Forest Ecology and Management 411 (2018) 82–89

Contents lists available at ScienceDirect

Forest Ecology and Management

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

Prescription side effects: Long-term, high-frequency controlled burning enhances nitrogen availability in an Illinois oak-dominated forest

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A R T I C L E I N F O

Keywords:
Stoichiometry
Carbon
Nitrogen
Phosphorous
Enzyme activity
Microbial biomass
Fire
Soil

A B S T R A C T

Controlled burning is a common management technique used to control invasive understory plants and promote oak regeneration throughout the prairie-forest ecotone and across eastern deciduous forests. However, prescribed burning effects on oak forest and woodland soils remain largely unknown. As fire has significant and often disproportionate impacts on soil carbon (C), nitrogen (N), and phosphorus (P), evaluating the impacts of repeated, low-intensity prescribed burning on oak forest soil is fundamental for understanding and predicting burning effects on vegetation dynamics and ecosystem functions. The goal of this study was to evaluate relative changes in soil C and nutrient dynamics in response to three decades of annual, low-intensity controlled burning in an oak-dominated forest in northern Illinois. We found that annual burning increased soil organic and inorganic N availability and microbial biomass N. Burning also increased N cycling rates and N-degrading enzyme activity in the 5–15 cm soil layer. Surprisingly, annual burning had little effect on soil P pools and fluxes, likely due to intrinsically high soil P availability. Similarly, though total soil C increased, burning did not alter available C concentrations, microbial biomass C, C-degrading enzyme activities, or C mineralization rates. As such, annual burning created a positive feedback on inorganic N production, altering relationships among C, N, and P. In contrast with management goals, controlled burning often fails to enhance oak proliferation or decrease the abundance of invasive plants. As Eastern forests were historically N-limited, our study suggests a potential mechanism behind these restoration outcomes: frequent, low-intensity burning may produce soil environments that are incompatible with restoration goals.

1. Introduction

Controlled burning is a common forest management technique used to promote oak regeneration and control invasive understory plants. Prior to European settlement, low-intensity surface fires were common throughout the Eastern US, but concurrent with fire suppression, oaks have increasingly been replaced by fire-intolerant tree species, such as maples, and invasive plants (Abrams, 1992; Bowles et al., 2003; Hutchinson et al., 2008). In order to restore oak forests and woodlands, land managers have increasingly incorporated prescribed burns into their management plans (Blake and Schuette, 2000; Brose et al., 2013; Gilbert et al., 2003; Kruger and Reich, 1997). Despite this increase in prescribed burning, restoration goals of enhancing oak regeneration and decreasing invasive species abundance are often not met (Bowles et al., 2007; Huddle and Pallardy, 1996; Hutchinson et al., 2005; Wendel and Smith, 1986). Furthermore, though fire commonly impacts soil carbon (C), nitrogen (N), and phosphorous (P) dynamics, prescribed burning effects on oak forest and woodland soils remain largely unknown. As such, evaluating the impacts of repeated, low-intensity prescribed burning on oak forest soils is important for understanding and predicting burning effects on vegetation dynamics and ecosystem functions.

Soil responses to fire vary with fire intensity, frequency, and vegetation type, making it challenging to predict how soils in oak forests and woodlands will respond to frequent, low-intensity fires (Certini, 2005; González-Pérez et al., 2004; Johnson and Curtis, 2001). For example, while soil organic matter frequently decreases in response to high-intensity wildfires (Certini, 2005; González-Pérez et al., 2004; Neary et al., 1999), low-intensity fires may result in increased soil organic matter due to increased recalcitrance of charred organic matter and incomplete volatilization of organic compounds at low temperatures (González-Pérez et al., 2004; Knicker, 2007; Marschner et al., 2008; Neary et al., 1999; Prieto-Fernández et al., 2004). Furthermore, though higher frequency burns have different impacts on forests than single or low-frequency burns (Chen et al., 2009; Hernández and Hobbie, 2008; Neill et al., 2007; Tester, 1989; Williams et al., 2012), most studies are

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https://doi.org/10.1016/j.foreco.2017.12.041
Received 3 November 2017; Received in revised form 21 December 2017; Accepted 22 December 2017

short-term or from ecosystems that have experienced limited burning. Finally, effects on soils have been found to vary dramatically among ecosystems (Boerner et al., 2008a; Wan et al., 2001). Oak forests represent the majority of forestland in the Eastern US (Smith et al., 2009), and the frequency and scale of controlled burning is projected to increase (Illinois TNC National Interagency Fire Center, 2012; Saxton et al., 2016). As such, it is important to evaluate how oak forest soils respond to long-term, frequent controlled burning.

Prescribed burning may alter C, N, and P quantity, quality, and availability. Because C and N volatilize at lower temperatures than P, low-severity fires may enhance C and N limitation relative to P (Certini, 2005; Prieto-Fernández et al., 2004; Toberman et al., 2014). However, in addition to changes in quantity, controlled burns also have the potential to alter the form and availability of soil nutrients (Menaut et al., 1993; Raison, 1979). For instance, although total N often decreases in a high temperature fire, a significant proportion of the remaining N may become more available to plants and microbes after a fire due to fire-driven conversion of organic N to inorganic N (Bell and Binkley, 1989; Certini, 2005; Hernández and Hobbie, 2008; Pyne, 2001). Furthermore, soil pH is generally higher in burned soils than unburned soils after many different types of fire (Certini, 2005; González-Pérez et al., 2004; Knicker, 2007; Miesel et al., 2012; Scharenbroch, et al., 2012). The increase in pH may affect soil nutrient availability, including phosphate, which is most available in higher pH ranges due to the release of hydrogen cations (Riley and Barber, 1971). Finally, shifts in the microbial community and changes in microbial biomass may alter nutrient cycling rates and relative nutrient demand (Boerner et al., 2008a; Miesel et al., 2012; Reich et al., 2001). For instance, C-, N-, and P-degrading enzymes may respond positively, negatively, or neutrally to controlled burning (Boerner et al., 2008a; Certini, 2005; Eivazi and Bayan, 1996; Miesel et al., 2012; Rietl and Jackson, 2012). As such, fire alters the nutrient composition of soil, although these effects vary based on location and type of fire.

The goal of this study was to evaluate relative changes in soil C and nutrient dynamics in response to repeated low-intensity controlled burning in an Illinois oak-dominated forest. Plant community dynamics and ecosystem functions may be more responsive to relative changes in nutrient and resource availability than to total changes (Midgley and Phillips, 2016; Toberman et al., 2014). As such, we used a stoichiometric approach to analyze soil biogeochemical responses to controlled burning in forest plots and to better understand the microbial dynamics of burned soils. Based on existing literature, we hypothesized that organic matter and pH would increase in burned soils. Because of the exhibited effects of burning on C, N, and P concentrations, availability, enzyme activities, and microbial biomass, we expected repeated burning to alter the stoichiometric ratios of these factors in the soil. Specifically, we hypothesized that the relatively low volatilization temperatures of C and N compared to P would lead to reductions of C and N relative to P in burned soils. Additionally, we hypothesized that the enzymes that degrade these elements would be affected by the burning practices and by the relative limitation of C and N, leading to increased activities of C- and N-degrading enzymes relative to P-degrading enzymes.

2. Methods

2.1. Site description

This study was conducted in the East Woods of The Morton Arboretum, a deciduous hardwood forest in DuPage County, Illinois, USA (41°49′N, 88°3′W). Prior to European settlement, the East Woods consisted mainly of burr and white oak savannah in addition to prairie and oak-dominated deciduous forest (Bowles et al., 1994). Following settlement, 'high grading' or selective cutting of large diameter trees occurred followed by some clear cutting in 1917 (Wilhelm, 1991). The East Woods is currently a dense forest (basal area of $28 \pm 3\,ha^{-1}$) dominated by a mixture of oaks (*Quercus* spp.), sugar maple (*Acer saccharum* Marsh), and American basswood (*Tilia Americana* L.; Carter et al., 2015). The region has a continental climate with temperatures as low as -6°C in January and up to 22 °C in July, with 800–1000 mm mean annual precipitation. Soils are deep and moderately- to poorly-drained Alfisols formed from a thin layer of loess (0.3–1 m) underlied by glacial till. The major soil series are Beecher and Ozaukee silt loams.

The Morton Arboretum has conducted annual controlled burns in the East Woods since 1985 (Wilhelm and Masters, 1994). Our study area consisted of adjacent ~25 ha management units that were either annually burned (south) or unburned (north). Treatments were separated by an east-west road that served as a firebreak. Most burns were conducted in fall, though weather conditions delayed some fires until spring. The most recent burn prior to data collection occurred in spring 2015, 14 months prior to sampling. Fuel is primarily leaf litter and persistent herbaceous vegetation, and previous studies have reported 30–100% loss of litter mass (Bowles et al., 2007; Scharenbroch et al., 2012). Multiple infrared thermometer assessments of fire temperatures in 2003 and 2009 indicate that mean surface fire temperatures consistently range from 120 to 230 °C (Jacobs et al., 2004; Scharenbroch et al., 2012). Fires at our site are considered to be low- to moderate-intensity fires (Franklin et al., 1997) or cool fires (Roberts, 2004). Soil organic C is volatilized around 100–200 °C, and N starts to volatilize above 200 °C whereas P does not volatilize at temperatures below about 760 °C (González-Pérez,et al., 2004).

As reported by Bowles et al. (2007), in 1988, shrub densities were 4600 stems ha^{-1}, and the shrub layer was dominated by *Prunus virginiana*, *Lonicera mackii*, and *Ribes missouriense* (Bowles et al., 2007). Sub-canopy trees were dominated by *Rhamnus cathartica*, *Tilia americana*, and *Ostrya virginiana*. By 2002, woody stem density was reduced by 97% in the burned area while shrub layer stem density remained at 4400 stems ha^{-1} in the unburned area. In addition, percent cover of the ground layer vegetation was initially ~80%, and ground layer vegetation was dominated by *Erythronium albidum*, *Smilaciana racemose*, and *Pathenocissus* sp. in both the burned and unburned areas. However, elimination of the shrub and small sapling layer reduced light competition from this strata; burning induced a significant shift in ground layer vegetation toward greater abundance of summer herbs without a decline in spring herbs, leading to an increase in the relative abundance of *Alliaria petiolata*, *Helianthus strumosus*, and *Impatiens* sp. in the burned area. In addition, by 2002, percent cover of the ground layer vegetation was reduced to ~ 70% in the burned area to ~ 40% in the unburned area.

We established 32 15 x 15 m plots, 14 in the burned area and 18 in the unburned area. Our goal was to establish a “mycorrhizal gradient” in each management unit as previous research shows that tree type, classified by tree mycorrhizal association, can mediate environmental change effects on the soil (Midgley and Phillips, 2016). Based on previous surveys of the site, we targeted plots with varying abundances of trees associated with either arbuscular (AM) or ectomycorrhizal (ECM) fungi in both the burned and unburned areas, which resulted in our unbalanced design. For trees > 10 cm diameter at breast height (DBH), we measured DBH and identified the trees to species. The most abundant tree species were sugar maple, white oak (*Quercus alba* L.), and ironwood (*Ostrya virginiana* Mill.). We calculated the percent ECM trees for each plot based on basal area and known tree species mycorrhizal associations (Brundrett et al., 1990; Wang and Qiu, 2006).

2.2. Soil sampling

All sampling was performed in a 12 × 12 m internal plot to avoid edge effects. The internal plot was divided into four quadrants for soil sampling. Each quadrant was further divided into 36 squares, and one square within each quadrant was randomly chosen for soil sampling for a total of 4 samples per plot. In June 2016, we collected four cores from the top 15 cm of the soil from each plot with a 5 cm diameter stainless