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RESEARCH

Patterns of Soil Dissolvable Matter Following Prescribed Fire Reintroduction in Hurricane-Damaged Southeastern U.S. Coastal Forests

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Abstract

Prescribed fire is a common management practice for restoring and preserving coastal forest ecosystems in the southeastern U.S., increasingly complicated by wind events that alter fuel distribution and loads, and potentially fire severity. We quantified variations in dissolvable matter of soil to identify how variations in fire severity influence carbon and nutrients within the first year following the re-introduction of prescribed fire to two recently hurricane-impacted, fire-dependent coastal forests along the Gulf of Mexico, USA: (1) Perdido River Preserve, Florida, a young (~20-year old) longleaf pine forest with an open herbaceous understory and (2) Weeks Bay National Estuarine Research Reserve, Alabama, a mature (60+-year old) slash pine forest with an understory containing shrubs and hardwood saplings. At the time of fire re-introduction, both forests had variable fuel biomass, likely driven by previous wind events. At Perdido, fire severity positively correlated with an immediate increase in water-extractable organic carbon (WEOC), nitrogen (WEN), and the carbon-to-nitrogen ratio (C: N). No pronounced change in WEOC was seen at Weeks Bay. Elevated WEN levels were measured at both sites one-year post-fire, resulting in a decreased C: N ratio from 15 to 8, indicating enhanced microbial-derived carbon. We observed a transient increase in dissolved phosphate and small pyrogenic carbon at Perdido, while humic-like fluorophores increased at Weeks Bay. Thus, varying fuel loads and, consequently, fire severity in fire-managed, wind-disturbed forests can temporarily alter patterns of dissolvable matter, potentially leading to more pronounced consequences for carbon and nutrient loss in young, open forests.

Highlights

- Fire severity increased with fine fuel load across wind-impacted coastal forest.
- Prescribed fire re-introduction to wind-impacted coastal forests modifies soil WEOM.
- Fire severity directly influences WEOC and WEN levels post-fire.
- UV-Vis and fluorescence spectroscopy track changes in WEOM post-fire.
- Comparative analysis of two sites shows distinct site-level soil responses.

Keywords Water-extractable organic matter · Fire severity · Eutrophication · Nitrogen cycling · Carbon cycling

Extended author information available on the last page of the article

1 Introduction

Preserving and restoring coastal forest ecosystems across the southeastern U.S. depends on frequent, low-severity prescribed fires. Maintaining their composition and structure provides vital ecosystem services, including carbon sequestration, and improves water quality (Song et al. 2013). However, the increasing frequency of hurricane disturbances modifies fuel loads by increasing leaf and woody debris (Suzuki et al. 2019) and enhancing biomass of live understory vegetation (Angulo-Sandoval et al. 2004). These variations in fuel load distribution can lead to differences in fire behavior (Berlin 1980), directly affecting standing carbon stocks by removing wood, vegetation, and soil organic matter. Variations in fire severity can be reflected by rapid alterations in the concentration and composition of water-extractable organic matter (WEOM) and nutrients. Understanding these changes provides insight into the impact of prescribed fire on carbon and nutrient cycling in wind-damaged forests.

Within the coastal forests of the southeastern U.S., the longleaf (*Pinus palustris*) and slash (*Pinus elliottii*) pine ecosystems are particularly noteworthy for their ecosystem services (Bracho et al. 2012; Samuelson et al. 2014). Longleaf and slash pine flatwoods, characterized by a sparse canopy and a rich understory of grasses and forbs, depend on frequent, low-severity surface fires to maintain an open structure and prevent mesophication (Fowler and Konopik 2007; Van Lear et al. 2005; Waldrop et al. 1992). Decades of fire exclusion and land-use change transformed open savanna-woodlands into closed forests dominated by shade-tolerant, and often fire-sensitive or opportunistic species, reducing key ecosystem services such as carbon sequestration and soil fertility (Varner et al. 2005). To restore and maintain these services, prescribed fire is commonly used in various fire-dependent ecosystems in the southeastern U.S. (Cummins et al. 2023). These low-severity surface fires mimic the natural fire regime and maintain relatively open conditions, thereby increasing herbaceous understory biodiversity and regeneration of desired pine and oak trees (Ryan et al. 2013). They also reduce fuel loads, thereby mitigating the risk of uncontrolled wildfires (Ivey et al. 2024). Contrary to intuition, prescribed fire can increase carbon stocks by reducing the likelihood of future high-severity wildfires, which are far more destructive and carbon-releasing (Hurteau and North 2010). Prescribed fire can also promote the formation of more stable forms of soil organic carbon, pyrogenic carbon (PyC). (Certini 2005; Paustian et al. 2016). Thus, frequent application of prescribed fire in southeastern U.S. forests is critical for their restoration and maintenance.

In addition to soil organic carbon, fire impacts the mobility and availability of nutrient elements like nitrogen (N) and phosphorus (P) (DeBano 2000). N and P are essential for plant growth but can contribute to eutrophication in adjacent water bodies if mobilized post-fire (DeBano 2000; Waters et al. 2023), which could be especially important in southeastern coastal forests. For instance, N, present as dissolved organic N (DON) or nitrate (NO_3^-), can increase immediately post-fire due to enhanced mineralization, but long-term N dynamics highly depend on vegetation recovery and microbial activity (Wan et al. 2001; Koyama et al. 2010). P, while less mobile than N, can be released in soluble forms following the combustion of organic matter and redistribution of ash, potentially impacting nutrient-sensitive waters (Certini 2005; Correll 1998; Waters et al. 2023). The frequency of fire events modifies the dynamics of ecosystem carbon storage and nutrient distribution by influencing soil carbon reservoirs and nutrient availability that underpins plant and microbial growth, thereby affecting the capacity of these soils to act as carbon sinks (Pellegrini et al. 2018). Properly managed prescribed fires can prevent the net decline in soil carbon levels caused by uncontrolled, high-severity fires, which rapidly release carbon stored in the soil back into the atmosphere (Pellegrini et al. 2018).

Prescribing fire in the southeastern U.S. is complicated by another form of disturbance: wind damage. Hurricanes have historically made landfall predominantly in southern coastal states, accounting for approximately 94% of U.S. hurricane impacts over the past century (NOAA 2023). These wind events can cause extensive forest damage through defoliation and branch loss and the uprooting and snapping of trees (Zampieri et al. 2025), placing southeastern U.S. forests at a heightened risk of transitioning from carbon sinks to carbon sources (Coulston et al. 2015). Tree death and damage also alter the fuel landscape directly by increasing dead fuels (e.g., leaf litter, woody debris) but also indirectly through increased live fuels (e.g., understory vegetation) in response to increased light availability (Cannon et al. 2019). Thus, the interplay between wind damage, changing fuel mosaics, and prescribed fire re-introduction following wind events to fire-dependent ecosystems poses unique challenges and uncertainties for forest management in the southeastern U.S.

Despite numerous studies on prescribed fire impacts on forest soils (e.g., Hobley and Prater 2019; Lucas-Borja et al. 2022), short-term (≤ 1 year) responses of water-extractable organic matter (WEOM; i.e., the dissolvable fraction extracted under laboratory conditions) and nutrient fractions in coastal pine forests with hurricane-driven fuel redistribution, especially those coupling field-quantified fire severity with molecular characterization, remain undocumented. Storm-altered fuel mosaics can modulate the severity and patchiness of prescribed burns (Cannon et al. 2019), yet direct links between those fuel patterns and short-term WEOM/nutrient responses are missing. WEOM is highly sensitive to such disturbances in forests and represents the most mobile and reactive portion of soil organic matter (Kalbitz et al. 2000; Jaffé et al. 2008). WEOM captures rapid shifts in carbon and nutrient fractions that drive post-fire biogeochemical responses. Cross-site comparisons that account for differences in hurricane-modified fuel loads remain limited, making it difficult to generalize how such disturbances affect carbon stabilization and the risk of nutrient exports to nearby aquatic systems (Klimas et al. 2020).

The objective of this study is to identify how prescribed fire severity affects patterns of WEOM and nutrients in wind-damaged forests. Addressing this question provides important insights into the risks and benefits associated with reintroducing fire to managed forest systems. We monitored changes in WEOM and nutrient fractions in two coastal forests that varied in age (~20- vs. 60+-year old) and dominant overstory tree (slash vs. longleaf pine) before and up to one year following the reintroduction of prescribed fire in recently hurricane-damaged forests, both of which exhibited a mosaic of fuel loads. We focused on the first-year post-fire to capture rapid, immediate changes in soil water-extractable solution, providing insights into the early stages of post-fire reintroduction of carbon and nutrient dynamics in forests with variable fuel loads. At these broadly comparable sites, we assessed how hurricane-modified fuel mosaics influenced prescribed fire severity and, in turn, the magnitude, composition, and persistence of WEOM, its molecular characterization, and associated nutrient responses. We aim to contrast trajectories between the two sites to evaluate how disturbances shape short-term processes. We hypothesized that (i) WEOM concentration will increase with fire severity, (ii) this increase will be transient and concentrations and composition of WEOM will revert to pre-fire conditions within one year, and (iii) fire severity will correlate with the release of small aromatic WEOM compounds. This design links measured fire severity to short-term, water-extractable soil organic matter chemistry using a coupled fuel-consumption–molecular approach and offers a concise basis for comparison across wind-impacted, fire-managed forests.

2 Materials and methods

2.1 Site Descriptions

The study was conducted in two coastal forests, Weeks Bay National Estuarine Research Reserve (Weeks Bay) (30° 25’ 9.3504” N, 87° 49’ 49.944” W) in Alabama, and Perdido River Preserve (Perdido) (30° 27’ 56” N, 87° 24’ 34” W) in Florida, USA (Fig. 1). Both sites are in a humid subtropical climate region, with warm summers (average high of 33 °C) and mild winters (average 16 °C), and occasional cold waves. Precipitation comes from a combination of winter storms, hurricanes, and tropical systems with an average annual accumulation of 170 cm (NOAA 2024).

Weeks Bay is a mature (60–80-year-old) stand (13.4 ha) of coastal slash pine (Pinus elliottii). Soils comprise the Okenee and Hyde series (fine-silty, mixed, active, thermic Typic Umbraquults; Web Soil Survey, USDA). At Perdido, we sampled a young (20-year-old) longleaf pine (Pinus palustris) stand (12.3 ha). Soils are primarily of the Hurricane (sandy, siliceous, thermic Oxyaquic Alorthods) and Albany (siliceous, subactive, thermic Aquic Arenic Paleudults), respectively (Web Soil Survey, USDA). Weeks Bay experienced

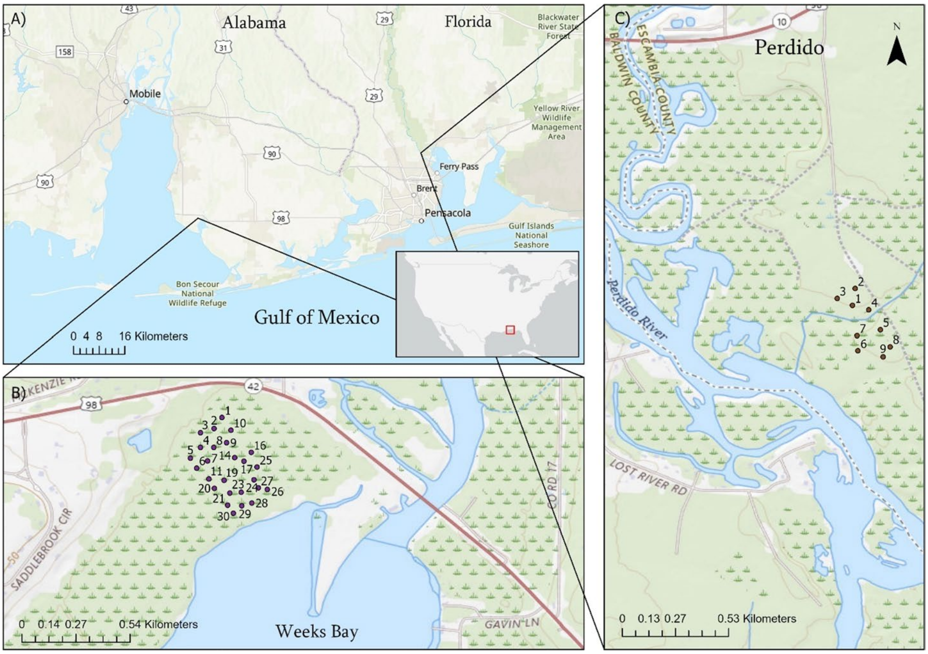

Fig. 1 Geographic depiction of the Weeks Bay (**B**) and Perdido (**C**) sampling sites situated in Southern Alabama and the Florida Panhandle, respectively, including the specific locations of sampling plots

Category 2 Hurricane Sally on September 15, 2020, and both sites experienced Category 3 Hurricane Zeta on October 23, 2020.

Both coastal slash and longleaf pine forests of the southeastern U.S. coastal plain historically experienced frequent, low-intensity surface fires. The historical fire return interval was longer in slash (6 to 10 years) compared to longleaf (2 to 4 years) pine (Landers 1991) because slash pines typically occupy wetter, poorly drained sites at lower elevation that flood during heavy rain events, while longleaf pines occupy drier, higher elevation sites (Long et al. 2004). Because these are fire-dependent systems, prescribed fires were reintroduced at both sites ~ 20 years ago, with fires occurring every 2–6 years. Recent prescribed fires prior to this study occurred at Weeks Bay in July 2016 and June 2020 and at Perdido in March 2019.

2.2 Plot Selection and Sampling Design

At both sites, we first delineated wind-disturbed areas receiving fire restoration using satellite imagery in Google Earth (Google Earth Pro Version 7.3). Within these areas, we selected study plots (15-m fixed radius) to represent a gradient of wind damage, as indicated by downed tree density evident on satellite imagery, as we anticipated that fuel loads, and consequently fuel consumption during fire (i.e., fire severity), would vary with the degree of wind damage. Plot number varied between sites due to differences in size, location, and configuration of wind-damaged areas within the prescribed fire units. At Weeks Bay, we determined plot location first using a stratified random sampling approach to ensure plots were distributed across the wind-disturbed area, which stretched from Weeks Bay to ~450 m inland. We subdivided the stand into six 70-m × 280-m sections running parallel to the bay using satellite imagery, then subdivided each section into 16 grid cells (35 m × 35 m). We randomly selected five cells within each of the six sections for plot locations ($n = 30$ plots, centered on the grid cell). For the most part, this method selected plots representing an array of wind-damage, but we re-selected a few plot locations to ensure representation across a wind damage gradient. At Perdido, we used satellite imagery to select nine plots within the central part of the burn unit dominated by longleaf pine (0.062 km^2), representing a gradient of wind damage. Because there were no quantitative values associated with our visual assessments of degree of wind damage using satellite imagery, we later confirmed that plots represented a gradient of wind damage by reconstructing the total basal area of trees that existed prior to hurricane damage by measuring the basal area of all live (standing and leaning live trees with signs of green leaves and/or live cambium) and dead (both standing and fallen trees, including snags, stumps, and trees that were uprooted completely, snapped along the trunk, or had entire loss of the crown). Using this approach, we confirmed that both sites have experienced a range of tree losses since the hurricanes. Wind damage was more severe and variable at Weeks Bay, ranging from 0.0 to 41.2 $m^2\ ha^{-1}$, with a mean basal area loss of 16.0 ± 11.7 $m^2\ ha^{-1}$. Perdido showed less tree loss, ranging from 1.0 to 7.5 $m^2\ ha^{-1}$, with a mean basal area loss of 3.6 ± 2.3 $m^2\ ha^{-1}$ (Shrestha 2024).

2.3 Fuel Biomass, and Fire Severity

To estimate fire severity (i.e., fuel biomass consumed by fire; Keeley 2009) across the wind-impacted plots, we measured fuel biomass before (< 1 month) and immediately after (within

1 week) the reintroduction of prescribed fire and subtracted the post-fire fuel biomass from the pre-fire biomass. We originally estimated fuel changes using all measured fuel types and loads. However, our original estimates sometimes showed fuel biomass additions, largely due to snags falling over and dead branches falling, thereby increasing biomass of coarse woody fuels. In some cases, we also observed increased biomass of the soil organic layer, which may be due to charred surfaces obscuring our ability to accurately estimate soil organic layer depth. Because of these anomalies, we based our fire severity estimates on changes in surface fine fuels (i.e., understory vegetation + fine woody debris + leaf litter), which were the major drivers of fire spread.

Understory vegetation biomass was quantified using a destructive harvesting approach. In each of three transects (N, SE, SW) originating from the center of each plot, all understory vegetation biomass (< 2 m tall) within a 0.5-m × 0.5-m quadrat was harvested at either the 9-m or 4-m location. Immediate post-fire harvesting was done, avoiding pre-fire harvested areas to ensure accurate estimation. Clippers were used to collect the understory biomass, which was then stored in labeled paper bags. In the laboratory, samples were dried at 60 °C for ~ 48 h to attain a constant weight. Oven-dried biomass was then used to calculate understory vegetation biomass per hectare. The plot-level understory vegetation biomass was obtained by averaging the biomass values from three transects within each plot.

Dead fine woody debris (FWD) biomass was estimated using Brown's planar intercept method along each transect line (Brown 1974). Wood pieces of different sizes and time lags (1-hour, 10-hour, and 100-hour) were tallied along the transects between 2-m and 7-m. Woody debris biomass was calculated using standard formulas (Brown 1974) and specific gravity values (Anderson 1978). Plot-level FWD biomass was determined by first summing the biomass across all size classes within each transect, then averaging these sums across the three transects per plot.

To quantify leaf litter biomass, leaf litter depth measurements were taken at the 4-m and 9-m marks along each transect. Leaf litter depth was then multiplied by site-specific leaf litter bulk densities obtained from leaf litter samples collected from 20 plots at Weeks Bay and 14 plots at Perdido using a 25-cm × 25-cm quadrat near the plot center. Collected samples were placed in labeled paper bags. In the lab, twigs (> 6 mm in diameter) were removed, and litter samples were oven dried at 60 °C for ~ 48 h to attain a constant weight for bulk density estimation, which was determined as the mean g dry weight of leaf litter per unit volume sampled. Bulk density values used were 28 kg m^{-3} for Weeks Bay and 24 kg m^{-3} for Perdido. Plot-level leaf litter biomass was determined by averaging the calculated biomass values from multiple sampling points within each plot.

Because pre-fire fuel biomass is a major driver of fire severity (Keeley 2009), we also attempted to link wind damage severity (i.e., basal area loss) to pre-fire fuel biomass but found little correlation for a variety of reasons. First, a plot can lose trees in a wind event, but dead trunks, branches, and leaves may not fall on that plot. Second, a plot can lose considerable basal area and still contain many live stems that contribute to fuel inputs. Thus, because of these disparities associated with methodological constraints, we focused our analysis on surface fuel biomass variability across plots and impacts on fire severity, and consequently, soil carbon and nutrient dynamics.

2.4 Re-Introduction of Prescribed Fire Following Wind Events

Re-introduction of fire at both sites occurred ~ 1.5 years after wind damage. Re-introduction of fire at both sites occurred ~ 1.5 years after wind damage. At Weeks Bay, the prescribed fire occurred on April 27, 2022. The fire was ignited using a combination of a drip torch by hand and all-terrain vehicle. Ignition started at 13:30 using a backing fire on the downwind side (i.e., southeastern edge) of the unit, which was then followed by fire laid down in progressive strips on the upwind side of the backing fire to move the fire through the unit. Fire was mostly out, except for smoldering dead wood, by 17:00. During the burn, winds were out of the NNW at 2.2–3.3 m s^{-1}. Relative humidity was 25–55%, and air temperature was 25–27 °C. Flame length ranged from 0.4 to 1.6 m. At Perdido, the burn occurred on March 21, 2022. A backing fire was first ignited at 10:40 using a drip torch by hand on the west-northwest (i.e., downwind) side of the unit, and the fire was progressed through the unit using a series of strip and point source ignitions. Winds were out of the ESE at 1.4–2.2 m s^{-1}. Relative humidity was 29–51%, and air temperature was 22–28 °C. Fires at both sites were surface fires that did not carry into the canopy.

2.5 Sampling Methods

Soil samples of the O and upper layer of the A horizons (0–5 cm depth) were excavated using a soil auger at the 9-m mark along each of three 15-m long transects radiating from the plot center at 0°, 135°, and 225° and placed in labeled Ziploc bags. Before sampling, surface leaf litter and live vegetation were carefully removed. All samples were stored in a cooler before processing in the lab within 48 h. The soil was air-dried, sieved (< 2 mm), and stored in the dark before subsequent analyses.

At Perdido, samples were gathered before the prescribed fire on March 8, 2022, immediately post-fire on March 22, 2022, and one year post-fire on March 7, 2023. At Weeks Bay, we sampled pre-fire conditions from March 5 to 7, 2022, immediately after the fire on April 26, 2022, one month after the fire on May 31, 2022, and one year after the fire on March 10, 2023.

2.6 Water Extraction

Soils were extracted by agitating 10 g of air-dried soil with 60 mL of ultrapure water (18.2 MΩ) in a 250 mL polypropylene bottle for 24 h. The suspension was agitated using a temperature-controlled orbital shaker (Lab-Line Instruments Inc., Iowa, USA) at 90 RPM and 25 °C. The suspensions were then decanted into 50 mL centrifuge tubes and centrifuged at 4000 rpm for 10 min at 25 °C. The supernatant was filtered through 0.45 µm polyether sulfone membranes. The filtered solutions were stored in pre-combusted amber glass bottles at 4 °C and analyzed within 24 h.

2.7 Water Extracts Quantification and Characterization

To assess changes in extractable solutes, we measured the electrical conductivity (EC) using an Accument Basic AB30 Conductivity Meter (Fisher Scientific, Waltham, MA). The effect of fire deposition on soil pH was determined using a Seven Compact S220-Basic pH/Ion benchtop meter (Mettler Toledo, Columbus, OH). The quantification of water-extractable organic carbon (WEOC) and WEN was conducted using a multi-C/N 3100 analyzer (Analytik Jena, Jena, Germany). Ultraviolet-visible (UV-vis) absorbance was recorded over a wavelength range of 250 to 700 nm at 0.5 nm intervals on a UV-1800 spectrophotometer (Shimadzu, Kyoto, Japan) using a 1.0 cm quartz cuvette. Fluorescence excitation-emission matrices (EEMs) were collected with an FP-8500 spectrofluorometer (JASCO, Easton, MD). To minimize inner filter effects, samples were diluted with ultrapure water until the UV-absorbance at 254 nm was below 0.10 (Gilchrist and Reynolds 2014; Lakowicz 2006). EEMs were acquired at an excitation range of 250 to 585 nm at 5 nm intervals and emissions from 260 to 600 nm at 2 nm intervals. Ultrapure water blanks were measured daily for background subtraction, Raman normalization, and Rayleigh scattering correction. (Murphy et al. 2014)

Cations and metals (Na, Mg, Al, Si, K, and Ca) were measured with Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES, Spectro Ciros, Spectro Analytical Instruments Inc. Mahwah, NH). Anions (NO_3^-, Cl^-, and PO_4^{3-}) were analyzed using the Dionex Ion Chromatography (IC) system (Dionex ICS 3000, Thermo Fisher Scientific, Waltham, MA). Ammonium (NH_4^+) concentration was measured using a colorimetric method (Mulvaney 1996), using MQX200 Microplate Spectrophotometer (Bio-Tek Instruments, Winooski, VT).

2.8 Data Analysis

We used linear regression to examine the relationships between pre-fire fine-fuel biomass and changes in fuel load between the immediate pre-fire and post-fire sampling periods. To better understand spatial patterns of fine fuel loads prior to and after fire re-introduction, we interpolated understory, fine woody debris, and litter biomass using Empirical Bayesian Kriging (EBK) for Perdido and Inverse Distance Weighting (IDW) for Weeks Bay in ArcGIS software (2023, Esri, Redlands, CA). The interpolation method was selected based on the Exploratory Interpolation Tool in ArcGIS Pro, which ranks methods based on Root Mean Square Error (RMSE).

The absorbance ratios E2:E3 (250 nm:365 nm) and E4:E6 (465 nm:665 nm) were calculated from the absorbance data. The E2:E3 ratio is a bulk spectroscopic indicator of WEOM molecular size. Smaller values of E2:E3 are indicative of larger WEOM size (Peuravuori and Pihlaja 1997; Hautala et al. 2000). The E4:E6 ratio has been attributed to the oxygen to carbon ratio (O: C) of WEOM (Chen et al. 1977). Specific UV-absorbance at 254 nm ($SUVA_{254}$ L mol^{-1} cm^{-1}) was calculated by normalizing the absorbance at 254 nm to the WEOC concentration, indicating WEOM aromaticity. (Minor et al. 2014; Weishaar et al. 2003). The principal fluorophores in the EEMs data set were identified and quantified with Parallel Factor Analysis (PARAFAC), using the staRdom R package (Pucher et al. 2019).

Statistical analyses were performed using R (R Core Team 2024) in RStudio (Posit Team 2024). Pearson's correlation coefficients (r), coefficient of determination (r^2) and p-values were used to assess the significance and explanatory power of the correlations between fire severity and the immediate change in concentrations of WEOC, WEN and C: N. We used one-way repeated measures ANOVA to determine the changes in the studied variables over time.

3 Results

3.1 Fuel Distribution and Load Change

Changes in fuel loads between the immediate pre-fire and post-fire sampling periods increased linearly with pre-fire surface fine fuel biomass at both sites (Fig. 2), with Perdido exhibiting a stronger correlation ($r^2 = 0.81, p < 0.001$) than Weeks Bay ($r^2 = 0.73, p < 0.001$). Understory vegetation and leaf litter exhibited similar degrees of change at both sites, declining from ~7 to ~1–2 tons/ha and from 10 to 13 tons/ha to ≤1 tons/ha, respectively, at both sites (Supplemental Table SM1). At Perdido, clear zones of relatively low, moderate, and high fire severity emerged, while patterns at Weeks Bay were less clear (Fig. 3).

3.2 Spatial and Temporal Dynamics of WEOM

An increase in WEOC, WEN, and C: N immediately after fire was highly correlated with fuel load change at Perdido (Fig. 4). A weaker negative correlation was seen between fuel load change and WEOC change at Weeks Bay ($r^2 = 0.2, p < 0.05$; Fig. 4). No significant correlation was observed between changes in fuel load and changes in WEN and C: N at Weeks Bay.

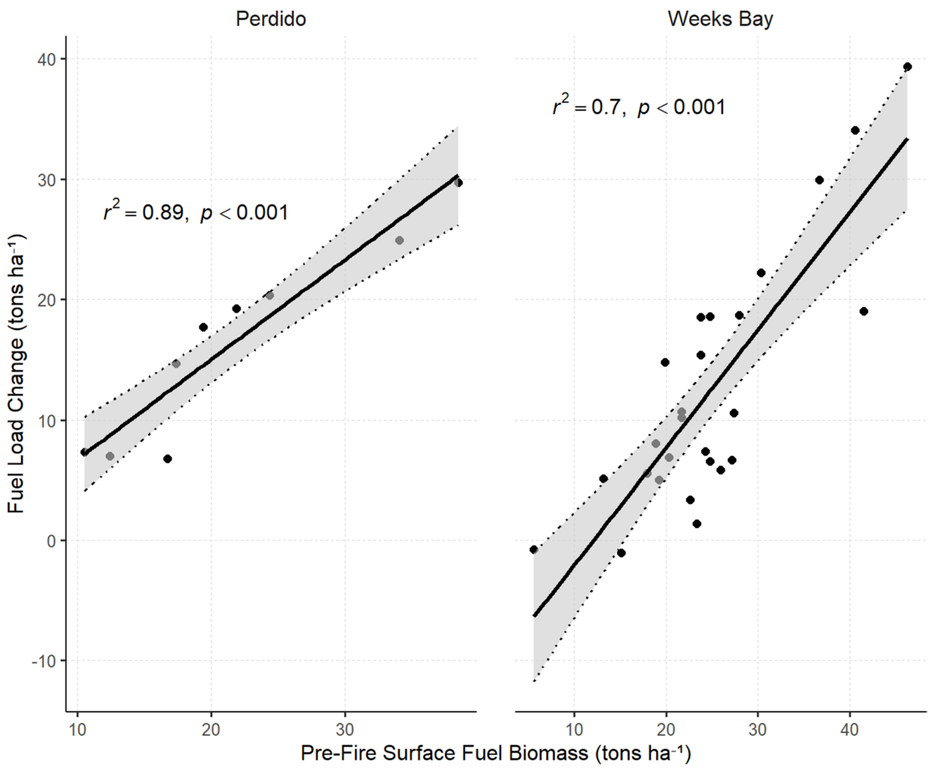

Fig. 2 Fuel load change as a function of pre-fire surface fuel biomass (comprising understory vegetation, fine woody debris, and leaf litter). Solid lines represent fitted regressions, shaded regions indicate 95% confidence intervals. Reported r^2 and p values correspond to each regression

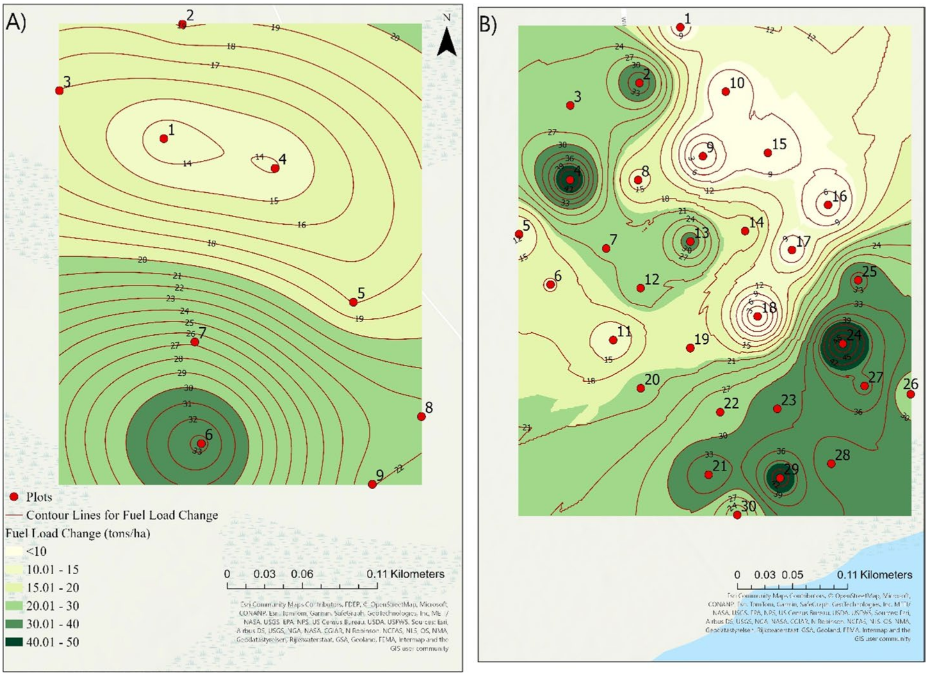

Fig. 3 Spatial interpolation maps displaying fuel load change (tons ha^{-1}) for Perdido (**A**) and Weeks Bay (**B**). Fuel load change, indicative of fire severity, was calculated as the reduction in surface fuel biomass (comprising understory vegetation, fine woody debris, and leaf litter) post-prescribed fire compared to pre-fire conditions

The increase in WEOC concentrations at Perdido was transient, and within one year, the concentration decreased back to pre-fire levels ($p<0.05$, repeated measures ANOVA; Fig. 5). WEN levels, however, increased from 4.1 mg L^{-1} pre-fire to 10.3 mg L^{-1} one year after the fire ($p<0.01$). This resulted in a two-fold decrease in the C: N ratio from 15.3 to 7.8 ($p<0.001$).

At Weeks Bay, WEOC levels showed no significant change post-fire (Fig. 5). However, WEN levels increased from 4.9 mg L^{-1} pre-fire to 6.7 mg L^{-1} one year after the fire ($p<0.05$, repeated measures ANOVA). Similarly to Perdido, the C: N at Weeks Bay decreased from 14.5 to 8.8 ($p<0.001$).

3.3 Changes in the Molecular Composition of WEOM

At Perdido, the E2:E3 ratio remained steady at 4.38 ± 0.13 ($p=0.3$, repeated measures ANOVA, Table 1). A significant change in E2:E3 was observed for the Weeks Bay samples ($p<0.001$, repeated measures ANOVA). The E2:E3 ratio decreased by 13% immediately after the fire, then increased one month later and rose by 65% one year post-fire. This trend suggested the release of WEOM with a larger molecular size immediately after the fire, followed by a continuous decrease within the following year. The E2:E3 ratio also indicates a larger size of WEOM at Weeks Bay compared with Perdido initially (pre- and immediately post-fire), but a lower size one year post-fire. The E4:E6 ratio exhibited a similar trend to

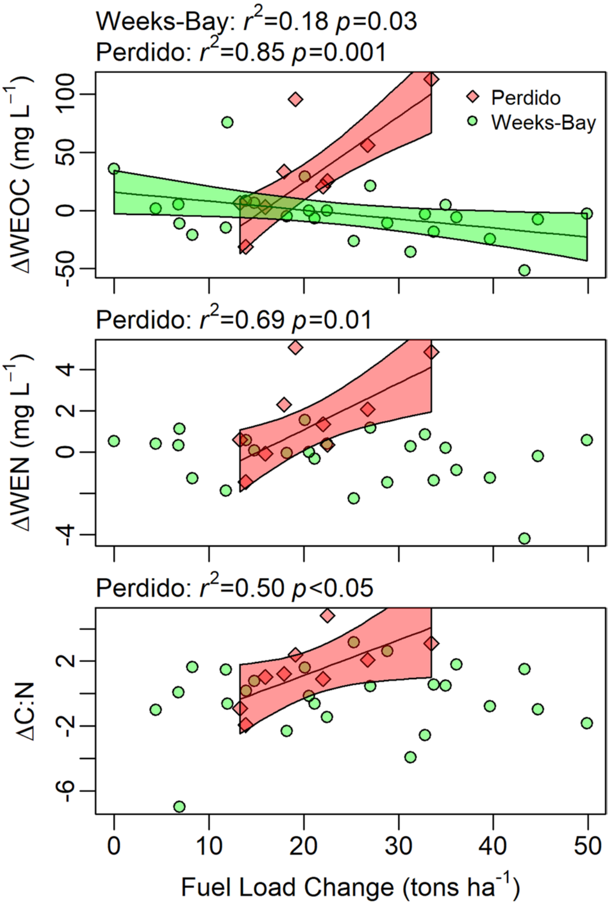

Fig. 4 Relation between fire severity (estimated as the change in fuel load) and the difference in WEOC, WEN, and C: N ratio, pre- and immediately post-fire values. Solid lines represent the linear regression, and shaded regions indicate 95% confidence intervals. The r^2 and p-values are reported for each line

E2:E3. At Perdido, the E4:E6 did not change immediately after fire but increased one year post-fire ($p<0.01$, repeated measures ANOVA). Similarly, at Weeks Bay, E4:E6 did not change initially after the fire, then decreased during the first month post-fire, and increased to higher values one year post-fire. The decrease in the E4:E6 ratio one month post-fire at Weeks Bay aligns with a significant decrease (88%) in $SUVA_{254}$, which is a strong predictor for WEOM aromaticity. At Perdido, E4:E6 exhibited a 25% increase within a year post-fire. No change was observed in $SUVA_{254}$ at Perdido.

Five fluorophore components have been identified in the PARAFAC analysis (Table 2; Fig. 6, and Supplemental Figure SM2). The first component (C1) is associated with humic-like substances (Jamieson et al. 2014; Zhang et al. 2022). A similar fluorophore has been described as a microbial-derived humic-like substance composed of relatively aliphatic compounds with low molecular weight (< 1000 Da) (Lado et al. 2023; Lambert et al. 2016; Podgorski et al. 2018). The second fluorophore component (C2) is reflective of humic acids characterized by a molecular weight exceeding 1000 Da (Huang et al. 2019; Ishii and Boyer 2012; He et al. 2016). C2, exhibiting longer Ex and Em wavelengths, likely encompasses complex structures with conjugated aromatic groups (Huang et al. 2018; Zhang et al. 2022). Its elongated emission wavelengths suggest a molecular structure potentially enriched in

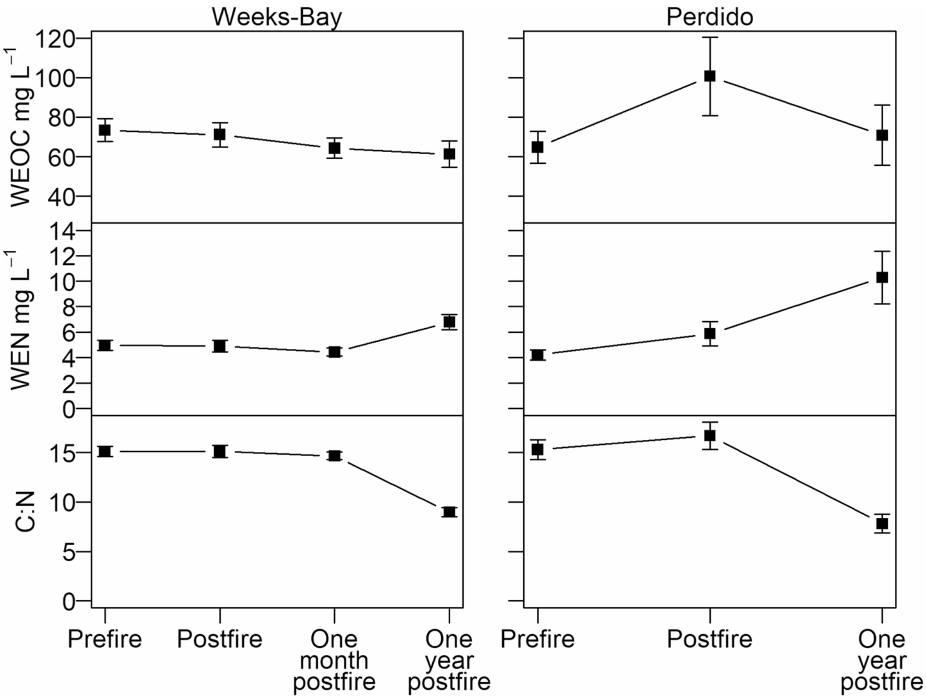

Fig. 5 Temporal changes in mean values of WEOC, WEN, and C: N in plots at Weeks Bay and Perdido. Error bars are the standard error

Table 1 Temporal changes in UV-Vis spectroscopic indicators of DOM. Mean values are presented, and standard errors are shown in parentheses

	Perdido			Weeks Bay			
	Pre fire	Postfire	One Year Postfire	Pre fire	Postfire	One Month Postfire	One Year Postfire
E2:E3	4.41 (0.15)	4.47 (0.19)	4.25 (0.32)	3.29 (0.20)	2.87 (0.21)	3.88 (0.30)	5.44 (0.27)
E4:E6	22.1 (1.32)	23.7 (1.70)	29.7 (1.27)	15.5 (0.35)	16.0 (0.64)	8.88 (1.42)	19.5 (2.33)
$SUVA_{254}$ (L mol-C^{-1} cm^{-1})	289 (28.0)	263 (19.9)	302 (40.6)	505 (38.2)	484 (39.9)	42.5 (9.70)	251 (24.5)

low-molecular-weight phenolic compounds and highly unsaturated aliphatic compounds (Uchimiya et al. 2013; Kellerman et al. 2015; Huang et al. 2019). The third fluorophore component (C3) is indicative of a humic-like substance (Lin and Guo 2020; Zhang et al. 2022), similar to those previously identified as humic-like components found in various environments, including soils and streams (Lado et al. 2023; Baghoth et al. 2011; Sharma et al. 2017). This component has also been characterized as having a large molecular size (Ishii and Boyer 2012) and described as organic matter derived from plants by microbial transformations (Hunt and Ohno 2007; Sharma et al. 2017). The fourth fluorophore com-

Table 2 Excitation (Ex) and emission (Em) range and maximum of the five components identified by PARAFAC analysis

Component (C)	Ex. Range nm	Em. Range nm	Ex. maxima nm	Em. maxima nm
1	225–275	300–380	260/310	422
2	225–275	380–480	270	490
3	250–300	420–520	360/265	438
4	225–300	300–380	285	304
5	250–300	420–520	250	290

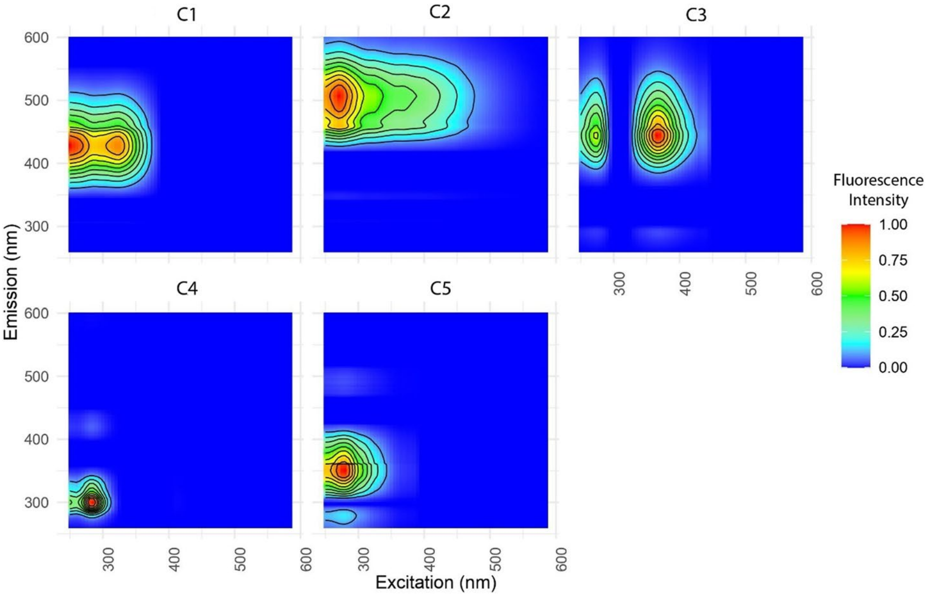

Fig. 6 The five fluorophore components of DOM were identified using PARAFAC analysis

ponent (C4) aligns with tyrosine-like fluorescence (Marhuenda-Egea et al. 2007; Dubnick et al. 2010). The fifth fluorophore component (C5) displays a shorter Ex peak than those typically associated with tyrosine and tryptophan fluorescence (Coble 1996; Yamashita et al. 2008) yet extends beyond the wavelength range noted by others (Marhaba et al. 2000). It has been previously described as a small-sized, poorly biodegradable component with limited adsorption to soil and sediments (Lado et al. 2023; Ishii and Boyer 2012; Sharma et al. 2017). Given the association of lower emission wavelengths with substances of lower molecular weight (Uchimiya et al. 2013; Kellerman et al. 2015), C4 and C5 may indicate the presence of smaller aromatic WEOM components.

At Weeks Bay, significant changes in the fluorescence components were not observed immediately or one month post-fire. However, notable changes became apparent after one year. Specifically, in alignment with the increase in E4:E6 and $SUVA_{254}$, the C1, C2, and C3 increased by 97%, 92%, and 104%, respectively, one year after the fire ($p<0.001$). At the same time, C4 decreased by 47% ($p<0.05$).

At Perdido, an immediate increase was observed in the C4 and C5 components post-fire, with C4 increasing by 49% and C5 by 50% ($p=0.11$ for both) (Table 3). While these changes were not statistically significant, they align with the transient increase in WEOC

Table 3 Temporal changes in fluorophore intensities at Perdido and weeks Bay. Values are the percent distribution of the Ramman intensity. Standard errors are shown in parentheses

	Perdido			Weeks Bay			
	Pre fire	Postfire	One Year Postfire	Pre fire	Postfire	One Month Postfire	One Year Postfire
C1	41.3 (0.2)	39.5 (0.4)	39.7 (0.2)	30 (1.4)	30.5 (1.3)	31.4 (1.4)	38.6 (0.3)
C2	28.8 (0.9)	27.6 (1.1)	32.4 (1.2)	19.4 (0.8)	20.6 (0.8)	22.3 (1)	25 (0.8)
C3	22.1 (0.4)	21.2 (0.5)	20.1 (0.7)	19.5 (0.9)	19.4 (1)	20.1 (0.9)	25.9 (0.5)
C4	3.9 (0.5)	5.9 (0.9)	3.1 (0.3)	18.4 (1.8)	17.8 (1.8)	15.7 (1.9)	5.2 (0.5)
C5	4 (0.7)	5.7 (0.7)	4.7 (0.7)	12.7 (1.1)	11.7 (1.1)	10.5 (1.2)	5.4 (0.5)

and WEN. Furthermore, a strong positive correlation was measured between pre- and post-fire changes in WEOC and C4 ($r^2=0.62$, $p=0.01$) or C5 ($r^2=0.65$, $p<0.01$) (Supplemental Figure SM3).

3.4 Temporal Variations in Soil Dissolvable Inorganic Ions

At Perdido, one year post-fire, NH_4^+ and NO_3^- concentrations increased by 88% and 168%, respectively ($p<0.01$, repeated measures ANOVA) (Supplemental Table SM4). Immediately after the fire, the water-extractable PO_4^{3-} increased from 4.2 ± 1.16 mg L^{-1} to 9.1 ± 3.18 mg L^{-1}, followed by a decrease to 0.9 ± 0.30 mg L^{-1} one year post-fire ($p=0.05$, repeated measures ANOVA). At Weeks Bay, one year post-fire, NH_4^+ decreased by 39% ($p<0.01$), while NO_3^- levels increased by 128% ($p<0.05$). No change in water-extractable PO_4^{3-} was measured at Weeks Bay.

4 Discussions

A key finding is that opposite trends were observed at the two study sites in the relationship between fire severity and WEOM change immediately post-fire. At Perdido, the decrease in WEOC concentration one year post-fire indicates a potential transport of dissolvable organic carbon to the Perdido River. Alternatively, this carbon fraction could be utilized by the soil's heterotrophs and transformed into less soluble organic carbon (De Vries et al. 2016). At Weeks Bay, the change in WEOC immediately post-fire showed a weaker and negative correlation with fire severity. This trend can be explained by differences in fire behavior between the two sites. At Perdido, the distribution of fuel load change was distinct, with clear low and high severity areas (Fig. 3), while at Weeks Bay, a patchy pattern, with isolated plots of high fire severity occurred adjacent to plots with low fire severity. This pattern likely results from complex fire-wind interactions (Cannon et al. 2017). The forest structure likely controlled the different fire-wind interactions in the two sites. Weeks Bay is an older stand with a predominantly woody understory and relatively closed canopy, whereas Perdido is characterized by a younger stand with a grassy understory and an open canopy. Hurricane-induced basal area loss in the mature stand of Weeks Bay created localized canopy openings, which promoted the growth of understory vegetation and accumulation of drier fuels, thereby amplifying local fire severity. At the same time, downed wood and tree throw created a fire discontinuity, impeding fire propagation. This divergence is consistent with wind-driven heterogeneity in fuel continuity, which can decouple severity and WEOM relationships by mixing smoldering and flaming phases at fine scales (Cannon et al. 2017). At Perdido, more continuous fuels likely promoted uniform heating and a coherent WEOM pulse (Roebuck et al. 2025), whereas the patchy severity at Weeks Bay diluted this signal.

Unlike WEOC, WEN increased significantly one year post-fire in both study sites. The rise in WEN with increasing nitrate and declining ammonium over the first-year post-fire likely reflects the nitrification sequence described by Wan et al. (2001), in which combustion and mineralization initially elevate soil NH_4^+ concentrations, followed by enhanced nitrification that oxidizes NH_4^+ to NO_3^-. The increase in WEN might be attributed to fire-induced reductions in plant uptake (Koyama et al. 2010) and to the suppression of microbial competition for N (Zhu et al. 2024). However, since the increase in WEN occurred after the first month post-fire (Fig. 5), a change in the activity of N-fixing bacteria, driven by alterations in vegetation and rhizosphere interactions post-fire, may better explain the increased WEN levels (Vitousek and Hobbie 2000). The elevated WEN concentrations in these N-depleted ecosystems may positively affect the accumulation of soil organic carbon (De Vries et al. 2006). However, soluble N may also be transported to connected waters, promoting eutrophication (Jani and Toor 2018). Importantly, the C: N ratio decreased in both sites from ~ 15, a value typically observed in stable SOM in forest soil, before and up to one month post-fire, to ~ 8, a value similar to average microbial biomass one year post-fire (Cleveland and Liptzin 2007), further supporting a shift toward microbial processing post-fire.

At Weeks Bay, $SUVA_{254}$, E4:E6, and E2:E3 increased between one month and one year post-fire, suggesting increased aromaticity and oxygen content alongside a decrease in molecular size. However, the decrease in E2:E3 immediately after the fire at Weeks Bay contradicts our hypothesis that fire would release smaller aromatic WEOM compounds. This pattern could reflect site heterogeneity and the patchy nature of burning at Weeks Bay, where locally higher burn severity may have caused stronger oxidation and polymerization of organic matter and/or soils and biomass-bonded organic matter, thereby reducing the abundance of aromatic molecules (Lado et al. 2023). Over time, microbial and oxidative processes can progressively transform these macromolecules, from high-molecular-weight material to smaller, more aromatic molecules (Goranov et al. 2022). The reduction in aromaticity at Weeks Bay, identified by a reduction in $SUVA_{254}$ in the first month post-fire, may indicate the leaching of small aromatic WEOM or the gradual release of less aromatic WEOM from biomass. Lado et al. (2023) found that heating soil to 300 °C increased $SUVA_{254}$, indicating higher aromaticity in some soils, while higher temperature (600 °C) reduced the proportion of aromatic compounds due to more intense oxidation and polymerization processes. The subsequent increase in $SUVA_{254}$, E4:E6, and E2:E3 suggests an overall increase in aromaticity and oxygen content, a year after the fire. This change in WEOM composition aligns with the increase in C1, C2, and C3 fluorophores, highlighting the dynamic nature of organic matter transformation in response to fire, even though WEOC concentration did not change significantly.

The rise in humic-like substances suggests that fire induces transformations in soil organic matter, contributing to the accumulation of more resistant organic compounds within the dissolvable fraction. C1, being microbially derived (Lado et al. 2023; Lambert et al. 2016; Podgorski et al. 2018), and C3, representing organic matter originally derived from plant material but transformed by microbial activity (Lado et al. 2023; Hunt and Ohno 2007; Sharma et al. 2017), together with the observed decrease in the C: N ratio at both sites one year post-fire, suggest enhanced microbial processing of post-fire organic inputs, leading to N retention in WEOM.

Post-fire, microbial activity plays an important role in decomposing remaining plant residues. Microorganisms metabolize these organic compounds, assimilating N into their biomass (Kaiser et al. 2015). Furthermore, as these microbes die, the remaining necromass contributes to the formation of stable soil organic matter and may increase the dissolvable N pool (Buckeridge et al. 2022). Moreover, the fact that C1, C2, and C3 are all humic-like substances suggests that post-fire decomposition and humification processes played a key role in N stabilization within WEOM. As plant and microbial residues decompose, they undergo humification, leading to the formation of humic substances that effectively bind N, creating a long-term N reservoir (Kögel-Knabner 2002).

Nitrate increased in both sites by one year post-fire. This increase aligns with the measured increase in WEN. However, most of the WEN was water-extractable organic N (WEON), with nitrate contributing only about 5% of the WEN at Perdido and Weeks Bay, one year post-fire. Ammonium increased immediately post-fire, contributing around 15% at Perdido and 13% at Weeks Bay out of the total WEN. The rise in nitrate and fall in ammonium at Weeks Bay one year post-fire suggest increased nitrification activity (Zhu et al. 2024, Wan et al. 2001).

Another key element monitored in this study is P. A transient increase in P concentration immediately after fire, followed by a decrease over the following year, is an important indicator of P leaching into connected waters. The immediate P pulse followed by a decline mirrors ash-derived solubilization and subsequent transport/sorption; similar post-fire nutrient export concerns are reported for receiving waters (Caroni et al. 2024; Bracewell et al. 2023). A change in the P: N ratio due to prescribed fire has been suggested as a driver of eutrophication (Correll 1998; Waters et al. 2023).

We observed a transient increase in both C4 and C5 at Perdido following the fire. This pattern suggests the deposition of low-molecular-weight pyrogenic WEOM during the fire, followed by their leaching or decomposition within the first year. A similar observation of small-sized pyogenic WEOM was reported in a controlled experiment (Lado et al. 2023). The decreased fluorescence intensity of components C4 and C5 at Weeks Bay one year after the fire may be due to the leaching of these smaller aromatic compounds (Shen et al. 2014).

Overall, Perdido exhibited a pronounced but transient increase in WEOC, phosphate, and smaller aromatic WEOM components (C4 and C5), suggesting increased leaching losses from the system. This enhanced leaching may be attributed to Perdido's younger forest structure, which lacks extensive root biomass and organic matter accumulation typically found in older forests, reducing retention capacity for dissolved solutes.

5 Conclusions

This study provides insights into the impact of prescribed fire on soil organic matter and nutrient dynamics in wind-damaged coastal forest ecosystems. The distinct responses observed between Perdido and Weeks Bay underline the role of site-specific factors, including forest age, structure, and initial fuel load, in governing post-fire WEOM and nutrient dynamics. At Perdido, where fire severity was comparatively more evenly distributed, we observed a transient increase in WEOM immediately post-fire. However, this pattern was not observed at Weeks Bay, which experienced more spatially heterogeneous fire severity.

The transient increase in dissolvable carbon and P at Perdido in response to prescribed fire, alongside the longer-term increase in dissolvable N species at both sites, indicates a potential source into nutrient-sensitive coastal waters and an increasing risk of episodic eutrophication, linking terrestrial fire effects to coastal water quality. While these findings offer insight into how the dissolvable fraction of organic matter behaves within the terrestrial system, future work tracing the fate and transport of released nutrients and associated stoichiometric shifts in adjacent aquatic environments would help establish a clearer link between terrestrial nutrient release and downstream water-quality responses. Parallel characterization of microbial community structure and function could further clarify the role of post-fire microbial processing in mediating nutrient stabilization and transformation within the dissolvable fraction. Because hurricane disturbance is altering fuel distributions and fire behavior, the site-dependent responses documented here provide a basis for anticipating carbon–nutrient dynamics under increasing climate-driven disturbance and for managing prescribed burns to sustain key ecosystem services.

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Data Availability The datasets generated during and/or analyzed during the current study are available in the AU Scholarly Repository, https://aurora.auburn.edu.

Declarations

Declaration of Competing Interest The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Competing Interests The authors declare no competing interests.

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