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Post-Fire Comparisons of Forest Floor and Soil Carbon, Nitrogen, and Mercury Pools with Fire Severity Indices

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Forest fires are important contributors of C, N, and Hg to the atmosphere. In the fall of 2011, a large wildfire occurred in northern Minnesota and we were able to quickly access the area to sample the forest floor and mineral soil for C, N, and Hg pools. When compared with unburned reference soils, the mean loss of C resulting from fire in the forest floor and the upper 20 cm of mineral soil was 19.3 Mg ha^{-1}, for N the mean loss was 0.17 Mg ha^{-1}, and for Hg the mean loss was 9.3 g ha^{-1}. To assess the influence of fire severity on the forest floor and mineral soils, we used an established method that included a soil burn severity index and a tree burn severity index with a gradient of severity classes. It was apparent that the unburned reference class had greater forest floor C, N, and Hg pools and higher C/N ratios than the burned classes. The C/N ratios of the 0- to 10- and 10- to 20-cm mineral soils in the unburned reference class were also greater than in the burned classes, indicating that a small amount of C was lost and/or N was gained, potentially through leaching unburned forest floor material. However, with a couple of exceptions, the severity classes were unable to differentiate the forest floor and mineral soil impacts among soil burn and tree burn severity indices. Developing burn severity indices that are reflective of soil elemental impacts is an important first step in scaling ecosystem impacts both within and across fire events.

Abbreviations: FIA, Forest Inventory and Analysis; THg, total mercury.

Fires liberate forest floor and upper mineral soil C, N, and Hg, which can influence atmospheric concentrations of greenhouse gases and the cycling of C, N, and Hg (Amiro et al., 2001; Kasischke et al., 2005). Losses of C, N, and Hg as a result of fire also affects short- and long-term soil productivity and watershed storage of Hg, with implications for bioaccumulation in aquatic systems (Gabriel et al., 2009; Woodruff and Cannon, 2010; Miesel et al., 2012). Because CO_2 is emitted to the atmosphere during fire, studies have shown that the short-term impacts of fire on the ecosystem system C balance are negative (e.g., Hurteau and Brooks, 2011). Nitrogen pools also tend to decrease immediately following fire because of the volatilization of N species during fire (e.g., Caldwell et al., 2002); however, in some cases, N availability and overall site productivity can increase shortly after fire (e.g., Kutiel and Naveh, 1987). Miesel et al. (2012) conducted a recent meta-analysis on the effects of fires on forest floor and soil C and N in the Lake States region of North America, including both boreal and temperate ecosystems. Forest floor and mineral soil C and N pools or concentration data from burned sites were compared with unburned reference sites. Their results indicated

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that forest floor C decreased from 19 to 47% in the short term (<5 yr following fire), while upper mineral soil C ranged from an increase of 16% to a decrease of 16% during the same time period (Miesel et al., 2012). Similarly, forest floor total N decreased from 3 to 61% following fire and ranged from an increase of 24% to a decrease of 16% in mineral soils. The forest floor layer is therefore much more impacted by fire than mineral soils and, in the short term, the forest floor layer is generally a source of C and N to the atmosphere (Certini, 2005). The response of mineral soils tends to be variable, but they are generally small sinks or sources of C and N following fire, the balance probably a result of the intensity of the fire (Miesel et al., 2012).

Mercury is of great concern in aquatic systems due to its transformation into the extremely toxic methyl mercury that bioaccumulates within the food chain. Studies in northern Minnesota have indicated that watersheds that have forest floor and upper mineral soil horizons with higher C concentrations and associated total mercury (THg) concentrations tend to have fish with higher THg concentrations (Gabriel et al., 2012). Hence, volatilization of C and associated THg as a result of fire could lead to long-term decreases in Hg in the aquatic food chain as THg is removed from the watershed; however, short-term impacts are less clear. Other studies in northern Minnesota have indicated that fire leads to enhanced local deposition (Witt et al., 2009) that, in the short term, has the potential to bioaccumulate in the food chain (Kelly et al., 2006). Others have found little or no change in accumulation in aquatic biota following fire (Garcia and Carignan, 1999, 2000; Allen et al., 2005). Several studies have measured soil Hg concentrations or pools before and after fire (Harden et al., 2004) or compared burned to nearby unburned areas after fire (Amirbahman et al., 2004; DiCosty et al., 2006; Engle et al., 2006; Biswas et al., 2008; Navratil et al., 2009; Mitchell et al., 2012) and generally found that fire has the largest impact on forest floor and surface mineral soil Hg concentrations and pools, with the effect of fire decreasing with soil depth.

The effects of fire on soil C, N, and Hg may depend on the fire severity. *Fire intensity* refers to the energy released by a fire, while *fire severity* is an index of the combined effects of both fire intensity and duration (Certini, 2005). Fire severity describes the magnitude of the impact to an ecosystem (Keeley, 2009). The terms *fire intensity* and *fire severity* are often used interchangeably and inconsistently applied (Jain et al., 2004). For example, researchers may describe fires as high or low intensity, but fire intensity alone is not directly relatable to fire severity impacts (Neary et al., 2008; Keeley, 2009). Furthermore, the immediate impacts to the vegetation often do not correspond with impacts to soils (Fraver et al., 2011). Linking ecosystem impacts to repeatable metrics of fire severity is a critical step forward in scaling fire impacts from plots to fire events (Parsons et al., 2010).

We took advantage of the 2011 Pagami Creek wildfire in northern Minnesota, which resulted in fire impacts ranging from surface fires to crown fires, to evaluate the response of fuels, plant communities, and soils to the fire severity level shortly after the fire. We compared C, N, and Hg pools and the C/N ratio for the forest floor and upper mineral soil in burned and unburned control sites and compared soil burn and tree burn severity indices with forest floor and mineral soil C, N, and Hg pools and C/N ratios.

MATERIALS AND METHODS

Study Site

The study was conducted within and near the Superior National Forest in northeastern Minnesota (Fig. 1). The area has a mean annual precipitation of ~71 cm, with mean July and January temperatures of 17 and −8°C, respectively. Undulating and glacially carved terrain is underlain by Precambrian Canadian Shield bedrock, with soils formed primarily from glacial till and outwash resulting from the Rainy lobe during the Wisconsinan glacial period. Upland soils are strongly influenced by variable depth to bedrock, and the landscape is embedded with abundant lakes and wetlands (Heinselman 1996). All upland soils are Entisols that are either well drained, shallow, with 20 to 50 cm of gravelly coarse sandy loam over bedrock or well drained, moderately deep, with 50 to 100 cm of gravelly sandy loam over bedrock. In the area of the fire, these broad categories of soils that are mainly based on depth to bedrock each represent about 50% of the area, with minor inclusions of slightly deeper soils (J. Barott, U.S. Forest Service, Superior National Forest, personal communication, 2014). The shallow to bedrock soils include Quetico (loamy, isotic, acid, frigid Lithic Udorthents) and Insula (loamy, isotic, frigid Lithic Dystrudepts), while the moderately deep soils include Conic (coarse-loamy, isotic, frigid Typic Dystrudepts) and Wahlsten (coarse-loamy, isotic, frigid Oxyaquic Dystrudepts). Map units generally range from 2 to

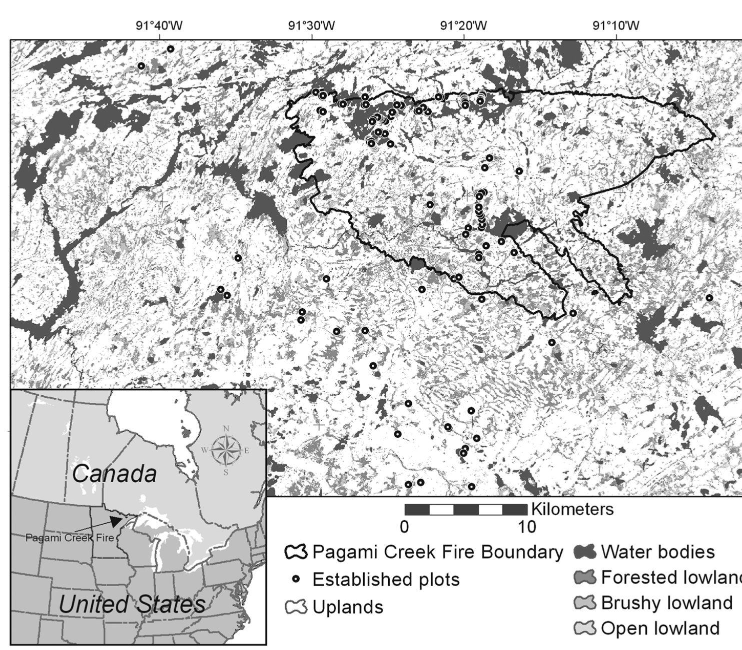

Fig. 1. Location of wildfire area and plot locations in northeastern Minnesota.

18% slope, with pines (e.g., *Pinus banksiana* Lamb. and *Pinus resinosa* Aiton) tending to be on shallow and sandier soils, deciduous species (e.g., *Populus tremuloides* Michx., *Populus grandidentata* Michx., *Betula papyrifera* Marshall, and *Acer rubrum* L.) on the deeper soils, and *Abies balsamea* (L.) Mill. and *Picea glauca* (Moench) Voss on the deepest and finest textured soils (J. Barott, personal communication, 2014).

On 18 Aug. 2011, a lightning strike initiated a wildfire in the northwest corner of what became the future burn area of the Pagami Creek Fire (Fig. 1). The fire expanded relatively slowly until 10 September, when windy and dry conditions led to a blowup where the fire traveled about 10 km to the south, and a subsequent change in wind direction on 12 September pushed the expanded fire front an additional 30 km to the east and southeast. The fire was essentially extinguished on 22 Oct. 2011. In total, 370 km^2 burned, making the Pagami Creek Fire the largest fire in Minnesota since the Cloquet Fire in 1918 which burned nearly 1000 km^2.

Experimental Design

To assess the responses of soil C, N, and Hg pools and C/N ratios to contrasting levels of fire severity, we established transects and an associated network of Forest Inventory and Analysis (FIA) style plots across expected gradients in fire severity based on post-fire remote sensing (relative differenced normalized burn ratio algorithm, according to Miller et al., 2009) and pre-fire knowledge of the vegetation communities (Wolter et al., 2009; Wolter and Townsend, 2011). A total of 31 transects with 133 12.6-m-radius plots were installed, including 24 unburned control plots established outside the Pagami Creek Fire boundary (Fig. 1), five of which were coincident with long-term plots established by the FIA program. In this study, we addressed upland mineral soils excluding wetlands, which decreased our sample size from 133 to 123 plots. Soils and ancillary data were collected during two periods. The first set of soil samples was collected in October to November 2011, immediately after the fire. During the 2011 sampling, we focused our sampling on fire-affected plots. Following our 2011 field campaign, we found that our initial assessment of fire severity underrepresented the highest soil burn severity classes. A second set of soil samples was collected in April to May 2012 and included unburned forested control plots but also added 25 burned plots that represented the higher soil burn severity classes that were underrepresented in the fall 2011 collection.

Fire Severity Estimates

Jain and Graham (2007) developed a consistent, justifiable, and repeatable approach through a synthesis of the literature for both overstory vegetation and soil burn severity classifications. The method has five categories of soil burn and tree burn severity ranging from 0 to 4, with 0 being unburned and 4 being the most severely burned.

The soil burn severity class system corresponds to the post-fire index discussed by Jain et al. (2012) and depends on the coverage of forest floor remaining and oxidation level (i.e., color) of the mineral soils. Because the index is based on the post-fire forest floor cover percentage, it is possible for forest floor mass to be lost when compared with the pre-fire condition without changing coverage. Soil burn severity was estimated on 10-m^2 circular subplots located at three azimuths (0 or north, 120, and 240°) 6.5 m from the plot center. Forest floor coverage was estimated visually as the percentage of total cover on each subplot. Digital photos (visual and near infrared) taken for each soil burn severity subplot were reviewed to ensure consistency in soil burn severity class assignment. Individual subplot soil burn severities were aggregated to develop a plot-level soil burn severity estimate. Primary soil burn severity categories are unburned (Category 0, $n = 24$ plots), lightly burned (Class 1, surface organic matter >85% present, $n = 15$ plots), moderately burned (Class 2, surface organic matter 40–85% present, $n = 27$ plots), severely burned (Class 3, surface organic matter <40% present, $n = 32$ plots), and very severely burned (Class 4, little to no surface organic matter present, $n = 25$ plots). While the classification key also includes subcategories to the primary categories based on mineral soil color, we used the primary burn severity categories based on the forest floor coverage to maintain adequate sample numbers for statistical analysis.

The tree burn severity class is dependent on the color of the tree crowns. Tree burn severity estimates were made at the plot center, classified visually for the whole 499-m^2 circular plot. Tree burn severity classes range from 0 (unburned, $n = 24$ plots), 1 (evidence of burn, with green tree crowns present, $n = 23$ plots), 2 (>98% of tree crowns brown, $n = 12$ plots), 3 (a plurality of brown and black crowns present, $n = 12$ plots), and 4 (>98% black crowns, $n = 52$ plots) (Jain and Graham 2007). Classes 1 and 3 have subcategories based on the relative proportions of green and brown (Class 1) and brown and black (Class 3), but like the soil burn severity classification, we restricted our analysis to the primary burn categories. Forest type was also assessed visually based on the tree species that remained standing or had probably fallen during or following the fire. These forest types were reclassified into coniferous, deciduous, or mixed for use as covariates in the statistical analysis.

Soil Sampling and Analysis

Triplicate forest floor and mineral soil samples were collected at each plot using identical methods to those used by FIA for comparison of long-term FIA plot data. The forest floor and soil were sampled on the same three azimuths used for assessing the soil burn severity class but at 3.7 m from the plot center. The forest floor was sampled with a 30-cm-diameter ring, and mineral soil cores were taken from the 0- to 10- and 10- to 20-cm depths (or to a limiting layer such as bedrock, which is common in this region) with a 6-cm-diameter hammer-driven bulk density corer. Soil cores were taken preferentially from inside the forest floor ring, but if the bedrock was too shallow for an adequate sample depth (minimum of 10 cm), we sampled the mineral soil at a maximum distance of 0.5 m from the forest floor sample location. Forest floor and mineral soil samples were frozen on return to the laboratory. Within 2 mo, the forest floor and mineral soil samples were thawed, and a subsample was oven dried at 65°C for the forest floor and 105°C for the mineral soils. The remaining sample was air dried, passed through a 2-mm sieve, and ground with a stainless steel Wiley mill, with complete cleaning between samples. The three replicates were aggregated by weight at the plot level and analyzed for total C and N concentrations on a Leco total elemental analyzer, with a separate subsample analyzed for THg concentration using a direct Hg analyzer (DMA-80, Milestone Inc.) and USEPA Method 7473 (USEPA, 2009). To determine the forest floor C, N, and Hg mass per unit area (i.e., pools), we used the oven-dried corrected weight of the entire forest floor sample after air drying and sieving. Although we measured the forest floor depth at the four cardinal directions in each of the 30-cm subplots, we believe using the oven-dry corrected weight of the entire sample is more accurate than calculating a density based on the mean forest floor depth. For the mineral soils, we used the oven-dried corrected weight of the entire 0- to 10- and 10- to 20-cm increments to calculate the overall soil bulk density (including the >2-mm fraction). The fine bulk density (including only the <2-mm fraction) was calculated following the removal of the >2-mm fraction, with the volume of that fraction estimated by using 2.65 g cm^{-3} (i.e., the density of quartz). The pools of mineral soil C, N, and Hg are based on the mean fine soil bulk density of the three replicates.

Statistical Analyses

For all statistical analyses and presentation of the data, we used C, N, and Hg pools and the ratio of C and N pools for the C/N ratio. Total losses and gains of forest floor and mineral soil C, N, and Hg pools as a result of fire were estimated by subtracting the mean mass of those elements in all burned plots (n = 99 plots) from the mean mass of those elements in the unburned reference plots (n = 24 plots). This approach assumes that we representatively sampled the gradient of burn severities and the proportional area of each burn severity in the fire and that the unburned reference plots sampled after the fire were representative of the burned plots before the fire. We used one-way general linear models (PROC MIXED, SAS Version 9.X) to determine if C, N, and Hg pools and C/N ratios differed according to the fire severity classes (Table 1). Before analysis of the tree burn severity indices and to meet assumptions of normality, we logarithmically transformed the data for forest floor C pools and C/N ratios for the 0- to 10-cm mineral soil C pools and the 10- to 20-cm mineral soil C and N pools. Data for the forest floor N and Hg pools, the 0- to 10-cm mineral soil C/N ratios, and the 10- to 20-cm mineral soil C/N ratios and Hg pools did not meet normality assumptions. As such, we ranked the data (i.e., measurement observations were converted to their ranks in the overall data set, such as the smallest variable receiving a rank of 1, the next larger receiving a rank of 2, etc.). This allowed us to conduct a nonparametric test on these variables as an analog to the parametric tests we performed on data sets for which normality assumptions were met. Ranking the data for a nonparametric test is a less powerful approach than one-way ANOVA, so this is a conservative approach not likely to increase the chance of a Type I error. For the soil burn severity indices, we logarithmically transformed the data for forest floor C pools and C/N ratios, the 0- to 10-cm mineral soil C pools, and 10- to 20-cm mineral soil C and N pools and ranked the data for forest floor N and Hg pools and the C/N ratios in both of the mineral soil layers before analysis. We included forest type (coniferous, deciduous, or mixed) as a covariate when it explained a significant amount of the model variance. Forest type explained a significant amount of variation in the data for forest floor C/N ratios for both the tree burn and soil burn severity indices and for Hg pools in the 10- to 20-cm mineral soil layer for the soil burn severity index. When differences among severity levels were significant, we performed means separation tests using Tukey's adjustment for multiple comparisons. For a comparison of C/Hg ratios between burned and control sites, we used a two-tailed t-test. We used $p \leq 0.05$ to determine statistical significance.

Table 1. Results of general linear models, showing F and p values for fire severity level and forest cover category for forest floor and 0- to 10- and 10- to 20-cm-depth mineral soil for C, N, and Hg pools and C/N ratios using tree burn and soil burn severity indices. Bold values indicate significance at the p < 0.05 level.

Soil layer	Response variable	Severity level F	Severity level p	Cover category F	Cover category p
	Tree burn (n = 123)				
Forest floor	C†	13.44	**<0.0001**	3.78	**0.012**
	N‡	6.23	**0.0001**	2.63	0.053§
	C/N ratio†	9.85	**<0.0001**	3.61	**0.030**
	Hg‡	11.77	**<0.0001**	2.54	0.059§
0–10-cm mineral	C†	2.96	**0.023**	–	–
	N	1.01	0.404	–	–
	C/N ratio‡	7.43	**<0.0001**	–	–
	Hg	2.76	**0.031**	–	–
10–20-cm mineral	C†	2.71	**0.033**	–	–
	N†	1.26	0.290	–	–
	C/N ratio‡	3.39	**0.012**	–	–
	Hg†	2.65	**0.037**	3.23	**0.04**
	Soil burn (n = 123)				
Forest floor	C‡	9.55	**<0.0001**	–	–
	N‡	3.43	**0.010**	–	–
	C/N ratio†	10.13	**<0.0001**	3.45	**0.035**
	Hg‡	8.74	**<0.0001**	–	–
0–10-cm mineral	C†	2.29	0.064	–	–
	N	0.73	0.576	–	–
	C/N ratio‡	8.15	**<0.0001**	–	–
	Hg	1.88	0.119	–	–
10–20-cm mineral	C†	2.53	**0.044**	–	–
	N†	0.44	0.777	–	–
	C/N ratio‡	5.03	**0.001**	–	–
	Hg	2.56	**0.042**	2.43	**0.093**

† Data were logarithmically transformed before analysis.

‡ Data were ranked before analysis.

§ Covariate was included based on marginal significance and improvement of model fit.

RESULTS

Carbon, Nitrogen, and Mercury Pool Changes from Fire

Across all plots, the mean loss of C in the forest floor was 9.7 Mg ha^{-1}. The 0- to 10-cm soil depth lost 5.5 Mg C ha^{-1} and the 10- to 20-cm soil depth lost 4.1 Mg ha^{-1}. For N, the forest floor lost 0.25 Mg ha^{-1}, while the 0- to 10- and 10- to 20-cm soil depths each gained 0.04 Mg ha^{-1}. For Hg, 3.0, 2.1, and 4.2 g ha^{-1} were lost from the forest floor, 0- to 10-cm soil depth, and 10- to 20-cm soil depth, respectively.

Relationship of Tree Burn Severity Index and Forest Floor and Soil Carbon, Nitrogen, and Mercury Pools and Carbon/Nitrogen Ratios

Forest floor C and Hg pools and C/N ratios were higher in the unburned reference plots than in any of the tree burn severity classes (Table 2). Among severity classes, Class 3 tended to have lower C and Hg forest floor pools than the other severity classes with the exception of Class 2. The forest floor N pool was highest in the unburned control class (Class 0), which differed from Classes 2 and 3 but not from the lightly burned Class 1 and somewhat surprisingly the very severely burned Class 4. The severely burned class (Class 3) had lower N than all other classes with the exception of the moderately burned class (Class 2). The C pool tended to be higher in the unburned reference class (Class 0) than the lightly burned class (Class 1) at the 0- to 10-cm soil depth, with no differences at the 10- to 20-cm soil depth. The C/N ratio was higher for the unburned reference class than all but the moderately burned class (Class 2) at both the 0- to 10- and 10- to 20-cm soil depths. No differences were found for N or Hg pools at either the 0- to 10- or 10- to 20-cm soil depths (Table 2).

Relationship of Soil Burn Fire Severity Index and Forest Floor and Soil Carbon, Nitrogen, and Mercury Pools and Carbon/Nitrogen Ratios

Forest floor C and Hg pools and C/N ratios were higher in the unburned reference class (Class 0) than in any of the soil burn severity classes, which themselves did not differ (Table 3). The forest floor N pool was only different between the unburned reference class (Class 0) and the severely burned class (Class 3). The C/N ratio was higher in the unburned reference class (Class 0) than all the burned treatments at 0 to 10 cm and higher than all but the severely burned treatment (Class 3) at 10 to 20 cm. No differences were found for C, N, or Hg pools in either the 0- to 10- or 10- to 20-cm soil depths (Table 3).

DISCUSSION

It is apparent from the data that the forest floor was most affected by the fire, with that effect decreasing with depth. Overall, we found that burned sites on average lost approximately 65% of C and 51% of N in the forest floor, lost 17% of C and gained 3% of N in the 0- to 10-cm depth, and lost 15% of C and gained 3% of N in the 10- to 20-cm soil depth when compared with the unburned controls. A synthesis of wildfire studies on temperate forests indicated that 67% of the C and 69% of the N pools were lost from the forest floor, with no change in the mineral soils (Nave et al., 2011). Bradford et al. (2012) found that blowdown

Table 2. Tree burn severity index of forest floor and 0- to 10- and 10- to 20-cm-depth mineral soil for C, N, and Hg pools and C/N ratio.

Tree burn index	C	N	C/N ratio	Hg
	Mg ha^{-1}			g ha^{-1}
		Forest floor		
0	14.3 (2.3) a†	0.47 (0.08) a	32.2 (1.5) a	4.0 (1.0) a
1	5.1 (0.7) b	0.25 (0.03) ab	22.8 (1.4) b	1.1 (0.2) b
2	3.4 (0.5) bc	0.17 (0.03) bc	21.8 (1.4) b	0.5 (0.1) bc
3	2.2 (0.6) c	0.12 (0.03) c	19.0 (1.5) b	0.5 (0.2) c
4	5.3 (0.5) b	0.26 (0.03) ab	21.9 (1.1) b	1.3 (0.2) b
		0–10-cm mineral soil		
0	32.6 (2.3) a	1.5 (0.12)	23.3 (1.6) a	31.3 (1.8)
1	22.5 (1.4) b	1.4 (0.08)	17.0 (0.7) b	32.5 (1.9)
2	28.3 (2.6) ab	1.6 (0.16)	18.5 (1.1) ab	25.6 (2.5)
3	28.6 (1.6) ab	1.6 (0.13)	17.8 (1.0) b	27.8 (1.8)
4	28.5 (1.7) ab	1.6 (0.08)	18.2 (0.5) b	30.9 (1.5)
		10–20-cm mineral soil		
0	26.8 (1.3)	1.4 (0.09)	19.2 (0.7) a	40.5 (2.8)
1	20.6 (1.0)	1.4 (0.09)	15.9 (0.9) b	34.6 (2.9)
2	22.2 (1.9)	1.4 (0.12)	16.4 (1.0) ab	32.6 (1.8)
3	26.5 (2.4)	1.7 (0.14)	15.8 (0.9) b	40.8 (2.4)
4	22.7 (1.1)	1.5 (0.07)	16.7 (0.7) b	35.9 (1.9)

† Data within a column followed by different letters are significantly different at the $p \leq 0.05$ significance level. Numbers in parentheses are standard errors.

Table 3. Soil burn severity index of the forest floor and 0- to 10- and 10- to 20-cm-depth mineral soil for C, N, and Hg pools and C/N ratio.

Soil burn index	C	N	C/N ratio	Hg
	Mg ha^{-1}			g ha^{-1}
		Forest floor		
0	14.3 (2.3) a†	0.47 (0.08) a	32.2 (1.5) a	4.0 (1.0) a
1	4.8 (0.7) b	0.26 (0.04) ab	20.7 (1.5) b	1.0 (0.2) b
2	4.7 (0.6) b	0.22 (0.03) ab	22.4 (1.1) b	0.9 (0.2) b
3	4.1 (0.6) b	0.19 (0.03) b	22.9 (1.4) b	0.8 (0.2) b
4	4.7 (0.9) b	0.25 (0.05) ab	20.1 (1.4) b	1.2 (0.3) b
		0–10-cm mineral soil		
0	32.6 (2.3)	1.5 (0.12)	23.3 (1.6) a	31.3 (1.8)
1	23.5 (1.4)	1.5 (0.08)	16.2 (0.6) b	26.4 (1.7)
2	26.1 (2.6)	1.4 (0.14)	18.9 (0.9) b	25.6 (2.3)
3	28.7 (1.6)	1.6 (0.09)	18.1 (0.7) b	31.3 (1.3)
4	28.1 (1.7)	1.6 (0.09)	17.8 (0.7) b	30.7 (2.3)
		10–20-cm mineral soil		
0	26.8 (1.3)	1.4 (0.09)	19.2 (0.7) a	40.5 (2.7)
1	22.1 (2.5)	1.6 (0.19)	14.8 (1.2) b	32.6 (3.0)
2	20.5 (1.3)	1.4 (0.11)	16.1 (1.3) b	35.3 (2.8)
3	23.6 (1.2)	1.4 (0.07)	17.1 (0.6) ab	34.1 (2.1)
4	24.3 (1.5)	1.5 (0.09)	16.5 (0.8) b	40.8 (2.1)

† Data within a column followed by different letters are significantly different at the $p \leq 0.05$ significance level. Numbers in parentheses are standard errors.

(a wind event that blew down trees) in combination with fire led to 74% decreases in forest floor C pools but no change in mineral soil C following a wildfire in northern Minnesota. In southwestern Wisconsin, the forest floor mass decreased 41% following a prescribed fire in two consecutive years in oak–hickory forests, with little change in organic C concentration in the mineral soil (Knighton, 1977). Rothstein et al. (2004) found a 66% decrease in the forest floor C pool in wildfire-affected jack pine (*Pinus banksiana*) forests in Lower Michigan, with little change or even small increases in C in mineral soils. Across a chronosequence of fire sites in the same region as this study, Woodruff and Cannon (2010) found that 65% of the soil C in the O and A horizons was volatilized back to the atmosphere during fires with "deep burning." Lynham et al. (1998) also measured small increases in the N pool in upper mineral soils following the experimental burning of young jack pine forests in northern Ontario. Alban (1977) found 12 to 47% decreases in forest floor soil organic matter pools and 12 to 33% decreases in forest floor N pools, similar to this study, and Lynham et al. (1998) found small increases in mineral soil N pools in red pine (*Pinus resinosa*) stands of north-central Minnesota following prescribed fire.

The C/N ratios of the unburned reference treatment were generally higher than those of the burned treatments for forest floor and both mineral soil depths. For the forest floor, it appears that C is released more readily than N, leading to lower C/N ratio organic material after fire. Mineral soils also appear to have lost small amounts C and/or gained small amounts of N, which led to lower C/N ratios. Graphical interpolations from Yermakov and Rothstein (2006) also showed markedly decreased C/N ratios for both forest floor and upper mineral soils in recent fires when compared with a 72-yr-old fire. The lower C/N ratios in the mineral soils of the burned plots possibly indicate N inputs from leaching of the forest floor or N_2–fixing plants (Kutiel and Naveh, 1987; Caldwell et al., 2002). Because we sampled shortly after the fire in late fall after a killing frost and shortly after snowmelt in the spring, inputs from N_2–fixing plants would be unlikely. Rain events did occur after the burn and before our sampling in fall 2011, and snowmelt occurred before our spring 2012 sampling. We suspect it could be microbes that are releasing N (and C) from the unburned forest floor or possibly N released from the decomposition of other fire residues that is being leached in dissolved form from the forest floor to the mineral soil below during these events.

Mercury generally behaved similarly to C, as Hg is known to have an affinity for binding to C (Yin et al., 1996), and gaseous Hg emissions (mainly in the form of elemental Hg) have a close relationship with CO_2 emissions during fire (Friedli et al., 2003). However, the C/Hg ratio of the forest floor was higher ($p = 0.01$) in burned plots than in the control plots, indicating that Hg is more volatile than C where the fire was presumably hottest. In the 0- to 10- ($p = 0.17$) and 10- to 20-cm ($p = 0.48$) mineral soil layers there was no difference in C/Hg ratios between burned and control plots. Combustion studies in laboratories on conifer and deciduous litter have indicated that nearly the entire Hg pool (>98%) is emitted back to the atmosphere, with little left behind in ash (Friedli et al., 2003). Although the bulk of the Hg emitted during wildfires is in elemental form (Hg^0) that will ultimately be deposited outside the watershed and potentially far from the site of the fire, about 13% of the Hg emitted during wildfires is in particulate form (Friedli et al., 2003), which has the potential to be deposited locally (Witt et al., 2009). In our study, 72% of the Hg pool was lost from the forest floor, 7% from the 0- to 10-cm soil depth and 10% from the 10- to 20-cm soil depth. Similar to our results, Mitchell et al. (2012) conducted a study in the same region and found that 84% of the Hg pool in the forest floor was emitted back to the atmosphere after a large wind event that blew down trees, after which the site was salvage logged and then had fire, but found no differences in the mineral soil pool after any fire treatments. Woodruff et al. (2009) also measured Hg loss as a result of fire in northern Minnesota and found 100% Hg pool losses in the forest floor and 87% from the A horizon of severely burned areas. In a chronosequence of fires that included some of the same sites, Woodruff and Cannon (2010) found that 88% of the Hg pool in the forest floor + A horizon was lost during fires that "deeply burned." In conifer forests of the Cascade Range, Biswas et al. (2008) found that 45% of the Hg pool was volatilized from the upper 8 cm of soil, which included both forest floor and mineral soil layers. Engle et al. (2006) found that 90% of the Hg pool in the forest floor was emitted back to the atmosphere during a prescribed fire in conifer forests of western Nevada, while 94% of the forest floor Hg pool was lost during a wildfire. In both cases, upper mineral soil layers gained a small amount of Hg. It is apparent from our data and that from the literature that much of the Hg in the forest floor is typically emitted to the atmosphere during fire and, if the heat is transmitted into the mineral soil, some Hg from mineral soil layers is also volatilized.

Neither the soil burn nor the tree burn severity classifications led to many differences among the burn classes in the forest floor or mineral soil C, N, or Hg pools or the C/N ratios (Tables 2 and 3). Of the few differences that did exist under the tree burn severity classification, the data tended to indicate that the severely burned category (Class 3) had the largest impact on the forest floor, with no differences among tree burn severity classes for the mineral soils. Although it might be expected that the very severely burned class (Class 4) would have the greatest impact, Class 3 tree burn severity fires tend to include both the bole and canopy, whereas Class 4 fires tend to be high-intensity canopy fires that can have little impact on soils (Jain and Graham, 2007).

Overall, however, it appears that if a plot was burned, forest floor C, N, and Hg were lost independently of the plot-level burn severity classification. Burn severity classification (either soil burn or tree burn) at the plot scale did not create divergence in forest floor or mineral soil losses of C, N, and Hg pools sampled at the subplot scale and aggregated at the plot scale. This was the first attempt to use the Jain and Graham fire severity methods to assess the relationships of post-fire forest floor and soil chemical properties. When we compared the paired plot-level classification of the two severity indices, they were correlated (ρ = 0.55) but certainly not identical. Other studies have demonstrated that fire impacts to the vegetation are often decoupled from impacts to the soil layer (e.g., Fraver et al., 2011). The lack of sensitivity in soil variables to the tree burn severity index was somewhat expected, yet the soil burn severity index was designed to capture direct impacts to the forest floor and upper mineral soils. We note a couple of complicating factors that probably influence the ability of the soil burn severity index to directly represent forest floor and mineral soil impacts. First, the index is based on what is remaining after the burn. Consequently, it relies on the spatial coverage of forest floor rather than the volume of forest floor lost, as the latter cannot be known apart from inference from nearby control plots. Second, within our methods, the soil burn class and the soil sampling location were not at identical positions within the sampling transect. Because of the heterogeneous nature of fires, it is possible that this difference in location led to some decoupling between the soil burn severity class and the forest floor present. It would be more appropriate to assess the soil burn severity class within the forest floor ring where the forest floor and mineral soils are sampled. This should lead to an improvement in the correlation among burned classes and the soil chemical properties.

CONCLUSIONS

The Pagami Creek Fire was the largest in Minnesota in nearly 100 yr. Fires of this magnitude are large sources of CO_2, oxidized forms of N, and gaseous Hg to the atmosphere. Based on the area of the fire and our mean pool loss rates when compared with unburned controls, we estimate that >500,000 Mg of C, nearly 5000 Mg of N, and >250 kg of Hg were emitted to the atmosphere as a result of the Pagami Creek Fire. For C, that amount is similar to the amount of C that 52,000 sport utility vehicles emit into the atmosphere every year. The amount of N is equal to about the fertilization rate of 31,000 ha of corn (*Zea mays* L.). The Hg emissions are comparable to about 18% of the 2010 annual emissions for the state of Minnesota.

The bulk of the elemental fluxes were from the combustion of the forest floor, with somewhat lesser amounts from the upper mineral soils. In the case of N, the upper mineral soils were small sinks. As the sites recover and vegetation regrowth commences, the forest floor and soil C and N will begin to accumulate at a rapid pace, but because Hg is atmospherically derived, it may not reach levels that occurred pre-burn. Because of current lower Hg emissions than in the past, the accumulation of Hg in soils is occurring at a much slower rate and Hg levels may never reach pre-fire concentrations, which, in turn, should lead to lower long-term Hg concentrations in the aquatic food chain. However, the Hg that was volatilized during the fire is a short-term source of deposition elsewhere.

Although we anticipated that our soil burn severity index and potentially our tree burn severity index would be able to differentiate the impacts to forest floor and soils, this was generally not the case. The fire-affected forest floor and in some cases upper mineral soil were statistically different than the unburned controls, but differences among the soil burn and tree burn severity classes were limited. Soil burn severity assessment is particularly challenging because it relies on what remains after a burn without full knowledge of what existed before the burn. Also, fine-scaled heterogeneity of the impacts on forest floor and mineral soil introduces further noise to the relationships. Despite these challenges, reliable and repeatable estimates of burn severity is a critical first step in scaling ecosystem impacts both within and across wildfire events.

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