

# Interacting effects of wildfire severity and liming on nutrient cycling in a southern Appalachian wilderness area

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Received: 31 May 2012 / Accepted: 3 August 2012 / Published online: 26 August 2012  
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## Abstract

**Aims** Wilderness and other natural areas are threatened by large-scale disturbances (e.g., wildfire), air pollution, climate change, exotic diseases or pests, and a combination of these stress factors (i.e., stress complexes). Linville Gorge Wilderness (LGW) is one example of a high elevation wilderness in the southern Appalachian region that has been subject to stress complexes including chronic acidic deposition and several wildfires, varying in intensity and extent. Soils in LGW are inherently acidic with low base cation concentrations and decades of acidic deposition have contributed to low pH, based saturation, and Ca:Al ratio. We hypothesized that wildfires that occurred in LGW followed by liming burned areas would

accelerate the restoration of acidic, nutrient depleted soils. Because soils at LGW had extremely low concentrations of exchangeable  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$  dolomitic lime was applied to further boost these cations. We evaluated the effectiveness of dolomitic lime application in restoring exchangeable  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$  and subsequently increasing pH and Ca:Al ratio of soils and making Ca and Mg available to recovering vegetation.

**Methods** Five treatment areas were established: severely burned twice (2000 & 2007) with dolomitic lime application (2xSBL); moderately burned twice with lime application (2xMBL); severely burned twice, unlimed (2xSB); moderately burned once (2000), unlimed (1xMB); and a reference area (REF; unburned, unlimed). In 2008 and 2009, we measured overstory, understory, and ground-layer vegetation; forest floor mass and nutrients; and soil and soil solution chemistry within each treatment area.

**Results** All wildfire burned sites experienced substantial overstory mortality. However, understory biomass doubled between sample years on the most recently burned sites due to the rapid regrowth of ericaceous shrubs and prolific sprouting of deciduous trees. Burning followed by lime application (2xSBL and 2xMBL) significantly increased shallow soil solution  $\text{NO}_3\text{-N}$ , but we found no soil solution  $\text{NO}_3\text{-N}$  response to burning alone (2xSB and 1xMB). Surface soil base saturation and exchangeable  $\text{Ca}^{2+}$  were significantly affected by liming;  $\text{Ca}^{2+}$  concentrations were greater on 2xMBL and 2xSBL than 2xSB, 1xMB and REF.

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Responsible Editor: Zucong Cai.

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There was a smaller difference due to moderate burning along with greater soil  $\text{Ca}^{2+}$  on 1xMB compared to REF, but no difference between 2xSB and REF. Surface and subsurface soil exchangeable  $\text{Al}^{3+}$  were lower on 2xSBL than 2xSB, 2xMBL, 1xMB, and REF. Liming decreased soil acidity somewhat as surface soil pH was higher on the two burned sites with lime (pH=3.8) compared to 2xSB without lime (pH=3.6). **Conclusions** Liming resulted in decreased soil  $\text{Al}^{3+}$  on 2xSBL coupled with increased soil  $\text{Ca}^{2+}$  on both 2xSBL and 2xMBL, which improved soil Ca/Al ratios. However, the soil Ca/Al ratio response was transitory, as exchangeable  $\text{Al}^{3+}$  increased and Ca/Al ratio decreased over time. Higher lime application rates may be necessary to obtain a substantial and longer-term improvement of cation-depleted soils at LGW.

**Keywords** Dolomitic lime · Exchangeable base cations · Nitrogen · Calcium · Aluminum · Fire severity · Forest floor · Soil solution nutrients · Acidic soils

## Introduction

Wilderness and other natural areas are threatened by large-scale disturbances (e.g., wildfire), air pollution, climate change, exotic diseases or pests, and a combination of these stress factors (i.e., stress complexes). Potentially sensitive forest ecosystems to these disturbance and stress complexes are found throughout the southern Appalachian region, particularly at high elevation. Linville Gorge Wilderness (LGW) is an example of a high elevation area in western North Carolina that has been subject to chronic acidic deposition (Elliott et al. 2008) and several wildfires, varying in intensity and extent (Newell and Peet 1995; Wimberly and Reilly 2007).

Wildfires are often landscape scale disturbances that have the potential to significantly impact hydrologic processes such as surface runoff, sediment yield, and sediment and nutrient transport to streams (Debano et al. 1998; Covert et al. 2005) and biogeochemical processes by altering pools and fluxes of carbon (C), nitrogen (N), phosphorus (P), calcium (Ca), magnesium (Mg), and potassium (K) (Knoepp et al. 2005). The typical impact of wildfire is an immediate change in vegetative cover and structure, forest floor surface, physical and chemical properties of the soil, followed by mid- and long-term changes in

above- and belowground biological pools and nutrient cycling processes. However, the magnitude and duration of these responses depend on the interactions among fire severity, post-fire precipitation regime, topography, soil characteristics, and vegetative recovery rate (Robichaud 2005). For example, a high severity fire could increase soil cations and pH and decrease soil organic matter, total N, and forest floor mass (Certini 2005; Knoepp et al. 2005). Nutrient responses are also impacted by changes in vegetation and forest floor, as well as changes in biological processes that regulate nutrient cycling during fire recovery (Knoepp et al. 2005).

In addition to wildfire disturbances, previous research in LGW reported acidic soils that are low in weatherable minerals, and with extremely low concentrations of exchangeable base cations, Ca, Mg, and K (Elliott et al. 2008). These low levels have been attributed to prolonged base cation leaching and accumulation in vegetation resulting in soils that have been depleted of Ca, Mg, and K. The sensitivity of LGW to acidic deposition was related to inherent soil factors (e.g., low pH, low Ca, K, and Mg concentrations, and low effective CEC) that reduce the soil's ability to neutralize acidic deposition from base-poor parent materials. Acidic deposition has contributed to declining availability of Ca, Mg, and K in the soils of the acid-sensitive forest ecosystem at LGW by leaching these nutrients from soil in the primary rooting zone (Elliott et al. 2008). Removal of base cations can also result in the mobilization of aluminum (Al) in soils affecting soil solution and drainage waters (NAPAP 2005) as well as plant root growth. High aluminum concentrations have been shown to reduce root growth and vegetation productivity partly due to alterations of soil calcium and magnesium availability (Joslin et al. 1992). Soil Ca replenishment through weathering is not likely because of the base-poor igneous and metamorphic parent material (Velbel 1992). Additionally, atmospheric inputs of Ca through deposition are low ( $<0.8 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ). Therefore, sustainability of forest productivity at LGW without Ca amendments is clearly in question (Elliott et al. 2008).

The purpose of this study was to evaluate the effectiveness of dolomitic lime application in restoring soil exchangeable  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$  concentrations in LGW. We hypothesized that the severe wildfires that occurred in LGW in 2000 and 2007 altered ecosystem nutrient availability by: (1) adding Ca via the 'ash-bed-effect' in which calcium, the dominant cation

found in ash, moves into the surface soil and is retained on cation exchange sites; (2) increasing soil pH and percent base saturation; (3) decreasing forest floor mass and nutrients; and (4) increasing soil nutrient cycling rates. We also hypothesized that liming these burned sites would accelerate the restoration of these acidic, nutrient depleted soils by: (1) adding base cations; (2) balancing soil pH; (3) reducing soil and soil solution Al; and subsequently, (4) make Ca and Mg available to the recovering vegetation

## Methods

### Study area

Linville Gorge Wilderness (LGW) is a 4,390 ha area in the Pisgah National Forest, Burke County, NC. Latitude ranges from 35°49'30" to 35°57'30" and longitude ranges from 81°55'30" to 81°52'30". Elevations range from 426 m at the bottom of the gorge to 1,250 m on upper ridges. Mean annual precipitation is 145 cm and mean annual temperature is 9.4 °C (Linville Falls, NC, National Climatic Database: [www.ncdc.noaa.gov](http://www.ncdc.noaa.gov)). LGW is within the Blue Ridge Geologic Province and soils are derived from high-grade metamorphosed sedimentary rocks, which are covered by unconsolidated Quaternary-aged colluvial and alluvial deposits (Lesure et al. 1977). The mica gneiss and lower quartzite parent materials at LGW results in the formation of soils with low Ca, Mg, and K that are potentially sensitive to acid deposition. More detailed descriptions of the vegetation, geology, and soils in this wilderness area can be found in Newell and Peet (1995).

Our study areas were confined to the upper ridges of LGW between 1,150 and 1,200 m elevations on relatively flat terrain (0–10 % slopes). Soils are in the Soco-Ditney complex, classified as coarse-loamy, mixed, mesic, Typic Dystrudepts (Soil Survey Staff, NRCS 2012). The vegetation on these upper ridges is classified as a pine/oak/heath community, which is comprised of *Pinus rigida* (Mill.), *P. pungens* (Lamb.), *P. virginiana* (Mill.), *Quercus montana* (Willd.), *Q. coccinea* (Münch.), and *Acer rubrum* (L.). Ericaceous (heath) shrubs, *Kalmia latifolia* (L.) and *Vaccinium* spp., occupy the midstory canopy.

In November 2000, the Brushy Ridge and Chimney wildfires burned 4,000 ha in and around LGW

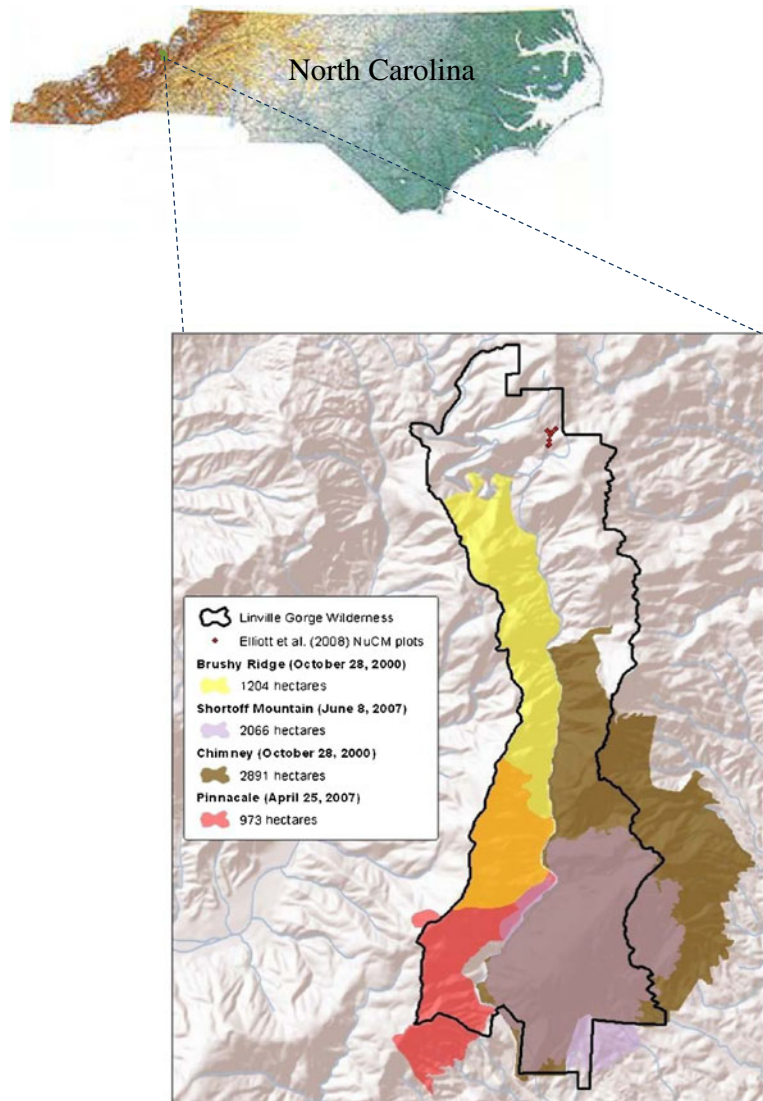
(Fig. 1); the fire ranged in severity across the northern portion of LGW from low elevation coves to higher elevation ridges and bluffs (Reilly et al. 2006; Wimberly and Reilly 2007). In May 2007, two fires occurred in the southern portion of LGW, the Pinnacle wildfire burned 973 ha of the west-side of the gorge and the Shortoff wildfire burned 2,030 ha of the east-side. A large portion of the Pinnacle fire overlapped the area burned in the 2000 Brushy Ridge wildfire resulting in much of the central section of LGW being burned twice in less than seven years (Fig. 1). Several low-intensity surface fires have also occurred, but their frequency and extent are not well documented. For example, a widespread surface fire occurred in the early 1950s, but the severity and extent of this fire was not recorded.

### Experimental design

Five treatment areas, ranging from 8 to 10 ha in size, were located on the west side of the Linville River in the southern portion of LGW and are similar in elevation, slope, soils, and pre-wildfire vegetation composition. The treatment areas were: severely burned twice (2000 & 2007) with dolomitic lime application in 2007 (2xSBL); moderately burned twice (2000 & 2007) with lime application in 2007 (2xMBL); severely burned twice (2000 & 2007), unlimed (2xSB); moderately burned once (2000), unlimed (1xMB); and a reference area (REF; unburned, unlimed). Severely burned areas were characterized by substantial overstory and understory mortality (i.e., crown fire) and minimal forest floor layer present (i.e., Oi+Oe consumed and only 2–3 cm of Oa layer remaining). Moderately burned areas were characterized by some overstory mortality (<30 %), substantial understory mortality, and most of the Oa layer (80–90 %) remaining intact. The most recent wildfire occurred in May 2007. Dolomitic lime was applied in October 2007. In March 2008, five 20-m×20-m permanent plots were established within each treatment area to measure overstory, understory, and ground-layer vegetation; forest floor mass and nutrients; and soil and soil solution chemistry.

For the 2xSBL and 2xMBL sites, dolomitic lime (55 % CaCO<sub>3</sub> and 40 % MgCO<sub>3</sub>, particle size 1–10 mm) was aerially applied at a rate of 1,120 kg ha<sup>-1</sup>. The lime was applied as an operational, management activity by the USDA Forest Service, Pisgah National Forest and no lime collectors were deployed to measure

**Fig. 1** Topographic map of Linville Gorge Wilderness in western North Carolina with locations of the 2000 and 2007 wildfires (GIS by: W.A. Jackson, National Forests of North Carolina, USDA Forest Service)



the plot-level application rate across the study area. The actual application rate likely varied over the treatment areas (2xSBL and 2xMBL), which would make it more difficult to detect significant differences among treatments. Hence, our results may provide a more conservative interpretation of responses and should be viewed in the context of large-scale operational applications.

For our study, where wildfires are an unplanned, landscape scale phenomenon, true treatment replication was not possible (Hargrove and Pickering 1992). Wildfires of this extent and magnitude are uncommon in the southern Appalachians and finding additional comparable sites (i.e., ridge-top fire followed by lime application) within the region was not possible. Plots

within each treatment area were replicated and we discuss only the statistical differences among our five areas, thus the data are interpreted with regard to the limitations of the experimental design (Oksanen 2001), such that our study represents a case study. Hurlbert (1984, 2004) himself recognized valid scientific contributions where replication was impossible, particularly in impact assessment studies.

#### Aboveground biomass and nutrients

Vegetation was measured in July (the time of peak biomass and full leaf expansion) 2008 and 2009. We measured all vascular plants in each permanent plot



using a nested plot design (overstory 400 m<sup>2</sup>, understory 25 m<sup>2</sup>, and ground-layer 4.0 m<sup>2</sup>). Vegetation was measured by layer: the overstory layer included all trees  $\geq 5.0$ -cm diameter at breast height (DBH, 1.37 m above ground); the understory layer included all woody stems  $< 5.0$ -cm DBH and  $\geq 0.5$  m height; the ground-layer included all woody species with stems  $< 0.5$ -m height and all herbaceous species.

Diameter of all overstory trees was measured to the nearest 0.1 cm and recorded by species in every plot. In the understory layer, basal diameter of trees and shrubs was measured to the nearest 0.1 cm and recorded by species in a 5.0-m $\times$ 5.0-m subplot located in the southeast corner of each 20-m $\times$ 20-m plot. Ground-layer vegetation was clipped in one 1.0 m<sup>2</sup> subplot per plot to estimate total biomass and nutrient concentrations. Ground-layer materials were placed in paper bags and transported to the laboratory for processing.

We estimated biomass of the overstory, understory, and forest floor for each plot at each site. To estimate aboveground biomass, tree diameter measurements were converted to biomass using species-specific allometric equations from Martin et al. (1998) for the hardwoods and Jenkins et al. (2003) for the pines. Understory tree and shrub measurements were converted to biomass using species-specific allometric equations from Boring and Swank (1986).

Four forest floor samples were collected in November 2008 (after leaf fall was complete) within each plot using a 0.3- $\times$ 0.3-m wooden sampling frame. Material within the 0.09-m<sup>2</sup> quadrat was separated into two components: litter (Oi) and fermentation plus humus (Oe+Oa). Small wood within the sampling frame was cut using pruning shears, and forest floor was removed by component after cutting along the inside of the sampling frame with a knife. Forest floor materials were placed in paper bags and transported to the laboratory for processing.

Foliage and wood samples were collected from overstory trees after full leaf expansion in July 2008 and 2009. On each plot within the pine/oak/heath community, samples were collected from the mid-crown of the four most dominant deciduous species (oaks [*Q. coccinea* and *Q. montana*], *Nyssa sylvatica* (Marsh.), *Oxydendrum arboretum* (L. De Candolle), and *A. rubrum*) and the evergreen species (*K. latifolia*). Forest floor, foliage, and ground-layer samples were dried at least 72 h at 60 °C, to a constant weight,

weighed to the nearest 0.1 g, and ground in a Wiley mill to pass through a 1-mm sieve. Loss-on-ignition (Nelson and Sommers 1996) was used to determine the ash-free weight of the Oa layer. This procedure consisted of incinerating a 0.5 g sample of forest floor for 12 h in a muffle furnace at 450 °C and then calculating by weight difference the organic and mineral fractions of the sample. Plant tissue and forest floor samples were analyzed for total carbon (C) and total nitrogen (N) with a Elementar Flash EA 1112 series (Thermo Scientific, Waltham, MA). Total phosphorus (P) and cations (K, Mg, Ca, and Al) were determined by ashing 0.5 g samples at 500 °C for 4 h and dissolving in 2.2 M nitric acid followed by analysis using a Jobin Yvon Ultima Inductively Coupled Plasma Spectrometer (Horiba Inc., Edison, NJ) at the Coweeta Hydrologic Laboratory using procedures outlined by Brown et al. (2009).

#### Soil and soil solution chemistry

Soil samples were collected in April and July of 2008 and 2009. Composite soil samples were collected by depth from all plots using an Oakfield soil probe. Composite samples consisted of 24–32 individual samples per plot. We collected both surface (0–10 cm depth) and subsurface (10–30 cm) soils. Within 2 h of collection, soils were mixed thoroughly, sieved to  $< 6$  mm, and a subsample (approximately 10 g) of soil was added to a pre-weighed 125 ml polyethylene bottles containing 50 ml 2 M KCl. The bottles plus soil were kept cool until returning to the laboratory and then stored at 4 °C. Upon returning to the laboratory bottles plus soil were weighed to determine the actual weight of soil extracted. Soil plus KCl samples were shaken and allowed to settle overnight (refrigerated); 15 ml of the clear KCl was pipetted into a sample tube and was analyzed for NO<sub>3</sub>-N and NH<sub>4</sub>-N on an Alpkem model 3590 autoanalyzer (Alpkem Corporation, College Station, TX) using alkaline phenol (USEPA 1983a) and cadmium reduction (USEPA 1983b) techniques, respectively. A subsample of soil ( $\sim 20$  g) was dried at 105 °C for  $> 12$  h to obtain oven-dry weight. All soil N data are reported on an oven dry weight basis. The remaining soil samples were air-dried and sieved to  $< 2$  mm for analyses of carbon (C), nitrogen (N), and exchangeable cations (Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>, and Al<sup>3+</sup>). Total C and N were determined by combustion as described above. Exchangeable

cations were extracted from 10 g of soil on a mechanical vacuum soil extractor using 50 ml of 1 M  $\text{NH}_4\text{Cl}$ . Solution concentrations of  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and  $\text{Al}^{3+}$  were determined with a Jobin Yvon Ultima Inductively Coupled Plasma Spectrometer (Horiba Inc., Edison, NJ) (Standard Methods 2000). Following the initial 12-h extraction excess  $\text{NH}_4\text{Cl}$  was removed from the soil interstitial spaces with 95 % EtOH.  $\text{NH}_4^+\text{-N}$  on the soil exchange sites was then extracted with 2 M KCl as a measure of effective soil cation exchange capacity (ECEC). The  $\text{NH}_4^+\text{-N}$  concentration in the KCl solution was determined using the alkaline phenol method described above. Soil pH was determined in a 1:1 soil to 0.01 M  $\text{CaCl}_2$  solution slurry using an Orion portable pH meter (model 250A) with a Thermo Scientific Orion pH probe. We determined dilute hydrochloric and sulfuric acid extractable  $\text{PO}_4^{2-}\text{-P}$  (Olsen and Sommers 1982; Brown et al. 2009) colorimetrically using ascorbic acid on an Alpkem autoanalyzer (Alpkem Corp., Wilsonville, OR). All soil nutrient data are reported on an oven dry weight basis. Base saturation (BS) was calculated as the percentage of exchangeable “base” cations in the ECEC:  $\text{BS} = 100 \times (\text{Mg}^{2+} + \text{Ca}^{2+} + \text{K}^+ + \text{Na}^+)/\text{ECEC}$ .

Soil solution samples were obtained by installing falling tension porous cup lysimeters on each plot of the five treatment areas. Lysimeters were placed at depths representing the bottom of the dominant soil horizons, generally the AB or BA (15 cm) and the lower B (>40 cm, maximum depth to saprolite) horizons. Horizon depths were identified from Natural Resources Conservation Service soil survey information and by soil probe reconnaissance (Jennifer Knoepp, personal observations in 2008). Two sets of lysimeters were installed at randomly selected locations within each plot (i.e., a total of 100 lysimeters). Lysimeters were allowed to stabilize for approximately 2 months before water samples were collected for chemical analyses. During the stabilization period, lysimeters were evacuated weekly; solution  $\text{NO}_3\text{-N}$  concentrations were monitored to ensure equilibration before data collection began. After lysimeters equilibrated, soil solution samples were collected from each lysimeter every 2 weeks, and 0.03 MPa of tension was applied to the lysimeter. Soil water samples were collected beginning May 2008 for a 16 month period (May 2008–August 2009).

Soil solution samples were composited monthly for analysis of,  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4^{2-}\text{-P}$ , and  $\text{SO}_4^{2-}\text{-S}$ ,  $\text{NH}_4^+\text{-N}$ ,

$\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and Al. Dissolved organic carbon (DOC), total dissolved nitrogen (TDN) and solution pH were determined once monthly. Soil solution  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4^{2-}\text{-P}$ , and  $\text{SO}_4^{2-}\text{-S}$  were determined on a Dionex 2500 Ion Chromatograph (Dionex Corporation, Sunnyvale, CA),  $\text{NH}_4^+\text{-N}$  was determined on an AlpKem model 3590 autoanalyzer (Alpkem Corporation, College Station, TX) using the alkaline phenol method (USEPA 1983a). Analyses for  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and Al were determined as described above and pH using an Orion research digital pH meter (model 611) with a Broadley James pH probe. Solution DOC and TDN content were determined using a Shimadzu TOC- $\text{V}_{\text{CPH}}$  TNM-1 (Shimadzu Scientific Instruments, Columbia, MD).

### Statistical analysis

The experimental design was based on five treatment areas (fixed factor) with the five plots (random factor) within each treatment area representing replication. We used analysis of variance and repeated measures analysis of variance (PROC MIXED, SAS Institute Inc. 2002–2003) to determine significant differences among the five treatment areas for plant biomass and nutrient contents, forest floor mass and nutrient contents, and soil and soil solution chemistry. We used the unstructured covariance option in the repeated statement because it produced the smallest value for the Akaike’s Information Criterion (AIC) and Schwarz’ Bayesian Criterion (SBC) (Little et al. 1996). We evaluated the main effects of time (sample dates), treatment (2xSBL, 2xMBL, 2xSB, 1xMBL, and REF), and time\*treatment interactions. If overall F-tests were significant ( $p \leq 0.05$ ) then least squares means (LS-means, Tukey-Kramer adjusted t-statistic) tests were used to evaluate significance among treatment and time interactions. We used the Satterthwaite option in the model statement to obtain the correct degrees of freedom (Little et al. 1996).

## Results

### Aboveground biomass and nutrients

All wildfire burned sites experienced substantial overstory mortality (> 300 stems  $\text{ha}^{-1}$ ). There were no live overstory trees on 2xSBL, followed by 2xSB which

had a larger number of dead trees than all of the other sites. Overstory biomass was lower on the severely burned areas (2xSBL and 2xSB) than the other treatments with no significant change between 2008 and 2009 (Table 1). Understory biomass was significantly higher on REF than all the burned treatments in both years (Table 1); and biomass doubled on the recently burned treatments (2xSBL, 2xMBL, and 2xSB) between 2008 and 2009. As expected, ericaceous (heath) species sprouted after fire; and their density increased between 2008 and 2009 for all recently burned treatments (data not shown). There were no significant differences in ground-layer biomass among treatments (Table 1).

Forest floor mass was lower on all burned sites compared to REF. The severely burned sites (2xSBL and 2xSB) had significantly less forest floor Oi-layer mass than the moderately burned sites (2xMBL and 1xMB) (Table 1). Forest floor Oe+Oa layer mass was greatest on 1xMB and REF but differed significantly only from 2xMBL (Table 1).

Nutrient content of forest floor components differed among treatments (Table 2). Forest floor Oi layer N

and Ca content was lowest in 2xSB and greatest in 2xMBL and 1xMB. Oi layer total P content was greatest in 2xMBL and least in 2xSB, while total Al was greatest in 2xSBL compared to all other sites. Oi layer Mg and K content did not differ among sites. Oe+Oa layer Ca content was significantly higher on 2xSBL than 2xSB and REF, while Mg content was significantly lower on 2xMBL compared to 2xSBL, 1xMB, and REF. Total P content of the Oe+Oa layer was least on 2xMBL and greatest on 1xMB and REF. Oe+Oa layer N and Al content did not differ among sites.

We found no differences in foliar N concentration among treatments for either evergreen or deciduous species (Table 3). Evergreen foliar Ca was greater on 2xMBL, 1xMBL, and REF than the severely burned sites (2xSBL and 2xSB); whereas deciduous foliar Ca was greater on 2xMBL than 2xSBL (Table 3). Total P was greatest on 2xMBL for both evergreen and deciduous foliage compared to 1xMBL and REF. There were no differences in N, Ca, and Mg concentrations of herbaceous plants among treatments; whereas, herbaceous plant K and P concentrations were greater on 2xSBL and 2xMBL than 2xSB and REF (Table 3).

**Table 1** Aboveground mass (mean±1 SE) of forest floor litter (Oi) and fermentation + humus (Oe+Oa); standing live and dead trees; live understory (shrubs and tree saplings <5.0 cm dbh, ≥0.5 m height); and ground layer (<0.5 m height) for the five treatment areas; severely burned twice (2002 & 2007) plus

dolomitic lime application (2xSBL); moderately burned twice plus lime application (2xMBL); severely burned twice, unlimed (2xSB); moderately burned once (2002), unlimed (1xMB); and a reference area (REF; unburned, unlimed)

	2xSBL	2xMBL	2xSB	1xMB	REF
Forest floor (kg ha <sup>-1</sup> )					
Litter (Oi)	664 bc (157)	1,184 a (102)	417 c (126)	1,383 a (135)	996 ab (116)
Oe+Oa	29,505 ab (4,071)	15,906 b (5,809)	29,140 ab (5,232)	36,704 a (3,287)	37,636 a (3,832)
Overstory layer (kg ha <sup>-1</sup> )					
Live trees					
2008	0 c	153,475 a (13,694)	27,085 c (10,583)	107,723 b (18,711)	100,558 b (16,854)
2009	43.6 c (12.6)	161,560 a (13,765)	16,145 c (8,527)	126,745 b (20,446)	101,108 b (16,470)
Dead trees					
2008	43,089 a (11,627)	53,365 a (7,798)	42,508 a (8,226)	11,344 b (4,190)	1,423 b (312)
2009	43,089 a (11,627)	55,262 a (7,943)	53,805 a (8,551)	12,176 b (3,935)	4,397 b (2,768)
Understory layer (kg ha <sup>-1</sup> )					
2008	3,946 b (969)	515 c (199)	1,875 bc (523)	3,043 b (878)	9,932 a (1,241)
2009	8,657 ab (2,133)	1,728 c (545)	3,592 bc (1,066)	5,425 b (2,082)	10,214 a (1,238)
Ground layer (kg ha <sup>-1</sup> )					
2008	360 (108)	235 (120)	914 (381)	322 (220)	360 (171)

Values in rows followed by different letters are significantly different ( $\alpha < 0.05$ ) among treatments (SAS 2002–2003). Standard errors are in parentheses.

**Table 2** Nutrient content (mean±1 SE) of forest floor litter (Oi) and fermentation + humus (Oe+Oa) for the five treatment areas in 2008. Treatments and code names are the same as in Table 1

	2xSBL	2xMBL	2xSB	1xMB	REF
Forest floor	N (kg ha <sup>-1</sup> )				
Oi	4.68 ab (1.28)	7.56 a (1.22)	3.10 b (1.23)	8.38 a (0.93)	5.46 ab (0.89)
Oe+Oa	343.26 (51.60)	208.67 (80.01)	336.28 (68.52)	429.82 (30.00)	409.19 (49.20)
	Ca (kg ha <sup>-1</sup> )				
Oi	6.60 ab (1.89)	10.96 a (1.34)	2.82 b (1.07)	9.90 a (1.43)	6.76 ab (1.52)
Oe+Oa	182.71 a (27.17)	135.22 ab (39.53)	80.96 b (14.50)	171.35 ab (23.91)	116.45 b (12.93)
	Mg (kg ha <sup>-1</sup> )				
Oi	1.31 (0.42)	1.34 (0.18)	0.92 (0.39)	2.14 (0.26)	1.69 (0.48)
Oe+Oa	31.95 a (6.08)	10.18 b (3.44)	17.46 ab (2.65)	35.86 a (6.60)	32.32 a (6.51)
	K (kg ha <sup>-1</sup> )				
Oi	1.76 (0.62)	2.34 (0.37)	1.05 (0.42)	1.60 (0.16)	1.35 (0.17)
Oe+Oa	15.14 b (3.88)	11.45 b (3.34)	16.12 ab (3.00)	22.67 ab (1.40)	26.99 a (1.82)
	P (kg ha <sup>-1</sup> )				
Oi	0.43 ab (0.12)	0.60 a (0.08)	0.21 b (0.08)	0.42 ab (0.04)	0.30 ab (0.04)
Oe+Oa	16.47 ab (2.75)	10.46 b (3.58)	13.89 ab (2.64)	20.70 a (1.03)	20.61 a (2.15)
	Al (kg ha <sup>-1</sup> )				
Oi	1.69 a (0.67)	0.32 b (0.06)	0.12 b (0.04)	0.42 b (0.09)	0.25 b (0.07)
Oe+Oa	104.79 (18.57)	58.05 (21.64)	52.97 (8.50)	82.34 (19.83)	71.09 (13.43)

Values in rows followed by different letters are significantly different ( $\alpha < 0.05$ ) among treatments (SAS 2002–2003). Standard errors are in parentheses

### Soil and soil solution chemistry

With few exceptions, we found significant date, treatment, and date \* treatment interaction effects for ECEC, base saturation, soil exchangeable cations (Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>, and Al<sup>3+</sup>), and extractable PO<sub>4</sub><sup>2-</sup>-P, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>+</sup>-N for both surface (0–10 cm depth) and subsurface (10–30 cm depth) soils (Table 4). For surface soils, ECEC was significantly greater on 2xMBL compared to 2xSB, 2xSBL, 1xMB, and REF; and significantly lower on 2xSBL than both 2xSB and REF (Table 4). Surface soil base saturation was significantly greater on 2xSBL, 2xMBL and 1xMB than 2xSB and REF. Surface soil exchangeable Ca<sup>2+</sup> was significantly affected by liming treatment; Ca<sup>2+</sup> concentrations were greater on 2xMBL and 2xSBL than 2xSB, 1xMB and REF (Table 4). Burning alone did not affect surface soil Ca<sup>2+</sup>. Patterns of exchangeable Mg<sup>2+</sup> differed, with significantly greater Mg<sup>2+</sup> in surface soils on 1xMB compared to 2xSB and REF. Concentrations of exchangeable K<sup>+</sup> were greater on 2xMBL than 2xSBL, 2xSB, 1xMB, and REF; and significantly lower on 2xSBL compared to 2xSB,

1xMB, and REF. Surface soil exchangeable Al<sup>3+</sup> was significantly greater on 2xMBL than 2xSB, 2xSBL, 1xMB, and REF; and significantly lower on 2xSBL than 2xSB, 1xMB, and REF (Table 4). The ratio of Ca:Al was greatest in 2xSBL and differed significantly from 2xSB, 1xMB and REF (Table 4)

Subsurface soil nutrient responses were similar. ECEC was greater on 2xMBL and 2xSB than 2xSBL, 1xMB, and REF; and significantly lower on 2xSBL than 1xMB, and REF (Table 4). Base saturation of subsurface soil was greater on 2xSBL and 2xMBL than REF. Exchangeable Ca<sup>2+</sup> was significantly greater on 2xMBL compared to 2xSB, 2xSBL, 1xMB, and REF; and greater on 2xSBL compared to REF. Subsurface soil exchangeable Mg<sup>2+</sup> was significantly greater on 1xMB than 2xSBL, 2xSB, and REF (Table 4). Exchangeable K<sup>+</sup> was significantly greater on 2xMBL than 2xSBL, 2xSB, 1xMB, and REF; and significantly greater on 2xSB than 2xSB+L, 1xMB, and REF (Table 4). Subsurface soil exchangeable Al<sup>3+</sup> was significantly greater on 2xMBL than 2xSB, 2xSBL, 1xMB, and REF; and significantly lower on 2xSBL than 2xSB and REF (Table 4).



**Table 3** Foliar nutrient concentrations (mean±1 SE) of evergreen *Kalmia latifolia* L., deciduous trees, and herbaceous plant species for the five treatment areas. Treatments and code names are the same as in Table 1

	2xSBL	2xMBL	2xSB	1xMBL	REF
N (g kg <sup>-1</sup> )					
Evergreen	8.56 (0.52)	9.92 (0.19)	9.42 (0.19)	9.46 (0.56)	8.58 (0.22)
Deciduous	16.61 (0.78)	18.62 (1.00)	16.37 (0.29)	17.54 (0.43)	16.82 (0.33)
Ca (g kg <sup>-1</sup> )					
Evergreen	6.69 b (0.37)	10.21 a (1.00)	7.18 b (0.31)	9.24 a (0.62)	10.83 a (0.64)
Deciduous	4.98 b (0.58)	6.69 a (0.16)	5.38 ab (0.49)	5.78 ab (0.55)	5.27 ab (0.07)
Mg (g kg <sup>-1</sup> )					
Evergreen	1.99 b (0.07)	2.25 a (0.04)	2.42 a (0.04)	2.26 a (0.08)	1.98 b (0.10)
Deciduous	1.89 b (0.20)	1.99 b (0.12)	2.70 a (0.25)	2.38 ab (0.16)	2.44 ab (0.16)
K (g kg <sup>-1</sup> )					
Evergreen	6.28 a (0.19)	5.93 a (0.23)	5.58 ab (0.17)	5.21 bc (0.21)	4.59 c (0.23)
Deciduous	9.03 ab (0.48)	9.73 a (0.51)	9.08 ab (0.25)	7.88 b (0.30)	9.82 a (0.56)
P (g kg <sup>-1</sup> )					
Evergreen	0.74 ab (0.03)	0.78 a (0.02)	0.71 ab (0.03)	0.65 b (0.05)	0.63 b (0.02)
Deciduous	1.52 ab (0.04)	1.61 a (0.05)	1.42 bc (0.04)	1.22 d (0.03)	1.36 c (0.04)
Al (g kg <sup>-1</sup> )					
Evergreen	0.05 b (0.01)	0.06 ab (0.005)	0.05 b (0.004)	0.07 a (0.002)	0.07 a (0.002)
Deciduous	1.04 (0.03)	1.04 (0.02)	1.52 (0.02)	1.36 (0.02)	0.09 (0.02)
Ca:Al ratio					
Evergreen	140 (16)	175 (15)	160 (9)	143 (8)	154 (9)
Deciduous	89 (32)	73 (11)	38 (5)	46 (7)	85 (15)
N:P ratio					
Evergreen	11.7 c (0.4)	12.5 bc (0.5)	13.4 ab (0.4)	14.9a (0.6)	13.6 ab (0.3)
Deciduous	11.0 b (0.6)	11.6 b (0.6)	11.6 b (0.3)	14.4 a (0.4)	12.4 b (0.2)
Herbaceous	2xSBL	2xMBL	2xSB	1xMB	REF
C (g kg <sup>-1</sup> )	443 b (6.40)	460 ab (5.25)	478 a (3.63)	465 ab (5.55)	463 ab (2.32)
N (g kg <sup>-1</sup> )	11.39 (1.00)	13.69 (1.33)	9.82 (0.47)	13.51 (1.48)	9.10 (0.62)
Ca (g kg <sup>-1</sup> )	8.00 (0.25)	7.30 (0.88)	7.03 (0.56)	8.97 (1.00)	6.33 (1.45)
Mg (g kg <sup>-1</sup> )	2.28 (0.30)	2.21 (0.15)	2.33 (0.16)	3.04 (0.18)	2.25 (0.42)
K (g kg <sup>-1</sup> )	14.79 a (2.74)	14.01 a (3.17)	6.23 b (0.64)	9.67 ab (1.08)	6.32 b (0.35)
P (g kg <sup>-1</sup> )	1.78 a (0.32)	1.37 a (0.14)	0.82 b (0.11)	1.16 ab (0.16)	0.76 b (0.03)

Evergreen is leaf tissue nutrient concentration of *Kalmia latifolia*. Deciduous is tissue nutrient concentration of deciduous tree species. Values in rows followed by different letters are significantly different ( $\alpha < 0.05$ ) among treatments (SAS 2002–2003). Standard errors are in parentheses

Ratio of Ca:Al was significantly greater in the subsurface soil on 2xSBL and 2xMBL compared to all other sites (Table 4).

We found little to no extractable NO<sub>3</sub><sup>-</sup>-N in surface or subsurface soils across the treatment areas, i.e., very few soil samples had NO<sub>3</sub><sup>-</sup>-N values above the detection limit. Exchangeable NH<sub>4</sub><sup>+</sup>-N was not significantly different among treatments for either surface or subsurface

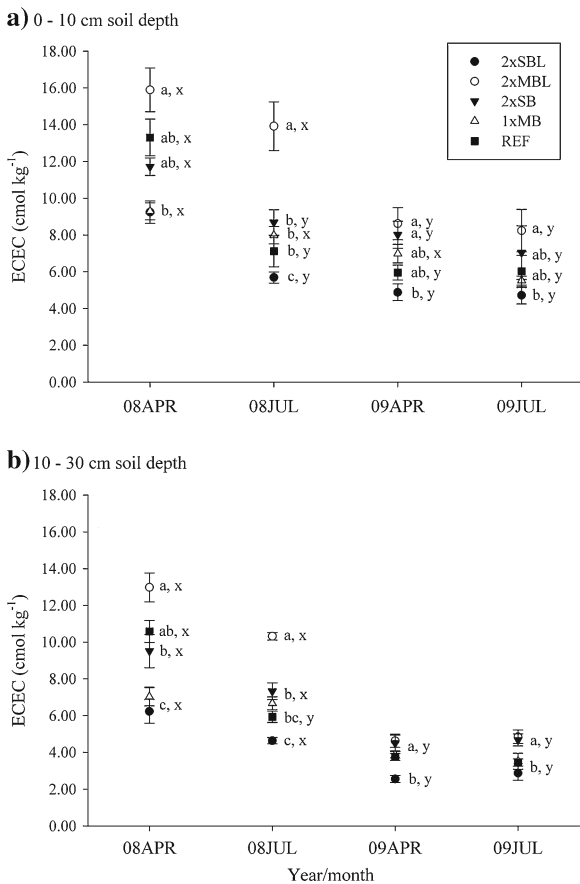
soils (Table 4). Surface soil extractable PO<sub>4</sub><sup>2-</sup>-P was greatest in 2xMBL and REF sites and differed significantly from 2xSB. Subsurface soil PO<sub>4</sub><sup>2-</sup>-P was greatest in REF compared to 2xSBL.

We examined changes in soil chemistry during the 2 years of sample collection and found that ECEC significantly declined over time for both surface and subsurface soil depths (Fig. 2a–b). While there were

**Table 4** Soil chemistry (mean $\pm$ 1 SE) for the five treatment areas. Treatments and code names are the same as in Table 1. All values are in cmol<sub>c</sub> kg<sup>-1</sup> except for pH, base saturation (BS, %), total carbon and nitrogen (g kg<sup>-1</sup>), and Ca/Al molar ratio

	Surface (0–10 cm) soil depth					Subsurface (10–30 cm) soil depth				
	2xSBL	2xMBL	2xSB	1xMB	REF	2xSBL	2xMBL	2xSB	1xMB	REF
ECEC	6.132 c (0.311)	11.665 a (0.454)	8.866 b (0.457)	7.362 bc (0.346)	8.098 b (0.469)	4.074 c (0.336)	8.146 a (0.186)	6.503 a (0.329)	5.270 b (0.172)	5.927 b (0.242)
pH	3.75 a (0.05)	3.80 a (0.04)	3.60 b (0.03)	3.77 a (0.03)	3.71 ab (0.02)	4.05 (0.02)	4.05 (0.03)	4.00 (0.01)	4.06 (0.04)	3.97 (0.03)
BS	19.49 a (1.22)	14.64 a (3.09)	6.67 b (0.71)	14.09 a (2.21)	5.91 b (0.52)	7.47 a (0.44)	7.23 a (0.77)	5.16 bc (0.33)	6.11 ab (0.69)	4.03 c (0.28)
Carbon	33.97 (6.48)	37.74 (2.05)	51.35 (6.91)	41.93 (1.30)	43.95 (1.85)	13.24 b (3.01)	13.94 b (1.56)	21.86 a (1.30)	14.91ab (0.60)	15.78 ab (1.03)
Nitrogen	1.07 b (0.21)	1.64 a (0.11)	1.43 ab (0.12)	1.45 ab (0.05)	1.39 ab (0.05)	0.46 b (0.09)	0.70 a (0.04)	0.78 a (0.06)	0.62 ab (0.03)	0.60 ab (0.03)
NO <sub>3</sub> -N	0.0001 (0.00004)	0.00003 (0.00003)	ND	ND	ND	ND	ND	ND	ND	ND
NH <sub>4</sub> <sup>+</sup> -N	0.012 (0.002)	0.011 (0.002)	0.007 (0.001)	0.010 (0.001)	0.007 (0.001)	0.004 (0.0005)	0.004 (0.0003)	0.005 (0.0003)	0.005 (0.0003)	0.004 (0.0002)
PO <sub>4</sub> <sup>3-</sup> -P	0.024 ab (0.001)	0.025 a (0.002)	0.020 b (0.001)	0.024 ab (0.001)	0.025 a (0.001)	0.011 b (0.001)	0.014 ab (0.001)	0.014ab (0.001)	0.014ab (0.001)	0.016 a (0.001)
Ca <sup>2+</sup>	0.732 a (0.050)	0.962 a (0.166)	0.185 c (0.036)	0.401 b (0.063)	0.063 c (0.006)	0.096 b (0.004)	0.152 a (0.025)	0.057 bc (0.015)	0.061 bc (0.006)	0.022 c (0.002)
Mg <sup>2+</sup>	0.281 ab (0.025)	0.228 ab (0.022)	0.147 b (0.012)	0.404 a (0.105)	0.179 b (0.020)	0.060 b (0.004)	0.074 ab (0.008)	0.056 b (0.003)	0.110 a (0.021)	0.056 b (0.006)
K <sup>+</sup>	0.124 c (0.006)	0.279 a (0.016)	0.221 b (0.020)	0.185 b (0.004)	0.189 b (0.014)	0.112 c (0.007)	0.247 a (0.006)	0.178 b (0.010)	0.120 c (0.007)	0.124 c (0.008)
Al <sup>3+</sup>	7.64 c (0.62)	16.76 a (0.82)	12.40 b (0.64)	11.01 b (0.48)	10.94 b (0.77)	5.78 c (0.49)	11.66 a (0.49)	8.75 b (0.65)	7.64 bc (0.37)	8.11 b (0.27)
Ca/Al	0.102 a (0.006)	0.072 ab (0.017)	0.015 c (0.003)	0.037 bc (0.006)	0.006 c (0.001)	0.018 a (0.001)	0.016 a (0.003)	0.006 bc (0.001)	0.009 b (0.001)	0.003 c (0.0002)

ND = samples below the detection limits of our methods. Values in rows within a soil depth followed by different letters are significantly different ( $\alpha < 0.05$ ) among treatments (SAS 2002–2003). Values were averaged across time; and then, mean values and standard errors were based on the five plots per treatment area ( $n=5$ ). Standard errors are in parentheses



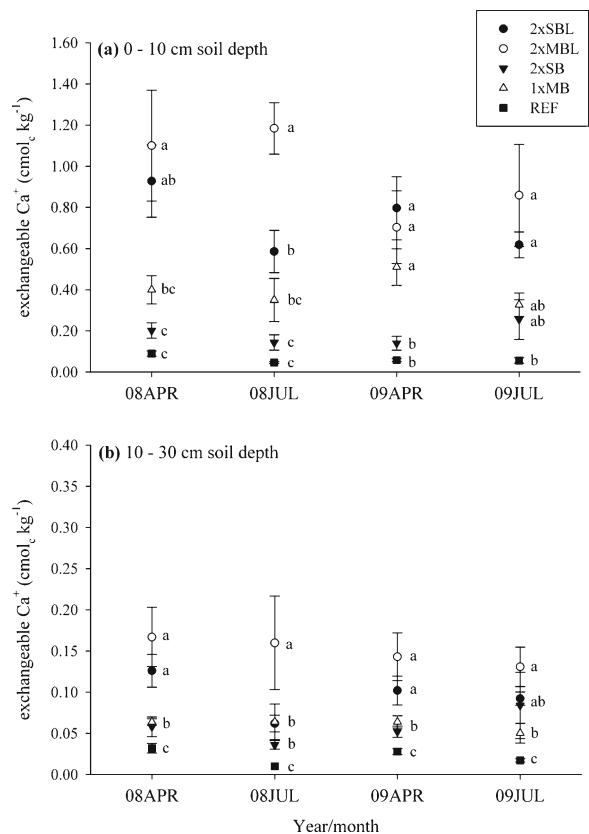
**Fig. 2** Soil effective cation exchange capacity (ECEC) at two soil depths: **a** surface (0–10 cm) and **b** subsurface (10–30 cm). Soils were sampled in April and July of 2008, 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2. Letters *a, b, c*, denote significant ( $\alpha=0.05$ ) differences among treatments within sample dates; letters *x, y, z* denote significant ( $\alpha=0.05$ ) differences among sample dates within treatments

no significant changes over time in exchangeable  $\text{Ca}^{2+}$  for any treatments at either soil depth (Fig. 3a–b), soil  $\text{Al}^{3+}$  concentrations increased over time on 2xMBL for both soil depths and on 2xSBL for the surface soil depth (Fig. 4a–b). The soil Ca:Al ratios declined during the 2 year sampling period in both the 2xSBL and 2xMBL sites (Fig. 5a–b).

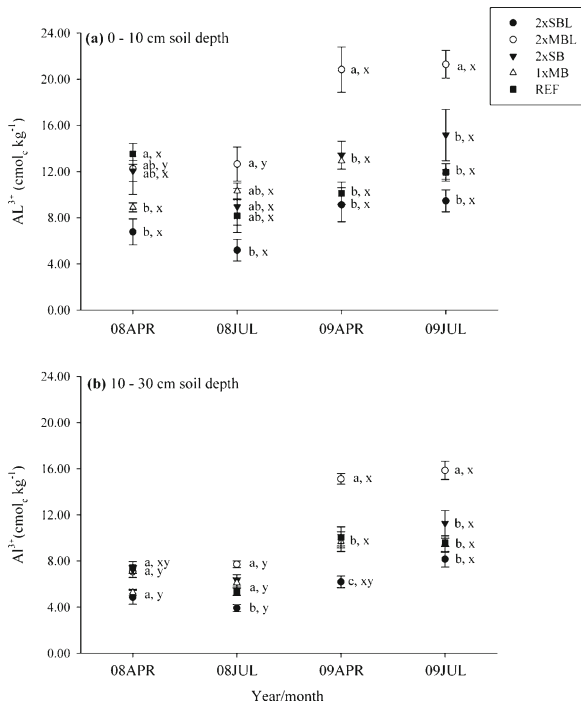
We found significant date, treatment, and date \* treatment interaction effects for soil solution chemistry collected shallow (15 cm) and deep (>40 cm) within the soil profile (Table 5). Shallow soil solution  $\text{NO}_3^-$ -N was greater on 2xSBL and 2xMBL than the other treatments; deep soil solution  $\text{NO}_3^-$ -N concentrations were greater on 2xMBL than the other treatments at the deeper soil

depth (Table 5), this response lasted for several months (Fig. 6a–b). There were no significant differences among treatments for  $\text{NH}_4^+$ -N or  $\text{K}^+$  concentrations in either shallow or deep soil solution (Table 5). Both shallow and deep soil solution  $\text{Ca}^{2+}$  concentrations were greatest on 2xSBL and 2xMBL compared to REF (Table 5). Soil solution  $\text{Ca}^{2+}$  concentrations of deep soil on 1xMB and REF were lower than the recently burned treatments. Neither shallow nor deep soil solution  $\text{Ca}^{2+}$  changed over time (Fig. 7a–b).

For the shallow soil solution, Al concentration was greater on REF than the burned sites (Table 5, Fig. 8). Solution Al concentrations for shallow and deep soil were lower on 2xSBL than the other sites (Fig. 8a–b). There were no patterns of changing Al concentrations



**Fig. 3** Soil exchangeable calcium ( $\text{Ca}^{2+}$ ) at two soil depths: **a** surface (0–10 cm) and **b** subsurface (10–30 cm). Soils were sampled in April and July of 2008, 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2. Letters *a, b, c*, denote significant ( $\alpha=0.05$ ) differences among treatments within sample dates; there were no significant differences among sample dates for any treatment



**Fig. 4** Soil exchangeable aluminum ( $Al^{3+}$ ) at two soil depths: **a** surface (0–10 cm) and **b** subsurface (10–30 cm). Soils were sampled in April and July of 2008, 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2. Letters *a*, *b*, *c*, denote significant ( $\alpha=0.05$ ) differences among treatments within sample dates; letters *x*, *y*, *z* denote significant ( $\alpha=0.05$ ) differences among sample dates within treatments

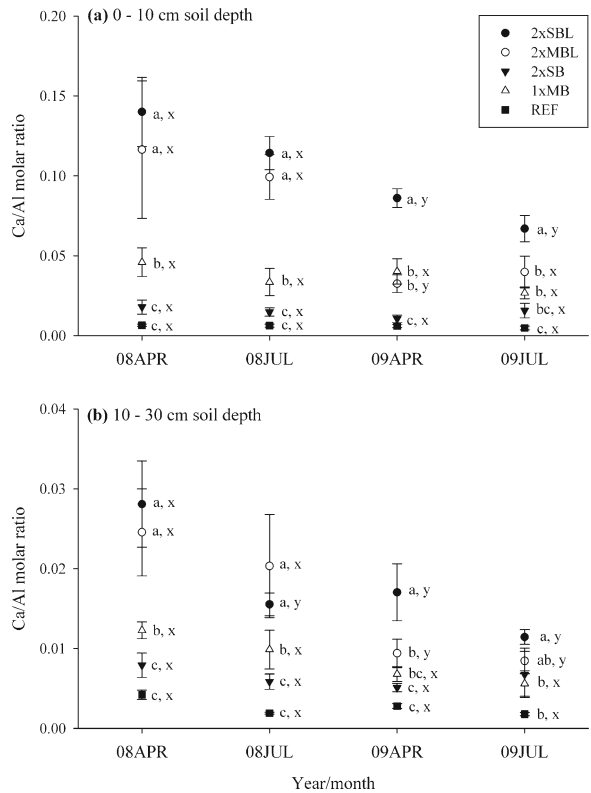
among months of sample collection. Shallow soil solution  $SO_4^{2-}$ -S concentrations were greater on 2xSBL than all other sites, but there were no significant differences among treatments for deep soil (Table 5).

**Discussion**

All wildfire burned sites experienced substantial overstory mortality. The large number of dead pines on all sites, including the reference, was due in large part to a southern pine beetle (*Dendroctonus frontalis* Zimm.) outbreak in 1999–2001 (Duehl et al. 2011); the three areas impacted by the 2007 wildfire had little to no live pine remaining in the overstory. Overstory dead tree mass was comparable among the three recent wildfire sites. However, 2xMBL had a much larger component of live trees than the other burned sites due to the presence of a few large (average dbh=59.3 cm; average density=70 trees ha<sup>-1</sup>) oaks (*Quercus alba* L.,

*Q. coccinea*, and *Q. montana*) that contributed substantially to the total aboveground biomass. Understory biomass doubled between 2008 and 2009 on the most recently burned sites; and 2xSBL had already accumulated biomass equivalent to the reference site due to the rapid regrowth of ericaceous shrubs and prolific sprouting of deciduous trees.

Soils in Linville Gorge Wilderness Area are inherently acidic and low in base cation concentrations (Elliott et al. 2008). This along with decades of acidic deposition has led to soils with low percent base saturation, low pH, and low Ca:Al ratios. Although there is variation among tree species in acid-sensitivity, Ca is an essential nutrient and Al is a potential toxin (Cronan and Grigal 1995); thus, depletion of soil Ca and increases in available Al due to acidic deposition may significantly alter the health and



**Fig. 5** Soil Ca/Al molar ratio at two soil depths: **a** surface (0–10 cm) and **b** subsurface (10–30 cm). Soils were sampled in April and July of 2008, 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2. Letters *a*, *b*, *c*, denote significant ( $\alpha=0.05$ ) differences among treatments within sample dates; letters *x*, *y*, *z* denote significant ( $\alpha=0.05$ ) differences among sample dates within treatments

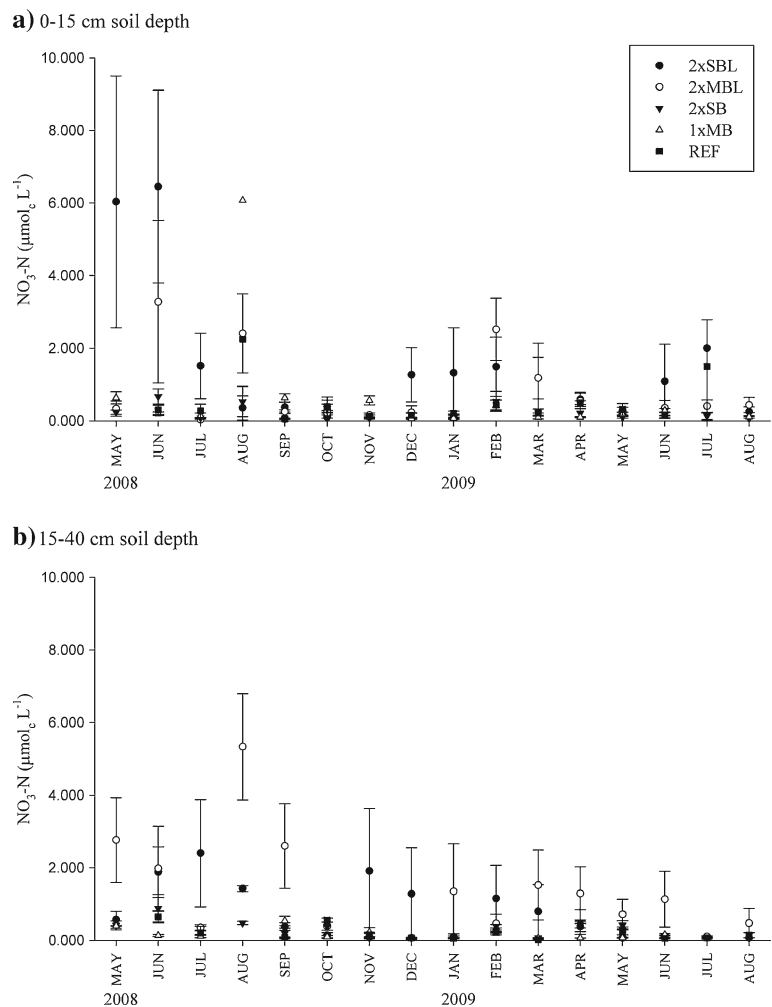
**Table 5** Soil solution chemistry (mean±1 SE) for the five treatment areas. Treatments and code names are the same as in Table 1. All values are in  $\mu\text{mol}_c \text{L}^{-1}$  except for pH and TDN, DOC, and Al are in  $\text{mg L}^{-1}$

	Shallow lysimeters (15 cm)					Deep lysimeter (> 40 cm)				
	2xSBL	2xMBL	2xSB	1xMB	REF	2xSBL	2xMBL	2xSB	1xMB	REF
pH	4.73 b (0.08)	4.97 a (0.12)	4.46 c (0.07)	4.81 b (0.06)	4.41 c (0.12)	4.61 b (0.08)	4.81 a (0.12)	4.46 c (0.03)	4.69 b (0.04)	4.62 b (0.02)
DOC	17.76 b (4.02)	7.31 c (0.93)	18.88 b (2.51)	12.31 c (1.62)	31.96 a (4.16)	3.18 bc (0.30)	4.11 b (1.31)	6.52 a (2.46)	2.70 c (0.13)	4.49 b (1.62)
TDN	30.30 b (5.11)	17.50 c (3.12)	22.25 c (2.66)	19.10 c (1.67)	36.44 a (3.96)	8.08 b (1.63)	12.17 a (3.35)	8.79 b (2.39)	5.93 b (0.84)	6.37 b (1.83)
NO <sub>3</sub> <sup>-</sup> -N	1.668 a (0.586)	0.834 a (0.274)	0.211 b (0.032)	0.343 b (0.043)	0.314 b (0.060)	0.800 ab (0.437)	1.345 a (0.487)	0.318 b (0.046)	0.163 b (0.015)	0.226 b (0.015)
NH <sub>4</sub> <sup>+</sup> -N	1.118 (0.242)	0.669 (0.123)	1.081 (0.265)	0.570 (0.223)	1.292 (0.237)	0.348 (0.024)	0.641 (0.219)	0.649 (0.186)	0.700 (0.380)	0.440 (0.112)
PO <sub>4</sub> <sup>3-</sup> -P	0.286 ab (0.043)	0.214 b (0.067)	0.402 ab (0.064)	0.275 ab (0.053)	0.465 a (0.074)	0.213 (0.063)	0.107 (0.025)	0.221 (0.025)	0.194 (0.023)	0.185 (0.026)
Ca <sup>2+</sup>	114.23 a (29.76)	82.12 ab (16.35)	42.87 bc (7.82)	44.45 bc (9.58)	35.07 c (2.43)	47.43 a (11.33)	60.68 a (24.81)	30.84 ab (7.16)	16.87 b (1.84)	20.85 b (1.45)
Mg <sup>2+</sup>	58.76 ab (5.16)	37.34 b (5.46)	44.03 ab (8.69)	68.02 a (14.52)	71.41 a (6.59)	34.95 (4.31)	31.88 (4.20)	28.70 (3.63)	39.47 (4.89)	37.46 (2.72)
K <sup>+</sup>	58.49 (13.35)	48.37 (20.82)	58.32 (7.37)	35.38 (9.72)	37.75 (4.42)	26.51 (3.73)	29.53 (3.34)	30.33 (3.92)	18.95 (2.14)	21.02 (2.41)
SO <sub>4</sub> <sup>2-</sup> -S	147.80 a (14.08)	98.95 bc (5.09)	117.96 b (11.08)	75.98 c (6.91)	89.42 c (5.09)	122.79 (16.65)	99.03 (9.81)	118.46 (10.86)	98.77 (5.16)	111.73 (8.22)
Al	0.633 b (0.182)	0.170 c (0.046)	0.998 b (0.084)	0.840 b (0.254)	2.104 a (0.162)	0.137 bc (0.053)	0.165 bc (0.055)	0.379 a (0.089)	0.101 c (0.010)	0.318 ab (0.102)

Values in rows within a soil depth followed by different letters are significantly different ( $\alpha < 0.05$ ) among treatments (SAS 2002–2003). Values were averaged across time; and then, mean values and standard errors were based on the five plots per treatment area ( $n=5$ ). Standard errors are in parentheses



**Fig. 6** Soil solution nitrate-nitrogen ( $\text{NO}_3^-$ -N) at two soil depths: **a** shallow (15 cm) and **b** deep (>40 cm). Soil solution was collected from May 2008 to August 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2



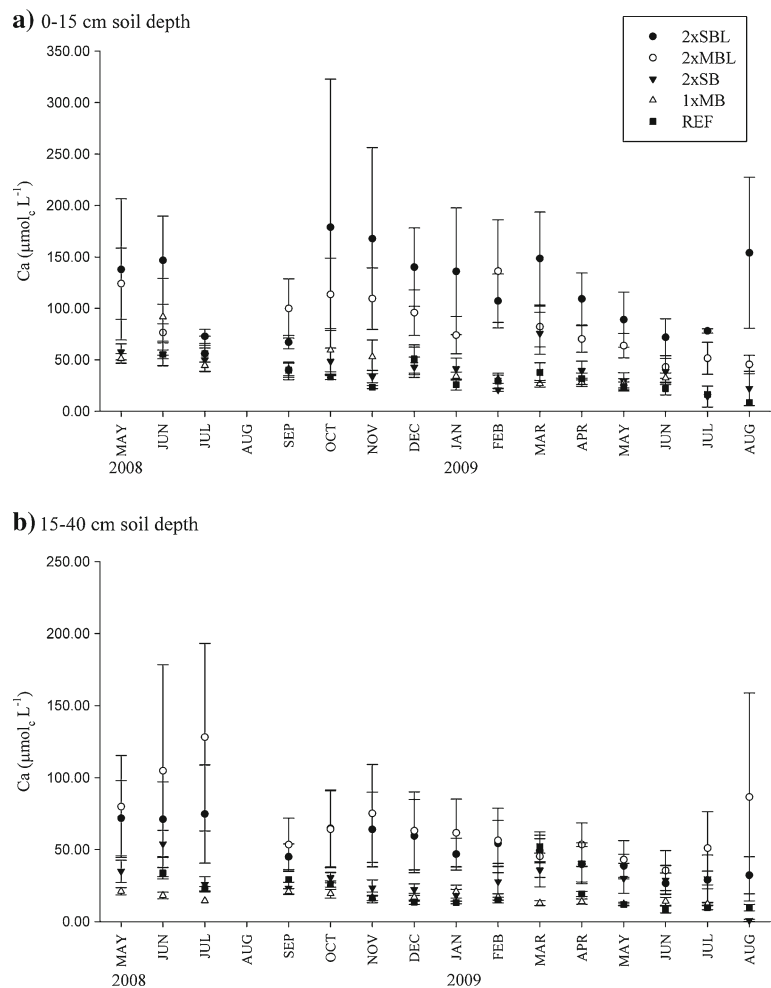
productivity of forest trees (Schaberg et al. 2006; Halman et al. 2011; Long et al. 2011).

Pools and fluxes of soil carbon and nutrients can be significantly impacted by wildfire (Debano et al. 1998; Johnson and Curtis 2001; Knoepp et al. 2005; Hebel et al. 2009), and severe burning can alter soil chemical and physical properties (Korb et al. 2004; Homann et al. 2011; Nave et al. 2011). Soil  $\text{NH}_4$ -N concentration often increases following fire due to organic matter combustion, denaturing of proteins, and condensation of  $\text{NH}_4$ -N in soil (Knoepp et al. 2005; Certini 2005), however, this increase is often short-lived, lasting less than one year (Knoepp et al. 2009; Lavoie et al. 2010). Data have also shown a pulse of  $\text{NO}_3$ -N in runoff and soil solution due to increased nitrification in response to environmental changes and the initial  $\text{NH}_4$ -N pulse (Korb et al.

2004; Certini 2005; Shakesby and Doerr 2006; Johnson et al. 2007). We found no differences in soil  $\text{NO}_3$ -N or  $\text{NH}_4$ -N among our treatment areas or compared to the reference. However, we did not initiate soil sampling until 10 months after the May 2007 Pinnacle Fire; thus, we may have missed the initial pulse of  $\text{NH}_4$ -N in soils at Linville Gorge.

While we measured no significant soil solution  $\text{NO}_3$ -N response to burning alone, shallow soil solution  $\text{NO}_3$ -N was significantly higher on 2xSBL and 2xMBL than the other sites, indicating that burning followed by lime application may have increased available nitrogen. However, we could not detect any significant differences in foliar N among treatment areas. Soil solution  $\text{NO}_3$ -N concentrations for the burned sites were within ranges of those reported by Knoepp and Swank (1993) after a fell-and-burn, but

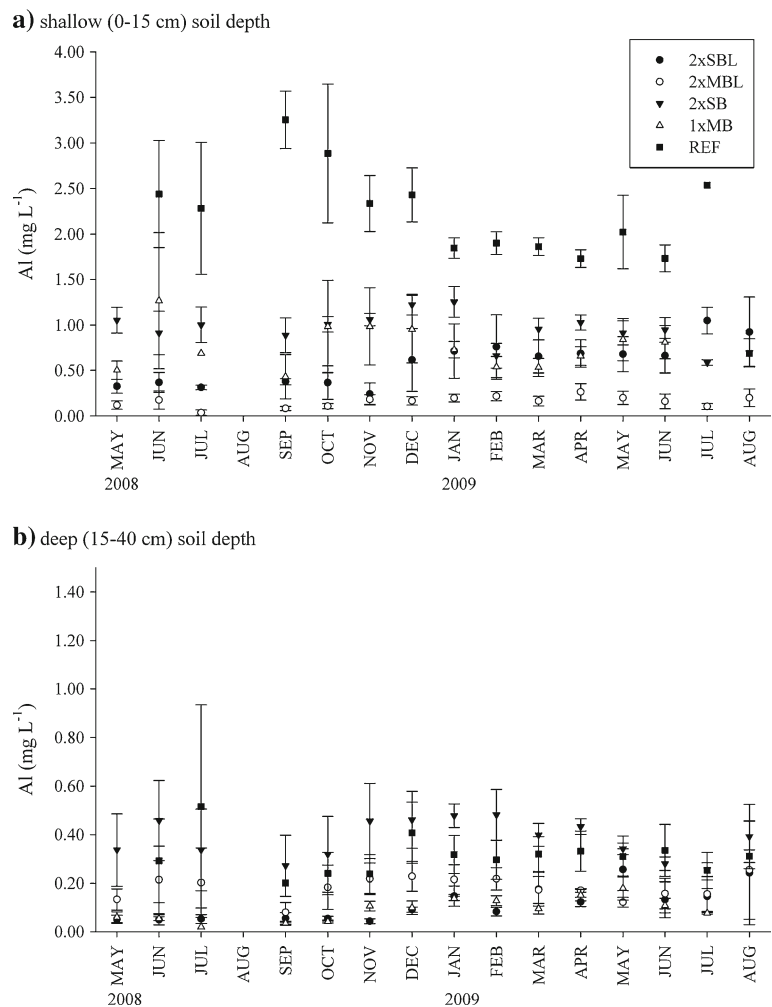
**Fig. 7** Soil solution calcium ( $\text{Ca}^{2+}$ ) at two soil depths: **a** shallow (15 cm) and **b** deep (>40 cm). Soil solution was collected from May 2008 to August 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2



not as high as those reported by Elliott et al. (2012) after restoration burning in a degraded shortleaf pine ecosystem. Further, Knoepp and Swank (1993) found no significant increases in soil solution  $\text{NH}_4\text{-N}$ , but they did find increases in soil solution  $\text{NO}_3\text{-N}$ . Knoepp and Swank (1993) explained that even though the response to treatment was significant,  $\text{NO}_3\text{-N}$  concentrations in solutions remained low, ranging from 0.02 to 0.50  $\text{mg L}^{-1}$ . In *Pinus echinata*-mixed oak ecosystems, Elliott et al. (2012) measured a greater response for several months after high severity fire, soil solution  $\text{NO}_3\text{-N}$  was above 3.0  $\text{mg L}^{-1}$  on cut +burn treatments and above 1.0  $\text{mg L}^{-1}$  on burn only treatments. Whereas, with low severity fire in *Pinus echinata*-mixed oak, Elliott and Vose (2005) found no detectable differences between control and burned sites for soil solution nutrients ( $\text{NH}_4\text{-N}$ ,  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4$ , Ca, Mg, and K) for 10 months after burning.

We expected that the effects of the 2007 wildfire followed by lime application would ameliorate the acidic, Ca depleted soils in LGW. Calcium and Mg are added via the ‘ash-bed-effect’ and surface lime application would further increase soil Ca and Mg making it available to the recovering vegetation (Knoepp and Swank 1997; Elliott et al. 2002). Liming has been used to remediate acidic soil conditions and base cation depletion resulting from long-term inputs of atmospheric deposition (e.g., Kreutzer 1995; Sharpe and Voorhees 2006; Moore and Ouimet 2010; Long et al. 2011; Cho et al. 2011). Surface soils (0–10 cm) and subsurface soils at LGW were acidic (pH of  $\leq 3.8$  and 4.0 for surface and subsurface, respectively). Liming decreased acidity somewhat and surface soil pH was significantly higher on the two burned sites with lime addition (pH=3.8) compared to the severely burned site without lime (pH=3.6). We

**Fig. 8** Soil solution aluminum (Al) at two soil depths: **a** shallow (15 cm) and **b** deep (>40 cm). Soil solution was collected from May 2008 to August 2009 on the five treatment areas. Treatments and code names are the same as in Fig. 2



measured soil exchangeable Ca concentrations an order of magnitude greater on limed sites compared to the reference site; soil Ca remained elevated into the second year of sampling.

We found that forest floor, soil and soil solution Ca concentrations on 2xSB were no different than concentrations on 1xMB or REF indicating that wildfire alone did not affect Ca pools or availability on this severely burned site. This finding does not support our hypothesis that wildfire would increase soil Ca via the “ash-bed-effect”, also it contrasts with studies that have reported increased soil Ca following burning (Certini 2005; Knoepp et al. 2005). Soil solution Ca of deep soil was 2–3 times higher on the limed sites than on 1xMB and REF. Solution Ca concentrations of deep soil were low on 1xMB and REF and comparable to the headwater stream ( $13.8 \mu\text{mol}_c \text{L}^{-1}$ ) of an undisturbed, low elevation watershed within LGW (Elliott et al. 2008).

In our study, the moderately burned site with lime application had greater foliar Ca than the severely burned sites while soil exchangeable and soil solution Ca did not differ, suggesting that the applied lime was subsequently taken up by vegetation. Deciduous leaf Ca for all treatments was comparable to foliar Ca concentrations found by Minocha et al. (2011) before addition of  $\sim 4.5 \text{ Mg ha}^{-1}$  wollastonite ( $\text{CaSiO}_3$ — $1.2 \text{ Mg ha}^{-1}$  of Ca; a more soluble form of Ca than dolomitic lime) to a small watershed at Hubbard Brook, New Hampshire. They reported a significant increase of more than  $3.00 \text{ g Ca kg}^{-1}$  for *Acer saccharum* (Marsh) foliage 5 years after the lime addition.

Other studies have found greater and long lasting improvement in soil pH and nutrient availability following the addition of a greater amount of dolomitic lime than was applied at LGW. For example, Long et al. (2011) showed that a single application of

22 Mg ha<sup>-1</sup> of dolomitic lime had a sustained effect on *Acer saccharum* (Marsh.) crown vigor, growth, and flower and seed production. Moore et al. (2008) found that dolomitic lime application in northern hardwood forests significantly altered soil chemical properties; soil pH, base saturation, and exchangeable Ca and Mg increased in a linear fashion as application rate increased from 2 to 20 Mg ha<sup>-1</sup> for 10 years after a single application. In a *Picea abies* (L. Karst.) stand, Kreutzer (1995) reported greater than two-fold increases in soil exchangeable Ca and Mg 4 years after an application rate of 4 Mg ha<sup>-1</sup> dolomitic lime. In comparison to these studies, application rate at LGW was relatively low ( $\approx$  1.1 Mg ha<sup>-1</sup> dolomitic lime) and may have been spatially variable due to the aerial application method.

Lime additions reduced soil exchangeable Al concentrations on 2xSBL, but not 2xMBL. The decrease in Al coupled with increased soil exchangeable Ca on both 2xSBL and 2xMBL, resulted in improved soil Ca/Al ratio due to lime addition on both sites. However, the soil Ca/Al ratio response was transitory, and soil exchangeable Al increased and Ca/Al ratio decreased over time for both the surface and subsurface soil depths. Liming may be a good tool to offset acidic deposition and improve cation availability in these soils; however, the Ca/Al ratios remained well below the toxicity threshold <1.0 (Cronan and Grigal 1995) in LGW where surface soil Ca/Al ratios ranged from 0.006 to 0.102 depending on treatment. Over all sites at LGW, soil exchangeable Ca was lower than reported for Ca depleted soils in the northeastern United States. For example, Bedison and Johnson (2010) showed decreased soil exchangeable Ca over a 70 year period in forests of the Adirondack Mountains; however, concentrations (1.5–1.6 cmol<sub>c</sub> Ca kg<sup>-1</sup>) were well above those at LGW. Soil exchangeable Ca concentrations at LGW were more comparable to those reported for acidic, soils in hardwood forests (0.7–0.8 cmol<sub>c</sub> Ca kg<sup>-1</sup>) in West Virginia (Farr et al. 2009), pine forests (0.05–0.23 cmol<sub>c</sub> Ca kg<sup>-1</sup>) in South Carolina (Markewitz et al. 1998), and hardwood forests (0.8 cmol<sub>c</sub> Ca kg<sup>-1</sup>) in Tennessee (Johnson et al. 2008).

Long et al. (2011) found that foliar Ca and Mg of *Acer saccharum* and *Prunus serotina* (Ehrh.) were approximately twice as high in sites with lime application compared to sites without lime; we found no foliar response to liming in comparison to reference sites.

However, their (Long et al. 2011) application rate was 22.4 Mg ha<sup>-1</sup> of dolomitic lime; 20 times the application rate in our study. In addition, we did not find significant relationships between soil exchangeable Ca and foliar Ca (evergreen  $R^2=0.0004$ ,  $P=0.9242$ ; deciduous  $R^2=0.1210$ ,  $P=0.0958$ ) or exchangeable Mg and foliar Mg (evergreen  $R^2=0.0499$ ,  $P=0.2939$ ; deciduous  $R^2=0.1219$ ,  $P=0.0945$ ) for samples collected in 2008, suggesting that lime application at this rate did not affect plant uptake of Ca and Mg.

## Conclusions

Linville Gorge Wilderness is one of numerous areas in the eastern United States that have been identified with depleted soil Ca and other base cations (e.g., Bailey et al. 2005; Elliott et al. 2008; Johnson et al. 2008; Warby et al. 2009; Bedison and Johnson 2010). Parent material low in cation-containing primary minerals along with chronic atmospheric deposition of N and S oxides, has resulted in removal of Ca via leaching and mobilization of soil Al (Fenn et al. 2006; Sullivan et al. 2011). Since 1990, there has been a substantial reduction in deposition of sulfur oxides and Ca, but little change in deposition of nitrogen oxides (Driscoll et al. 2001; Mitchell et al. 2011). We hypothesized that the wildfires that have occurred in LGW followed by application of dolomitic lime would add Ca and Mg, and consequently increase soil pH and reduce soil exchangeable Al. However, lime application rates were not sufficient to increase soil Ca and Mg availability to mitigate soil Al availability, increase the Ca/Al ratio above the toxicity threshold, or substantially increase plant uptake of Ca and Mg.

Higher lime application rates (e.g., 10–20 Mg ha<sup>-1</sup> dolomitic lime) may be necessary to obtain the substantial and long-term improvement of Ca-depleted soils at LGW, seen in other studies (Moore et al. 2008; Long et al. 2011). Lime application rate at LGW was relatively low and may have been spatially variable due to the aerial application method. Lime additions did increase soil exchangeable Ca and reduce exchangeable Al, which resulted in improved soil Ca/Al ratio. However, the Ca/Al ratio response was small and declined by the second year of sampling. These results show that liming could improve cation availability of these acidic soils, if the application rate was higher or lime was applied repeatedly.

**Acknowledgements** We thank the Grandfather Ranger District, Pisgah National Forest for their cooperation in establishing field sites. Special thanks to Patsy Clinton, Chris Sobek, Neal Muldoon, and Craig Stickney for assistance in field sampling and Cindi Brown and Carol Harper for chemical analyses of samples. Drs. Mary Beth Adams and Andrew Scott and two anonymous reviewers provided helpful comments on the manuscript. This research was supported by a Burned Area Emergency Response grant to William Jackson, Air Resource Specialist, Region 8, USDA Forest Service; Coweeta Hydrologic Laboratory, USDA Forest Service; and the Coweeta LTER project funded by National Science Foundation grant DEB-0823293. The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

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