

SOIL CHEMISTRY AS AFFECTED BY FIRST-TIME PRESCRIBED BURNING OF A GRASSLAND RESTORATION ON A COASTAL PLAIN ULTISOL

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While grassland restorations have gained recent popularity throughout the United States, few restorations have been initiated on the highly weathered soils of the Mid-Atlantic Coastal Plain; thus potential soil response to grassland management practices, such as prescribed burning, are not clearly understood in this region. The objective of this study was to evaluate the impacts of prescribed burning on the soil chemistry of a young grassland restoration established on agricultural fields in eastern Maryland. The soil is a slightly acidic highly weathered Typic Hapludult with a sandy loam surface texture and low cation exchange capacity (CEC). In spring 2003, soil was collected in five intervals to a depth of 20 cm 1 d before and 11 d and 1 yr after a first-time prescribed burn. By 11 d after the burn, and following 4.6 cm of rainfall, soil pH had increased significantly in all depth intervals relative to pre-burn values. Prescribed burning also significantly changed the relationship of CEC, Ca, and Mg with organic matter within 11 d. Results indicated that the infiltration and dissolution of base-cation-rich, alkaline ash from several days of rain after the burn significantly changed the soil chemistry within a few weeks after burning. By 1 yr after the burn, the soil pH and organic matter did not differ from pre-burn conditions. The impacts of prescribed burning of a grassland on a highly weathered Ultisol, therefore, involve short-term changes in soil chemistry, followed by dissipation within a year due to long-term buffering of the soils. (Soil Science 2005;170:913-927)

Key words: Prescribed burn, cation exchange capacity, prairie restoration, soil chemical properties, pH

GRASSLAND restoration projects on environmentally sensitive or poor agricultural lands have been developed in recent decades in many regions of the United States. Many of the projects have been supported by the Conservation Reserve Program (CRP) and Conservation Reserve Enhancement Program (CREP), which have been major incentives from the federal and state governments to encourage farmers to restore natural wildlife habitat and to reduce soil

erosion potential. These projects have added to the wealth of knowledge obtained from over 50 years of grassland restoration and conservation work in the United States (Collins and Wallace, 1990; Knapp et al., 1998; Risser et al., 1981).

Management of CRP and CREP grassland sites often involves prescribed burning or mowing to reduce invasion of woody species and to improve conditions for growth. These management tools remove standing litter and usually change limiting conditions of light, nitrogen availability, and moisture (Baer et al., 2003; Hulbert, 1969; Hulbert, 1988; Knapp and Seastedt, 1986). The majority of CRP sites have been developed in the Midwestern United States and in the Great Plains (USDA, 2004) where prairies were historically prevalent on the landscape.

One region of the United States that has not received much attention in terms of restoration efforts is the MidAtlantic Coastal Plain. Evidence

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exists to suggest that the vegetation of the MidAtlantic Coastal Plain, as well as that of coastal New England, consisted primarily of grasslands, pineland savanna, and heath land during various periods in geologic history, including pre-colonial times (Kulikoff, 1996; Tyndall, 1992). This vegetation persisted into recent historical times primarily due to natural fires and fires set by Native Americans (Anderson, 1982; Pyne, 1983; Vickery and Dunwiddie, 1997). Rapid agricultural development and urban growth in the nineteenth century, however, led to significant loss and near elimination of all grassland habitat in the MidAtlantic Coastal Plain (Askins, 1997).

Grassland restoration projects have been conducted in a few areas of the East Coast, primarily on military grounds and airports in the Northeast (Vickery and Dunwiddie, 1997) or on coastal islands of New England (Vickery and Dunwiddie, 1997). In the MidAtlantic States, grassland projects have been developed on agricultural land through the CRP program, including plantings to perennial grasses, however the land enrolled is <1% of the total CRP contracts in the United States (USDA, 2004). Moreover, the percentage planted to perennial grasses has decreased over the last few years (USDA, 2004). Grasslands have also been managed in several National Wildlife Refuges in the area (USFWS, 2003). Prescribed burning is conducted as basic annual management of a majority of the warm-season grasslands in the MidAtlantic Coastal Plain (M. Mitchell, personal communication, 2004). The impact of the prescribed burns on the vegetation, not the soil, has been the primary focus of these studies.

The low-nutrient status and acidity of most of the soils in the MidAtlantic region are in contrast to the nutrient-rich soils of many regions of the Midwest and Great Plains, where most grassland restoration projects have been undertaken. Many of the soils, in particular in eastern Maryland, are highly weathered Ultisols (Shields and Davis, 2002). The input of carbon-rich and base-cation-rich, alkaline ash from burning should positively affect numerous soil chemical properties in these soils. Soil pH (Ehrenreich and Aikman, 1963; Ojima, 1987; Owensby and Wyrill, 1973; Picone et al., 2003; Tester, 1989) has been shown to be increased by prescribed burning in a few grasslands in the Midwest and Great Plains. Reports of increases in soil organic carbon (Ueckert et al., 1978) and exchangeable cations (Christianson, 1976;

Owensby and Wyrill, 1973) in response to burning are rare, however. Very few studies of the cation exchange capacity (CEC) of soils, an inherently important fundamental property of soils, have been conducted; of those reported, no changes in CEC were measured after prescribed burns (Picone et al., 2003; J. Blair, personal communication, 2004).

The objective of this study was to evaluate the impact of first-time prescribed burning on the soil chemistry of a grassland restoration on a highly weathered Ultisol of the MidAtlantic Coastal Plain. We hypothesized that ash infiltration after a prescribed burning of a grassland restoration on soil of the MidAtlantic Coastal Plain would increase near-surface soil pH to a greater degree than found in other regions after burning, as well as increase the base cation contents of the weathered soils. In addition, we hypothesized that the soil CEC would increase from ash incorporation and from the increase in soil pH.

MATERIALS AND METHODS

Site Description

A grassland restoration was initiated on 92 ha of land previously cropped for more than 60 years in a corn (*Zea mays* L.)-soybean (*Glycine max* L.)-wheat (*Triticum aestivum* L.) rotation at the Chester River Field Research Center (CRFRC) in Queen Anne's County, Maryland (76° 01' W, 39° 14' N). The average annual precipitation in the region is 108 cm and the average winter and summer air temperatures are 1.9 and 23.5 °C, respectively (Shields and Davis, 2002).

After being entered into the CRP in 1998, numerous 1- to 13-ha, low-production agricultural fields were drill-seeded with various mixtures of warm season grasses in Spring 1999 (Schwartzman et al., 2002). The fields at this site have been managed to optimize conditions for grassland bird habitat. As part of this management scheme, prescribed burning has been conducted every three years on a staggered rotational basis, usually in March or April. By the end of the second growing season (i.e., Fall 2000), warm-season grasses had been successfully established in all fields (Schwartzman et al., 2002).

A 13-ha field that had never been burned was chosen as the study field to investigate the impacts of first-time prescribed burning. The field had been seeded to big bluestem (*Andropogon gerardii* Vitman), little bluestem [*Schizachyrium*

scoparium (Michx.) Nash], and eastern gamagrass (*Tripsacum dactyloides* L.). Despite variable weather since the time of seeding (i.e., a drought in 1999, a wet cold summer in 2000, an average summer in 2001, and severe heat and a drought during Summer 2002), the overall percentage of live vegetation cover was 42% in 2002, with approximately 25% being composed of warm-season grasses (Schwartzman et al., 2002). The soil underlying the field is a well-drained Unicorn-Sassafras loam (40% coarse-loamy, mixed, semiactive mesic Typic Hapludult and 45% fine-loamy, siliceous, semiactive, mesic Typic Hapludult) (USDA, 2004). These soils formed in Atlantic coastal plain parent materials and are highly weathered and well drained (USDA, 2004).

Experimental Design

For this study, a representative 0.4-ha area (60 m long \times 60 m wide) was selected within the 13-ha field. Preliminary analyses of the soil and the vegetation in the field in a smaller 200 m² plot before the first burn revealed a large degree of natural variability of soil chemical properties and a good representation of the variability of vegetation in the 200-m² plot as compared with the entire field (Swartzman et al., 2002). We therefore assumed that the 0.4-ha study area had a mix of vegetation and a similar degree of variability in soil chemical properties as the majority of the field. Moreover, the slope, which was uniform and $<0.5\%$, was representative of the field. The size of the study area was similar to that used by Brye et al., (2002) who characterized the fate of litter-contained nutrients after multiple burning frequencies.

To conduct a detailed evaluation of the effect of burning on the soil surface chemical properties of the field, a 25-point sampling grid was established in the 0.4-ha study area before burning. All sampling points in the grid were evenly spaced 15-m apart and were assumed to be independent observations characterizing the variability of the study area. Soil surface properties can be spatially correlated, up to large distances under some circumstances. However, the assumption of independent observations at a sample spacing of at least 15 m is supported by numerous previous studies in native prairies (Brye, 2003; Brye et al., 2004b; Brye and Slaton, 2003; Brye and West, 2005; Brye et al., 2004d) and prairie restorations (6 m sample spacing, Brye et al., 2002), in which sampling was conducted based on the same assumption of sample

independence. The experimental unit, within which soil properties were assumed uniform, was established as a 0.1-m² area surrounding each sampling point to accommodate soil sampling before and after burning. Replicate analyses were run on subsamples from each core collected in the study area. In addition to analyzing the data for correlations between soil properties using the 25 samples, the mean of the samples collected was calculated to compare the average change in the soil properties before and after the burn.

The effects of burning were evaluated based on a comparison of soil samples collected after the burn with samples collected just before the burn. Since this study was conducted within the context of a larger bird habitat restoration project, a nonburned control was not a practical nor available option to use in the experimental design. If a field is left for more than 3 years, the necromass becomes too dense for the grassland birds. The pre/postsoil disturbance, grid-sampled approach used in this study was modeled after that used by Brye et al. (2002, 2003, 2004a, 2004c, 2005). Brye et al., (2002), in particular, evaluated the effects of burn frequency on necromass-contained nutrients in a prairie restoration without a nonburned control. We collected samples the day before the burn, to characterize the site under conditions before the first burn of the field, serving as the reference point for comparison of the impact of the burn treatment.

Prescribed Burn

The first prescribed burn of the 13-ha field containing the 0.4-ha study area was conducted on April 4, 2003. The timing of the burn was based on the need to burn before the beginning of the bird nesting season on April 15 and delayed somewhat from several weeks of rain in March. The relative humidity was $\sim 85\%$, the air temperature was 20.6 °C, and the wind speed was 16 to 19 km h⁻¹ at the time of the burn (NCDC, 2003). By visual observation, approximately 90% of the vegetation was effectively burned, though some woody vegetation remained. Grass crowns were not completely consumed but were significantly charred.

Soil Sampling

Soil samples were collected from each of the 25 sample points within the 0.4-ha study area 1 day before and 11 days and 1 year after burning. Ash was still visibly present on much of the soil

surface at the time of the 11-day postburn soil sampling. A short time period was chosen to collect the 11-day postburn samples to minimize changes in the soil due to factors other than the immediate treatment impacts of ash deposition on the soil chemistry, the focus of our hypothesis, such as increased aboveground or below-ground production or increased microbial activity. The 11-day postburn sampling was a compromise between minimizing the elapsed time after the burn and allowing sufficient rainfall to cause some ash infiltration into the soil. This time period is similar to the 7-day postburn sampling used by Ojima (1987) to characterize the effects of burning in the Konza and Aldous prairies of Kansas.

Soil samples were collected with a split corer, 5 cm in diameter and 30 cm long. Soil cores were sectioned into 5 depth intervals: 0 to 2.5, 2.5 to 5, 5 to 10, 10 to 15, and 15 to 20 cm. Since the soil of the Ap horizon, roughly the top 0 to 20 cm, represents that which was influenced the most by the prior agricultural land use history, soil at depths below 20 cm was not collected. Soil samples were air dried for 48 hours and ground and sieved to <2 mm. Two subsamples from each depth interval for each core were analyzed and the results of these replicates were averaged. Replicate ash samples were collected by hand from the soil surface immediately after burning. An additional set of soil cores 5 cm in diameter were collected from the same depth intervals within each of the 25 experimental units after the first growing season after burning and sectioned by depth for soil bulk density determination (Blake and Hartge, 1986).

Soil Physical Analyses

Soil cores collected for bulk density determination were dried for 24 hours at 105 °C and weighed (Blake and Hartge, 1986). Since bulk density has been shown to be unaffected by the burning of warm-season grasslands (Owensby and Wyrill, 1973; Schacht et al., 1996), it was assumed that bulk density measurements after the growing season represented those immediately before and after the prescribed burn. In addition, the five depth intervals sampled at five of the 25 grid points were combined into one composite sample per sample point for particle-size analysis by the 2-hour hydrometer method (Arshad et al., 1996). The sand fraction (i.e., 0.02- to 2-mm diameter range) was not wet sieved to confirm 40-second readings.

Soil Chemical Analyses

Air-dried and sieved soil samples were analyzed in duplicate for pH, organic matter (OM), CEC, and extractable cations in the pre-burn and 11-day postburn samples and for pH and OM in the 1-year postburn samples. All methods followed standard procedures (Sparks, 1996), at times with modifications as given in Recommended Soil Testing Procedures for the Northeastern United States (NRCCST, 1995). Soil samples were analyzed for pH in a 1:1 wt/vol solution of 0.01M calcium chloride (CaCl_2) (Thomas, 1996) to mimic the ionic strength of the natural soil solution. Subsamples of soil ground to pass a 0.4-mm sieve were used to determine OM by loss-on-ignition at 400 °C (Nelson and Sommers, 1996). The subsamples were heated in porcelain crucibles for 24 hours at 105 °C in a drying oven and then for 16 hours at 400 °C in a muffle furnace (Fisher Scientific Isotemp Muffle Furnace 550, Pittsburgh, PA). The difference in weight of the sample after heating at 105 °C and at 400 °C was used to calculate the OM content. Studies of black carbon in soil and sediment samples resulting from vegetation burns indicate that char carbon is not thermally stable at 400 °C (Nguyen et al., 2004). Hence, combustion by loss-on-ignition at 400 °C will be a measure of natural soil organic matter and of char carbon incorporated into the soil after the burn.

Soil CEC was measured by the method of compulsive exchange (Sumner and Miller, 1996) with some minor modifications (NRCCST, 1995). Due to the rather laborious procedure and the likelihood that only the near-surface soil CEC was affected by ash infiltration, soil CEC was only measured on the top three depth intervals (i.e., 0 to 2.5, 2.5 to 5, and 5 to 10 cm). The soil was initially saturated with barium ion (Ba^{2+}) using 0.1M barium chloride (BaCl_2) extracting solution. The soil was then washed three times with 2 mM BaCl_2 to reduce the ionic strength to one appropriate for highly weathered soils (NRCCST, 1995). To compulsively exchange magnesium (Mg^{2+}) for Ba^{2+} and precipitate the released Ba^{2+} , the soils were equilibrated with 0.05M magnesium sulfate (MgSO_4). The final solutions were adjusted to the pH of the rinsed soil solution using 0.05M sulfuric acid and to the ionic strength of the reference solution with 0.1M MgSO_4 or distilled water, if needed. The solutions were filtered through a 0.45- μm filter (Millipore HA)

and acidified to $\text{pH} < 2$ with nitric acid for analysis (Clesceri et al., 1998).

The Mg^{2+} concentrations in the resultant solutions from the compulsive exchange were measured by atomic absorption spectrophotometry (Perkin Elmer Analyst 100, Wellesley, MA). Lanthanum (La^{2+}) was added to standards and diluted samples in a 1:10 volume ratio of La^{2+} to Mg^{2+} to minimize solution matrix interferences according to Clesceri et al. (1998). The amount of Mg^{2+} adsorbed to the soil was calculated from the difference between the amount remaining in solution and the initial amount of Mg^{2+} added. The total CEC was calculated from the total amount of adsorbed Mg^{2+} .

Soil samples were extracted with Mehlich-3, dilute acid extracting solution for labile phosphorus (P) and extractable cations (NRCCST, 1995). Mehlich-3 solution was added in a 1:10 (wt/vol) soil:solution ratio. Samples were placed on a reciprocating shaker at 200 r.p.m. for 5 minutes and then filtered through quantitative paper (Whatman 44). Phosphorus was measured colorimetrically (Genesis Spectronic 20, Spectronic Instruments, Rochester, NY) within 4 hours of the extraction by the ascorbic acid method (Kuo, 1996). Extractable cations [i.e., calcium (Ca, 183.801 nm), magnesium (Mg, 285.213 nm), potassium (K, 766.490 nm), sodium (Na, 589.592 nm), iron (Fe, 259.940 nm), manganese (Mn, 257.610 nm), zinc (Zn, 213.856 nm), and copper (Cu, 324.754 nm)] were measured by inductively coupled argon-plasma (ICAP) spectrophotometry (CIROS CCD ICP, Spectro Analytical Instruments, Inc., Fitchburg, MA).

The soil buffering capacity was measured using a modification of a technique used for Atlantic Coastal Plain soils (Weaver et al., 2004). Soils were sampled before the burn at the same 5 grid points as were used for particle-size analysis and combined across depth increments into one composite sample per sample point. Initial soil pH was measured in six 1:10 (wt/vol) soil water suspensions for each sample point. Saturated calcium hydroxide [$\text{Ca}(\text{OH})_2$] was added to the suspensions in 0.5 mL increments in batch mode, creating a range of 0.5 to 3.0 mL added. After 1 h of continuous stirring to allow for equilibration of the soil suspension, the pH of the solution was recorded. The slope of the linear regression of pH and mmol OH^- added was used to determine the soil buffering capacity, expressed as $\text{cmol} (+) \text{kg}^{-1} \text{pH}^{-1}$.

Ash samples were analyzed for total elemental content. The ash was digested in nitric acid and resultant cations (i.e., Ca, Mg, K, Na, Fe, Mn, Zn, and Cu) were determined by ICAP spectrophotometry.

Data Conversions and Statistical Analysis

All measured elemental concentrations were converted from a mass-basis to a volume-basis using the appropriate measured soil bulk density value to better represent soil elemental contents at the study site.

For soil chemical properties, homogeneity of variance between pre- and postburn samples was evaluated using Levene's test (Minitab 13.31, Minitab Inc., State College, PA). A two-sample *t* test for unpaired samples, with and/or without equal variances, was used to ascertain the effect of the prescribed burn on soil chemical properties at each depth interval sampled (Minitab). Linear correlations were also performed between selected pairs of soil chemical properties with data from all depth intervals (Minitab). In addition, the effect of the prescribed burn on the linear relationship among selected soil chemical properties with data from all depth intervals was determined using analysis of covariance (SAS Version 8.1, SAS Institute, Inc., Cary, NC). Significance was established at the 0.05 level for all statistical analyses. Mean values \pm standard errors are reported.

RESULTS AND DISCUSSION

Soil Physical and Chemical Properties

As was expected due to the cultivated agricultural land-use history, soil bulk density across the study site was relatively uniform at any given depth and tended to increase with depth in the top 20 cm (Fig. 1). Soil bulk density averaged 1.13 ($\text{SE} = 0.03$) g cm^{-3} in the 0 to 2.5 cm depth interval, 1.40 ($\text{SE} = 0.02$) g cm^{-3} in the 2.5 to 5 cm depth interval, and 1.59 ($\text{SE} = 0.03$) g cm^{-3} in the 15 to 20 cm depth interval. Although the study site was mapped as having a loam-textured surface, the average particle-size distribution in the top 20 cm was 73.7% sand, 18.5% silt, and 7.8% clay, which resulted in a slightly coarser, sandy-loam surface texture.

Before burning, the soil was slightly acidic, low in organic matter, cation exchange capacity, and extractable cations (Figs. 2 through 5). The soil buffering capacity in the composite 0 to 20 cm depth interval was $2.0 \text{ cmol} (+) \text{kg}^{-1} \text{pH}^{-1}$,

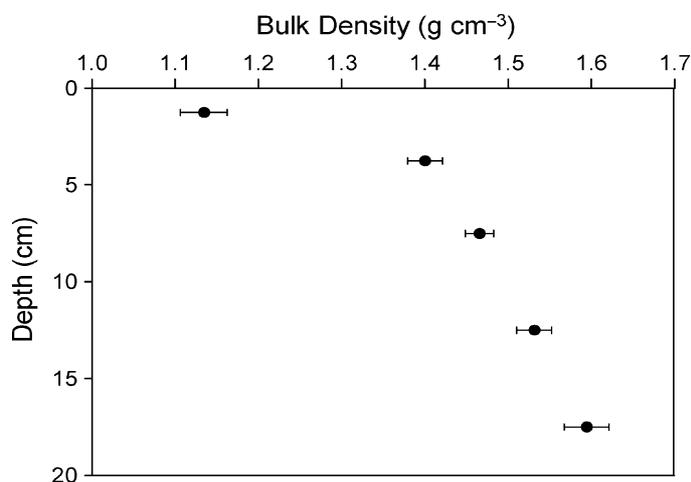


Fig. 1. Mean (\pm standard error) soil bulk density profile ($n = 5$ per depth) representing the 0.4-ha tallgrass prairie study area.

which is comparable to values determined for other Atlantic Coastal Plain soils (Liu et al., 2004). As was expected, numerous strong ($P < 0.01$) correlations existed between many of the measured soil chemical properties before burning (Table 1). Both pre-burn soil OM and CEC were strongly positively correlated with extractable Ca, Mg, and K contents as well as with each other ($r = 0.75$). Extractable Ca was strongly positively correlated with Mg and K ($r = 0.92$ and 0.73 , respectively). Correlations of pH with OM, CEC and the cations were weaker, but still significant.

Effects of First-Time Burning

Soil pH

Despite a relatively large degree of variability in soil pH among individual samples both before and after burning, the average soil pH increased significantly ($P < 0.05$) at all soil depths within the study area by 11 days after the prescribed burn (Fig. 2). Before burning, the average soil pH was 5.72 (SE = 0.06) in the 0 to 2.5 cm depth interval and was 5.39 (SE = 0.09) in the 15 to 20 cm interval. After burning, the average soil pH was 5.94 (SE = 0.05) in the 0 to

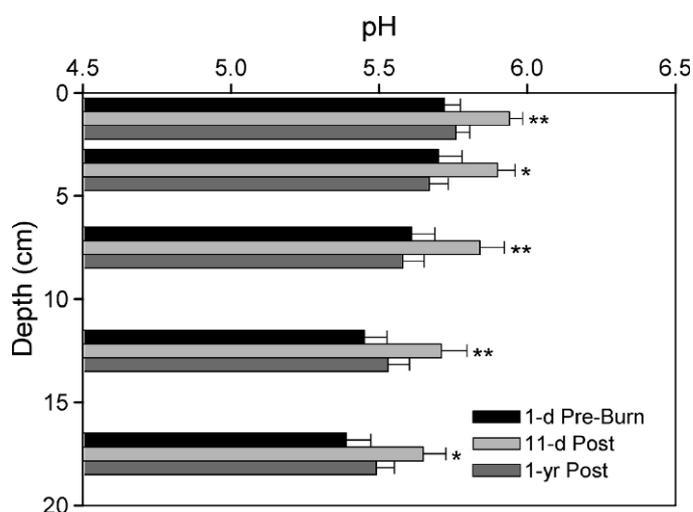


Fig. 2. Mean (\pm standard error) soil pH profile ($n = 25$ per depth) 1 day before (i.e., 1-day pre-burn), 11 days and 1 year after prescribed burning (i.e., 11-days post and 1-year post, respectively). Asterisks denote significant difference from 1-day pre-burn levels at $P < 0.05$ (*) and $P < 0.01$ (**).

TABLE 1

Summary of pre- and post-burn correlations (r) among soil organic matter (OM), cation exchange capacity (CEC), extractable Ca, Mg, and K, and soil pH for all sample depths combined

Soil Variable	CEC	Ca	Mg	K	pH
Pre-burn correlations (r)					
OM	0.865 [§]	0.882 [§]	0.768 [§]	0.708 [§]	0.249 [‡]
CEC		0.946 [§]	0.873 [§]	0.772 [§]	0.406 [§]
Ca			0.924 [§]	0.725 [§]	0.525 [§]
Mg				0.576 [§]	0.627 [§]
K					0.239 [‡]
Post-burn correlations (r)					
OM	0.906 [§]	0.923 [§]	0.824 [§]	0.752 [§]	0.135
CEC		0.942 [§]	0.991 [§]	0.827 [§]	0.276 [†]
Ca			0.919 [§]	0.775 [§]	0.383 [§]
Mg				0.736 [§]	0.496 [§]
K					0.164

Symbols denote significant correlations at $P < 0.05$ [†] $P < 0.01$ [‡] and $P < 0.001$ [§]

2.5 cm depth interval and 5.65 (SE = 0.08) in the 15 to 20 cm interval. Burning did not affect the measured variability associated with soil pH. The increase in pH in each depth interval was similar and averaged 0.23 (SE = 0.01) pH units. After 1 year, soil pH was no longer significantly different than the pre-burn soil pH (Fig. 2).

The relatively low buffering capacity of the highly weathered Ultisol in this study makes it sensitive to basic inputs. Burning of grassland vegetation converts mineral components of the grasses into soluble salts (Daubenmire, 1968)

and, upon contact with water, the salts undergo hydrolysis, which is an alkalinity-producing reaction. The several days of rain that occurred in the 11 days after the prescribed burn would have allowed the base-cation-rich, alkaline ash from the burn to penetrate the near-surface soil at the CRFRC site. A total of 4.63 cm of rain occurred during the 11 days, primarily on days 3 through 8 after the burn (NCDC, 2003). Based on average measured soil bulk densities by depth (Fig. 1), assuming the volumetric soil water content was relatively high (i.e., ~ 33%) from the rain, and assuming little or no runoff occurred, considering the well-drained soils with 2 to 5% slope and a low rainfall intensity (NCDC, 2003), the 4.63 cm of rain between burning and 11-day postburn soil sample collection would have penetrated to a depth of nearly 15 cm if the soil layers above were saturated. However, it is very likely that given the sandy-loam soil texture, the upper soil layers did not saturate and the rainwater penetrated to an even greater depth than 15 cm. Some ash and ash-dissolution products probably penetrated to a similar depth into the soil as the rainwater. At each depth sampled to 20 cm, the increase in pH was similar (Fig. 2), supporting the calculation that the water had in fact infiltrated to at least 20 cm.

Increases in soil pH after first-time burns of grasslands have only been reported in two other studies. In a silt-loam soil of the Hayden Prairie in Iowa, Ehrenreich and Aikman (1963) reported a 0.9 pH unit increase in the surface 2 cm

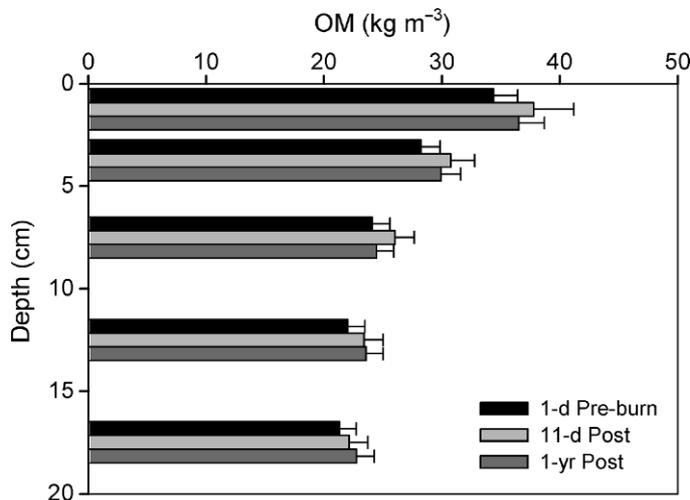


Fig. 3. Mean (\pm standard error) organic matter (OM) profile ($n = 25$ per depth) 1 day before (i.e., 1-day pre-burn), 11 days and 1 year after prescribed burning (i.e., 11-days post and 1-year post, respectively).

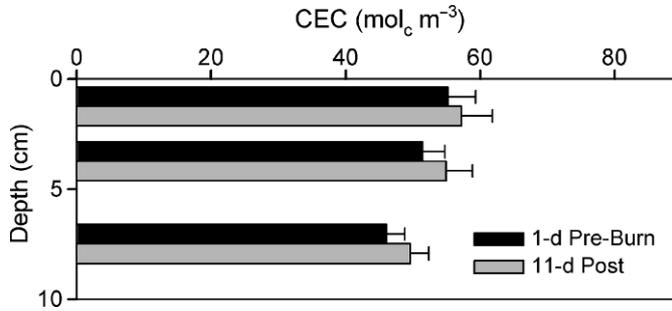


Fig. 4. Mean (\pm standard error) cation exchange capacity (CEC) profile ($n = 25$ per depth) 1 day before (i.e., 1-day pre-burn) and 11 days after prescribed burning (i.e., 11 days post).

several months after first-time burning in the spring of the year. As was the case at the CRFRC site, they found that the pH increase dissipated by 1 year after the burn. Similar to

the pH change observed in this study, Picone et al. (2003) reported a soil pH increase of 0.38 units in the surface 5 cm, 1 day after winter burning a grassland composed of warm-season

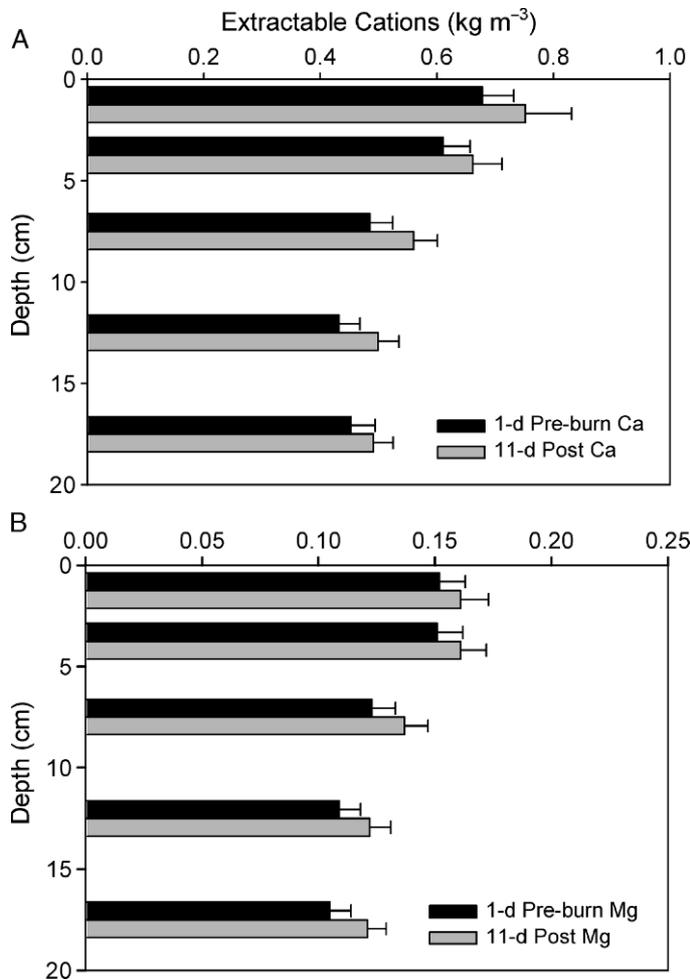


Fig. 5. Mean (\pm standard error) extractable Ca (A) and Mg (B) profiles ($n = 25$ per depth) 1 day before (i.e., 1-day pre-burn) and 11 days after prescribed burning (i.e., 11 days post).

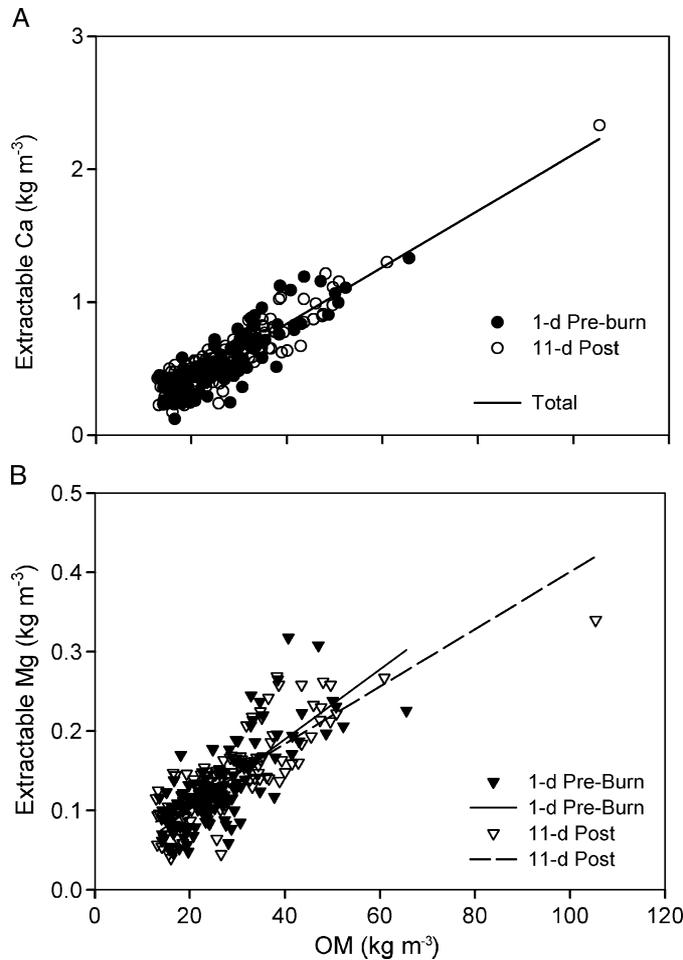


Fig. 6. Relationships between extractable Ca (A) and Mg (B) and organic matter (OM) in the top 20 cm before (i.e., 1-day pre-burn) and/or 11 days after prescribed burning (i.e., 11-days post). Regression equations are as follows: extractable Ca = $0.0213 \cdot \text{OM} - 0.012$ ($P < 0.001$, $r^2 = 0.82$); extractable Mg(1-day pre-burn) = $0.0044 \cdot \text{OM}(1\text{-day pre-burn}) + 0.014$ ($P < 0.001$, $r^2 = 0.59$); extractable Mg(11-day post) = $0.0036 \cdot \text{OM}(11\text{-days post}) + 0.039$ ($P < 0.001$, $r^2 = 0.68$).

grasses on a poorly-drained Mollisol in Argentina. The pH increase was attributed in part to ash deposition to the soil surface. On the other hand, no pH changes were found at Big Meadows Shenandoah Park Virginia (Christensen, 1976) and in tobosagrass fields on Vertisol soils in Texas (Ueckert et al., 1978) several months after the burns.

Interestingly, soil pH increases comparable to those of the CRFRC site have also been measured at a few sites in response to repeated annual burning of grasslands. In an oak savanna restoration in Minnesota on acidic sandy-loam soils, which are similar to those of this study, soil pH increases in the top 10 cm (0.3 to 0.9 pH units depending on burn frequency) were found

several months after prescribed burns (Tester, 1989). At the Aldous Prairie in Kansas, which had been burned annually for almost 50 years, a 0.24 unit increase in soil pH in the top 20 cm was measured 1 week after a Spring burn (Ojima, 1987). In another tallgrass prairie site in Kansas, subjected to 38 years of annual burning, Owensby and Wyrill (1973) reported soil pH increases of up to 0.25 units in each 7.6 cm depth increment, sampled to a depth of 91 cm, 5 to 8 months after winter, early-spring, and late-spring burning. Ehrenreich and Aikman (1963) also measured a significant increase in soil pH, 0.3 pH units several months after the third of 3 years of annual burning Hayden Prairie in Iowa. In contrast, repeated burning of a portion

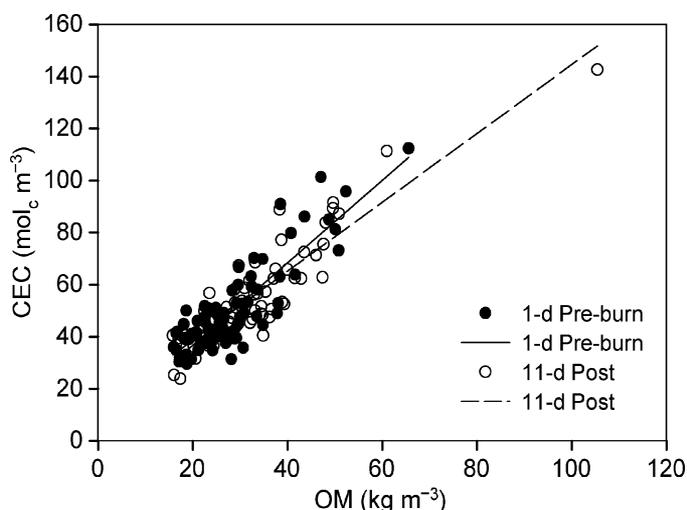


Fig. 7. Relationship between cation exchange capacity (CEC) and organic matter (OM) in the top 20 cm before (i.e., 1-day pre-burn) and 11 days after prescribed burning (i.e., 11-day post). Regression equations are as follows: $CEC(1\text{-day pre-burn}) = 1.58 \cdot OM(1\text{-day pre-burn}) + 8.68$ ($P < 0.001$, $r^2 = 0.75$); and $CEC(11\text{-days post}) = 1.32 \cdot OM(11\text{-days post}) + 8.68$ ($P < 0.001$, $r^2 = 0.82$).

of the Konza Prairie of Kansas (Ojima, 1987) and a grassland on an acid sandy loam in Connecticut (Niering and Dreyer, 1989) showed no change in soil pH.

Despite differences in soil and climate, soil pH increases reported at sites subjected to both first-time or repeated burning were comparable or slightly greater than the increases found at the CRFRC site. A net impact of site-specific conditions of burn intensity and fuel load, ash incorporation and rainfall, as well as soil and climate conditions could account for the similarity between results for soil pH in these studies. The CRFRC soil has a low buffering capacity and hence is more sensitive to basic inputs, however the net amount of ash incorporated into the surface soil and the amount dissolved could have been less than the amount at the other sites, due to lower initial litter amounts or different climatic conditions. The litter amount of a grassland field next to the study site at CRFRC, which was destructively sampled after 3 years of growth, was 1191 kg ha^{-1} . In contrast, sites with a higher fuel load, but a higher buffer capacity could have similar pH changes. Fuel loads at the Hayden Prairie of Ehrenreich and Aikman (1963) were 2236 kg ha^{-1} at the repetitively burned site and 4942 kg ha^{-1} in the site burned for the first time, 2 to 4 times the amount at the CRFRC site. Ehrenreich and Aikman attributed the lower pH increases in response to repeated burning at

their Iowa Prairie to the smaller fuel load. The net effect of fuel loads and soil buffering appears to be a very important component of the short-term impact of prescribed burning on soil pH.

The length of time before which pH changes dissipate will depend on the soil buffering capacity and the amount of leaching. Owensby and Wyrill (1973) attributed pH increases throughout the soil profile to annual deposition of salts from ash replacing those leached from surface layers over the previous year. The results of the CRFRC site suggest a similar soil pH profile can develop in response to short-term leaching of ash, but which then dissipates within 1 year. The similarity in reported pH changes after first-time burning and repeated burning could be explained by the dissipation of the soil pH from one burn to the next. Although the short-term increase in soil pH was significant at the CRFRC site, the small magnitude of the increase probably resulted in no significant increase in nutrient availability and hence no impact on plant growth, especially as compared with other major impacts of the burn such as increasing light availability due to removal of plant litter.

Soil Organic Matter, Cation Exchange Capacity, and Extractable Cations

Numerical increases in OM, CEC, extractable Ca and Mg were observed by 11 days after

the prescribed burn throughout the top 20 cm, but, in contrast to soil pH, the increases, assumed to be a result of the prescribed burn, were not statistically significant (Figs. 3 through 5). Similar to soil pH, burning did not affect the measured variability of these properties at each depth. The prescribed burn did not significantly affect extractable K, Na, Fe, Mn, Zn, Cu, S, and P (data not shown). By 1 year after burning, similar to soil pH, OM did not differ from pre-burn values (Fig. 3).

Statistically significant changes in the relationships between OM, CEC, and soil properties did occur following 11 d after burning, which support the numerical increases, and which may be a sign of future change with repetitive burning. For chemical properties that were linearly correlated with pH before and 11-d after burning, namely CEC and extractable Ca and Mg (Table 1), the relationship did not change significantly after the burn, as indicated by statistically equivalent slopes and intercepts of the regression equations before and after the burn. The increase in soil pH detected must have been matched by increases in exchangeable Ca and Mg and CEC, as suggested in the depth profiles, in order to maintain these same relationships. An increase in OM is supported by the statistically equivalent relationship between OM and Ca before and 11 d after burning (Fig. 6A). Either no change occurred with OM and Ca due to burning, or, more likely, considering the increase in Ca suggested by the relationship with pH and

the numerical increases in OM and Ca in the depth profiles, both increased in a manner to maintain the same relationship.

Another line of evidence indicating that burning increased OM, CEC, and extractable cations in the soil is the 11-day postburn change in the slope of the regression equations characterizing the relationships between CEC and OM (Fig. 7), Mg and OM (Fig. 6B), and Mg and Ca (Fig. 8). The slope ($P = 0.046$) characterizing the linear relationship between CEC and OM decreased 11 days after burning (Fig. 7). In addition, the linear relationship between extractable Mg with OM changed (Fig. 6B), in contrast to that between Ca and OM (Fig. 6A); the slope decreased ($P = 0.024$) and the intercept increased ($P = 0.046$) after burning. Moreover, the linear relationship between extractable Mg with Ca the slope decreased ($P = 0.006$) and the intercept increased ($P = 0.009$) (Fig. 8) 11 days after burning. In these cases, the change in relationships could have resulted from an increase or decrease in one or both of the properties. The changes are most likely due to increases in all of the properties, considering the numerical increases in the depth profiles and the regression analyses discussed above, with differing magnitudes that led to changes in the relationships between the properties.

Infiltration of ash into the soil after the burn probably caused observed trends in OM, as well as CEC, extractable Ca and extractable Mg

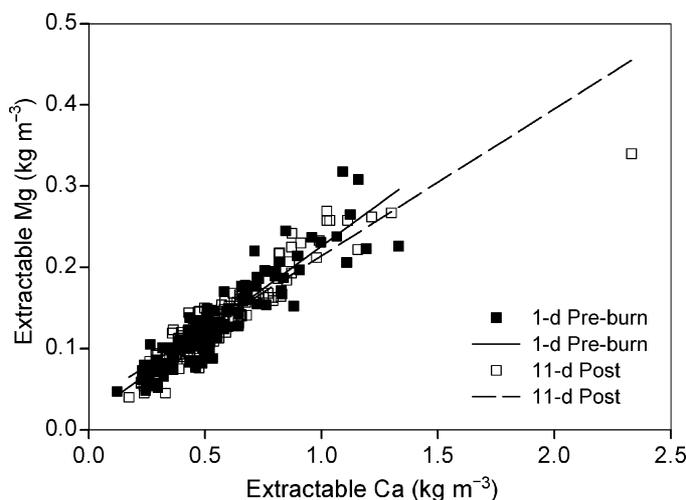


Fig. 8. Relationships between extractable Mg and Ca in the top 20 cm before (i.e., 1-day pre-burn) and/or 11 days after prescribed burning (i.e., 11-days post). Regression equation is as follows: extractable Mg(1-day pre-burn) = $0.210 \times \text{extractable Ca}(1\text{-day pre-burn}) + 0.016$ ($P < 0.001$, $r^2 = 0.85$); extractable Mg(11-days post) = $0.181 \times \text{extractable Ca}(11\text{-days post}) + 0.033$ ($P < 0.001$, $r^2 = 0.85$).

shortly after burning. In the short time period of 11 days, it is unlikely that OM in the soil increased due to increased root growth or increased microbial activity. Rather, the increases observed are probably due to ash deposited on the soil surface that had slowly infiltrated into the soil as a result of the precipitation events. The ash, a form of black carbon, in particular char carbon, is measured by the loss-on-ignition method used to quantify soil OM. Even a small influx of ash can provide a large input of OM relative to the low background levels of soil OM in the soil at the CRFRC site (Fig. 3), in contrast to sites with much higher natural soil OM levels.

Numerical increases in soil CEC and OM suggest a contribution of the ash to CEC levels. Soil CEC and OM were strongly correlated both before the burn and 11 days after burning (Table 1; Fig. 7). Studies of forest soils mixed with a large percentage of ash from hardwoods and pines indicate a direct correlation between an increase in ash in the soil and an increase in CEC (Tyron, 1948). Laboratory studies of black carbon, indicate the presence of negative functional groups amidst the mostly nonpolar character of black carbon molecules (Skjemstad et al., 2002; Yang and Sheng, 2003), which can be a source of cation exchange sites. The smaller slope of the 11-day postburn linear relationship between OM and CEC (Fig. 7) suggests that the ash does not provide as many CEC sites as does natural soil OM. The similarity of the relationship between pH and CEC before and 11 days after burning, although only weakly correlated, suggests that the increase in pH increased CEC values, as would occur by deprotonation of exchange sites.

Increases in soil OM or CEC after burning have not been commonly reported in grasslands, most likely due to relatively high initial levels of both parameters. Increases in OM in soil have been reported in a site of grasses under longleaf pines in Mississippi (Greene, 1935) and in the Kansas prairie site of Owensby and Wyrill (1983), both of which have relatively low levels of organic matter in the soils and which have been repeatedly burned. Ueckert et al. (1978) reported an increase in soil organic carbon (OC) in tobosagrass fields on Vertisols after several months after a first-time burn. Ueckert et al. (1978) attributed the increase to surface cracking during a dry period, such that relatively large amounts of ash entered the soil preferentially during subsequent rainfall events rather than by

infiltration through matrix flow, as would be the case at the CRFRC site. In contrast, Ehrenreich and Aikman (1963) reported no significant change in soil OM following first-time burning, but the soil OM was five times greater than that of the soil of the CRFRC site in this study. Picone et al. (2003) measured a decrease in soil OC 7 d after the burn, attributing the loss to combustion in the surface soil; no change in soil CEC was detected. Similarly, soil CEC did not change in a long-term study in the annually burned Konza Prairie, most likely due to high initial CEC levels (J. Blair, personal communication, 2004).

The increase in extractable cations observed at the CRFRC site 11 days postburn probably resulted from the production of cations during ash dissolution and resultant salt hydrolysis, with storage of a portion of these cations on soil cation exchange sites. The primary cations in the ash were Ca (1.12%), Mg (0.17%) and K (0.24%); Na, Fe, Mn, Zn, and Cu concentrations were each < 0.1%. Although the relationship between Ca and OM did not change in response to burning (Fig. 6A), the slopes characterizing the linear relationships between OM and Mg decreased (Fig. 6B). Since the Ca concentration in the ash itself was more than five times higher than that of Mg, dissolution of the ash probably increased Ca more than Mg. The change in relationship is also reflected in the decrease in slope characterizing the linear relationship of Ca and Mg 11 days postburn (Fig. 8). The correlations between extractable cations and CEC in pre-burn and 11-day postburn samples (Table 1) suggest that areas that have greater CEC retain more of the cations released from the ash.

Reports of increases in extractable soil cation concentrations due to burning are few. Owensby and Wyrill (1973) reported that cation concentrations increased in all depth intervals sampled several months following burning, along with increases in pH and soil OM. They attributed the parallel pH and cation increases to dissolution of soluble salts from ash deposition. The cation increases were greater, even against higher background cation contents, most likely due to a higher fuel load or greater ash incorporation. The comparable soil pH increase to that at the CRFRC site, however, could be due to a greater buffering capacity of the prairie soils. In contrast, no change in exchangeable cations was observed at the annually burned Konza Prairie, which also had higher background

soil cation contents than this study (J. Blair, personal communication, 2004). On the other hand, Christianson (1976) detected increases in soil cations two months after first-time burning of grasses, sedges and herbs in acidic soils of Big Meadows Shenandoah National Park. In a laboratory study, Lloyd (1971) added ash formed from burning herbaceous vegetation to calcareous and brown-earth soils of England and observed an increase in exchangeable K, but not Ca. However, Lloyd (1972) did not observe a change in exchangeable K or Ca due to ash addition to field plots. Differences found between sites could not only be due to background cation contents, but the degree of leaching of the soluble cations.

The short-term effects of burning on soil chemical properties of a grassland restoration on a highly weathered Ultisol in the MidAtlantic Coastal Plain suggest that burning will probably serve as an important means to recycle organic matter and major cations and to increase soil pH and CEC early in the growing season. The similarity in the magnitude of the changes between these results and those of other studies, in particular for soil pH, indicates the combined impact of burning conditions, weather after burning, soil properties, and climate in determining the impact of burning on soil chemistry. The dissipation of short-term changes after 1 year suggests that long-term changes in the soil chemistry of a grassland restoration on a highly weathered Ultisol in the MidAtlantic Coastal Plain will probably only occur with repeated burning and with greater retention of ash and ash-dissolution products. The balance between short-term changes and the long-term buffering capacity of a typical, highly weathered Ultisol in the MidAtlantic Coastal Plain suggests that grassland restoration can be successful and that sustainable long-term ecological productivity probably can be achieved.

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