doi: 10.4996/fireecology.1301034

Research Article

Carbon Emissions during Wildland Fire on a North American Temperate Peatland

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ABSTRACT

Northern temperate zone (30° to 50° latitude) peatlands store a large proportion of the world's terrestrial carbon (C) and are subject to high-intensity, stand-replacing wildfires characterized by flaming stage combustion of aboveground vegetation and long-duration smoldering stage combustion of organic soils. Coastal peatlands are a unique region in which long-duration wildfire soil combustion is responsible for the majority of total annual emissions from all wildfires in the North American coastal plain. We developed a new method and approach to estimate aboveground and belowground C emissions from a 2008 peatland wildfire by analyzing vegetation C losses from field surveys of biomass consumption from the fire and soil C losses derived from the Soil Survey Geographic Database, a digital elevation model derived from airborne optical remote-sensing technology and

RESUMEN

Las turberas ubicadas en la zona templada (30° a 50° de latitud norte) almacenan una gran proporción del carbono (C) total del mundo, y está sujetas a incendios de gran intensidad que producen el reemplazo de rodales y que se caracterizan por un estado de combustión por llamas en la vegetación aérea y de combustión incandescente y de larga duración en suelos orgánicos. Las turberas costeras representan la única región en la cual la combustión incandescente de larga duración es la responsable de la mayoría de las emisiones totales anuales de todos los incendios en las planicies costeras de Norte América. Nosotros desarrollamos un nuevo método y enfoque para estimar emisiones de C, tanto aéreas como subterráneas, de un incendio de turberas de 2008 a través del análisis de las pérdidas de C mediante relevamientos a campo de combustión de biomasa por el fuego, pérdidas de C derivados de la base de datos del Relevamiento Geográfico de Suelos, un modelo de elevación digital derivado de tecground elevation surveys using a Global Navigation Satellite System receiver. The approach to estimate belowground C emissions employed pre-fire LIDAR-derived elevation from ground return points coupled with post-fire survey-grade GPS elevation measurements from co-located ground return points. Aboveground C emission calculations were characterized for litter, shrub foliage and woody biomass, and tree foliage fractions in different vegetation classes, thereby providing detailed emissions sources. The estimate of wildland fire C emissions considered the contribution of hydrologic regime and land management to fire severity and peat burn depth. The peatland wildfire had a mean peat burn depth of 0.42 m and resulted in estimated belowground fire emissions of 9.16 Tg C and aboveground fire emissions of 0.31 Tg C, for total fire emissions of 9.47 Tg C (1 Tg = 10^{12} grams). The mean belowground C emissions were estimated at 544.43 t C ha^{-1}, and the mean aboveground C emissions were 18.33 t C ha^{-1} (1 t = 10^6 grams).

nologías ópticas de sensores remotos para evaluar partículas aéreas, y relevamientos de elevación del terreno usando un receptor del Sistema Global de Navegación. El enfoque para estimar emisiones de C subterráneo empleó puntos pre-fuego (derivados de LIDAR) acoplados con relevamientos de estos puntos post-fuego, y a los cuales se retornaba y re-medía con GPS. Los cálculos de las emisiones se caracterizaron para mantillo, follaje de arbustos y biomasa leñosa, y fracciones de follaje de árboles en diferentes clases, los cuales proveían detalles de las fuentes de emisión. La estimación de las emisiones de C del incendio consideró la contribución del régimen hidrológico y el manejo de la tierra en la severidad y la profundidad de la quema de las turberas. El incendio de la turbera tuvo una profundidad media de 0,42 m y resultó en una estimación de la emisión subterránea de 9,16 Tg de C y una emisión de la biomasa aérea de 0,31 Tg C, dando un total de 9,47 Tg de C (1 Tg = 10^{12} gramos). Las emisiones medias de C subterráneo fueron estimadas en 544,43 t C ha^{-1}, y las emisiones medias de C de la biomasa aérea fueron de 18,33 t C ha^{-1} (1 t =10^6 gramos).

Keywords: burn severity**,** carbon emissions, coastal plain, hydrologic regime, land management LIDAR, organic soils, temperate peatlands, wildfire

Citation: Mickler, R.A., D.P. Welch, and A.D. Bailey. 2017. Carbon emissions during wildland fire on a North American temperate peatland. Fire Ecology 13(1): 34–57. doi: 10.4996/fireecology1301034

INTRODUCTION

The global contribution of biomass burning to carbon (C) emissions in the atmosphere was identified as an important source of radiatively and photochemically reactive trace gases in 1980 (Seiler and Crutzen 1980); however, the direct and indirect contributions of wildland and prescribed fires to total C emissions and the positive feedback on the climate system remain difficult to assess. Crutzen *et al.* (1979) investigated the atmospheric budgets of biomass-burning trace gases in the atmosphere that included carbon monoxide (CO), molecular hydrogen (H_2), nitrous oxide (N_2O), nitric oxide (NO), nitrogen dioxide (NO_2), and carbonyl sulfide (COS). Seiler and Crutzen (1980) estimated net fluxes of carbon in forest and grassland wildland fires and showed that trace gases were emitted into the atmosphere in large quantities by measuring emission rates of trace gases relative to carbon dioxide (CO_2) in the smoke plumes. Total global emission rates of trace gases were approximated from CO_2 released into the atmosphere during wildfires and prescribed burning across different ecosystems and land management practices. A general model—consisting of total land area burned, the average organic matter per unit area, the average aboveground biomass relative to the total average biomass, and the burning efficiency of the aboveground biomass—was used to estimate the biomass burned annually in global biomes. Seiler and Crutzen (1980) calculated annual global C emissions from wildland fires of 2 Pg C to 4 Pg C (1 Pg = 10^{15} grams). However, their emissions model did not include the contribution of belowground soil C released during wildland fires, which have the potential to release much higher C emissions than the aboveground C emissions on a per unit land area basis. Wong (1978) estimated wildfire soil C emissions at 0.4 Pg C yr^{-1} and Seiler and Crutzen (1980) subsequently revised Wong's estimate downward to 0.1 Pg C yr^{-1}. A recent study (van der Werf *et al.* 2010) excluded fuelwood burning from the C emissions estimate of Seiler and Crutzen (1980) and calculated annual global C emissions of 2.6 Pg C yr^{-1} (ranging from 1.7 Pg C yr^{-1} to 3.5 Pg C yr^{-1}).

Recent advances in satellite-derived fire products using moderate-resolution imaging spectroradiometer data from the Terra and Aqua satellites, the Advanced Very High Resolution Radiometer sensor on the National Oceanic and Atmospheric Administration Polar Operational Environmental Satellite, and the Geostationary Operational Environmental Satellite to quantify burned areas in near-real-time have enhanced our ability to estimate regional and global wildland fire emissions (Gregoire *et al.* 2003, Simon *et al*. 2004, Giglio *et al.* 2006). Remotely sensed data and their products have been used in combination with biogeochemical and terrestrial ecosystem models to estimate emissions (van der Werf *et al.* 2003, 2004, 2006; Hoelzemann *et al.* 2004; Ito and Penner 2004; Jain *et al.* 2006). These studies estimated annual global wildland fire C emissions that ranged from 1 Pg C yr^{-1} to 3 Pg C yr^{-1}, with large interannual variability associated with global fire activity and ~20% uncertainty in C emissions (Field and Shen 2008, van der Werf *et al.* 2010).

Quantifying the magnitude and spatial extent of C storage in peatlands is an important first step toward predicting C emissions during wildland fires and the changes in regional C balance in response to land use and land cover change. Peatlands are wetlands with an organic soil layer of at least 30 cm, which may extend to a depth of 15 m to 20 m (Turunen *et al.* 2002). They store more C than any other terrestrial ecosystem per square meter, and exert a net cooling effect on the global radiation balance (Dise 2009). Peatlands comprise less than 3%, or 400 Mha, of Earth's land area and store an estimated 15% to 30% of the global soil C stocks (Solomon *et al.* 2007, Limpens *et al.* 2008). Boreal and subarctic peatlands cover 346 Mha and comprise a C pool that ranges from 273 Pg C (Turunen *et al.* 2002) to 455 Pg C (Gorham 1991), to 473 Pg C to 621 Pg C (Yu *et al.* 2010); temperate peatlands cover an estimated 35.0 Mha and store 455 Pg C (Moore 2002); tropical-subtropical peatlands in southeast Asia cover 27.1 Mha and store 42 Pg C to 55 Pg C (Hooijer *et al.* 2010, Yu *et al.* 2010), and South American peatlands cover 4.5 Mha and have accumulated 13 Pg C to 18 Pg C (Yu *et al.* 2010). The spatial heterogeneity of peatland soils and vegetation poses challenges to quantifying C storage and wildland fire emissions, and to estimating interannual variability and uncertainty. Global organic C is estimated to be 684 Pg C to 724 Pg C in the upper 30 cm of soil, 1462 Pg C to 1548 Pg C in the upper 100 cm, and 2376 Pg C to 2456 Pg C in the upper 200 cm (Batjes 1996).

In the continental United States, temperate peatlands are located primarily in the Great Lakes region and in the coastal plain of the southeastern and Gulf states (USDA NRCS

2007). The total area of organic soils in the contiguous United States is estimated at 6.08 Mha, and the amount of organic C held in organic soils is estimated at 1714.7 Tg at the 0 cm to 30 cm depth, 5088.4 Tg at the 0 cm to 100 cm depth, and 7590.0 Tg at the 0 cm to 150 cm depth (Johnson and Kern 2003). The reported total C stored in temperate peatlands in one field study in the northern state of Minnesota was 1286 ±125 Mg C ha^{-1}, with 90 % to 99 % of that C found in peat soils that ranged from 1 m to 5 m in depth (Weishampel *et al.* 2009). In the southeastern state of North Carolina, the estimated total area of peatland soils is 0.27 Mha, with a total C pool of 327 Tg (Ingram and Otte 1981).

Wildland fire emissions in the contiguous United States vary considerably year to year, with average annual C releases estimated at 58 Tg yr^{-1} (Wiedinmyer and Neff 2007). Examinations of surface fires have attributed approximately two-thirds of the fire extent and emissions in the southeastern United States to prescribed fires (Wiedinmyer and Neff 2007, NIFC 2010). However, wildfires that ignite organic peat soils in this geographic area are frequent phenomena when drought conditions prevail. Despite their potential for large positive feedback to the climate system through sequestration and emission of greenhouse gases, peatland fires and their C emissions are not explicitly included in global climate models. In North America, modeling efforts in boreal regions have begun to address ground fire contributions to C emissions, but temperate peat fires have received much less attention (Poulter *et al.* 2006; de Groot *et al.* 2007, 2009). This information gap is in part due to uncertainties in soil depth, composition, and physical properties of the region's peat soils.

In the southeastern United States, peat soils are common in the Coastal Plain province. Techniques to detect and quantify ground fire emissions accurately are difficult to implement because of the heterogeneity in spatial and temporal soil loss (Poulter *et al.* 2006). Post-fire vegetation and soil measurements, as well as rapid visual estimates, can provide robust estimates of soil consumption and resulting emissions from wildfires in certain circumstances (Boby *et al.* 2010). However, precise quantitative measures of soil loss are difficult to obtain because of a dearth of pre- and post-wildland fire soil elevation information. Field examination of post-fire vegetation can provide some estimate of soil loss with a significant investment of time and resources. Hummocky mounds of litter and organic soil, as well as generally uneven micro-terrain features, predominate in many southeastern peatland systems, making such field measures more difficult. The more rapid approach of direct measurement of soil loss may provide a more practical solution to the problem of estimating fire-induced peat loss. In view of the ongoing discussions on climate change, it is important to know how much C is emitted into the atmosphere during peatland wildfire events.

In this study, we developed a new method and approach to estimate aboveground and belowground C emissions from a 2008 peatland wildfire in eastern North Carolina,USA, by combining burn intensity model output, field surveys, and remotely sensed information. Belowground C emissions were estimated by first determining the change in soil-surface elevation of randomly selected bare ground points using pre-wildfire Light Detection and Ranging (LIDAR)-derived soil elevation and co-located post-fire ground-surveying elevation measurements. Soil organic C was calculated for the depth of soil consumed for each soil horizon within each of the soil series. Aboveground C emission calculations employed estimates of area burned, fuel loading, and consumption proportions that were characterized for litter, shrub, and tree foliage fractions in different vegetation classes, coupled with tree and shrub density measures. The estimate of wildland fire C emissions considered factors contributing to peatland emissions such as hydrologic regime, land management decisions, and remotely sensed estimates of fire severity.

METHODS

Site Description

The Evans Road Fire, reported on 1 June 2008, was located approximately 24 km northwest of Fairfield in the New Lake vicinity in eastern North Carolina, USA, and 5 km south of Pocosin Lakes National Wildlife Refuge, on private farmland. The lightning ignition occurred at 35°37′7.8″N, 76°27′3.6″W and, by 14 June 2008, spread to 16 814.2 ha in three counties in eastern North Carolina: Hyde, Tyrrell, and Washington (Figure 1). The fire was declared to be out on 9 January 2009. The wildfire perimeter encompassed portions of the US Fish and Wildlife Service's Pocosin Lakes National Wildlife Refuge (10 509 ha), North Carolina Pettigrew State Park (243 ha), and private lands (6061 ha).

Vegetation consisted of two National Vegetation Classification ecological communities: Woodland and Shrubland (Anderson *et al.* 1998). These plant communities are formed on ombrotrophic peat bogs, dominated by evergreen coniferous and sclerophyllous species. Woodland overstory trees include pond pine (*Pinus serotina* Michx), loblolly bay (*Gordonia lasianthus* [L.] Ellis), and red bay (*Persia borbonia* [L.] Spreng). Understory shrubs in-

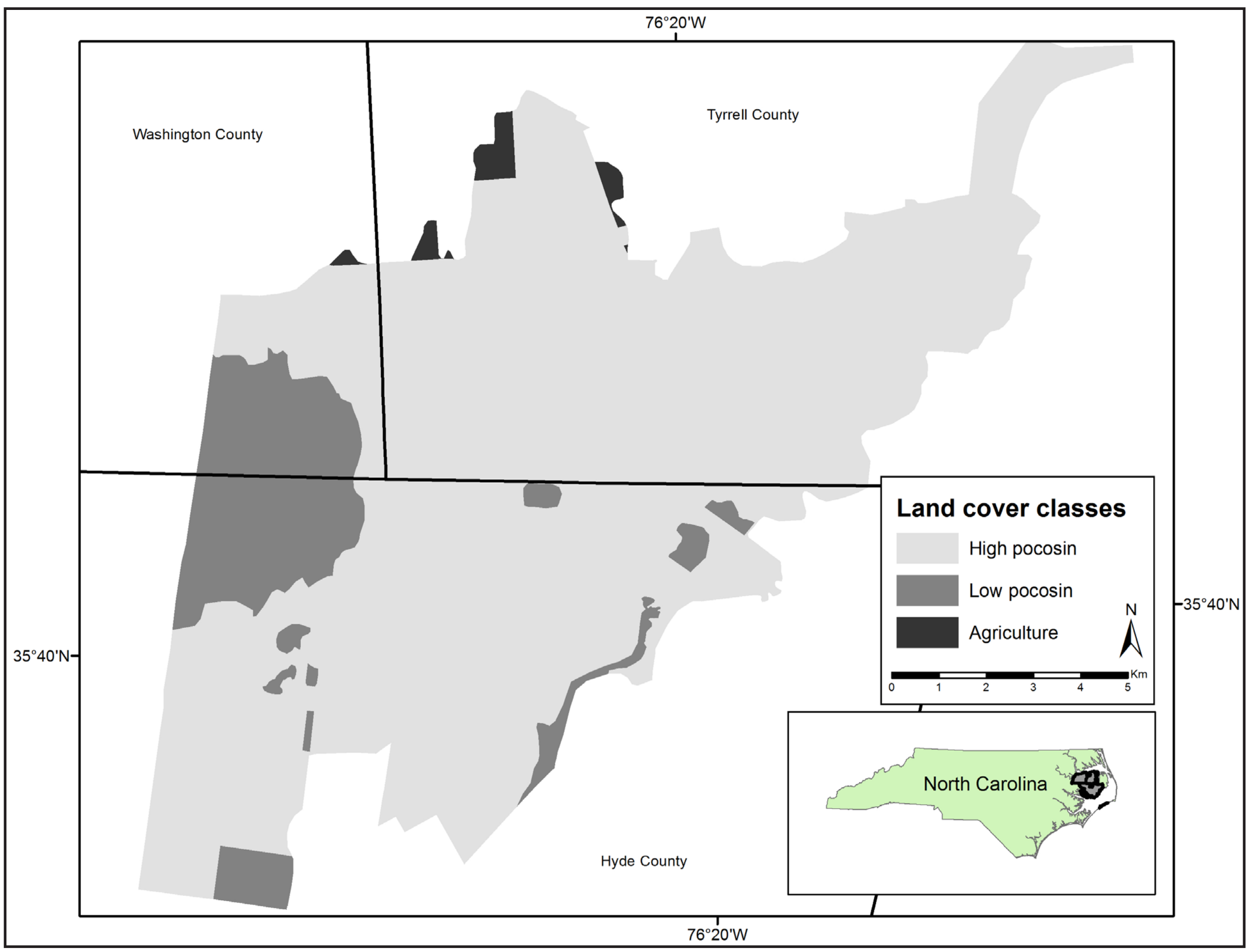

Figure 1. Fire perimeter land cover. Extent of high pocosin (14 417 ha), low pocosin (2223 ha), and agriculture (173 ha) in the eastern North Carolina counties of Tyrrell, Washington, and Hyde.

clude gallberry (*Ilex glabra* [L.] Gray), tall gallberry (*Ilex coriacea* [Pursh] Chapm.), fetterbush (*Lyonia lucida* [Lam.] K. Koch), and titi (*Cyrilla racemosa* L.). Herbaceous diversity is low and is dominated by Virginia chainfern (*Woodwardia virginica* [L.] Sm.). Since the Holocene period and before commercial logging operations, which ended in the early 1900s, vegetation was dominated by Atlantic white cedar (*Chamaecyparis thyoides* [L.] B.S.P.) and bald cypress forests (*Taxodium distichum* [L.] Rich.). Historical fire return intervals for the area are 30 yr to 60 yr (Wilber and Christensen 1984).

High and low pocosin, oligotrophic freshwater evergreen shrub bogs, historically covered one million hectares of North Carolina's coastal plain. These bogs began forming during the last 10 000 years, during the mid-Holocene. Peat deposits range in depth from 0.3 m to 2.4 m in interfluvial areas, to 5 m deep in Pleistocene channels and Carolina bays (Ingram 1987). Extensive drainage for agriculture, forestry, and peat mining reduced the natural wetland area to 281 000 hectares by 1980 (Richardson 1982). Seventy percent of the nation's pocosins occur in North Carolina, where they make up 50 % of the state's freshwater wetlands. Approximately 2700 km^2 of this area is below 1 m in elevation (Poulter and Halpin 2008).

Hydrological management is a key feature in the study site. Previous landowners established a ditch network system to drain most of this area in order to harvest and manage timber, and later to establish agricultural fields in the organic soil. Private lands and a portion of the public lands in the study area retain this ditch network (Figure 2). Additionally, the US Fish and Wildlife Service actively manages surface hydrology by impounding water through the use of water control devices. Little or no water impoundment existed on private lands because of agricultural crop requirements and wildlife management goals. Areas of natural hydrology occur on lower elevation portions of the public land.

From 1971 to 2000, the mean annual temperature at the study site was 16.7 °C, with winter monthly means ranging from 6.3 °C to 11.7 °C, and summer monthly means ranging from 20.6 °C to 26.6 °C. Average precipitation is 126.5 cm yr^{-1} (NCSU 2000). Drought conditions prevailed throughout 2007 and much of 2008 prior to the wildland fire ignition, but had dissipated by late April in the study area (NCDMAC 2011). The Keech-Byrum Drought Index (KBDI), a key indicator of 1000-hour fuel and organic soil moisture for firefighting, was 261 on the day of fire ignition, and rose to a maximum value of 490 on 13 June 2008. Keech-Byrum Drought Index values between 201 and 400 are indicative of late spring or early growing season, when litter and duff layers are drying and begin to contribute to severe fire behavior. Keech-Byrum Drought Index values between 401 and 600 are typical of late summer and early fall in the southeastern United States. At these KBDI values, litter and organic (O) horizons of histosol soils burn intensely (Melton 1996).

The Evans Road Fire was contained within three land cover types: high pocosin (14 417 ha), low pocosin (2223 ha), and agricultural land (173 ha) (Figure 1). The majority of the aboveground vegetation-flaming combustion occurred from the start of the lightning ignition on 1 June 2008, until 5 July 2008. The period from 5 July until the fire was declared to be out was dominated by smoldering organic soil combustion. The land area that comprises the study site was burned previously during the Allen Road Fire that occurred on 7 April 1985, consuming ~40 000 ha. This fire had estimated total C emissions of 1 Tg C to 3.8 Tg C, and a heterogeneous burn pattern resulting in a C loss of 0.2 kg C m^{-2} to 1 kg C m^{-2} (Poutler *et al.* 2006).

Fire Severity Classifications

Differences in pre-fire and post-fire vegetative reflectance were categorized into four burn severity classes in Landsat Thematic

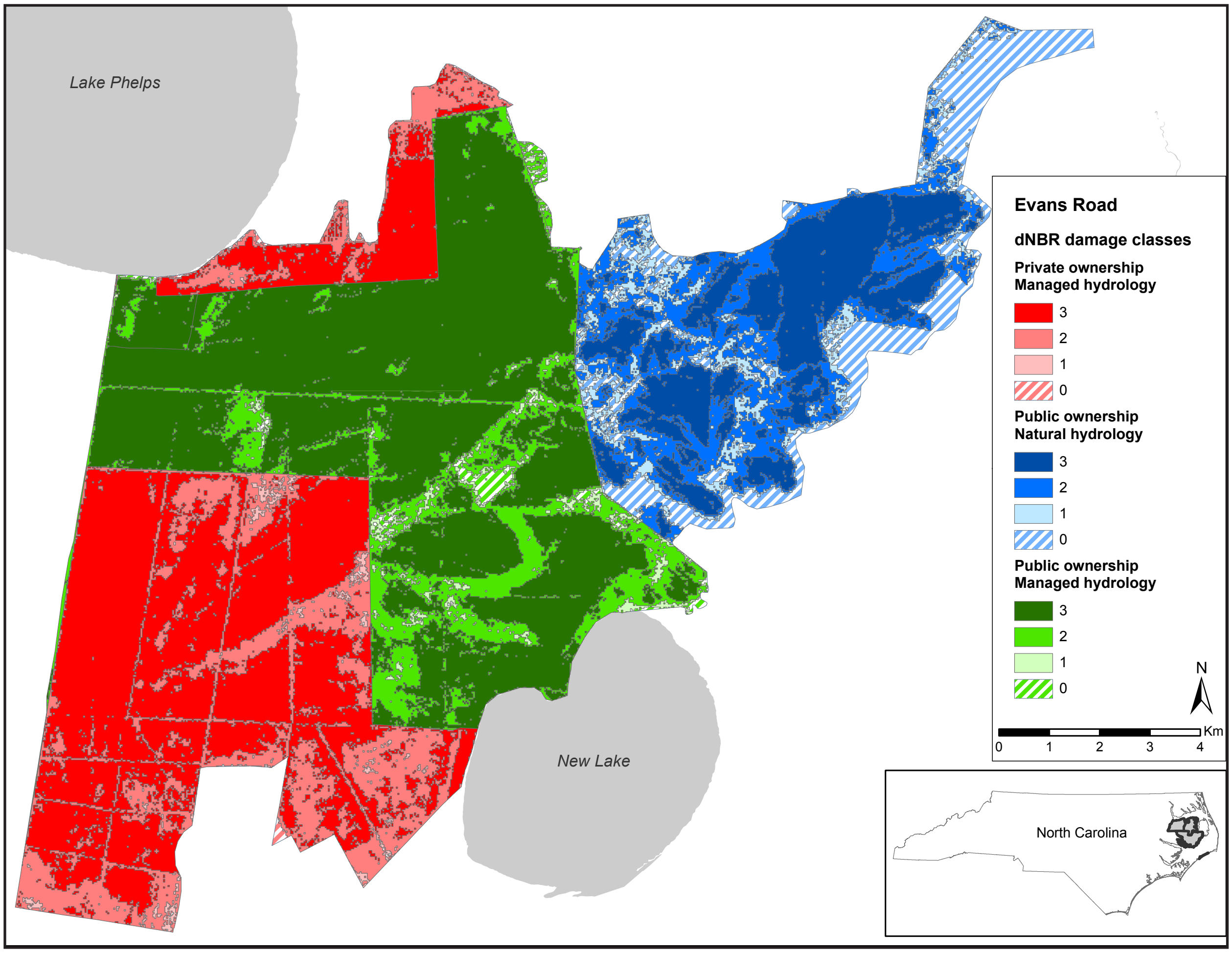

Figure 2. Location of differenced Normalized Burn Ratio (dNBR) burn severity classes. dNBR severity classes (0 = unburned to 3 = high severity), hydrological regime (natural and managed surface hydrology), and private and public ownership.

Mapper imagery according to the differenced Normalized Burn Ratio (dNBR) approach (Key and Benson 1999, 2006). The dNBR algorithm, derived from Landsat imagery, has been used extensively throughout the wildland fire community to identify vegetation burn severity classes. The dNBR is a ratio of reflectance values in the near- and mid-infrared bands, and pre-fire and post-fire differences exhibit differences in reflectance associated with vegetation. The dNBR values were classified into four Burned Area Reflectance Classification-Adjustable product groups using the Jenks Natural Breaks Classification method. This method is designed to determine the best arrangement of values into different classes by seeking to minimize each class's average deviation from the class mean, while maximizing each class's deviation from the means of the other groups. For our study site, the thresholds for our dNBR severity classes were 0 to 134 (unburned, Class 0), 135 to 168 (low severity, Class 1), 169 to 246 (moderate severity, Class 2), and 247 to 255 (high severity, Class 3).

There are two ownership categories within the study area: public and private. Public ownership consisted of Pocosin Lakes National Wildlife Refuge owned by US Fish and Wildlife Service and Lake Pettigrew State Park. Private lands consisted of privately owned property managed for agriculture and forestry land uses. The Evans Road Fire lightning ignition occurred on private lands on property south and west of the burned public lands that had been previously drained to lower the water table. Managed hydrology included land with maintained ditch networks and water control structures on public and private lands. Natural hydrology existed only on lower elevation public land (Figure 2). The selection of randomized field sampling sites was based on burn severity, ownership, and hydrology management combinations that had a land area of >200 ha. Of the 12 unique combinations of burn severity, ownership class, and hydrological management, the eight combinations with the largest area covered 97.6% of the entire burned area. These eight categories were selected for pre-fire and post-fire elevation measurement sampling. Land cover categories <200 ha were not sampled due to spatial constraints from polygon buffers between adjacent land cover categories and roads. These unsampled land cover categories comprised 2.4% of the burned area.

Land Cover Classification

A LIDAR-derived canopy height layer coupled with aerial photographic analysis informed a land cover classification of the study area. LIDAR canopy height was derived through a two-stage operation that initially calculated minimum-maximum differences of LIDAR returns in 6.1 m cell size using GRASS 6.4 (D. Newcomb, US Fish and Wildlife Service, Raleigh, North Carolina, USA, personal communication). Neighborhood analyses using a 3 m × 3 m area further refined canopy estimates to obtain 18.3 m cell size for the completed dataset (D. Newcomb, personal communication). A threshold of 4.0 m in the canopy height layer distinguished high from low pocosin. High and low pocosin and agricultural patches >5 ha were confirmed using 2006 to 2009 aerial photographs and were digitized (Figure 1).

Pre-Fire and Post-Fire Elevation Measurements

Pre-fire and post-fire elevation difference measurements determined organic soil loss. Pre-fire elevation data were derived using bare-earth points from North Carolina Floodplain Mapping Program (2003) LIDAR returns data sets acquired January to March 2001 (North Carolina Floodplain Mapping Program 2007*a*). Vertical accuracy for the study area was 10.1 cm to 15.0 cm Root Mean Square Error (North Carolina Floodplain Mapping Program 2007*b*). Randomly selected LIDAR bare-earth points were located in the field (Sawada 2004), and then co-location field measurements of post-fire elevations were collected using survey-grade Global Positioning System (GPS) equipment. This system employed two Trimble® R8 RTK Global Navigation Satellite System receivers, a TSC2 handheld data logger, and an HPB 450 radio. Field measurements were corrected to the National Spatial Reference System using the Online Position User Service (NOAA 2010). Static GPS field surveying vertical and horizontal accuracy was ±5 mm (Trimble 2004). Similar approaches to assess LIDAR elevation accuracy have been conducted in other contexts (Hodgson and Brensnahan 2004, Peng and Shih 2006).

Random soil-surface elevation points were selected >50 m and <500 m from road polygons to eliminate edge-induced fire and fuel effects (Richardson 1991). Geographic Information Systems (GIS) procedures were used to establish a 450 m strip for random point population (ArcGIS "erase" command) and polygon road buffers of 50 m and 500 m (ArcGIS "buffer" command). The dNBR damage class polygons were intersected with the 450 m strip to establish an equal number of random points for burn severity, ownership class, and hydrological management combinations. Randomly generated points were snapped to the nearest LIDAR bare-earth point.

Estimating Carbon Release

An estimation of C emissions from wildfires was first described by Seiler and Crutzen (1980) and modified by French *et al.* (2000). We modified the multistep process to include determining the area burned by relating the fire perimeter to land cover classification and C stocks associated with land cover classifications geospatially linked to the C stocks in soil series and their horizons from the USDA Natural Resources Conservation Service (NRCS) Soil Survey Geographic (SSURGO) database. We then estimated the aboveground and belowground C consumption and emissions. Consumed C fractions were defined for both aboveground biomass and ground layers, which were associated with land cover type and soil drainage class, respectively. The equation (1) modified by French *et al.* (2002) and Kasischke and Bruhwiler (2003) from that of Seiler and Crutzen (1980) was used to estimate the total C release (Ct) from burning of both aboveground biomass and ground layers:

$$Ct = A(C_a\beta_a + C_g\beta_g), \tag{1}$$

where A is the total area burned (ha); C_a is the average C density of aboveground biomass (kg C m^{-2}), assuming that the C fraction of the aboveground biomass is about 0.50; β_a is the fraction of aboveground biomass consumed during a fire; C_g is the C density (kg C m^{-2}) of ground layers exposed to a fire; and β_g is the fraction of the organic layers consumed by the fire.

Vegetation Carbon Pool Estimation

The general approach to estimating aboveground C emissions employed estimates of area burned, fuel loading (biomass per unit area), and consumption proportions following various studies that addressed biomass combustion (Seiler and Crutzen 1980, French *et al.* 2002, French *et al.* 2011). Biomass calculations were then multiplied by 0.5 to attain C estimates.

Land cover classification estimates informed (1) the area burned, and (2) the basis of the fuel loading figures. Tree foliage, litter, and shrub biomass estimates coupled with tree density measures comprised the specific components of aboveground C emissions (Tables 1 and 2).

Table 1. Aboveground carbon content (t C ha^{-1}).

Land cover	Litter[a]	Shrub[b]	Foliage[c]	Total
High pocosin	4.32	21.63	0.68	26.63
Low pocosin	2.68	11.21	0.02	13.91
Agriculture[d]	0.00	0.00	0.00	4.50

[a] Litter and shrub biomass measures were derived from Ottmar and Vihnanek 2000.
[b] Shrub and tree density measures were derived from Ottmar and Vihnanek 2000.
[c] Foliage biomasses were derived from equations developed by Schroder *et al.* 1997, MacLein and Wein 1976 (cited in Jenkins *et al.* 2003).
[d] Agriculture biomasses were estimated from a South Carolina coastal plain wheat field in mid-May (Bauer *et al.* 1998).

A nearby unburned high pocosin, similar to the study area, had 4.32 t C ha^{-1} of litter and 21.63 t C ha^{-1} of shrub, and a nearby unburned low pocosin had 2.68 t C ha^{-1} and 11.21 t C ha^{-1} of litter and shrub, respectively (Tables 1 and 2). Foliage biomass estimates employed allometric equations multiplied by a foliage ratio equation for Eastern US conifers and red maple (*Acer rubrum* L.): the dominant evergreen and deciduous species in the study area system (Table 3). Only the high pocosin vegetation type had measurable red maple saplings.

Fire burned 173 ha of agricultural fields in Tyrrell County. The county's major crops by area planted were soybean (*Glycine max* [L.] Merr.; 40%), corn (*Zea mays* L.; 30%), and wheat (*Triticum aestivum* L.; 25%) (USDA 2007). In early June, only wheat had significant aboveground biomass; estimates for coastal plain wheat in mid-May were 4.5 t C

Table 2. Tree and sapling density (ha^{-1}) and diameter (cm)[a]

Vegetation type	Tree density (trees ha^{-1})	Sapling density (trees ha^{-1})	Average tree dbh (cm)	Average sapling dbh (cm)
Low pocosin	7	7	10.9	0.5
High pocosin	58	15	20.6	6.9

[a] Density and diameter values derived from stand numbers P-S 06 and P-S 04 in Ottmar and Vihnanek 2000.

Table 3. Biomass equations used in this study. The Eastern US conifer equation is from Schroder *et al.* (1997) and was used for the following species: eastern hemlock, eastern white pine. The red maple equation is from MacLean and Wein (1976) as cited in Jenkins *et al.* (2003). Variables are B = biomass (kg m^{-2}), FR = foliage ratio (dimensionless), d = diameter at breast height.

Vegetation type	Biomass equation	Foliage ratio equation
Eastern US conifer	$B = 0.5 + \left[15000\left(\frac{2.7d}{2.7d + 364946}\right)\right]$	$FR = e^{\left[-2.9584 + \left(\frac{4.4766}{d}\right)\right]}$
Red maple	$\log_{10}(B) = -0.8602 + 1.7963\log_{10}(d)$	$FR = e^{\left[-4.0813 + \left(\frac{5.8816}{d}\right)\right]}$

ha^{-1} (9 t biomass ha^{-1}; Bauer *et al.* 1998; Table 1).

We adjusted land cover biomass estimates of the three cover types (high pocosin, low pocosin, and agriculture) to reflect varying fire severity measures (dNBR). Aboveground fire severity varied spatially and resulted in heterogeneous fuel consumption in pocosin litter, shrub, and foliage biomass. Qualitative field procedures estimated these differences for each of the four dNBR values through visual examination of the fuel types in multiple, randomly selected georeferenced locations. Higher dNBR values were associated with higher aboveground combustion fraction. Litter, shrub, and foliage combustion fractions ranged from 90% to 100% for severity Class 3; severity Class 0 had combustion fractions ranging from 0% to 10% (Table 4).

Table 4. Aboveground combustion fraction.

dNBR severity class	Litter	Shrub	Foliage
3	100%	90%	100%
2	60%	60%	30%
1	20%	20%	10%
0	10%	0%	0%

Soil Organic Carbon Loss Estimation

We calculated belowground soil C emissions by intersecting GIS-derived area of land cover, hydrology modification category (managed and natural), and ownership category (private and public) combinations for the North Carolina counties of Hyde, Tyrrell, and Washington, with the SSURGO soil database elements (soil series, organic soil horizon depth, bulk density, C content; Soil Survey Staff 2011*a*) and calculated mean depth of organic soil horizon depth changes based on the difference of pre-burn LIDAR data points and post-burn field survey of co-located points.

Soil within the fire perimeter consisted of five muck soil series (Soil Survey Staff 2011*b*) with organic layers of 129.5 cm to >203.2 cm thick and two loamy soil series with surface mineral horizons ranging from 40.6 cm to

101.6 cm thick and underlying organic horizons ranging from 20.3 cm to more than 132.1 cm thick:

- Belhaven (loamy, mixed, dysic, thermic Terric Haplosaprists), 774.7 ha;
- Ponzer (loamy, mixed, dysic, thermic Terric Haplosaprists), 265.8 ha.
- Pungo (dysic, thermic Typic Haplosaprists), 6945.2 ha.
- Roper (fine-silty, mixed, semiactive, acid, thermic Histic Humaquepts), <1 ha.
- Scuppernog (loamy, mixed, dysic, thermic Terric Haplosaprists), 644.4 ha.
- Fortescue (fine-silty, mixed, active, acid, thermic Cumulic Humaquepts), 7.7 ha.
- Weeksville (coarse-silty, mixed, semiactive, acid, thermic Typic Humaquepts), 25.9 ha.

Soil Data Extraction and Calculations

Soil Survey Geographic soils tabular and spatial data were downloaded from the United States Department of Agriculture Natural Resource Conservation Service Soil Data Mart (http://sdmdataaccess.nrcs.usda.gov/). The information in the SSURGO database provides representative chemical and physical characteristics for each unique soil series to aid in the identification and classification of soils. Land management activities (i.e., conversion of woodland to a plowed and drained agricultural field) may alter some of the properties described in the official soil series description. The NRCS Soil Data Mart contains digital representations of the county-level soil maps published in NRCS Soil Survey books. The soil data were clipped to the fire perimeter boundary, and intersected with the severity and ownership classes to create analysis polygons. Microsoft Office Access® database software was used to import county-level soil database tables for map unit, component, and horizon, using procedures from the NRCS Soil Survey template geodatabase (available at http://sdmdataaccess.nrcs.usda.gov/). These tables are related in the following manner: each map unit has one or more components, and each component has one or more horizons. The area of the map unit made up by a single component is represented by a percentage value in the component table. In some cases, other soil series that are too small to map as components may be found within a map unit, resulting in component percentage values for a map unit that do not sum to 100%. In those cases, component percentages were scaled to ensure that 100% coverage was attained.

We developed a Structured Query Language query to extract horizon-level values and summarize them at the component and map unit levels. We calculated averages for representative, low, and high bulk density in g cm^{-3} (dbovendry_r, dbovendry_l, and dbovendry_h from the soil data, respectively), and representative, low, and high percent organic matter (om_r, om_l, and om_h from the soil data, respectively) within components from the data for all burned horizons within each component. Horizon depth was calculated by subtracting depth to the top of the horizon from depth to the bottom of the horizon (hzdepb_r – hzdept_r). The summed values for burned horizons within each component were used to calculate weighted averages for each map unit using the component percentage value (comppct_r). The map unit-level calculations were joined to a GIS shapefile of the geographic extent of each soil map unit using ArcGIS 10.2.2 software (ESRI 2014). The join was conducted using the common map unit key (MUKey) column in each data set. After calculating area for the clipped and intersected polygons, soil organic C was scaled up to the polygon area by multiplying the amount of C consumed per cubic meter by the number of cubic meters in the polygon, and converting units from kilograms to metric tons.

Soil organic C was calculated as adapted from Rasmusssen 2006 and Tan *et al.* 2007 as follows:

$$SOC_i = 0.05 D_i \rho_{bi} OM_i \ , \qquad (2)$$

where SOC_i is the soil organic C content (kg m^{-2}), *i* is either the O or Oa horizon (the Oa soil horizon is a very dark layer of highly decomposed humus [sapric]); D_i is the depth of soil consumed (cm); ρ_{bi} is the soil bulk density (g cm^{-3}); and OM_i is the organic matter weight percentage. Soil C emissions were estimated for each of the eight severity classes above 200 hectares that comprised ~97% of the area delineated for the Evans Road Fire. Carbon emissions were summed at high, representative, and low values by dNBR class, ownership category, and hydrology management category for each soil series within the burned area.

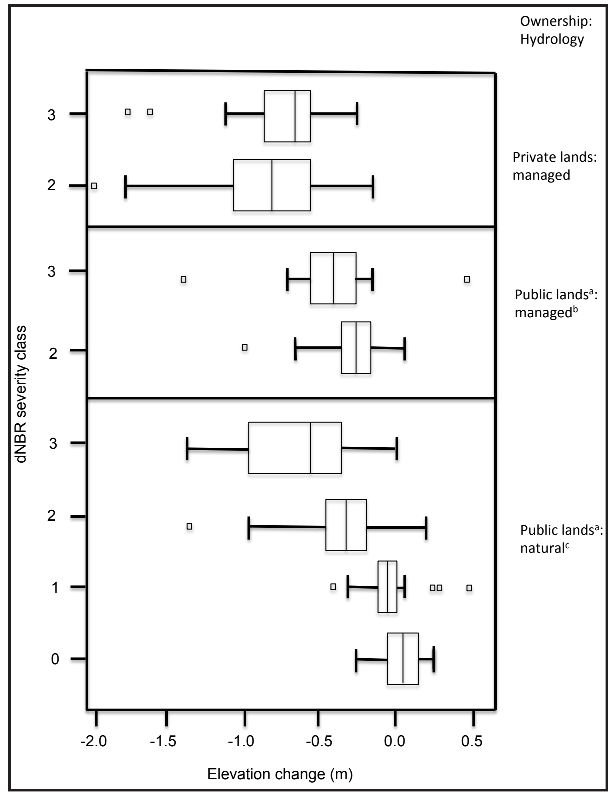

Figure 3. Boxplots of field elevation change measures categorized by dNBR severity class, ownership, and hydrology. Box-and-whisker plots illustrate box with median and 20th and 80th percentiles, and the whiskers with 10th and 90th percentiles. Outliers are plotted separately (□). US Fish and Wildlife Service and North Carolina State Parks constitute public ownership. Managed hydrology is characterized by a regular and extensive system of maintained ditches and gated culvert water control structures. Natural hydrology is characterized by unmanaged surface hydrology, absence of ditches, and water control structures. [a] US Fish and Wildlife Service and North Carolina State Parks constitute public ownership. [b] Managed hydrology is characterized by a regular and extensive system of maintained ditches and gated culvert water control structures. [c] Natural hydrology is characterized by unmanaged surface hydrology, absence of ditches, and water control structures.

RESULTS

Organic Soil Combustion

The study area had a mean burn depth of 0.42 m (Figure 3). Increased peat burn depth was correlated with more severe dNBR classes on private ownership lands with altered hydrology. Fire severity classes with low severity or no fire impact (dNBR = 0 or 1) were correlated with public lands with high water tables and closed canopies of loblolly bay, pond pine, and red maple. Sites dominated by hardwood species typically had closed canopies with a contiguous layer of hardwood litter and were most often observed in areas with unaltered hydrology, standing surface water, and dNBR = 0 and 1. Pond pine was found in areas with slightly higher elevations and altered hydrology. At these sites, high pocosins consisted of pond pine forest dominated stands with medium to closed canopy densities and deep litter layers, and low pocosins consisted of shrub dominated vegetation with sparse pond pine densities and minimal conifer litter. Peat burn depth was dependent on distance to the top of the water table. One indicator of areas with minimal peat burn depth was the immediate sprouting of bracken fern in the weeks after the flaming front of the fire.

dNBR Severity Classes and Ownership and Hydrology Regimes

The majority of the 16 813 ha burn area was in the most severe dNBR class of 3 (10 700 ha). Burn severity in Class 3 constituted nearly 100 % consumption of all aboveground small- and medium-diameter fuels, leaving only tree boles and larger branches. The intermediate burn severity classes (1, 2) had a mosaic of combusted litter fuels, organic soil, and aboveground foliage and fine woody fuels. Fires in areas of Class 0 burned only a fraction of the litter, leaving organic soil and aboveground vegetation intact. Private- and public-managed hydrology land cover categories had roughly the same area burned (6100 ha and 6800 ha respectively), and the burned area over public natural hydrology covered 4000 ha (Figure 4). Most of the vegetation burned was high pocosin (14 417 ha); patches of low pocosin existed in the eastern portions of the study area, totaling 2223 ha. The fire burned a relatively small area (173 ha) of agricultural lands along the northern borders (Figure 1).

Aboveground Carbon Emissions

Aboveground C emissions ranged from approximately 25 t ha^{-1} for the highest severity class to <1 t ha^{-1} for the lowest severity class (Figure 5). Differences in C emissions in groups sharing the same dNBR value reflect different low and high pocosin fractions among the ownership and hydrology categories. High pocosin vegetation type had the highest fuel loadings and aboveground C emission rate: 24.5 t ha^{-1} (Table 5).

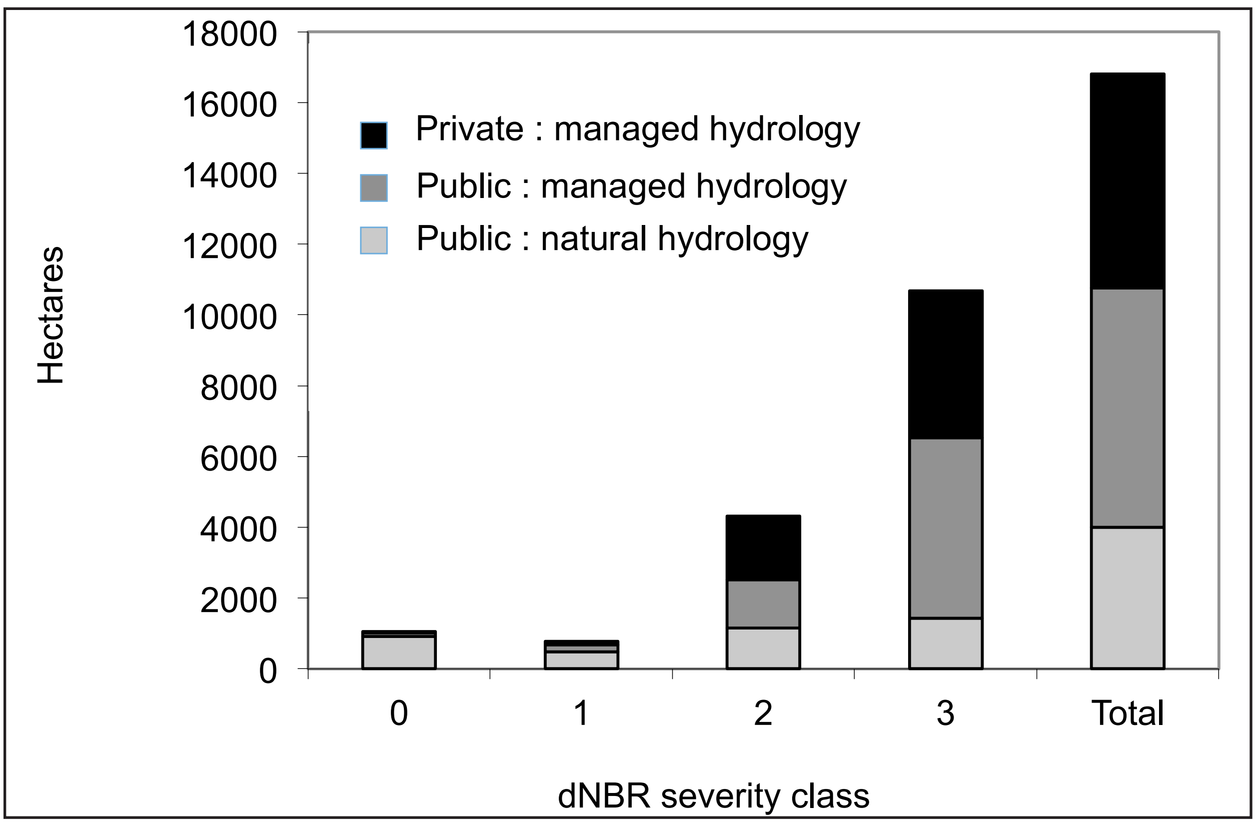

Figure 4. dNBR severity class by ownership and hydrological regime (ha). dNBR classes ranging from class 0 (indicates limited litter consumption only) to class 3 (indicates nearly 100 % aboveground fuel consumption).

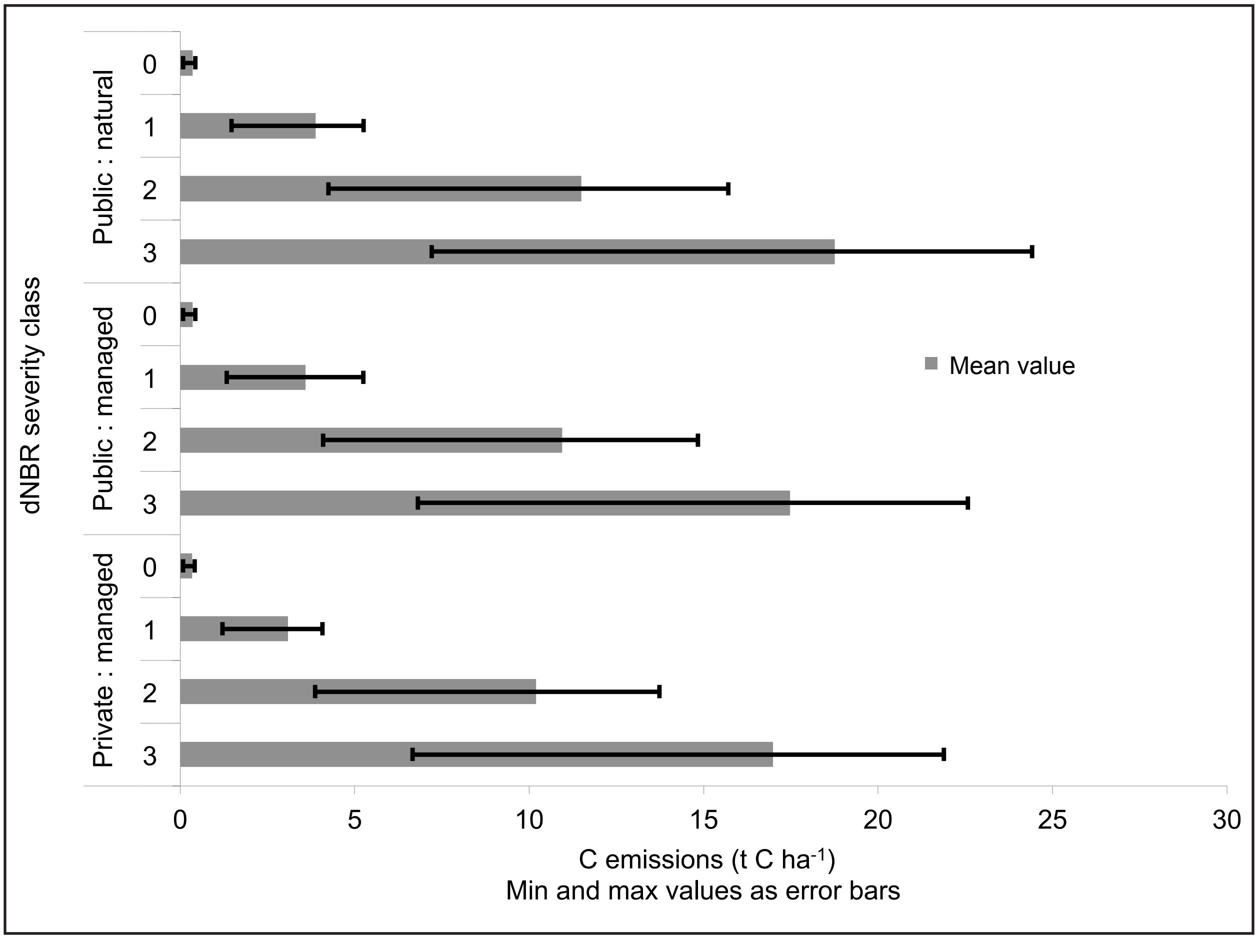

Figure 5. Aboveground carbon emissions by ownership and hydrology (t C ha^{-1}). Minimum and maximum values are illustrated as error bars.

Aboveground C estimates were determined by applying field biomass measures (Ottmar and Vihnanek 2000, Jenkins *et al.* 2003) and dNBR classes ranging from unburned (indicates limited litter emissions only) to class 3 (indicates nearly 100 % aboveground emissions). The ownership and hydrology values are metric tons of C consumed. Average aboveground C emissions over the entire study area totaled 18.33 t C ha^{-1}.

Aboveground C emissions based on vegetation class incorporated separate calculations for litter, shrub, and foliage in pocosins, with areas of taller, thicker vegetation (high pocosins) emitting higher rates of C relative to the other two vegetation types (Table 5). The ecosystem's shrub layer contributed the highest proportion of C in dNBR classes 1 to 3.

Belowground Carbon Emissions

Belowground carbon emissions ranged from 15 t C ha^{-1} in areas of natural hydrology and dNBR = 0 to a high of 1062 t C ha^{-1} in areas of managed hydrology, private ownership, and dNBR = 2 (Figure 6). The highest land cover C emission value was 805 t C ha^{-1} in low pocosin and dNBR = 2, and the lowest measured emission level was 15.5 t C ha^{-1} in high pocosin with dNBR value = 1 (Table 6). The entire study area belowground emissions averaged 544 t C ha^{-1}.

Factors Affecting Belowground Emissions

A number of factors affected soil loss due to combustion. Soil loss differed among

Table 5. Aboveground carbon emissions by land cover class (t C ha^{-1}).

dNBR severity class	High pocosin	Low pocosin	Agriculture[a]
0	0.43 (litter)	0.27 (litter)	0.45 (crop)
1	5.26 0.86, 4.33, 0.07 (litter, shrub, foliage)	2.78 0.54, 2.24, <0.01 (litter, shrub, foliage)	0.90 (crop)
2	15.78 2.59, 12.98, 0.2 (litter, shrub, foliage)	8.34 1.61, 6.73, 0.01 (litter, shrub, foliage)	2.70 (crop)
3	24.47 4.32, 19.47, 0.68 (litter, shrub, foliage)	12.79 2.68, 10.09, 0.02 (litter, shrub, foliage)	4.50 (crop)

[a] Fuels burned were crop biomass.

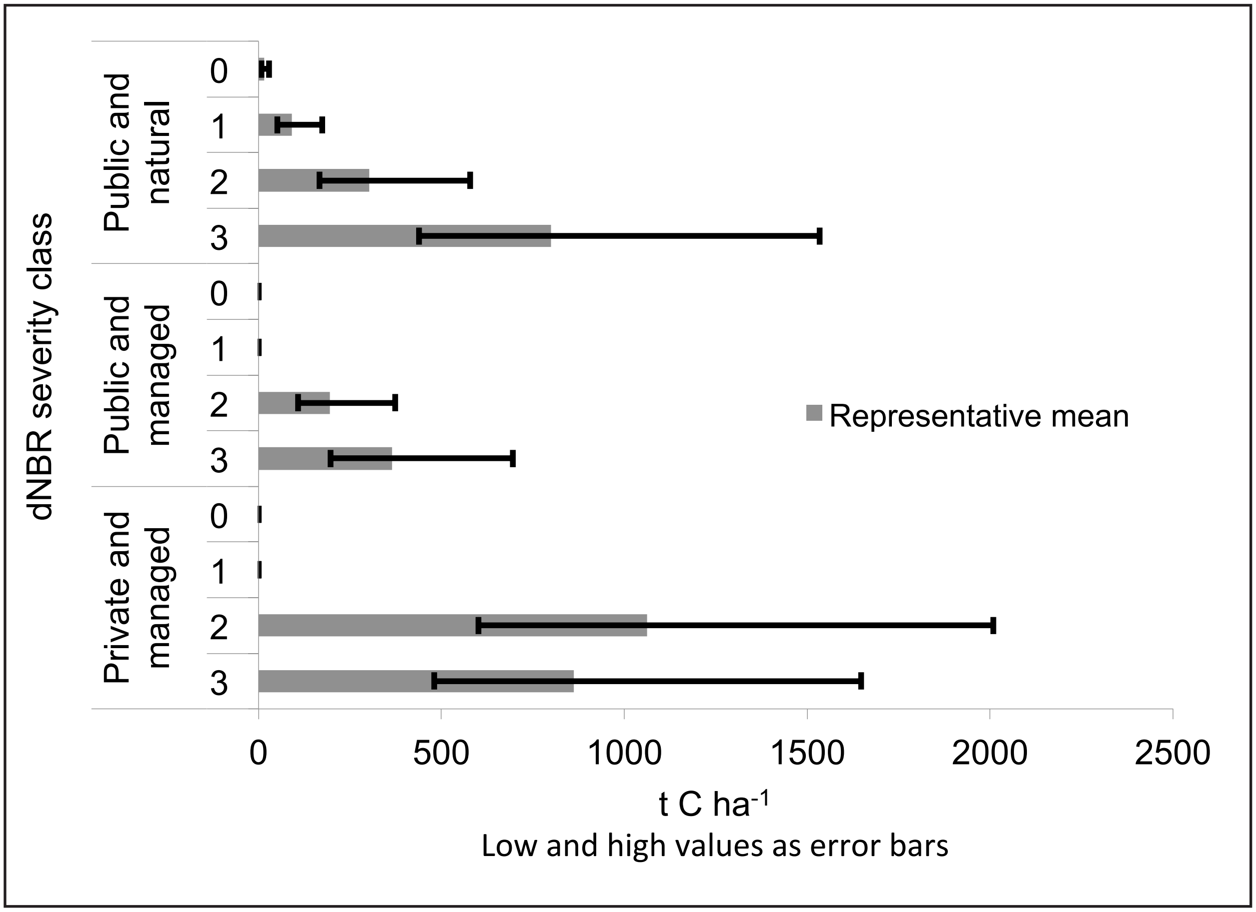

Figure 6. Belowground carbon emissions in dNBR classes (t C ha^{-1}). Minimum and maximum values are illustrated as error bars.

Table 6. Belowground carbon emissions by land cover class (t C ha^{-1}) in land cover categories >200 ha.

dNBR severity class	High pocosin	Low pocosin	Agriculture
0[a]	0.00	0.00	-[b]
1	15.50	-	-
2	557.36	805.84	-
3	623.79	625.29	-

[a] dNBR severity Class 0 had no measurable belowground C emissions.
[b] Land cover categories <200 ha were not sampled and are designated by (-).

dNBR classes and ownership and hydrology regimes with means ranging from 1 cm to 74 cm; standard deviation for all classes was 0.42 m (Table 7). Analysis of covariance (ANCOVA) results indicated that all factors (drainage, ownership, and dNBR) had a significant effect on pre-burn and post-burn elevation change (dependent variable). These factors explain 34 % of the variation in soil elevation change. Tukey-Kramer adjustment for multiple comparisons indicated that managed and natural hydrology as well as private and public ownership had different elevation loss values (P < 0.001). Pairwise comparisons between mean elevation differences of dNBR classes were significantly different ($P \leq 0.0013$) except for dNBR classes 0 and 1 and dNBR classes 2 and 3 (Table 8).

Table 7. Mean soil loss (cm) by dNBR class and ownership and hydrology in land cover categories >200 ha (- designates categories <200 ha).

dNBR severity class	Private managed hydrology	Public managed hydrology	Public natural hydrology
0	-[a]	-	1
1	-	-	6
2	74	13	20
3	57	24	52

[a] Land cover categories < 200 ha were not sampled and are designated by (-).

Table 8. Pairwise comparisons of mean dNBR values for Tukey-Kramer adjustment for multiple comparisons.

Pairwise comparisons of dNBR values	*P* values
0 versus 1	0.9625
0 versus 2	0.0003
0 versus 3	<0.0001
1 versus 2	0.0013
1 versus 3	0.0010
2 versus 3	0.6774

DISCUSSION

Carbon Emissions

Carbon emissions from the Evans Road Fire ranged from 0.03 kg C m^{-2} to 107.24 kg C m^{-2}. We estimated belowground fire emissions of 9.16 Tg C (544.43 t C ha^{-1}) and aboveground fire emissions of 0.31 Tg C (18.33 t C ha^{-1}), for total emissions of 9.47 Tg C (562.76 t C ha^{-1}). These values are higher than those reported for the Allen Road Fire that occurred 23 years earlier. The Allen Road Fire encompassed the Evans Road Fire perimeter, and had C emissions ranging from 0.2 kg C m^{-2} to 11 kg C m^{-2} and total emissions of 1 Tg C to 3.8 Tg C (Poutler *et al.* 2006). The lower C emissions reported for the Allen Road Fire are the result of the author's use of lower peat burn depths (0.01 m, 0.05 m, and 0.10 m), based on the literature (Hungerford *et al.* 1994) for similar peat wildfires. In contrast, the Evans Road Fire had a mean peat burn depth of 0.42 m and a maximum burn depth of 1.7 m, which were based on post-fire field surveys. Historically, during periods of prolonged drought and low water table, coastal peatlands have long-duration, high-severity wildfires. Rodriguez *et al.* (2012) and others (Richardson and Gibbson 1993) have suggested that formation and evolution of some Carolina bays in the coastal plain of the southern

United States are a result of deep-burning wildfires in peatlands that decreased surface elevation by >1 m. Lake Mattamuskeet, a 162 km^2 Carolina bay with a mean depth of 1 m, is reported to be a result of a wildfire that burned for 13 months. During the past century, drainage for agriculture and forest management has altered the wildfire regime, leading to increased fire frequency and severity during periods of drought. Large peatland wildfires (10 000 ha to 100 000 ha) in North Carolina have occurred frequently when the Palmer Drought Severity Index, an indicator of dryness, is in the mild to severe drought stage (Hungerford *et al.* 1994).

Role of Temperate Peatlands in the US and Global Wildfire Carbon Emissions

Temperate peatlands have recently been recognized as a significant source of atmospheric C during long-duration organic soil wildfires. Land use and land cover changes have dramatically altered temperate forested wetlands, disrupting ecosystem services through deforestation, surface hydrology manipulation, and organic soil management for agriculture and forestry. During periods of drought, temperate peatlands experience shortened wildfire return intervals (<20 yr to 80 yr), and increased wildland fire duration and behavior, depth of organic soil smoldering combustion, and C flux and fire emissions. Poulter *et al.* (2006) have suggested that temperate peatland fires may globally contribute 0.32 Pg C annually during drought years. In contrast, tropical peatland fires in Indonesian during 1997 had less variable depths (0.18 m) and lower C emission rates (260 t C ha^{-1} to 315 t C ha^{-1} from soil and 68 t C ha^{-1} from vegetation; Page *et al.* 2002, Ballhorn *et al.* 2009). Page *et al.* (2002) and Ballhorn *et al.* (2009) estimated that between 0.81 Gt and 2.57 Gt of C could have been released from peatland soils and vegetation. These rates are much higher than the 11 t C ha^{-1} to 60 t C ha^{-1} for boreal peatland (Kasischke and Stocks 2000, Amiro *et al.* 2001, Turetsky and Wieder 2001, Boby *et al.* 2010). North American ecosystems have estimated wildfire C emissions of 2.3 t C ha^{-1} to 60 t C ha^{-1}, with the upper estimate in northern conifer forests (French *et al.* 2011). Lower C emission rates in boreal regions are in part due to shallower soil consumption depths: typically <10 cm, with maximum values of approximately 1 m (Dyrness and Norum 1983, Zoltai *et al.* 1998). Poulter *et al.* (2006) have reported that temperate peatland fires with a return interval of 30 yr to 80 yr may emit 0.03 Pg C to 0.32 Pg C during drought years. This contribution represents as much as 30 % of the interannual variability of atmospheric C from global biomass burning (~1 Pg C yr^{-1}).

Application of LIDAR and GPS in Wildfire Carbon Emissions

The use of LIDAR elevation data were first reported by Poulter *et al.* (2006) as a method to determine the severity of wildfire organic soil combustion. In their study, LIDAR elevation data was used to constrain the estimates of organic soil combustion in the absence of historical burn depth data for the wildfire. The authors estimated that the vertical accuracy of the LIDAR elevation data was ±0.15 m, and the horizontal resolution was 6 m, resulting in detectable burn depths of 0.15 m to 0.30 m. The Poulter *et al.* (2006) study illustrates the difficulty in estimating wildfire C emissions from organic soil combustion. Histosol soils present challenges to the establishment of permanent survey markers because of the deep deposits of mainly organic materials (>12 % to 18 % C by weight) formed during the Holocene epoch, under conditions of saturation and flooding, that persist today. Stainless steel sectional rod monuments are one method to mark elevations in histosols, but the metal rods themselves introduce survey point bias through the additional heat conduction and resulting soil combustion during a wildfire. In the absence of permanent survey points within the perimeter of the Evans Road Fire, we developed a novel method and approach to estimate C wildfire emissions from organic soil combustion utilizing existing pre-burn LIDAR ground point elevations (vertical accuracy ±0.10 m to 0.15 m) and co-located post-burn field points using static survey-grade GPS equipment (horizontal and vertical accuracy ±5 mm). Additional studies are needed to address the detectable burn depths from wildfires using technologies with differing horizontal and vertical elevation accuracies.

Land Management Impacts on Peatland Fires

One land management strategy implemented on public and private peatlands is the use of water control structures to control the amount of water discharged through roadside channels, and thereby raise or lower the water table. Gated culverts were constructed and managed by the US Fish and Wildlife Service to control surface water elevation on public lands before the Evans Road Fire. The US Fish and Wildlife Service manages the water table to reduce wildland fire risk and to improve migratory waterfowl habitat and forage. In contrast, the adjacent private lands where the lightning ignition occurred were being managed to lower the water table (>2 m) for agriculture and hunting. Figure 3 illustrates the difference in burn depth between managed hydrology on public versus private lands. Poulter *et al.* (2006) estimated that, in order to maintain the region's peatlands as a C sink at the current rate of C sequestration (0.109 kg C m^{-2} yr^{-1} to 0.127 kg C m^{-2} yr^{-1}), the fire return interval must be >20 yr to 80 yr. The Allen Road and Evans Road wildfires demonstrate that peatland wildfires are a C source when the ignitions occur during periods of repeated drought and when land mangers lower the water table through manipulation of water control structures to increase surface water discharge.

Conclusions

Wildland fire C emissions for peatlands are being estimated despite the dearth of data associated with soil variables (extent and spatial pattern of temperate peatlands, peat depths, changes in bulk density with depth, and changes in C content with depth) and wildland fire characterization data (fire location, fire weather, fire perimeter, fuel biomass, fuel consumption biomass, and emission factors available to estimate changes in C stocks and non-CO^2 emissions). The difficulty in acquiring real-time temperate peatland wildfire C emissions data underscores the importance of estimating C emissions using a combination of field-based and remote-sensing techniques to refine aboveground and belowground combustion estimates, as well as the incorporation of land management factors such as surface hydrology and land use that contribute to the depth of wildfire peat combustion. Previous studies that estimated C emissions from peatland fires have been constrained by the lack of pre-burn and post-burn paired point data to accurately determine the depth of soil consumption across the fire area (Page *et al.* 2002, Poutler *et al.* 2006); however, a more recent study has utilized a pre-burn and post-burn LIDAR method (Reedy *et al.* 2015). Reedy *et al.* (2015) concluded that the LIDAR elevation error was not a significant contributor to uncertainty in C emissions from fires with substantial peat consumption. In our study, a limited research budget prevented a LIDAR-to-LIDAR data point comparison to determine burn depth. We used publicly available pre-burn LIDAR data combined with field survey soil-surface elevation data to determine burn depth.

It is likely that the combined impacts from surface hydrology management and a multiyear drought enhanced the severity of the Evans Road Fire by substantially lowering the water table, thereby enhancing the depth of soil combustion. The hydrological restoration of drained peatlands to manage and raise the water table contributed to the lower peat burn depths on public lands during the Evan Road Fire. The reinstatement of natural hydrology conditions in drained agricultural and forestry lands and the use of water control structures to store surface waters during periods of drought has been shown to be potential land management strategies to reduce the rate and extent of soil carbon consumption in temperate peatlands fires. This study suggests that a similar peatland management strategy could result in a significant reduction in global peatland wildfire C emissions and interannual variability of atmospheric C.

ACKNOWLEDGEMENTS

We are grateful for the expert advice of US Fish and Wildlife Service wildland fire staff members: S. Wilder, V. Carver, and W. Stanton, and Pocosin Lakes National Wildlife Refuge Manager H. Phillips. We thank the staff of the Pocosin Lakes National Wildlife Refuge for field assistance. Funding was provided by the US Fish and Wildlife Service order number F11PX05109 and the Joint Fire Science Program under US Fish and Wildlife Service contract number GS-35F-4721G. The use of trade or firm names in this publication is for reader information and does not imply endorsement by the Association for Fire Ecology of any product or service.

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