Trace Metal Distribution and Mobility in Soils after Silvicultural Thinning and Burning

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Abstract

Silvicultural thinning and burning are common management practices that are widely used to address ecosystem problems such as tree stocking and general forest health. However, high-severity fire has variable effects on soils, resulting in damages which are directly or indirectly reflected on the trace metal chemistry of the soil. This study was conducted to evaluate the trace metal variation at the Bankhead National Forest in Northern Alabama following the silvicultural thinning and burning. The experimental site had treatments consisting of two burning patterns and three levels of thinning as part of an overall treatment of three burning patterns and three levels of thinning applied to nine treatment plots to fit a completely randomized block design experiment. Four treatments sites were used for this study and samples were collected from soil profile pits excavated at representative plots within each treatment. The samples were analyzed for trace metals—As, Cu, Ni, Zn and Pb—using Perkin Elmer 2100 ICP-OES. Post treatment samples indicated that the trace metal concentrations generally decreased with soil depth. Copper, Ni, and Zn at the Pre-burn site gradually increased with depth to a maximum concentration at about 50 cm below the soil surface. Arsenic in the surface horizons increased by 156% in the burn-only sites, 54% in the thin-only treatment, 30% for the burn and thin treatments. Such differences were unlikely due to differences in the geochemistry of the parent material, but likely due to anthropogenic activities and possibly the forest management practices in question.

Keywords: burning, forest soils, geochemistry, silviculture, thinning, trace elements

1. Introduction

Silvicultural practices such as mechanical or forest thinning and prescribed fires are common forest management practices that are widely used, either separately or in combination, to address ecosystem problems such as tree stocking, species composition, and wildland fire concerns in forests (Peterson et al., 2005; Hessburg, Povak, & Salter, 2008; Close, Davidson, & Swanborough, 2011) as well as restoration projects (North, Innes, & Zald, 2007; Mitic, Stankov Jovanovic, Ilic, & Nikolic Mandic, 2015).

Mechanical thinning has received a lot of attention as a savanna restoration treatment because it can be used to rapidly and selectively modify overstorey canopy structure, and increase understorey light availability thereby favoring establishment and increased abundance of light-demanding grasses and forbs (Peterson, Reich, & Wrage, 2007; Close et al., 2011; Mitic et al., 2015). Prescribed burning on the other hand has been described as a key tool used in prairie and savanna restoration projects (Emery & Gross, 2005). This is because biologists and land managers often use prescribed burning as a tool in prairie and savanna restoration projects because of the historical dependence of these systems on fire (Axelrod, 1985; Richardson, Friedland, Kaste, & Jackson, 2014; Burton et al., 2016).

Prescribed fires have been an important management practice for maintaining forest ecosystem health (Dietrich, 1971; USDA/USDI, 1995; Brown, 2000; Richardson, Donaldson, Kaste, & Friedland, 2015). It is the preplanned and controlled application of fire to fuels in either a natural or modified state under prescribed conditions of

weather, soil moisture, etc. (SAF, 1984). The prescription establishes the conditions needed to confine the fire to a predetermined area, and create heat intensity to achieve the desired burning objectives. While fire intensity is a measure of the amount of energy released by combustion over a given time interval, fire severity is a measure of the impact fire has on soils and vegetation. Severity is a function of burn duration, fuel loading, degree of oxidation, vegetation type, weather, topography, soil texture and moisture, soil organic matter content, time since last burn, the area burned, and burn intensity (Neary, Klopatek, DeBano, & Ffolliott, 1999; Santin, Doerr, Otero, & Chafer, 2015).

Following intense burning, soil's micro and macrofauna as well as watersheds can suffer immediate and long-term damage (USDA Forest Service, 2005; Wanthongchai, Goldammer, & Bauhus, 2011). This damage is directly or indirectly reflected on the trace metal chemistry of the soil (Kutiel & Shaviv, 1993; Gaines & Creed, 2003; Neary, Ryan, & DeBano, 2005). This is because burning has the potential to indirectly accelerate weathering, loss of soil as sediment, alter organic matter formation, and the permeability and water infiltration capacity of the soil (Debano & Conrad, 1978; Flannigan, Stocks, & Wotton, 2000; Santin et al., 2015) and invariably the trace metal dynamics. Burning also affects the soil productivity through loss of organic matter, and changes in soil physical properties, including the disappearance or death of soil biota through heating (Wright, 1976; Gaines & Creed, 2003). The loss of organic matter invariably results in the loss of nutrients and in the increases of the soil susceptibility to erosion. The heat could affect the soils structure and could indirectly affect the soils capacity to absorb water (Amaranthus & Trappe, 1993; USDA, 2005).

Wood ash or biomass ash happens to be one of the byproducts of the forest floor after a burn. While there is a consensus regarding the general composition of wood ash, there is hardly enough relating it to the ecosystem redistribution of trace metals due to the chemistry of wood ash. However, there is evidence that biomass ash contains valuable nutrients as well as environmentally harmful heavy metals and organic compounds, e.g. poly-cyclic aromatic hydrocarbons (PAH) (Tillman, 1994; Richardson et al., 2015). It has been shown that wood ash has a good acid neutralizing capacity and ability to supply the soil with base cations (Ca^{2+}, K^+, Mg^{2+}, Na^+), various concentrations of readily soluble neutral salts (sulphates and chlorides of K and Na), but also contains heavy metals (Cu, Zn, Mn, Pb, Cd, Cr) (Fritze, Smolander, Levula, Kitunen, & MaÈlkoÈnen, 1994; Eriksson, 1998; Levula, Saarsalmi, & Rautavaara, 2000; Saarsalmi, Mälkönen, & Piirainen, 2001).

Trace metals distribution and mobility in soils have also been shown to be influenced by the soil parent material, and soil acidity (Li & Shuman, 1996; De Oliveria, Fontes, Da Costa, & Horn, 2000; Bodi et al., 2014). In terrestrial ecosystems, trace metals are released from soil parent materials under the influence of chemical weathering (Alloway, 1990). Moreover, the chemical forms of the metals in the soil depend on the soil environment (*e.g.*, pH, redox potential, presence of organic matter, and colloidal clay particles). For example, a lowering of the pH of the soil may increase the solubility of trace metals. Water logging of soils, followed by reduction, may increase solubility of Fe and Mn, hence all metals associated with ferric or manganic oxides under aerobic conditions (McBride, 1994). Therefore, the chemical forms of the metals in contaminated soils play a significant role in their transport and bioavailability to both native and agronomic plants.

While several studies have been reported on soil properties in reference to nutrient changes to ecosystem management (Neary et al., 2005; Campos, Abrantes, Keizer, Vale, & Pereira, 2016), very little information is available in the literature concerning the impacts of prescribed burning on heavy or trace metal changes, particularly trace metal distribution, mobility and re-distribution. Forest fires influence heavy metal contents and bioavaliability through wood ash, which is a product of forest biomass burning.

The effects of these management practices have been studied extensively in relation to soil's nutrient dynamics (Neary et al., 1999; Carter & Foster, 2004; Wanthongchai, Bauhus, & Goldammer, 2008), carbon cycling, and climate change (Flannigan et al., 2000; Lal, 2005), as well as the demography of most forest plants (Emery & Gross, 2005; Hessburg et al., 2008). Amongst these studies, information regarding the dynamics of trace metals is scarce. The limited research relating to the forest ecosystem and trace metal dwells on the influence of the soil parent material and the application of wood ash as a fertilizer and its apparent effects on the concentration of some trace metals as part of its components (Demeyer et al., 2001; Miller et al., 2002; Bladon, Emelko, Silins, & Stone, 2014).

At the Bankhead National Forest in Alabama where ecosystem management practices such as prescribed burning and thinning are used to control undergrowth species population, the fate of trace metals in that environment is uncertain. Due to the fact that factors that control the chemical forms of the trace metals (pH, organic matter content, mineral composition and charge characteristics) could themselves be affected by the conditions of prescribed burning, it became necessary to investigate the impacts of the fire managed environment on the

distribution and chemical forms of trace metals. Therefore, the objective of this study was to evaluate the trace metal distribution and mobility in a forest ecosystem after a silvicultural management practice.

2. Materials and Methods

2.1 Site Description

The William B. Bankhead National Forest is located on the Southern Cumberland Plateau and extends all the way through three Alabama counties of Winston, Lawrence, and Franklin at latitude 34°30′N and longitude 87°30′W. The Bankhead National Forest is one of four national forests within Alabama and covers 73078.84 hectares (180,581 acres) in northwest Alabama. The Bankhead is managed for multiple uses, including recreation, timber, wildlife and fish, water and soil, wilderness and range. It is the largest remaining tract of unfragmented deciduous forest in the state, and continues to protect water quality and serve as a watershed to local municipalities. The Bankhead National Forest has cultural and historical significance to residents and surrounding communities, including its spiritual significance to the Native American tribal members, and scientific and research significance to the scientific community notably the Natural Resources Leadership Institute (NRLI).

The management of the forest dates back to the beginning of the 1960s when the Forest Service initiated efforts to improve forest economic production by replacing some upland hardwood forests with faster growing loblolly pine—*Pinus taeda* (Gaines & Creed, 2003). This practice resulted in about 31,970 hectares (79,000 acres) of the 73,653 hectares (182,000 acres) of the forest covered with loblolly pine. While loblolly pine is a native tree species to the Southern Appalachian region, the dominance of pure stands of loblolly pine is not typical of native landscape of oak forests and fire dependent woodlands that occur in the uplands of the Cumberland Plateau (Gaines & Creed, 2003).

Over the past decade, the Bankhead National Forest experienced Southern pine beetles—*Dendroctonus frontalis*—infestations at epidemic levels, primarily in loblolly pine forests. An estimated 7,527 hectares (18,600 acres) of pine forest have been damaged by this epidemic. Most of the mortality occurred within the Sipsey wilderness and other areas where suppression efforts did not take place. This epidemic has resulted in large hectares of standing dead trees that are a public safety hazard thus escalating the risk of catastrophic wildfires in the future (Gaines & Creed, 2003).

The soils at the research sites were classified as Typic Hapludults of the Sipsey (fine-loamy, siliceous, semiactive, thermic Typic Hapludults) series in the USDA-NRCS preliminary soil map of Lawrence County (Nobles, Dillon Jr., & Mbila, 2009), and are known to contain sandstone, siltstone and shale (USDA Soil Conservation Service, 1959).

2.2 Treatments and Sampling

The treatments at the sites of this study consisted of two burning patterns (no-burn and 3 year-burn) and three levels of thinning (no-thin, 50% thin and 50% thin). The no-burn treatment means that the plot was not treated to any form of prescribed fire while the 3 year-burn indicates that the plot was intermittently burned after three years. The no-thin treatment constitutes no silvicultural logging of any form in the plot, while 50% thin have half of the plot logged according to silvicultural thinning. Treatment 1 (control), located inside the Sipsey wilderness area, was neither burned nor thinned (Table 1). This study was part of an overall treatment of three burning patterns (no-burn, 3 year-burn, and 10 year-burn) and three levels of thinning (no thin, 25% thin, and 50% thin) applied to nine treatment plots to fit a completely randomized block design experiment. Each treatment plot was about 10 hectares (25 acres). In all, four treatments were selected for this study. The selection was based on the research timing and interest. At the time of the research, the 3-year burn had been carried out twice while the pending 10-year burning was part of another study. Since we were more interested in seeing the effect of the burning than the logging, we chose a higher logging treatment (50% logging) over the lower treatment (25% logging) to be compared with control (No burning; No Logging). The research focus was therefore; (1) Control: No logging, No burning; (2) No logging, 3-year burning; (3) 50% Logging, No burning; and (4) 50% logging, 3-year burning. These were treatments 1, 3, 4, and 6 highlighted in Table 1. All treatments were performed by the Forest Service personnel and private logging companies.

Table 1. Lists of Prescribed Treatment at the study sites in Bankhead National Forest, AL

Treatment Area Number	Name	Thin Year	Burn Year
1	**Control**	**No Thin**	**No Burn**
2	No-Thin—10 year Burn	No Thin	2006
3	**No-Thin—3 year Burn**	**2005**	**2006**
4	**50 % Thin—No Burn**	**2005**	**No Burn**
5	25 % Thin—No Burn	2005	No Burn
6	**50 % Thin—3 year Burn**	**2005**	**2006**
7	25 % Thin—3 year Burn	2005	2006
8	50 % Thin—10 year Burn	2005	2006
9	25 % Thin—10 year Burn	2005	2006

At each treatment plot, a soil profile pit was excavated at representative locations within the treatment plot and described according to methods of the Soil Survey Staff (2006). Two sets of soil samples were collected by horizons from the soil profile pits—the pre-treatment and post-treatment samples—for analyses. Additional auger core samples were taken according to the depths of the soil horizons. The soil surface and the subsurface horizon soil samples were collected to monitor chemical differences. Bulk samples were also collected and used to run routine characterization of soils. Three of the four experimental blocks were sampled for this study.

2.3 Soil Sample Preparation

The soil samples collected were spread out on clean moisture-proof papers, placed on clean laboratory tables to air-dry. The dried samples were poured in a porcelain mortar and gently crushed with a porcelain pestle, then passed through a 2 mm standard soil sieve. The sieved samples were re-bagged, properly labeled, and stored for the necessary laboratory analyses.

2.4 Laboratory Analyses

The soil samples that were previously prepared were analyzed for trace metal contents by utilizing modified techniques from Miller and McFee (1983), and the Microwave Accelerated Reaction System (MARS) using the EPA method #3052 for complete digestion. With Miller and McFee, using open vessel digestion, total Cd, Zn, Cu, and Pb were solubilized from soil samples by digesting duplicate subsamples (0.5-2 g) in 250-mL flasks with 15 ml of concentrated HNO_3 on a hot plate (100 °C) overnight. We accomplished total solubilization from soil samples by utilizing a combination of different reagents (EPA method #3052) including 30% Hydrogen Peroxide (H_2O_2), Nitric Acid (HNO_3), and Hydrofluoric Acid (HF) in a closed vessel MARSXpress digestion within a total time of about 20 minutes and desired temperature of about 180 °C. The samples were cooled, filtered through #42 Ashless Whatmann Filter paper and stored for analyses. The elemental concentrations of the extracts from the soil samples were determined with a Perkin Elmer 2100 ICP-OES (inductively coupled plasma-optical emission spectroscopy) following the standard operating procedures required for the ICP.

2.5 Statistical Analyses

All statistical analyses were performed with the general linear model (GLM) procedure of the statistical analyses systems (SAS) package version 9.3 using a significant level of 0.05 (SAS Institute, 2008). The analysis of variance (ANOVA) procedure was employed to calculate the mean concentrations of the assessed parameters focusing on different class information as required for the main aspect of the ecosystems analyzed. The Tukey Studentized Range (HSD) Test was used for the mean separations. The Tukey test was chosen because we wanted to compute means of dependent variables which had equal sample size. Since multiple comparisons were being made simultaneously to the treatments, Tukey was preferred over T-Test.

3. Results and Discussion

3.1 Trace Metal Distribution in Control Site

Geochemical levels of trace metals in soils at the research sites were determined in soils of the control sites (Treatment 1), where no-burning and no- thinning treatments were applied. Soils from the control plots and the burning and thinning-managed soils were sampled at the same times, that is, during and after treatment (Figure 1).

In the control sites, the mean soil surface horizon concentrations of trace metals for the blocks ranged from 15-25 μg g^{-1} (As), 1-3 μg g^{-1} (Cu), 3-5 μg g^{-1} (Ni), 11-49 μg g^{-1} (Zn), and 12-22 μg g^{-1} (Pb). Post treatment

samples contained 15-24 µg g^{-1} (As), 0.4-6 µg g^{-1} (Cu), 3-7 µg g^{-1} (Ni), 11-26 µg g^{-1} (Zn), and 10-26 µg g^{-1} (Pb). There was no clear trend about the vertical distribution of the trace metals with depth along the soil profile, but the trace metal concentrations generally decreased with soil depth. There was no apparent difference between the trace metal concentrations from the pre and post treatment soil samples except for Zinc. Zinc showed an obvious reduced range from 11-49 µg g^{-1} to 11-26 µg g^{-1}. Soil Zn depletion could be due to Zn uptake by forest vegetation. Zn is a major component of plant growth hormone production and seed development. Plant Zn need and uptake from soil solution could explain the differences between the pre and post treatment period in Zn concentrations. There are no known trace metal threshold levels for forest soils but some studies have reported concentrations that are comparable to our results. For instance Kabata-Pendias and Pendias (2001) reported the range of trace metal contents of world forest soils as 25-156 µg g^{-1} (Zn), 7-150 µg g^{-1} (Cu), 7-100 µg g^{-1} (Ni), 1-93 µg g^{-1} (As), and 10-50 µg g^{-1} (Pb). Thus the levels of trace metals in the untreated soils were within the range previously reported for forest soils.
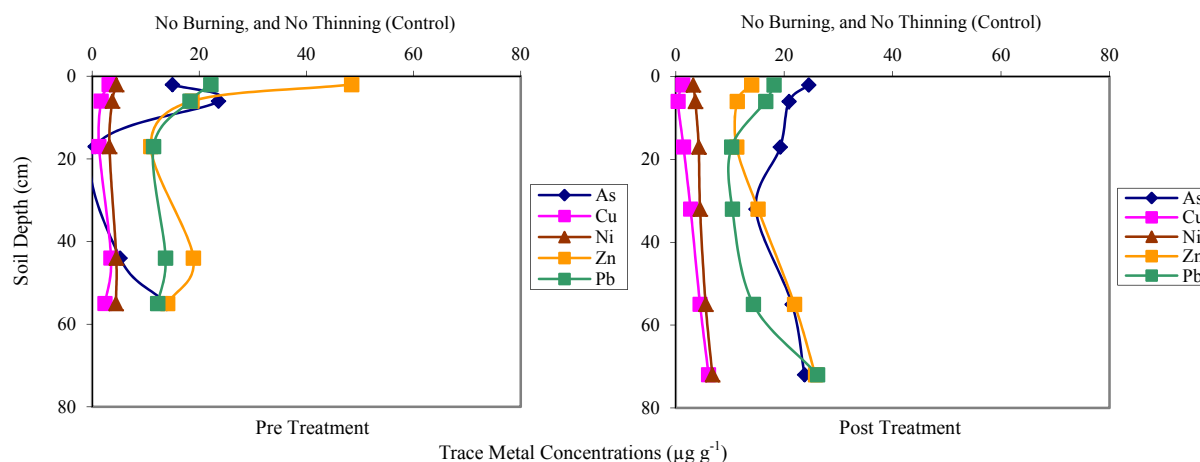

Figure 1. Trace metal distributions in the control plot at Bankhead National Forest

3.2 Vertical Distribution of Trace Metals at the Pre-Treatment Plot

Prior to burning and thinning treatments at the sites, soil samples were collected from the sites for analyses that were to be compared with samples collected after burning and/or thinning. Analyses of the samples from pre-treated plots (pre-burned, pre-thinned, and pre - burned and thinned) were based on averages of data from the blocks of the study. Firstly, for the burned only site (Figure 2a), trace metal concentrations ranged from 2-25 µg g^{-1} for As, 2-8 µg g^{-1} for Cu, < 1-8 µg g^{-1} for Ni, 9-14 µg g^{-1} for Pb, and 4 -15 µg g^{-1} for Zn. Copper, Ni, and Zn gradually increased with depth to a maximum concentration at about 45 cm below the soil surface. Secondly, lead concentration indicated a dual peak maximum with depth; one maximum at about 10 cm, and the other, at about 45 cm below the soil surface.

Several studies have reported the organic fraction and organic matter as being an important storehouse of trace metals (Mbila, Thomson, Mbagwu, & Laird, 2001; Campos et al., 2016). The property of the soil is also important in the behavior of the metals. Burning reduces the organic matter fraction and the retentive ability of the metals ensuring their mobility and redistribution down the profile. However, the calcareous nature of the soil could explain the uneven mobility and distributions of the individual metals.

The As levels in the soil were much higher than those of Cu, Ni, Pb, and Zn with the highest concentrations at about 25 cm, and the lowest concentrations at the lowest soil layer. While it is normal for trace metals to have individual patterns of reactions and mobility, the obvious accumulation of As down the soil profile and further tapering at the 30 cm point could indicate a connection to the prescribed burning.

As seen in Figure 2b, pretreatment samples from the no-burning and 50% thinning sites showed very similar trends for Cu, Ni, Zn, and Pb, with soil surface horizons concentrations of the metals that increased with soil depth to a maximum at 30-40 cm. The As concentrations sharply increased from the soil surface to a maximum concentration at about 20 cm from the soil and decreased with soil depth as was observed in the burning only treatment. The thinning treatment for Figure 2b increased the downward mobility of the metals except As by more than 15cm from Figure 2a to 75 cm. This could be due to the reduced organic matter and erosion caused by

logging, thus reducing the natural geochemical barrier of the surface and subsurface soil. Again, As is an anomaly, exhibiting a greater undulating accumulations from the 5 cm to the 25 cm point and ending at the 50 cm.
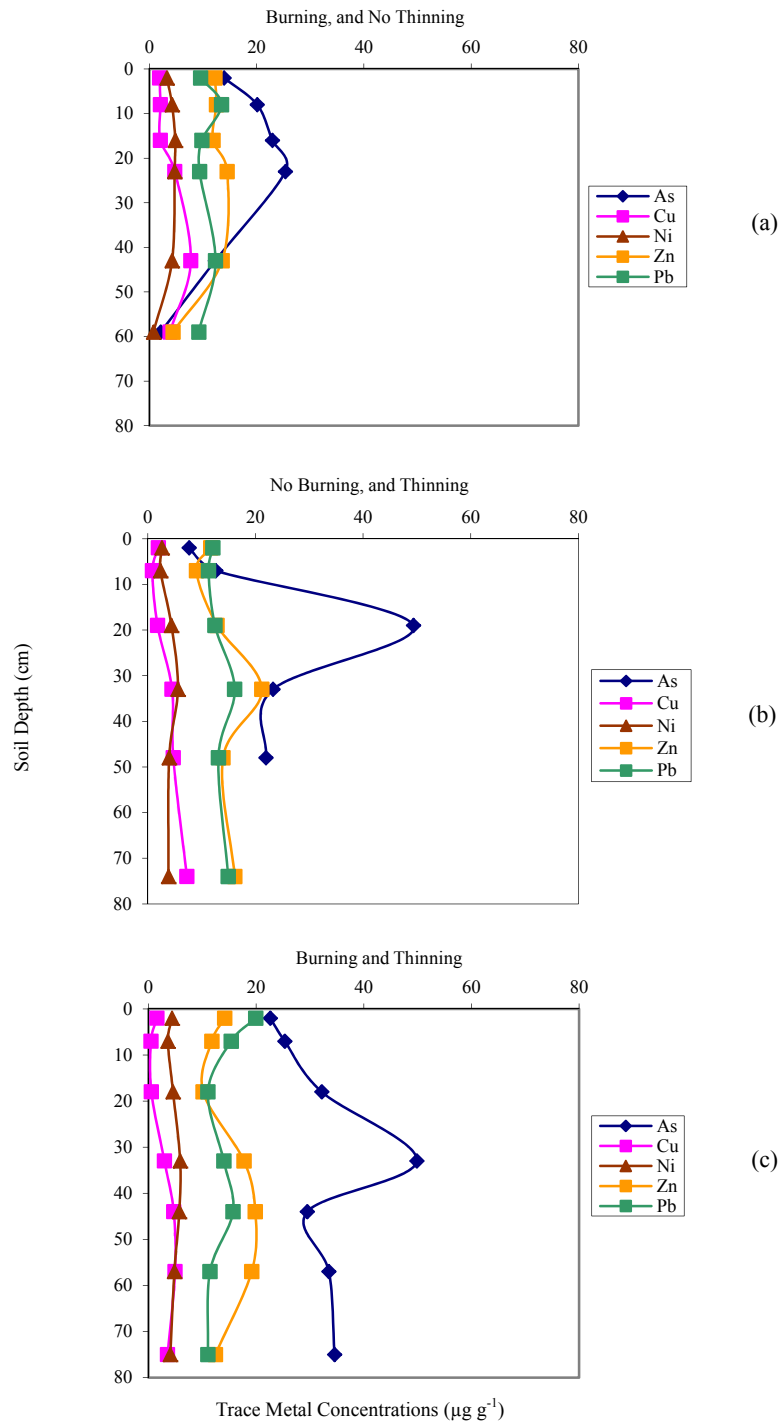
Figure 2. Vertical distribution of the metals at the pre-treatment plot

For the combined burning-thinning treatment, Cu, Ni, Zn, and Pb concentrations decreased slightly from the soil surface horizon, and then increased to their maximum concentrations at about 35-50 cm below the soil surface horizon (Figure 2c). The Arsenic concentrations sharply increased from the soil surface horizon concentrations to

a peak at about 35 cm soil depth and decreased irregularly with depth. Consequently, the distribution of the metals in the sites prior to treatments was highly variable in concentrations, according to the elements, but indicated a general trend of higher concentrations below the soil surface horizon. The burning and thinning treatment provided the most downward mobility for As relative to Figures 2a and 2b signifying a possible As loading relating to the treatments.

Compared to the distribution of the trace metals in soils of the control (no burning, no thinning) site, whereas the As concentration exhibited sharp decreases in soil surface horizon concentrations with depth in the control soils, it showed very sharp increases in concentrations with soil depth in the pretreatment soil samples. The differences in the distribution especially of As concentration could be suggestive of differences in the geochemistry of the geologic formations that formed the soils at the research sites. While there are very wide differences in the As concentrations and other elements reported in literature for soils and rocks (Brown & Thomas, 2014; de Caritat & Reimann, 2012), attributing the high As concentrations to the parent material cannot be substantiated by our data. Alternatively, high As content in the pretreatment samples could be due to differences in the management practices within the forest sites that may involve the use of agrochemicals with some As concentrations. In this sense, Arsenic is used in forestry as wood preservative to prevent rot and insect damage such as the southern pine beetle infestation that has been a problem at the research sites.

3.3 Vertical Distribution of Trace Metals at the Post-Treatment Plot

The post-treatment plot exhibited a vertical distribution that was somewhat similar to the pre-treatment plot for Cu, Ni, Zn and Pb. The only obvious difference was the distribution of As for the variants in Figures 3a, 3b, and 3c. Arsenic in the soil surface horizons increased after the treatment by 156% in the burn-only sites, 54% in the thin-only treatment, and 30% in the burn and thin treatment (Figures 3a, 3b, and 3c). Such differences in concentrations were unlikely due to differences in the geochemistry of the parent material, but were likely due to anthropogenic activities and possibly the forest management practices in question. Moreover, the effects of fire on trace elements may be long lasting due to the strong affinity for trace elements of ash and fine sediments (Blandon et al., 2014).
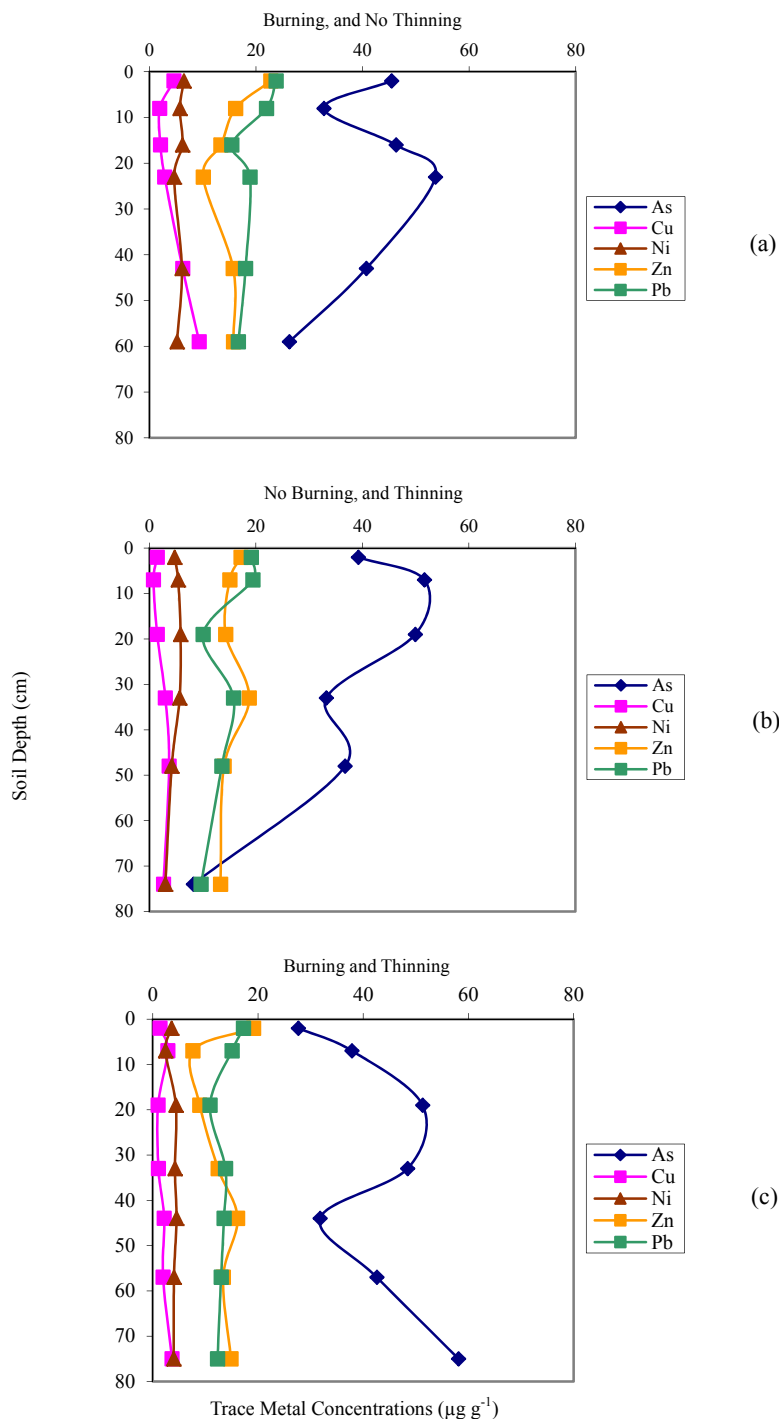
Figure 3. Vertical distribution of the metals at the post-treatment plot

3.4 Comparing Metal Redistributions Patterns at the Sites

The data comparison between the post treatment soil samples and the pretreatment data showed that the trace metal concentration of soils at the sites was not significantly different ($p < 0.1$) as shown in Table 2. However, a temporal variability of the post treatment data compared with the pretreatment data (time comparison), indicated that the As concentration was significantly different ($p < 0.1$) among the sites, while the Pb and Ni concentrations were significantly different ($p < 0.5$) among the sites. Copper and Zn concentrations were not significantly different among the sites (Table 2).

We originally hypothesized that trace metal concentrations of the control plots could provide a good measure of the geochemical levels of trace metals in the soil, and therefore, it could be compared with the concentrations in the treated plots. None the less, the range of concentrations and the degree of variability in soil profile distribution of the trace metals in the control soil suggested that this soil might have concentrations of some of the elements, *e.g.*, Zn and Pb which were much higher than those of the treated plots. We attribute those differences to slight differences in the geochemistry of the parent material of the soils at the site.

The significant differences in the concentration of As, Pb, and Ni in the pretreatment and post treatment samples suggested that a major source of the trace metals at the research sites seemed to be due to the anthropogenic treatments of prescribed burning and/or thinning.

Table 2. Statistical analysis of the variations in the treatments at the Bankhead National Forest

Class	As	Pb	Ni	Cu	Zn
Treatments	NS	NS	NS	NS	NS
Time	**	*	*	NS	NS
Horizons	NS	**	NS	***	NS

Note. NS = Not Significant; *, **, *** = Significance at the 0.05, 0.01, 0.001 level.

In a separate study that was carried out at the research site in order to evaluate the response of soil C and N pools to prescribed burning in the forest, plots subjected to burning alone or a combination of burning and thinning experienced decrease in Oi horizon weight due to the loss of forest floor material during the fire (Nobles et al., 2009). Thus a combination of burning and thinning led to organic matter combustion and ash addition during the fire. This possibly contained trace metals that were redistributed in the soil profile. Such observations about trace metals distribution elsewhere have been reported by others (Saarsalmi et al., 2001; Mitic et al., 2015; Campos et al., 2016).

The observation that burning and thinning treatments resulted in trace metal loading of soils suggested that trace metal dynamics at the sites probably depended on the vegetation biogeochemical cycling patterns. For instance, trace metals that originate from the soil parent material and in equilibrium with the soil solution supply trace metals to the tree stands. In turn, the trace metals are translocated to the biomass that ultimately burned and ashed during treatments. Thus we suggested that in such ecosystem, there will be a net increase of trace metals in the ecosystem via rock and mineral weathering and plant uptake to organic matter decomposition and cycling. Such might be the case for Pb and Ni concentrations.

4. Conclusion

We investigated the impacts of prescribed burning and silvicultural thinning on trace metal release and redistribution in forest soils at the Bankhead National Forest. In this national forest, forest management to improve and maintain the overall forest health, and productivity, required over populated pine stands to be thinned and other areas to be burned. With the tendencies to generate trace metals and contaminate both soil and surrounding waterways, the study evaluated trace metal dynamics in the soil of the forest ecosystem.

In the control sites, there was no clear trend about the vertical distribution of the trace metals in the soil profile, but the metals generally tended to decrease with depth. The levels of the trace metals in the untreated soils were within the range previously reported for forest soils. The metal distribution in the sites prior to treatments was highly variable in concentrations, and according to the different elements, but a general trend of higher concentrations below the soil surface horizon was indicated. In particular, Arsenic exhibited sharp decreases in soil surface horizon concentrations with depth in the control soils, but it showed very sharp increases in concentrations with soil depth in the pretreatment soil samples.

There were significant differences in the concentration of As, Pb, and Ni in the pretreatment and post treatment samples that suggested a major loading of the trace metals at the research sites: this was probably associated with the organic matter combustion and ash addition during the prescribed burning that possibly contained trace metals redistributed in the soil profile. The research showed that practices employed to manage the forest health could increase the concentration and redistribution of trace metals in the forest or wildland soils. A progressive analysis combining the results of the 10 year burn with these could give a comprehensive effect of prescribed burning on trace metal dynamics in the long run.

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