the study period. Riparian vegetation was predominantly forbs and grasses. All streams had similar proportions of erosional and depositional habitats, with an average of 75% depositional and 25% erosional across sites (Table 1).

Two years of pretreatment sampling on all watersheds preceded experimental treatments. Treatments were randomly assigned and initiated in May 2011. All watersheds were burned, and cattle were added to four of the six watersheds to examine the effects of grazing with (two watersheds) and without riparian protection (two watersheds); the remaining two watersheds were used as controls with no cattle. Study streams were identified according to the treatment applied to the watershed: C2 and C3 (ungrazed control), F1 and F4 (grazed with riparian exclusion fencing), and NF5 and NF6 (grazed without riparian fencing). Cattle density averaged 1 animal unit (au) (227–363 kg) per 2.4 ha in each treatment watershed. Watershed areas ranged 8.1 to 47.3 ha. Stocking rate was comparable to those of other managed rangelands (Towne et al., 2005). Cattle used in the experiment were yearlings averaging 295 kg each. Fencing was used to create a 15-m buffer on either side of the stream channel for a total of 30 m of cattle exclusion based on Fischer et al. (2000).

Each study watershed encompassed the stream up to its source and was enclosed within Osage Prairie, thus preventing upstream impacts beyond the stream reach examined. Prescribed burns were identical for all watersheds. Watersheds were divided into thirds, and spring burning occurred in a 3-yr rotation beginning May 2011 at the downstream portion of the study reach (Fig. 1). A small amount of private land adjacent to NF6 within the watershed was grazed throughout the treatment sampling periods similar to the pretreatment grazing regimes. Watersheds were grazed from May to July 2011 (82 d) and from April to July 2012 (88 d).

Benthic Organic Matter and Macroinvertebrate Sampling

Pretreatment and treatment samples were collected monthly when water was present from March 2009 to May 2012. All study streams dried by June of each year and began to flow again in the fall between October and early December. During each monthly visit (pre-PBG, n = 9; PBG, n = 6), six samples were collected from a 100-m reach within each watershed. Samples were taken at the downstream location of each stream on each sample date: three samples from depositional areas (i.e., pools) and three from erosional areas (i.e., riffle/runs). If no riffle/run areas were present, five pool samples were collected. A total of 301 and 204 samples were collected during the pre-PBG and

Table 1. Mean physical characteristics of streams before cattle addition and during the treatment period. Measurements were taken before the addition of cattle (pre-patch-burn grazing; n = 18) and during the treatment period (patch-burn grazing; n = 12).

Characteristic	Control†		Fenced		Unfenced	
	Pre-PBG‡	PBG	Pre-PBG	PBG	Pre-PBG	PBG
Erosional, %	16.7 (6)§	14.2 (4)	29.4 (3)	23.6 (3)	32.8 (5)	29.4 (4)
Pool, %	83.3 (6)	85.8 (4)	71.6 (3)	76.4 (3)	67.2 (5)	70.6 (4)
Substrate,¶ %						
Vegetation	15.7 (1)	32.0 (1)	7.15 (1)	18.4 (1)	18.5 (1)	20.9 (1)
Boulder/bedrock	7.3 (1)	1.6 (0)	25.5 (0)	17.5 (1)	15.8 (1)	1.4 (0)
Cobble/pebble/gravel	25.6 (1)	25.6 (1)	31.3 (1)	33.9 (1)	17.0 (0)	15.9 (0)
Fine#	51.4 (2)	42.1 (2)	36.1 (1)	30.2 (1)	48.3 (1)	61.7 (1)

† Control, C2 and C3 watersheds; Fenced, F1 and F4 watersheds; Unfenced, NF5 and NF6 watersheds.

‡ PBG, patch-burn grazing.

§ Values in parentheses are 1 SE.

¶ Substrate estimates were made using a modified Wentworth Scale (Cummins, 1962).

Fine = sand, clay, and silt combined.

PBG periods, respectively. The percentage of pool and riffle habitat in each 100-m stream reach was visually estimated so that data could be habitat weighted based on the proportions of each habitat available in each stream. Habitat-weighted values for each sample date were the sum of the average values from core samples multiplied by % pool habitat and average values from Surbers multiplied by % riffle and run habitat in each stream.

Pool habitats were sampled without intentional bias for organic matter and benthic macroinvertebrates using a 20-cm-diameter stovepipe core (sampling area, 314 cm^2; n = 3). Organic matter and sediments within the stovepipe core were removed to a depth of ~10 cm and deposited into a 20-L bucket. The volume of sample within the bucket was recorded, and the contents were elutriated and poured through a 250-μm sieve until all organic matter was removed from the bucket and retained on the sieve. Very fine particulate organic matter (VFPOM) samples were collected in a 200- to 250-mL sample cup as material passed through the 250-μm sieve. The VFPOM samples were stored on ice and processed within 48 h. Materials retained on the sieve were stored in a clear plastic bag preserved with ~8% formalin solution. The substrata composition of each individual stovepipe core and Surber (see below) sample (percent boulder, cobble, pebble, sand, silt, clay, bedrock) was visually estimated using a modified Wentworth Scale (Cummins, 1962).

Samples were collected from riffle/run habitats using a mini-Surber sampler with an area of 0.023 m^2 and a 250-μm mesh net (n = 3). The mini-Surber was placed evenly onto the substrate, allowing water to flow through the net. Substrata within the mini-Surber perimeter were disturbed with a scrub brush and allowed to flow into the mesh net. Substrata composition was visually estimated for each sample using the method previously described. Contents from the mesh net were rinsed into a plastic bag and preserved in ~8% formalin. Substrata composition was recorded as described above, and VFPOM samples were collected adjacent to the mini-Surber sample location using a stovepipe core as described above.

Laboratory analysis of VFPOM began within 48 h of sample collection following the methods of Whiting et al. (2011). The contents of the sample cups were resuspended with deionized water, and a subsample was vacuum filtered onto pre-ashed/pre-weighed glass fiber filters (47 mm; particle retention size, 1.6 μm). Filters were placed in a drying oven for 48 h, weighed to the nearest 0.1 mg, and combusted for 1 h at 450°C in a muffle furnace. After combustion, filters were rewetted with deionized water and placed in a drying oven for 24 h. Filters were then weighed to the nearest 0.1 mg. Ash-free dry mass (AFDM) was calculated by correcting for the original volume of sample collected, yielding g AFDM m^{-2}.

Laboratory separation of organic materials >250 μm was performed using stacked 1-mm and 250-μm sieves. Organic matter was divided in to coarse particulate organic matter (CPOM; >1 mm) and fine particulate organic matter (FPOM; <1 and >250 μm) fractions following the methods of Whiting et al. (2011). Macroinvertebrates were visually removed from CPOM using a dissecting microscope. Samples were dried for 48 h at 50°C and then weighed to the nearest milligram to estimate dry mass. Samples were then ashed in a muffle furnace at 500°C for ~1 h to estimate AFDM; correcting for sample area allowed for estimation of mg AFDM m^{-2}. Fine fractions were subsampled using a Folsom plankton splitter until 75 to 100 macroinvertebrates were counted. Remaining FPOM was dried and ashed using the same methods as for CPOM.

Macroinvertebrate Abundance and Biomass

Insects were identified to the lowest practical taxonomic level, usually genus, using the method described by Merritt et al. (2008), with the exception of Chironomidae, which were classified as either non-Tanypodinae or Tanypodinae. Noninsect groups were generally identified to family. Macroinvertebrates were placed into functional feeding groups based on Merritt et al. (2008) and Sarver (2005).

Body length (carapace length for crayfish) was measured to the nearest millimeter. Abundance was standardized to individuals m^{-2} based on the sampling area of the device used for collection. Fine sample abundances were multiplied by the subsample fraction and added to the coarse sample abundances to estimate total abundance. Biomass was estimated using length–mass regressions following the procedures of Benke et al. (1999).

Macroinvertebrate Bioassessment Metrics and Diversity

A modified Hilsenhoff Biotic Index (HBI) was used to evaluate overall community tolerance to organic pollution. Each taxon was assigned a tolerance value ranging from 0 to 10 (Hilsenhoff, 1987; Huggins and Moffett, 1988; Barbour et al., 1999). Tolerance values for each individual taxon were multiplied by the abundance of that specific taxon; these values were them summed and divided by the total individuals in the sample. The EPT index was calculated as the total number of taxa from the orders Ephemeroptera, Plecoptera, and Trichoptera in each sample (Barbour et al., 1999). Taxa richness and Shannon diversity (H′) were calculated based on numbers of taxa in samples.

Changes in organic matter standing stocks, physical characteristics, and macroinvertebrate abundance, biomass, functional structure, and diversity were assessed before and after implementation of PBG using a Before-After-Control-Impact (BACI) design with α = 0.05 (Stewart-Oaten et al., 1986). Analyses were run using SAS (version 9.3; SAS Institute Inc.); P values between 0.05 and 0.1 were considered marginally significant. The BACI approach requires sampling a control and an impact location simultaneously before and after a treatment; any preexisting differences between the two systems are accounted for in the development of the premanipulation statistical relationship between sites. Each sampling event is represented as the difference between the impact and control samples. It is assumed that the differences between control and impact sites are constant through time and that any change in these differences is due to the treatment effect. The mean difference between the control and impact sites before and after the treatment was analyzed as a one-way ANOVA design, with the number of observations equal to the number of sampling events before and during the treatment. Estimates of macroinvertebrate abundance and biomass, as well as organic matter, were combined and then averaged based on treatment for this analysis (control = C2 and C3; fenced = F1 and F4; unfenced = NF5 and NF6). The BACI analyses were performed between each treatment: control vs. fenced, control vs. unfenced, and fenced vs. unfenced.

Primer V6.1.3 (Clarke and Gorley, 2006) was used to run nonmetric multidimensional scaling (NMDS) to assess potential changes in community structure based on abundance and biomass.

Results

Inorganic and Organic Substrata

Fine inorganic substrata decreased during the treatment period in control and fenced watersheds by 18 and 16%, respectively, but increased 28% during treatment in the unfenced streams with cattle. The increase in fine inorganic substrata in unfenced watersheds relative to fenced watersheds was marginally significantly different ($F_{1,34} = 3.93$; $p = 0.056$) but was significantly different relative to the control watersheds ($F_{1,34} = 4.97$; $p = 0.032$) (Fig. 2). Estimates of cobble, pebble, and gravel showed little change in any of the streams during treatment.

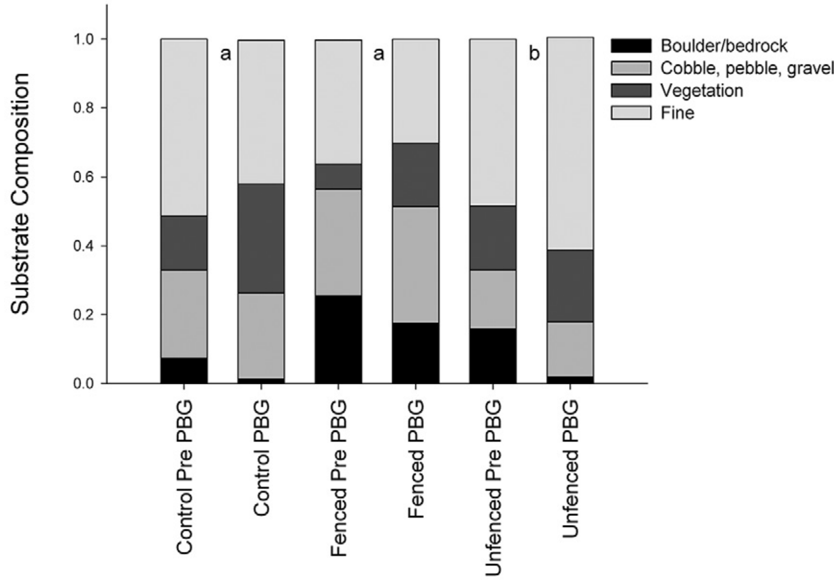

Fig. 2. Mean substrata composition in six headwater streams on Osage Prairie, Nevada, MO, before patch-burn grazing treatment (Pre-PBG) and during the treatment period (PBG). Lowercase letters denote significant differences in the magnitude of change from the pre-PBG year to the treatment year between control, fenced, and unfenced watersheds at $\alpha = 0.05$.

Standing stocks of VFPOM and FPOM in control watersheds were similar before and during the treatment period, whereas CPOM increased 19%. Unfenced watersheds showed the greatest change in FPOM during the treatment relative to the control and fenced watersheds, with a 67% increase, whereas FPOM in control and fenced watersheds was similar before and during the treatment. The change in FPOM in unfenced watersheds during the treatment period was significantly different from the control watershed ($F_{1,27} = 8.35$; $p < 0.008$) (Fig. 3). The slight increase in VFPOM in fenced watersheds was similar to that of control watersheds, but VFPOM in unfenced watersheds increased >2-fold, and this was significantly different from the lack of change in control watersheds ($F_{1,50} = 7.88$; $p = 0.007$) (Fig. 3). The change in VFPOM in the unfenced watersheds was also significantly different from the fenced watersheds ($F_{1,50} = 6.74$; $p = 0.012$).

Macroinvertebrate Abundance

Total habitat-weighted macroinvertebrate abundance in control streams showed trends of increasing (15%), whereas abundance in the fenced and unfenced streams declined by 25 and 47%, respectively (Table 2). Collector-gatherers dominated abundance in the study streams and increased in all streams during the treatment period, with the abundance in unfenced streams showing the greatest increase of 37% (Table 2; Fig. 4). Trends of decreasing abundance of collector-filterers were observed in all streams during the treatment, with the greatest decrease in unfenced watersheds (7.7-fold decrease) (Fig. 5). Predators also showed trends of decreasing relative abundance (contribution to total abundance) in all streams, with unfenced streams again showing the largest decrease during treatment (3.8-fold decrease). No consistent patterns were observed in the abundance of grazing or shredding macroinvertebrates.

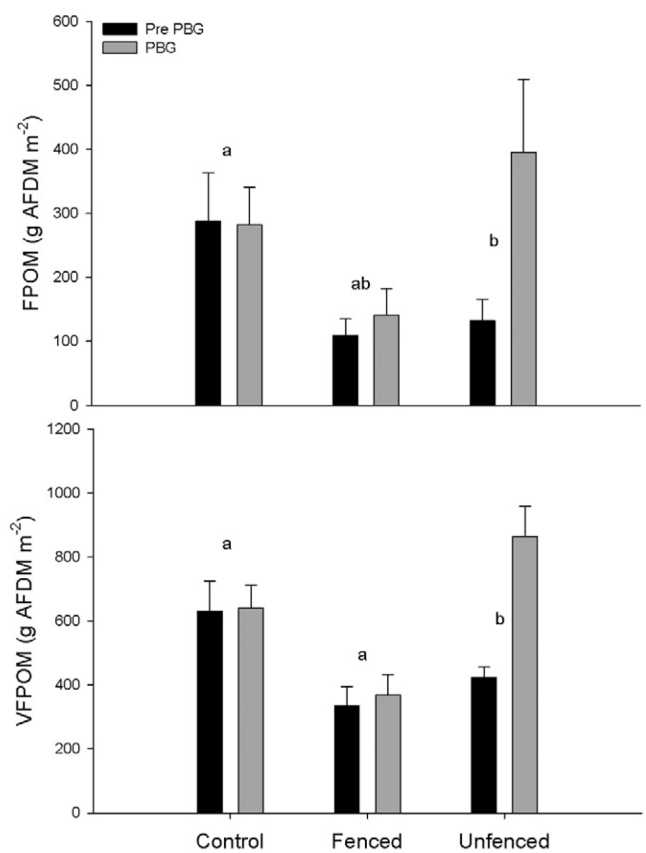

Fig. 3. Mean (±1 SE) standing stocks of fine particulate organic matter (FPOM) and very fine particulate organic matter (VFPOM) in g ash-free dry mass (AFDM) m^{-2} in six headwater streams on Osage Prairie, Nevada, MO, before patch-burn grazing treatment (Pre-PBG) and during the treatment period (PBG). Lowercase letters denote significant differences in the magnitude of change from the PBG year to the treatment year between control, fenced, and unfenced watersheds at $\alpha = 0.05$.

Macroinvertebrate Biomass

Total habitat-weighted macroinvertebrate biomass declined in all streams during the treatment (Table 2). Similar decreases in biomass were observed in fenced and unfenced streams (51 and 63%, respectively), with control streams showing the greatest decline in biomass (87%). Habitat-weighted biomass of FFG groups followed similar trends as abundance (Table 2; Fig. 4), with the exception of collector-gatherers; relative biomass of collector-gatherers decreased in all streams, with the relative biomass in control streams decreasing 40%. The decrease in relative biomass of collector-gatherers in control streams was significantly different from unfenced streams ($F_{1,28} = 5.01$; $p = 0.034$), where collector-gatherer relative biomass decreased by 40%. The relative biomass of collector-filterers followed similar trends as abundance, decreasing in all streams by 56 to 64%. Shredder relative biomass showed trends of increasing in all streams, with the fenced and unfenced streams increasing 2.7- and 3.5-fold, respectively, and with the relative biomass in control streams increasing 7.2-fold. The relative contribution of predators to the overall biomass in control and fenced streams remained similar during treatment, whereas those in unfenced streams decreased by 40%. The relative biomass of scrapers in control streams was similar before and during the treatment period (~3%). In fenced streams, relative scraper biomass decreased by ~4%, whereas an increase of 5% was observed in unfenced streams. The change in relative biomass of scrapers in fenced versus unfenced streams was significantly different ($F_{1,28} = 6.10$; $p = 0.021$).

The NMDS analysis showed significant community level responses in biomass of taxa to the treatments in fenced ($p = 0.05$) and unfenced watersheds ($p < 0.01$). The response in unfenced watersheds was driven by increases in *Siphlonurus*, non-Tanypodinae midges, and *Physa*, along with decreases in *Cordulegaster*, *Orconectes*, and *Agabus*. The significant change in fenced watersheds was a result of increases in *Siphlonurus*, *Agabus*, and *Laccophilus*. Similar to unfenced watersheds, *Cordulegaster* and *Orconectes* biomass decreased in fenced watersheds.

Indicator Taxa and Bioassessment Metrics

The relative abundance of Oligochaeta increased by 5% in fenced and 4% in unfenced streams, with control streams showing similar values of ~14% before and during the treatment period (Table 3). The relative abundance of Chironomidae showed trends of decreasing in control and fenced streams (1.2- and 1.1-fold, respectively). The 2.7-fold increase in Chironomidae relative abundance in unfenced streams was significantly different from the control ($F_{1,28} = 16.07$; $p < 0.001$) and fenced ($F_{1,28} = 14.13$; $p < 0.001$) streams (Fig. 4). The relative abundance of EPT increased 39% in control streams during the treatment period but decreased somewhat in fenced (5%) and unfenced (77%) streams during the treatment period (Fig. 4). The overall abundance of EPT taxa decreased 11.1-fold in unfenced streams (Fig. 5). The decrease in EPT relative abundance in unfenced streams was significantly different from control streams ($F_{1,28} = 8.31$; $p = 0.008$) and was marginally significantly different compared with fenced streams ($F_{1,28} = 3.73$; $p = 0.066$).

The relative biomass of Oligochaeta showed similar trends of decline in fenced and unfenced streams (3 and 2%, respectively), whereas control streams showed similar values before and during the treatment period (Table 3). The relative biomass of Chironomidae decreased in control streams (52%) but increased by 83% in unfenced streams. The increase in relative biomass of Chironomidae in unfenced streams was significantly different from the decrease in control streams ($F_{1,28} = 13.54$; $p = 0.001$)

Table 2. Habitat-weighted mean abundance and biomass of benthic macroinvertebrates in Osage Prairie streams before the addition of cattle (pre-patch-burn grazing; *n* = 18) and during the treatment period (patch-burn grazing; *n* = 12).

Functional feed group	Control†		Fenced		Unfenced	
	Pre-PBG‡	PBG	Pre-PBG	PBG	Pre-PBG	PBG
Gatherer						
Abundance, individuals m⁻²	47,388.2 (8,350)§	67,733.2 (25,115)	48,740.7 (12,033)	32,323.4 (6,354)	41,185.8 (9,042)	28,675.1 (6,928)
Biomass, mg AFDM¶ m⁻²	6,859.2 (4,263)	778.2 (263)	2,276.3 (741)	1,100.1 (351)	2,197.6 (402)	1,229.0 (315)
Filterer						
Abundance, individuals m⁻²	10,855.7 (4,853)	8,631.5 (7,192)	4,842.4 (1,840)	3,462.9 (1,593)	23,918.3 (8,639)	3,097.8 (2,011)
Biomass, mg AFDM m⁻²	58.4 (31)	5.9 (3)	35.4 (17)	7.9 (3)	107.0 (38)	21.6 (15)
Shredder						
Abundance, individuals m⁻²	370.5 (193)	1,611.3 (439)	745.1 (253)	1,111.2 (259)	298.2 (53)	1,454.3 (518)
Biomass, mg AFDM m⁻²	153.2 (51)	309.5 (87)	279.4 (87)	522.0 (126)	325.6 (124)	474.6 (135)
Predator						
Abundance, individuals m⁻²	10,282.5 (1,831)	4,071.5 (1,062)	9,107.2 (1,752)	10,784.9 (7101)	10,141.2 (1,489)	2,657.4 (524)
Biomass, mg AFDM m⁻²	5,515.6 (4,564)	630.9 (166)	3,181.4 (1,690)	1,996.9 (682)	2,719.5 (715)	898.5 (324)
Scraper						
Abundance, individuals m⁻²	232.0 (94)	291.9 (195)	517.3 (264)	417.9 (264)	1,581.2 (676)	1,329.0 (866)
Biomass, mg AFDM m⁻²	114.9 (48)	59.6 (33)	430.3 (263)	176.7 (78)	427.4 (135)	637.5 (245)
Overall						
Abundance, individuals m⁻²	71,522.2 (14,158)	82,414.8 (32,018)	64,439.3 (13,407)	48,113.9 (13,473)	76,802.1 (14,160)	40,753.9 (2,791)
Biomass, mg AFDM m⁻²	14,130.6 (9,015)	1,784.0 (328)	7,045.9 (2,033)	3,461.0 (900)	9,785.9 (2,791)	3,621.1 (722)

† Control, C2 and C3 watersheds; Fenced, F1 and F4 watersheds; Unfenced, NF5 and NF6 watersheds.

‡ PBG, patch-burn grazing.

§ Values in parentheses are 1 SE.

¶ Ash-free dry mass.

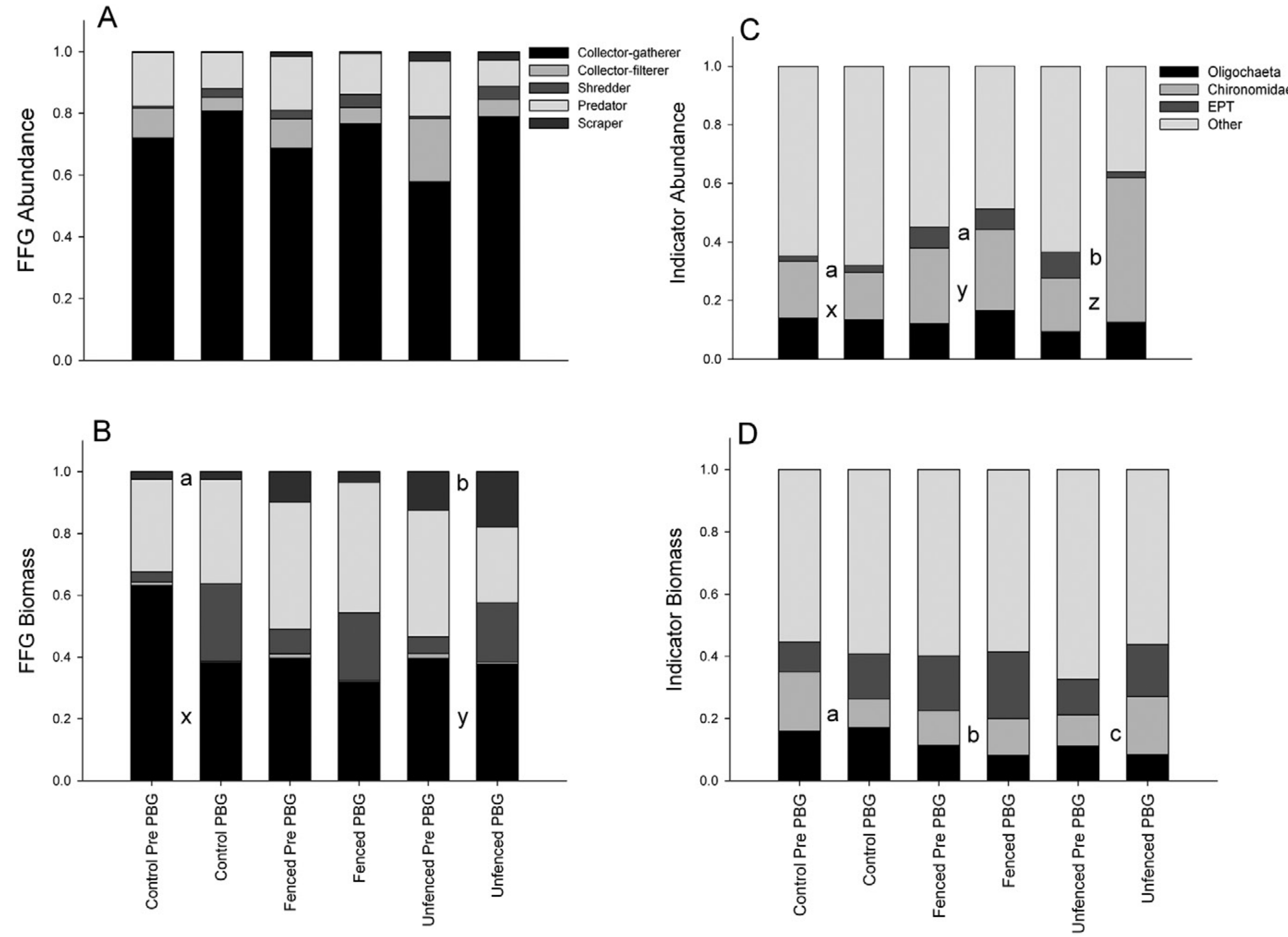

Fig. 4. Percentage contributions of functional feeding group abundance (A) and biomass (B) and contribution of indicator taxa abundance (C) and biomass (D) to total macroinvertebrates in six headwater streams on Osage Prairie, Nevada, MO, before patch-burn grazing treatment (Pre-PBG) and during the treatment period (PBG). Lowercase letters denote significant differences in the magnitude of change from the PBG year to the treatment year between control, fenced, and unfenced watersheds for the functional groups they are next to (α = 0.05). EPT, taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera; FFG, functional feeding group.

as well as the lack of change in fenced streams ($F_{1,28} = 4.90$; $p = 0.035$) (Fig. 4). The changes in relative biomass of Chironomidae in fenced and control streams were also different ($F_{1,28} = 4.61$; $p = 0.042$). Although the relative biomass of Chironomidae increased in the control and unfenced streams, biomass in terms of actual mg AFDM m^{-2} decreased in all streams during the treatment (Table 3). The relative biomass of EPT changed similarly in all streams, with each treatment showing increasing trends of 4 to 5% during the treatment period.

Shannon diversity (H′) decreased somewhat in all streams during treatment (Table 4), with the greatest decrease in unfenced streams (2.0–1.4). The trend of declining EPT index was similar across all sites, with the greatest decline of 1.9 observed in fenced streams. Control and unfenced streams decreased similarly (1.2 and 1.3, respectively). Taxa richness also followed trends of declining in all streams. Pretreatment richness estimates for control streams were 24.8 and decreased to 20.0 during the treatment, whereas fenced and unfenced streams had initial taxa richness estimates of 31.9 and 30.8, respectively, and fell to 29.2 and 26.8. Although increases in the HBI were observed in all streams, a statistically significant change was detected between unfenced and control streams ($F_{1,28} = 4.31$; $p = 0.051$); control streams increased from 7.5 to 7.8, whereas unfenced streams increased from 7.1 to 7.8. Control streams did not change in HBI category, but fenced and unfenced moved from "poor" to "very poor."

Discussion

Our study demonstrates that PBG can affect tallgrass prairie streams in a variety of ways, although negative impacts may be mitigated by fencing to restrict cattle from riparian habitats and the stream channel. Multiple indicators of stream integrity decreased after cattle grazing in the unfenced streams, but similar changes were not evident in the control or fenced streams. Thus, PBG resulted in an overall decline in biotic integrity of unfenced streams, and the biological responses we observed appeared to be linked to increases in fine organic and inorganic sediments in the stream channels.

Inorganic and Organic Substrata

Increases in fine inorganic substrata in the unfenced watersheds were consistent with our hypothesis that cattle grazing in riparian areas would lead to increased sedimentation. These results support those of previous studies assessing the effects of cattle on stream substrata and the positive influence riparian vegetation may have on streams (McKergow et al.,

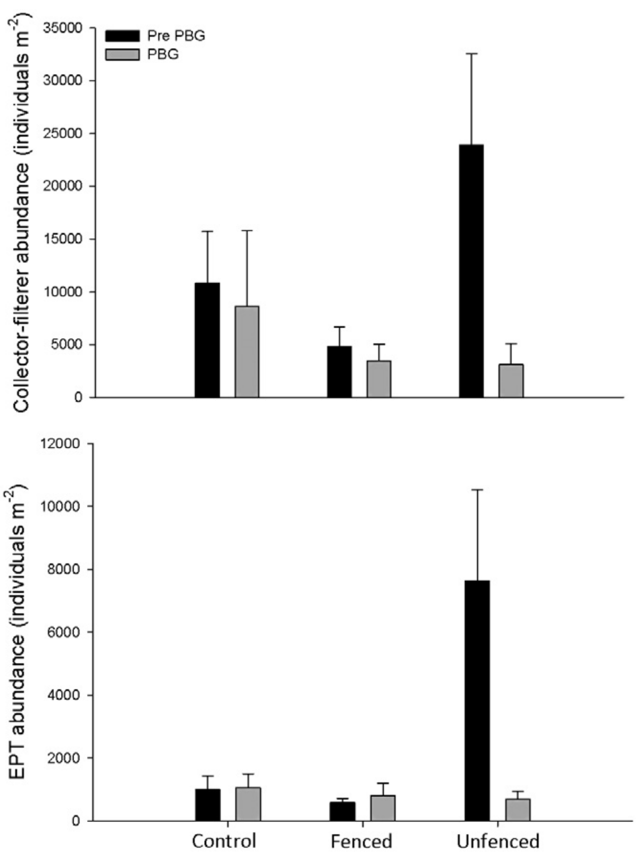

Fig. 5. Mean (±1 SE) abundance (individuals m^{-2}) of collector-filterer (A) and EPT taxa (B) in six headwater streams on Osage Prairie, Nevada, MO, before patch-burn grazing treatment (Pre-PBG) and during the treatment period (PBG). EPT, taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera.

2003; Raymond and Vondracek, 2011; Herbst et al., 2012). Excess sedimentation can harm aquatic organisms. For example, Suttle et al. (2004) found that increases in fine sediment led to decreased growth and survival of aquatic vertebrates. Increases in fine sediments can also reduce overall abundance and richness of macroinvertebrates (Larsen et al., 2011). Sedimentation may affect macroinvertebrates via multiple mechanisms. For example, Culp and Davies (1983) reported increased drift and loss of stable substrata as fine sediments increased. Interstitial spaces within the substrata and overall substrate heterogeneity decrease with increasing fine sediments, reducing important refugia for macroinvertebrates and spawning and nursery habitats for many fishes (Geist and Dauble, 1998; Gardner, 1999). Sedimentation can also affect system productivity by decreasing light availability for photosynthesis (Graham, 1990; Davies-Colley et al., 1992; Larson et al., 2013).

As with fine inorganic sediments, grazing in riparian areas also appeared to be linked to increases in fine organic particles in the unfenced streams. Excess fine organic sediments can increase turbidity and hinder light availability, reducing productivity (Graham, 1990; Davies-Colley et al., 1992; Castro and Reckendorf, 1995). Although potential initial increases in filtering macroinvertebrates reliant on suspended fine organic matter for food could occur, an excess of particles can clog or otherwise interfere with the feeding devices (e.g., silk nets of hydropsychid caddisflies and cephalic fans of black flies) and fill interstitial spaces that serve as important habitats for benthic organisms (Cushing et al., 1993; Hamm et al., 2011). Bison activities (i.e., native grazers in tallgrass prairie) can also increase fine sediments in streams, but bison spend less time in the water, and their effects tend to be localized to bison-crossing areas (Fritz et al., 1999).

Contrary to our hypothesis that CPOM would decrease as cattle grazed unfenced riparian areas, there were no significant patterns with CPOM before and after PBG. Cattle grazing in riparian areas can lead to decreases in CPOM and allochthonous inputs (Kauffman and Krueger, 1984). The lack of response in the unfenced streams on Osage may be related to the size of the burn parcels relative to the 100-m study reaches. The 100-m reaches did not extend past the portions of watersheds burned in 2011, with the remaining areas burned in 2012 and 2013 farther upstream. Cattle generally spend ~75% of grazing time within the most recently burned areas (Fuhlendorf and Engle, 2004). It is thus likely that the cattle in our study spent much of their

Table 3. Habitat-weighted mean abundance and biomass of Oligochaeta, Chironomidae, and EPT taxa in Osage Prairie streams before the addition of cattle (pre-patch-burn grazing; *n* = 18) and during the treatment period (patch-burn grazing; *n* = 12).

Taxa	Control†		Fenced		Unfenced	
	Pre-PBG‡	PBG	Pre-PBG	PBG	Pre-PBG	PBG
Oligochaeta						
Abundance, individuals m^{-2}	8551.8 (2343)§	11,610.6 (5687)	7475.4 (2306)	6994.9 (2272)	6037.3 (1941)	5175.4 (1905)
Biomass, mg AFDM¶ m^{-2}	500.3 (144)	242.9 (80)	691.4 (319)	170.5 (37)	429.2 (110)	306.6 (169)
Chironomidae						
Abundance, individuals m^{-2}	9727.4 (2390)	6693.6 (1678)	17,433.4 (5892)	12,701.1 (3804)	14,637.3 (3879)	18,337.4 (4694)
Biomass, mg AFDM m^{-2}	852.4 (280)	176.6 (65)	828.5 (427)	304.6 (100)	727.5 (269)	465.5 (100)
EPT#						
Abundance, individuals m^{-2}	997.7 (429)	1063.0 (436)	2692.3 (812)	1631.0 (398)	7635.4 (2898)	687.1 (253)
Biomass, mg AFDM m^{-2}	824.7 (451)	368.8 (203)	586.0 (130)	780.1 (360)	890.2 (260)	801.7 (368)

† Control, C2 and C3 watersheds; Fenced, F1 and F4 watersheds; Unfenced, NF5 and NF6 watersheds.

‡ PBG, patch-burn grazing.

§ Values in parentheses are 1 SE.

¶ Ash-free dry mass.

Taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera.

time in riparian areas much further upstream of the study reaches during the 2012 and 2013 grazing seasons.

Macroinvertebrate Functional Structure

Functional group responses in the unfenced streams suggest that cattle may shift the system to increasing importance of autochthonous food resources through direct and indirect pathways. As hypothesized, there was a significant increase in the relative biomass of scrapers in unfenced streams compared with fenced streams, and this was likely driven by increased light availability and nutrient inputs (e.g., Sabater et al., 2005; Bowman et al., 2007). Although we did not measure light availability in the channels, nutrient inputs were measured in a companion study. Results showed that total nitrogen values were significantly greater in the unfenced streams compared with control and fenced streams during the grazing manipulation (Larson, 2014). In particular, ammonium (NH_4^+) concentrations increased in the unfenced streams from 46.4 μg L^{-1} before grazing to 95.4 μg L^{-1} during grazing (Larson, 2014). Additionally, visual observations of stream channel geomorphology, as well as bank structure, indicated that grazing influenced bank width and exposure to light.

The increase in the relative biomass of scrapers in unfenced streams was driven primarily by a 3.3-fold increase in biomass of *Physa* snails. *Physa* are pulmonate snails that are tolerant to a variety of stressors including sediments, warm water, low dissolved oxygen, and nutrients. Although *Physa* are sometimes abundant in relatively undisturbed prairie streams (Stagliano and Whiles, 2002), they generally do not dominate the scraper functional group, as is often the case in eutrophic streams influenced by agricultural activities (Yuan, 2006; Stone et al., 2005).

The trend of lower collector-gatherer biomass in unfenced streams during treatment was due primarily to reduced numbers of non-Tanypodinae midges and the case-making hydroptilid caddisfly *Ochrotrichia*. Although the relative contribution of Chironomidae to total biomass in unfenced streams increased during treatment because of decreases in other taxa, the decrease in actual biomass of chironomids was unexpected. Chironomidae are generally considered relatively tolerant of pollution and disturbance, including sedimentation (Relyea et al., 2000). However, Chironomidae is an extremely diverse family in terms of taxonomic diversity and sensitivity. For example, some genera may have tolerance values comparable to sensitive EPT taxa (Bode et al., 1996); the particular taxa of midges that declined in our study streams may have been less tolerant of disturbances associated with cattle. Over time, shifts in the composition of the chironomid assemblages (e.g., increases in more tolerant midge taxa) in the grazed unfenced streams may occur.

Although not statistically significant, the trend of reduced abundance of filtering macroinvertebrates in the unfenced streams during PBG is consistent with prior studies investigating responses to sedimentation (Bryce and Lomnick, 2010; Larsen et al., 2011). Declines in filterers in the unfenced streams were primarily driven by ~90% decreases in chydorid cladocerans. Although individual chydorid taxa feed in a variety of ways and some may not be filter-feeders (Thorp and Covich, 2009), we considered the group as a whole to be filter-feeders for this study. Simuliid black flies also declined by ~10-fold, a pattern consistent with a prior study demonstrating that black flies are sensitive to increasing fine sediments (Larsen et al., 2011). Substantial decreases in the filter-feeding caddisfly *Hydropsyche* also contributed to the overall patterns with collector-filterers, and this negative response to sedimentation is consistent with a prior study in Kansas tallgrass prairie streams (Fritz et al., 1999). *Hydropsyche* densities were >2000 individuals m^{-2} in stream NF6 before treatment, but none was encountered in this stream during treatment. Declines in *Hydropsyche* during treatment may be related to burial of stable substrata, clogging of filtering nets, or a combination of the two; sedimentation has been shown to not only disrupt feeding but also to alter how *Hydropsyche* construct nets (Runde, 2000).

The lack of a response by shredding macroinverterbrates may have been related to the lack of a decline in CPOM in the PBG streams during treatment. Shredder populations are generally limited by CPOM (e.g., Wallace et al., 1997; Straka et al., 2012), and the availability of this resource did not change with PBG. As noted above, in 2012 and 2013 the burned parcels, which are most attractive to grazing cattle, were located upstream from the 100-m sampling reach. This may have reduced potential cattle impacts on CPOM inputs. Future applications of PBG may consider the exact location and timing of burn treatments in the context of riparian vegetation and organic matter inputs to streams. It is worth noting that woody riparian vegetation was relatively sparse in all treatments compared with nearby forested regions, so allochthonous input rates were relatively low in all these streams.

Biotic Integrity

Although actual numbers and biomass of Chironomidae decreased, increases in the relative abundance and biomass of this group in unfenced streams supported our prediction that the

Table 4. Macroinvertebrate community metrics for six headwater streams on Osage Prairie before the addition of cattle (pre-patch-burn grazing; *n* = 18) and during the treatment period (patch-burn grazing; *n* = 12).

Metric†	Control‡		Fenced		Unfenced	
	Pre-PBG§	PBG	Pre-PBG	PBG	Pre-PBG	PBG
H′	1.9 (0.1)¶	1.4 (0.1)	2.0 (0.1)	1.6 (0.1)	2.0 (0.1)	1.4 (0.1)
HBI	7.5 (0.3)	7.8 (0.1)	7.4 (0.2)	7.6 (0.1)	7.1 (0.3)	7.8 (0.1)
EPT	4.2 (0.8)	3.0 (0.7)	8.2 (1.0)	6.3 (0.9)	5.6 (0.8)	4.3 (0.7)
Richness	24.8 (1.6)	20.0 (2.1)	31.9 (1.3)	29.2 (1.2)	30.8 (1.3)	26.8 (1.1)

† H′ = Shannon Diversity; HBI = Hilsenhoff Biotic Index; EPT = taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera.

‡ Control, C2 and C3 watersheds; Fenced, F1 and F4 watersheds; Unfenced, NF5 and NF6 watersheds.

§ PBG, patch-burn grazing.

¶ Values in parentheses are 1 SE.

relative importance of tolerant taxa would increase in unfenced PBG streams. This prediction was also supported by declines of some sensitive EPT taxa and an overall decline in biotic integrity in unfenced streams. Declines in biotic integrity in the unfenced streams were linked to reduced relative abundance of two EPT taxa, *Paraleptophlebia* (Leptophlebiidae) and *Polycentropus* (Polycentropodidae). Fine sediments can have adverse effects on Leptophlebiidae mayflies (Broekhuizen et al., 2001), suggesting that the negative response we observed was linked to sedimentation from the cattle.

Responses of EPT taxa to unfenced PBG were similar to those observed in a previous study examining the effects of bison crossings on the biotic integrity of tallgrass prairie streams. Fritz et al. (1999) found that average EPT taxa richness fell from 3.0 to 1.8 at bison crossings, which is similar to the degree of change we observed (6.0–4.0) during PBG treatments in unfenced streams. However, the effects of bison on EPT taxa were limited to small areas where they cross the streams (Fritz et al., 1999), whereas we observed responses to cattle at the stream-reach scale.

Although overall EPT taxa responded negatively to PBG without fencing, NMDS analyses showed an increase in biomass of one mayfly taxon, *Siphlonurus*, in fenced and unfenced watersheds. The positive response by *Siphlonurus* was likely linked to their general tolerance to fine sediments (e.g., Nelson, 2010). *Siphlonurus* also feed primarily on fine organic sediments (Voshell, 1982).

Some measures of biotic integrity, such as taxa richness, did not change significantly in the unfenced streams during PBG. Shannon diversity did show a trend of declining in the unfenced streams during PBG, but this same trend was evident in the other study streams. Moderate or intermediate levels of natural hydrologic disturbance (e.g., intermediate disturbance hypothesis [Connell, 1978]) can sometimes enhance freshwater macroinvertebrate diversity and richness (Whiles and Goldowitz, 2001; Danehy et al., 2012). However, the prolonged dry period during the summer of the treatment year may have affected diversity and richness in all sites (Clarke et al., 2010; Martinez et al., 2013), possibly obscuring some PBG effects. For example, Ledger et al. (2012) found that high-frequency drought disturbance resulted in dominance by Chironomidae and Oligochaeta and reduced overall macroinvertebrate diversity. Other researchers have shown that stream invertebrate taxa richness increases with increasing flow duration (Boulton and Lake, 1992).

Changes in the community composition of aquatic insects may have far-reaching effects beyond streams because many riparian predators rely on prey species that originate in the water (Polis et al., 1997; Baxter et al., 2005; Ballinger and Lake, 2006). Increases in the relative abundance of taxa such as *Physa* snails and Oligochaeta, which do not have terrestrial life stages, in the unfenced streams, paired with reductions of EPT taxa, which emerge from the water as volant adults, may reduce energetic subsidies to the surrounding prairie. For example, Gray (1993) showed that insectivorous birds responded positively to emergences of adult insects from tallgrass prairie streams. Increases in the relative abundances of small-bodied insects such as Chironomidae, which essentially represents a shift to communities of smaller species in the unfenced streams, could further reduce prey availability for riparian predators. Heinrich et al. (2014) found that insectivorous birds were most attracted to emergences of larger-bodied adult aquatic insects (EPT taxa) from a midwestern river. Prairie conservation and management activities should account for the contributions of streams to prairie food webs and how management practices such as PBG may affect them.

Current management regimes focus primarily on terrestrial components of tallgrass prairies. However, protecting aquatic ecosystems is vital to maintaining a healthy and intact terrestrial ecosystem and watershed. Our results indicate that PBG can have adverse effects on tallgrass prairie streams but that these impacts are mitigated with 15-m riparian exclusion fencing to prevent cattle access. These results need to be interpreted in the context of documented benefits of PBG to terrestrial species and can provide a foundation for future decisions regarding management and conservation of the last <1% of original tallgrass prairies and streams that remain.

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