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# Impacts of fire and flooding on sediment carbon storage in a large, forested floodplain

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## Summary

Natural disturbances influence wetland carbon cycling, and fire is a key driver of terrestrial carbon stocks. However, the influence of fire on wetland carbon cycling remains poorly understood. Here, we investigated how prescribed fire and wildfire impact soil carbon storage in a forested floodplain of south-eastern Australia. We sampled four areas within Murray Valley National Park, the world's largest river red gum (*Eucalyptus camaldulensis*) stand, and compared soil carbon (C), nitrogen (N) and C:N ratios between control (unburnt in the 50 years prior to sampling), prescribed burn and wildfire-impacted floodplain areas. Mean soil C and N concentrations were 4.7% ± 0.32% and 0.36% ± 0.02%, respectively, and mean C:N ratios were 14.23 ± 0.33. Carbon concentrations and C:N were highest in control areas of the floodplain, while N concentrations were highest at wildfire-impacted areas. However, flood frequency was a stronger driver of soil C than fire disturbance. Soils at more frequently flooded areas had higher C concentrations compared to less frequently flooded areas, suggesting that resilience to C loss through fire could be enhanced through hydrological restoration. We believe this warrants further research as a potential nature-based climate measure. Mean C density data indicate soil C stocks of 9.4 Tg across Barmah-Millewa Forest, highlighting the significant carbon storage value of this ecosystem.

## Introduction

Understanding how to manage natural carbon sinks to meet climate change mitigation goals is increasingly important as global temperatures rise. Freshwater wetlands are among Earth's most effective ecosystems for carbon biosequestration (Villa & Bernal 2018), achieved as a result of low plant decomposition rates during inundation leading to long-term sediment carbon storage (e.g., Bernal & Mitsch 2011, Batson et al. 2015, Anderson et al. 2016). However, disturbance can result in wetlands releasing carbon stored in sediments (hereafter 'soils', to allow comparison between the carbon pools of wetlands and other terrestrial ecosystems) as greenhouse gases, and potentially shift them from functioning as net carbon sinks to net carbon sources (Herbst et al. 2013). Because of their importance in the global carbon cycle, wetlands and their management present a key opportunity for developing 'nature-based solutions' to climate change.

The role of fire in wetland carbon cycling is not well understood, but inferences can be drawn from terrestrial environments such as forests and grasslands. Fire impacts a wide range of ecosystems across different climatic regions, for which it is a key driver of carbon stocks (Knapp et al. 2015, Krishnaraj et al. 2016). At a global scale, wildfires are often considered carbon neutral over the long term (decades to centuries), as carbon uptake through vegetative regrowth is approximately equal to carbon emissions created through biomass combustion (Bowman et al. 2009). However, in some ecosystems, up to a third of biomass carbon can be retained in soils as pyrogenic organic matter (i.e., matter produced through the partial combustion of biomass), leading to a net increase in soil carbon storage (Kuhlbusch & Crutzen 1995, Johnson & Curtis 2001, Santín et al. 2015). This recalcitrant 'black carbon' is now considered an important contribution to global carbon sinks, where its resistance to microbial breakdown renders it an effective addition to long-term soil carbon pools (Knicker 2007, Kuzyakov et al. 2009). Globally, black carbon contributes 5–50% of total organic carbon in rivers, soils and sediments, equating to an estimated total of 300–500 Gt of carbon (Hockaday et al. 2007). However, between different ecosystems, the long-term impacts of fire on soil carbon are highly variable; they may be positive (Johnson & Curtis 2001, Zhao et al. 2012), negative (Certini et al. 2011) or neutral (Moghaddas & Stephens 2007, Nave et al. 2011, Kim et al. 2016). Therefore, insights from fire studies cannot always be reliably transferred between ecosystems for informing natural resource management.

Australia is the most fire-prone continent, and fire has been the primary land management tool used by First Nations people in Australia for tens of thousands of years (McCaw 2012, Krishnaraj et al. 2016). In recent decades, prescribed burns have been introduced by government agencies, most commonly to reduce the severity of inevitable wildfires, and less often to promote forest regeneration and biodiversity conservation (Pyne 2003, Burrows 2008). Typically, prescribed burns are carried out in the cooler months and are both less intense and less severe than wildfires (Bradstock et al. 2012, Bradshaw et al. 2013). Fire intensity is a key driver of carbon cycling; it determines the degree to which organic material is combusted and emitted as greenhouse gases (Neary et al. 2005, Bradshaw et al. 2013) or transformed from bioavailable carbon into recalcitrant pyrogenic (black) carbon, which is resistant to decomposition and is retained in the long-term carbon pool (Johnson & Curtis 2001, Lasslop et al. 2019). In many cases elsewhere, prescribed fire has been shown to be effective at minimizing carbon losses when compared to wildfire (Wan et al. 2001, Úbeda et al. 2005, Nave et al. 2011, Alcañiz et al. 2016), and it is therefore a critical tool to consider in the management of wetlands as carbon sinks.

Perhaps counterintuitively, some wetlands are commonly subjected to fire, and in some cases they burn more frequently than nearby uplands due to their high primary productivity following flooding, which can lead to increased fuel loads (Heinl et al. 2006, 2007, Ramberg et al. 2010). Furthermore, fire in wetlands can significantly alter the sediment seedbank, shifting vegetation abundance, richness and composition after fire (Kimura & Tsuyuzaki 2011, Arruda et al. 2016, Kohagura et al. 2020), which, in turn, can influence primary productivity and rates of soil carbon sequestration. Forested wetlands are characterized by large aboveground fuel loads and are therefore vulnerable to the high-severity fires typical of other forests (McCaw 2012). Where impacted by fire, wetlands may become more susceptible to repeat burning, where dead biomass becomes a large fuel source (Flores & Holmgren 2021). Australia has an estimated 240 000 km$^2$ of wetlands (Spiers & Finlayson 1999), which are mostly ephemeral and therefore susceptible to fire during their dry phase. Worldwide, the influence of fire on wetland carbon cycling is relatively understudied, with the exception of peatlands (Garnett et al. 2000, Clay et al. 2010, Turetsky et al. 2015). In south-eastern Australia, wetlands are commonly burnt by wildfire and/or prescribed burning; however, the effects of fire on wetland carbon cycling remain effectively unknown.

In this study, we investigated the impacts of fire in floodplains on long-term soil carbon storage, with the aim to inform management of these large, natural carbon sinks. Specifically, we compared burnt and control (unburnt in the 50 years prior) floodplain areas, and we also compared areas burnt by prescribed fire and wildfire (as a proxy for low-intensity and high-intensity burns). We sampled wetland soils across Murray Valley National Park (formerly Millewa Forest) in south-eastern Australia, the world's largest river red gum (*Eucalyptus camaldulensis*) floodplain. We quantified soil carbon (C) in four areas of the floodplain. Nitrogen (N) is heavily impacted by fire, and soil N concentrations can influence soil carbon sequestration (De Vries & Posch 2011); thus, we also quantified soil N% and carbon-to-nitrogen ratios (C:N), which can be used to infer soil carbon recalcitrance (Zak et al. 2017). We hypothesized that: (1) fire-impacted areas would have a higher carbon concentration than control areas due to contributions from recalcitrant pyrogenic (black) C; and (2) areas burnt by wildfire would differ in their C and N concentrations compared to those burnt by prescribed fire (i.e., long-term C and N levels change with fire intensity). We also used the Wetlands Insight Tool (WIT; Dunn et al. 2023) to understand the influence of hydrology as a driver of wetland biogeochemistry in fire-impacted floodplain areas. Our secondary aim was to estimate soil carbon stocks across Barmah-Millewa Forest (comprising Murray Valley National Park in New South Wales and Barmah National Park in Victoria).

## Methods

### *Site description and experimental design*

Data collection was carried out in June 2019 in Murray Valley National Park (144.89°, −35.81°; 92–97 m above sea level), located between the towns of Moama, Deniliquin and Tocumwal in the state of New South Wales, Australia (Fig. 1). The southern boundary of the National Park is the Murray River and the state border between New South Wales and Victoria. The Murray–Darling Basin is a heavily regulated riverine system and the largest and most economically significant water catchment in Australia (Pittock & Finlayson 2011). Together with the adjacent Barmah Forest in Victoria, Barmah-Millewa is the largest remaining river red gum (*E. camaldulensis*) floodplain in the world, covering 66 000 ha. The forest is located in the Riverina region, and the surrounding land use is predominately agricultural, being used for a mixture of grazing, dryland cropping and irrigated cropping. The climate in the region is semi-arid, with mean annual rainfall of 442.8 mm and mean maximum temperatures ranging from 12.9°C in July to 31.4°C in January (average from 1949 to 2014; Bureau of Meteorology 2019). Vertosols (cracking clays) represent the dominant soil type in the region (Agriculture Victoria 2021, ASRIS 2021).

Murray Valley National Park is a heterogeneous floodplain ecosystem, dominated by river red gum (*E. camaldulensis*) forest, interspersed with areas of woodland (*Eucalyptus* spp.), open wetland and marsh systems (dominated by *Juncus ingens*, *Phragmites australis* and *Pseudoraphis spinescens*) and riverine grasslands (Lynch et al. 2017). Historically, Barmah-Millewa Forest was regularly inundated by natural inflows, where high river levels resulting from upstream precipitation flowed over the riverbanks and into the forest. River regulation in the Murray River commenced with the construction of large dams and smaller locks and weirs between 1915 and 1974, as well as water regulators on creeks within the forest that control inflows from the Murray River (Parks Victoria 2020). With most water in the Murray River system now diverted to consumptive use for agriculture and irrigation, flows reaching floodplain wetlands have decreased significantly. Since 1993, Barmah-Millewa Forest has been allocated 'environmental water', whereby flows can enter the forest only when water regulators have been opened, to improve the flood regime and enhance the floodplain ecosystem (Parks Victoria 2020). Since 2009–2010, the range of these allocations has been 18–428 GL water per year, inundating 17–90% of the floodplain (Parks Victoria 2020). Inundation extent varies across the floodplain based on topography, with the required flow rates in the Murray River being 13 000–60 000 ML day$^{-1}$, which are necessary to reach different floodplain areas (Water Technology 2009). River regulation has also led to a shift in the timing of forest inundation; natural flooding, which has declined in frequency, typically occurs in winter–spring when precipitation rates are highest, while summer–autumn floods have increased corresponding to the high river levels needed to meet peak irrigation demand, with consequences for floodplain ecology and biogeochemistry (Chong & Ladson 2002, Treby & Carnell 2023).

The forest is regularly impacted by both wildfires and prescribed fires, and mapping by the New South Wales National Parks and Wildlife Service (2012, updated on request in 2019) was

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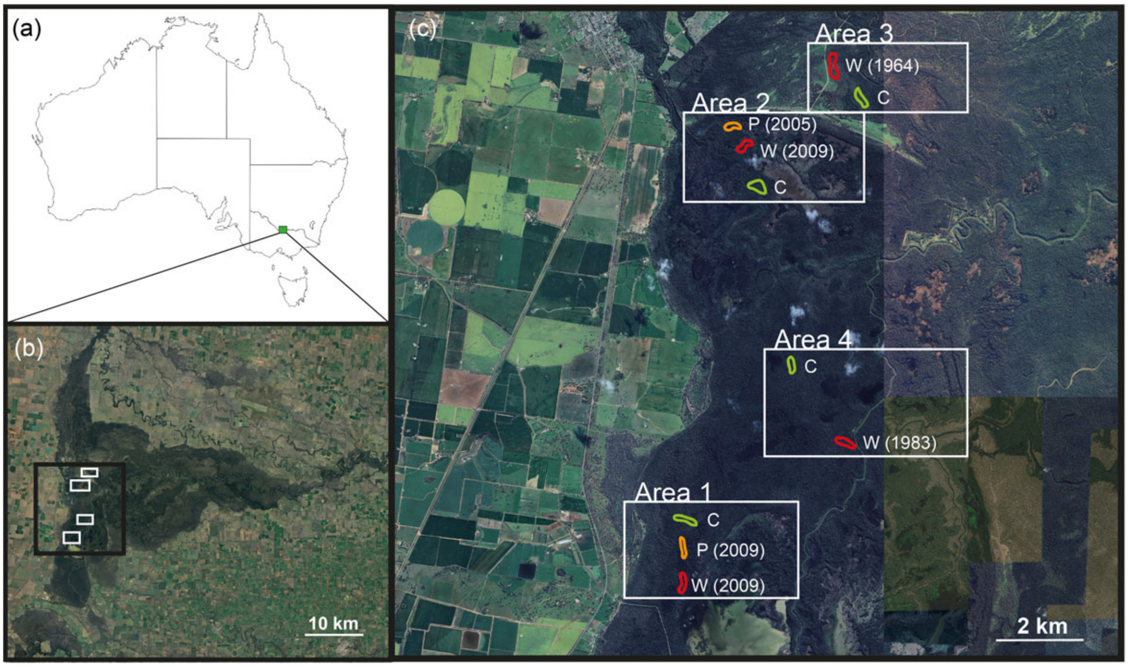


**Figure 1.** Soil sampling areas with different fire histories in Murray Valley National Park, Australia. (a) Map of Australia showing state and territory boundaries and the location of Murray Valley National Park. (b) Barmah-Millewa Forest, showing sampling locations. (c) Site map showing fire type and year of burn in parentheses. C = control (unburnt in the 50 years prior to sampling); P = prescribed fire; W = wildfire. Satellite images from Google Earth (2019).

used to determine past fire boundaries and dates. Each area sampled in this study included a combination of areas with different fire histories: either control (unburnt in the past >50 years) and wildfire, or with an area burnt by prescribed fire (for fuel reduction; Fig. 1 & Table 1). Independent areas were selected based on the overall proximity between them (to minimize differences driven by landscape heterogeneity) and fire treatment (i.e., areas where all three treatments were closely located were selected as a priority). Due to the stochastic nature of wildfires, the selected areas resulted in an unbalanced experimental design, where two areas had only control and wildfire-impacted areas, with no nearby area that had been burnt by prescription. The statistical analyses applied were appropriate for the resulting data. Similarly, there were insufficient areas available for us to exclusively select floodplain areas with the same or similar time since a fire had occurred (see 'Discussion' section). Areas 1 (74 ha) and 2 (69 ha) were burnt by wildfires 10 years prior to sampling (2009), and Areas 3 (32 ha) and 4 (45 ha) were burnt by wildfires >35 years prior to sampling (1964 and 1983, respectively; Fig. 1 & Table 1). At Areas 1 and 2, prescribed burns were carried out in 2009 and 2005, respectively (Fig. 1 & Table 1). All four areas were located within 12 km of one another (Fig. 1 & Fig. S1, Appendix S1).

### Soil sampling and laboratory analysis

Four soil cores were taken per fire type at each of the sampling areas (n = 40). Soils were sampled manually by hammering a 50-mm-diameter polyvinyl chloride (PVC) pipe to a depth of 20–30 cm from the ground surface. Replicates were spaced 75–100 m apart. We aimed to consistently collect soil to 30 cm depth, but this was not possible for all samples. We processed soil cores to a maximum depth of 28 cm, sectioned as follows from the ground surface: 0–1, 1–2, 2–3, 3–4, 4–5, 5–6, 6–8, 8–10, 10–12, 12–14, 14–16, 16–20, 20–24 and 24–28 cm (total soil subsamples = 499). A correction factor (Equation 1) was used to adjust sample depth (Equation 2) and volume (Equation 3) affected by soil compaction during sample collection (e.g., Smeaton et al. 2020), wherein the difference between the soil depth within and outside of the PVC soil core was added to the depth of the sample collected:

$$a = \frac{b - c}{b - d} \quad (1)$$

$$e = \frac{f}{a} \quad (2)$$

$$g = \pi r^2 (e) \quad (3)$$

where $a$ is the compaction correction factor, $b$ is the length of the PVC core, $c$ is the depth of air space within the core (i.e., the PVC length less the compacted total sample), $d$ is the depth of air space outside the core (i.e., the PVC length less the uncompacted reference point), $e$ is the corrected segment depth, $f$ is the original (uncorrected) segment depth and $g$ is the corrected volume of the segment, based on the radius ($r$) of the PVC pipe/soil sample.

Soil samples were dried at 60°C until a stable mass was reached, then homogenized and ground using a RM-200 electric mortar grinder (Retsch, Haan, Germany). C and N concentrations were measured using a MicroElemental CN analyser with *Callidus* v5.1 software (EuroVector, Pavia, Italy).

### Calculations

Dry bulk density (DBD) was obtained by multiplying the dry weight of each sample against the compaction-corrected segment volume, using the method described above. Soil carbon density (mg C cm$^{-3}$) was calculated by multiplying carbon concentration (%) by DBD for each sample. The molar C:N ratio was calculated by converting C and N concentration values to an arbitrary mass of 1 kg each, dividing each by the atomic mass of the element

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**Table 1.** Fire characteristics for each sample area in Millewa Forest.

| Area | Fire type | Fire date (season) | Burn area (approximate) |
|---|---|---|---|
| 1 | Prescribed burn (fuel reduction) | 15 July 2009 (winter) | 1252 ha |
| | Wildfire | 4 May 2005 (autumn) | 110 ha |
| 2 | Prescribed burn (fuel reduction) | 30 August 2009 (winter) | 410 ha |
| | Wildfire | 31 October 2009 (spring) | 1024 ha |
| 3 | Wildfire | 2 January 1964 (summer) | 237 ha |
| 4 | Wildfire | 2 January 1983 (summer) | 114 ha |

(12 mmol C $kg^{-1}$; 14 mmol N $kg^{-1}$), then dividing mmol C $kg^{-1}$ by mmol N $kg^{-1}$ (Lawless 2012). Soil organic C stocks were calculated to Mg $ha^{-1}$ by extrapolating the average C density from the samples collected to the total area of the floodplain, to 30 cm soil depth. As maximum core depths ranged from 10 to 28 cm, stocks were estimated to an additional 2 cm (from 28–30 cm) to allow comparison between our results and those of similar studies (IPCC 2014). For the purpose of this estimation, we assumed C content would be similar from 24–28 to 28–30 cm. Thus, we applied the average C density data calculated from samples at 24–28 cm depth (12.02 mg C $cm^{-3}$) to an additional 2 cm (total 30 cm) depth, then we added this to our stock calculations, wherein the mean C density of the 30–cm cores was converted to t C $cm^{-2}$, then multiplied by 66 000 ha to provide a total stock estimate.

### Analysis

Three separate univariate models were run in *R* version 3.6.0 (https://www.R-project.org/) using the *glmer()* function within the *lme4* package (Bates et al. 2015) for: (1) soil N concentration (%); (2) soil C concentration (%); and (3) C:N ratio. C:N ratio data were not transformed, but N and C concentration data were arcsine square-root transformed prior to analysis to meet model assumptions. N and C concentrations and C:N data were each analysed using a generalized linear mixed effects model with a gamma distribution (identity link), as each dataset was positively skewed towards large values, and all values were above 0. Each model included fire type (control, prescribed or wildfire) as a fixed factor and soil depth as a random factor. Because of the correlation between fire type and time since fire (i.e., there was only one fire date per fire type), time since fire was not included in the analysis.

### Wetlands Insight Tool

To understand our results in the context of heterogeneity across the floodplain, we used data from the WIT, an open-source workflow that converts satellite imagery data into biophysical parameters meaningful for wetland ecosystems (Dunn et al. 2023). The WIT uses the Landsat satellite archive from 1987 onwards to generate a fortnightly spatiotemporal summary of a wetland at 30-m resolution based on a user-defined polygon (Dunn et al. 2019). Using a combination of Water Observations from Space to identify areas with open surface water, Digital Earth Australia (DEA) Fractional Cover algorithms to identify areas of photosynthetic vegetation, non-photosynthetic vegetation and bare soil and the Tasseled Cap Wetness Index to identify wet vegetation areas, the WIT was used to find the percentage of area in the polygon dominated by each of the following: open water, wet vegetation, green vegetation, dry vegetation and bare soil, and their changes over time (Dunn et al. 2019). WIT data for the entire Millewa Forest floodplain are in Dunn et al. (2023).

Here, data were generated for polygons around the core sampling area for each fire type at each area (encompassing all four cores and ranging from 2.0 to 6.3 ha per polygon; n = 10; Fig. 1 & Table S1, Appendix S1). Because of the 30–m resolution of the imagery, it was not possible to generate WIT data for the floodplain area of each core individually. The time series data for each polygon (i.e., each fire type within each area) are presented in Appendix S1. Three additional generalized linear models were run to determine the relationship between the WIT data and soil C, N and C:N ratio. For these models, we first assessed collinearity between the WIT classes (open water, wet vegetation, green vegetation, dry vegetation and bare soil) and determined the correlations between proportional cover of dry vegetation and wet vegetation ($R = 0.97$), as well as between open water and green vegetation ($R = 0.89$). We therefore included only open water, wet vegetation and bare soil, with fire type as a fixed factor in these models.

## Results

Soil N concentrations were 0.003–4.04% (mean 0.36% ± 0.02%). Soil N significantly increased with fire intensity; at areas burnt by wildfire, soils were 21% higher and 18% higher in N concentration than in control and prescribed burn areas, respectively (Fig. 2 & Table 2). Area and fire type had a significant interactive effect on soil N; in Areas 1 and 2, N increased with fire intensity, Area 3 had similar levels of soil N across fire types and in Area 4, N decreased with fire intensity (Fig. 2). Soil N generally decreased with soil depth, although this pattern was less consistent in areas burnt by wildfire (Fig. 2).

The soil C concentration range was 0.09–41.99% (mean 4.70% ± 0.32%). Overall, soil C significantly decreased with fire intensity (i.e., at control areas, soil C concentrations were 33% higher than at prescribed burn areas and 36% higher than at wildfire areas; Fig. 2 & Table 2). However, area and fire type had a significant interactive effect on soil C; in Areas 1 and 2, C increased with fire intensity, Area 3 had similar levels of soil C across fire types and in Area 4, C decreased with fire intensity (Fig. 2 & Table 2). Soil C generally decreased with soil depth from the surface humus (O layer) to the topsoil (A layer), except in Area 3, where it was similar in wildfire samples across all depths (Fig. 2).

Soil C:N ratios ranged from 3.59 to 62.02 (mean 14.23 ± 0.33). Fire type had a significant impact on C:N ratios; in control areas, it was 19% and 17% higher than at prescribed burn and wildfire areas, respectively, although there was no significant difference between the two fire types (Fig. 2 & Table 2). Area alone had a significant effect on C:N ratios; it was lower in Area 1 than in all other areas, and in Area 4 it was higher than in Area 3 (Fig. 2 & Table 2). No significant interaction between area and fire type was found, and C:N ratios generally decreased with soil depth (Fig. 2 & Table 2).

Carbon density ranged from 1.78 to 504.07 mg $cm^{-3}$ (mean 54.01 ± 62.11 mg $cm^{-3}$). The mean estimated C stock in the top 30 cm of soil of 74.4 Mg C $ha^{-1}$ suggests that total soil C stock across the 66 000 ha of the Barmah-Millewa Forest were 4.91 Tg C.

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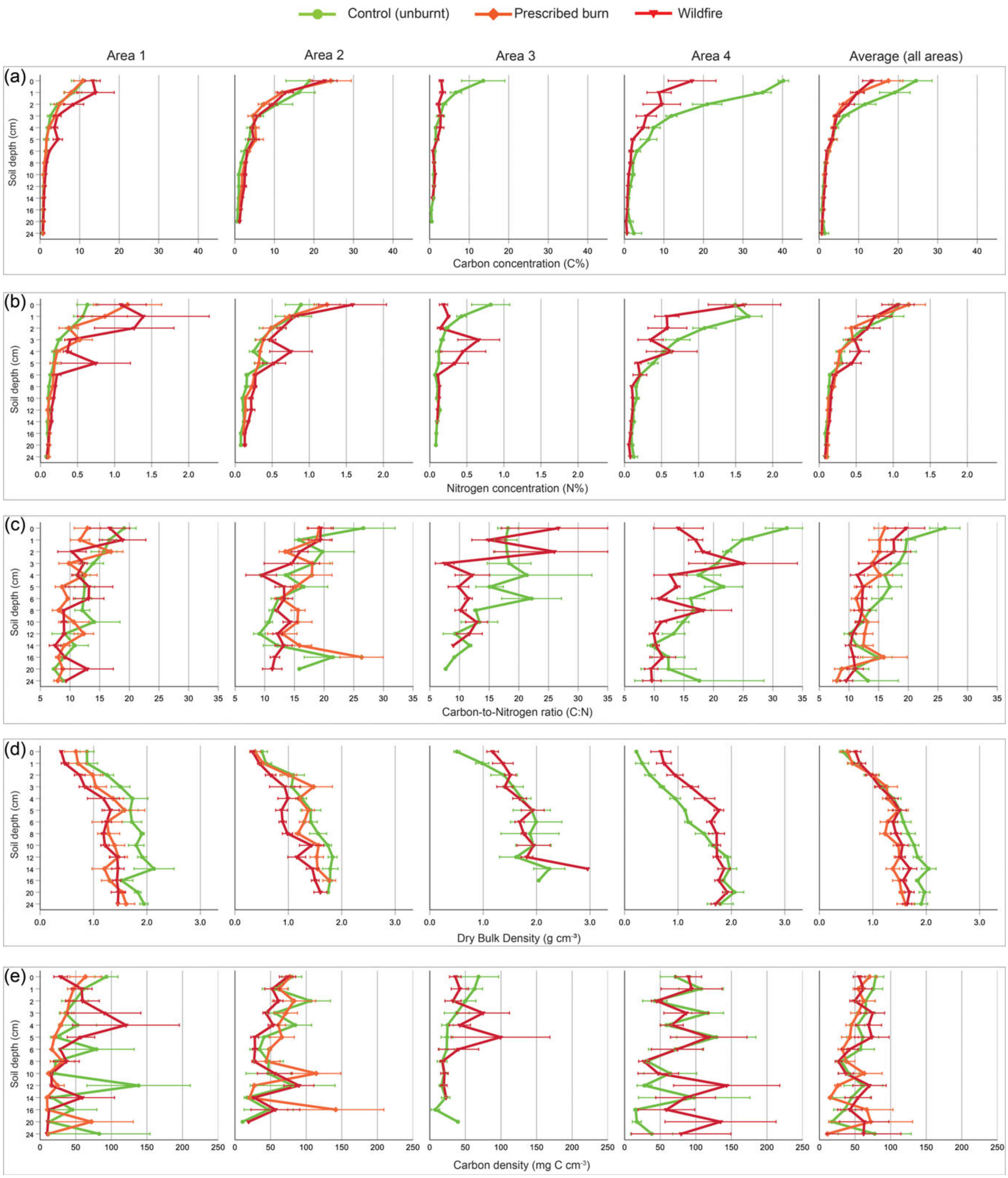


**Figure 2.** Soil responses to fire at Murray Valley National Park: (a) soil carbon (C) concentration, (b) soil nitrogen (N) concentration, (c) C:N ratio, (d) dry bulk density and (e) C density. Soil depth labels refer to the upper sampling limit (i.e., 0 refers to 0–1 cm, 24 refers to 24–28 cm; see 'Methods' section for all increments). Markers indicate mean value for each depth and error bars represent one standard deviation.

## *Wetland cover and soil C and N*

Area 1 had a higher proportion of open water than the other areas, and Areas 1 and 2 had greater proportions of wet vegetation than Areas 3 and 4, which both had a higher cover of dry vegetation than Areas 1 and 2 (Fig. 3 & Appendix S1).

The proportion of open water cover was significantly correlated with soil C concentration ($df = 493$; $t = -2.5$; $p = 0.013$); C concentrations were higher in open water areas than bare soil areas, and there was a near-significant relationship between the proportion of wet vegetation and higher soil C concentrations

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**Table 2.** Statistical significance of soil responses to fire between sampling sites at Murray Valley National Park using generalized linear mixed-effects models.

| | Nitrogen (%) | Carbon (%) | C:N ratio |
|---|---|---|---|
| Fire type | 6.27*** | 6.03*** | 0.22** |
| Site | 4.39*** | 6.26*** | 2.67*** |
| Site × fire type | −5.73*** | −6.67*** | −1.57 |

**$p \leq 0.01$, ***$p \leq 0.001$.

(df = 493; t = 1.7; p = 0.087). The proportion of wet vegetation was significantly correlated with soil N concentration; in wet soil areas, N concentrations were higher than in both open water and bare soil areas (df = 493; t = 2.29; p = 0.02) None of the WIT classes included in the model, however, were significantly correlated with soil C:N ratio. In all three models where the WIT data were included, fire type was not significantly correlated with soil C or N concentrations, nor with C:N ratio.

## Discussion

We found significant effects of fire type (i.e., control, prescribed or wildfire) on soil C and N concentrations when variations in flood frequency were not accounted for in the 66 000-ha river red gum forest in south-eastern Australia. Soil C concentrations and C:N ratios were higher in control areas than in both prescribed fire and wildfire-impacted areas. Soil C and N concentrations and C:N ratios varied significantly between areas, highlighting substantial landscape heterogeneity across the floodplain. When flood frequency (indicated by the relative proportional cover of open water, wet vegetation and bare soil at each area over time) was considered, fire type became an insignificant factor for driving soil C and N concentrations and C:N ratios. These findings suggest that flood frequency is a more important determinant of soil C storage in these floodplains than fire, which highlights that water management has the potential to remediate potential C losses from fire in these ecosystems.

The influence of fire type on soil C and N concentrations and C:N ratios depended on the sampling area and varied substantially even between unburnt (control) areas of the floodplain. Soil C and N concentrations are inherently variable both spatially and temporally (Homann et al. 2011, Nave et al. 2011, Carnell et al. 2018), and floodplain biogeochemistry is especially heterogeneous due to variable flood frequency across the floodplain (Blazejewski et al. 2009, Appling et al. 2014, Lininger et al. 2018). The WIT data showed that from 1987 to 2020 Areas 1 and 2 had much higher relative coverage of wet vegetation than Areas 3 and 4, suggesting that Areas 1 and 2 are more frequently inundated parts of the floodplain. Flood frequency influences vegetation cover through sediment deposition, ecological succession and channel movement (Zehetner et al. 2009, Appling et al. 2014). Organic matter decomposition rates and related microbial activity are also intrinsically linked with floodwater and depend on flood frequency, duration and timing (Garcia-Navarro et al. 2018, Bai et al. 2020). If, during flooding, substantial volumes of sediment are deposited onto the floodplain, flooding can also act to bury carbon-rich surface sediments, promoting long-term sequestration (D'Elia et al. 2017) and potential entrapment of pyrogenic material. More frequent flooding may also enhance allochthonous carbon contributions to floodplain soils, in addition to promoting carbon sequestration through reduced aerobic microbial respiration and expedited vegetative recovery (Andersson et al. 2004, Zehetner et al. 2009, Batson et al. 2015).

Our data suggest that flood frequency can influence whether soil C is lost or gained following fire. In the two areas more characteristic of wetlands (Areas 1 and 2), soil C increased from control areas to prescribed burn areas to wildfire areas, whereas in the two drier areas (Areas 3 and 4), soil C was lower in wildfire-impacted than control areas. This may occur as a result of flood frequency driving C sequestration at rates high enough to compensate for any losses caused by fire. Alternatively, this may be a case of carbon-rich pyrogenic material being contributed to the ground surface following fire, then being better retained in waterlogged soils during flooding (Baldock & Smernik 2002), although we did not detect any visible indication of pyrogenic material (i.e., charcoal fragments) in samples. Typically, post-fire landscapes are vulnerable to wind erosion (Boerner 2006), and in steep valleys topsoil and litter are readily lost during post-fire precipitation events (Hupp et al. 2019, Wohl et al. 2020). Topsoil water repellency caused by heating can also enhance erodibility (González-Pérez et al. 2004, Shakesby & Doerr 2005, Cerdà & Doerr 2008, Nave et al. 2011). However, in a low-relief floodplain such as Barmah-Millewa Forest, low-velocity flooding may cause little or no erosion and, as with precipitation, may enhance pyrogenic organic matter and nutrient retention within the system (Knicker 2007). Post-fire precipitation enables leaching of nutrients from the ash layer into the soil, which can then enhance microbial activity and vegetation growth and, ultimately, the accumulation of new organic matter (Boerner 2006). On a low-relief floodplain, these C accumulation processes could be triggered similarly through flooding, indicating the potential for floodplain fire and water management to be optimized together for C sequestration.

The quality of soil organic matter was lowest in control areas, as evidenced by C:N ratio data, and it was similar between prescribed fire and wildfire-impacted areas. Average C:N ratio values were below 20, comparable to similar wetland types in the Northern Hemisphere and low enough to indicate that N was not limiting microbial activity across the sampled areas (Craft & Chiang 2002, Craft et al. 2018). The lower C:N ratios of soils at prescribed burn and wildfire-impacted areas suggests that a post-fire shift in the quality of litter has occurred into soils and has been maintained over the long term (González-Pérez et al. 2004, Certini et al. 2011, Krishnaraj et al. 2016). Post-fire deposits onto the forest floor are commonly lower in terms of their C:N ratios than litterfall due to increased N volatilization (e.g., Coetsee et al. 2010) and microbial activity (Homann et al. 2011). Since microbial activity in soils demands more N, soils with higher C:N ratios typically have lower rates – or a reduced extent – of organic matter decomposition and more stable C stocks (Averill et al. 2014, Zak et al. 2017). Contrary to our first hypothesis that fire-impacted floodplain areas would have greater quantities of recalcitrant pyrogenic matter, which would facilitate long-term C sequestration, our C:N ratio data suggest that organic matter in control areas of the floodplain is more resistant to decay than in fire-impacted areas.

Our data suggest that floodplain soils in Barmah-Millewa Forest are a significant C pool. The 74-Mg $ha^{-1}$ C density of this floodplain is 174% greater than that of the Victorian state-wide wetland average of 31 mg C $cm^{-3}$ (Carnell et al. 2018). While we measured only long-term soil C stocks, the total C pool of the forest is also probably substantial. Additional contributions of 63 and 5 Mg C $ha^{-1}$ of stored C for coarse woody debris and leaf litter,

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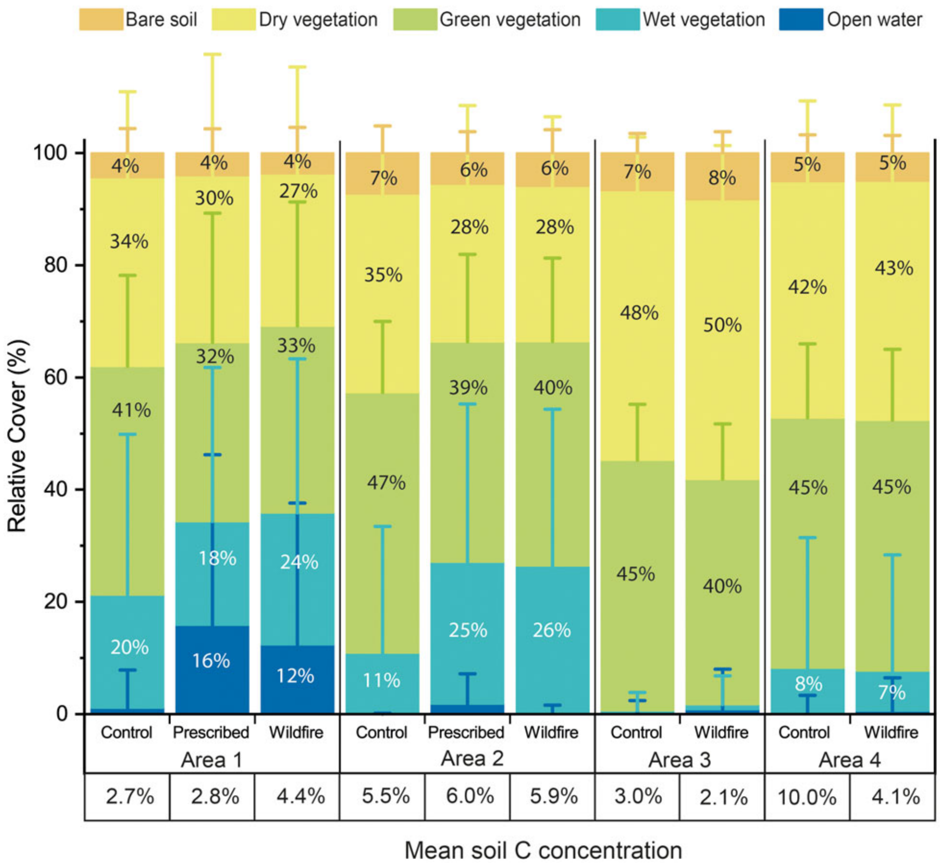


**Figure 3.** Mean soil carbon concentrations and relative proportional cover of open water, wet vegetation, green vegetation, dry vegetation and bare soil, shown for each area and fire type from 1987 to 2020. Stacked columns represent the mean value for each cover type from 1987 to 2020, and error bars represent one standard deviation from the mean. Relative cover values <4% are not labelled.

respectively (Robertson et al. 1999), would equate to an estimated total of 142.4 Mg C $ha^{-1}$ or 9.4 Tg C across the entire 66 000-ha forest floor (Table 3). Using sediment accretion rates obtained from Barmah Forest, we estimate soil carbon sequestration to be 0.87 Tg C $yr^{-1}$ across all of Barmah-Millewa Forest (Table 3; Thoms et al. 1999). Annual primary productivity from river red gums and aquatic macrophytes may contribute a further 2.05 Tg C to the Barmah-Millewa floodplain each year (Table 3; Robertson et al. 1999). This highlights the need to protect and manage this important floodplain ecosystem for its carbon storage value.

For floodplain land managers seeking to protect or enhance natural C sinks, our findings suggest that this could be achieved with fire management tailored to the specific conditions of the area. For optimizing C sequestration, frequently inundated floodplain areas may be resilient to, or even benefit from, periodic fire events, whereas less frequently flooded areas may benefit from fire suppression. River regulation in the Murray River allows for control over the timing and duration of inundation events during managed flows (water for the environment), which may facilitate further testing and optimization of flood and fire interactions in red gum floodplains. Given the large amount of C stored in Barmah-Millewa Forest, understanding how land management techniques influence C cycling in large, forested floodplains in this region could lead to more effective C uptake and create potential incentives and opportunities in C markets, such as the Emissions Reduction Fund (Clean Energy Regulator 2016).

Our findings on the influence of fire type on carbon storage in floodplain soils will benefit from further research. Because black carbon formation, water repellency and microbial communities in soils all change with fire intensity and are strong drivers of post-fire C and N storage, knowing the temperature range and degree of soil heating during each fire (i.e., comprehensive characterization of each fire type, including severity and intensity) would increase understanding (e.g., Baldock & Smernick 2002, Santín et al. 2015). As time since fire is an important factor in post-fire soil C responses (Johnson & Curtis 2001, Hamman et al. 2008, Hurteau & Brooks 2011, Sawyer et al. 2018), a chronosequence study, starting prior to fire disturbance, would also enhance understanding of C cycling and fire relationships. Additional soil physical and chemical properties that drive C sequestration should also be characterized (Leifeld et al. 2005, Watts et al. 2006, Powlson et al. 2011, Kravchenko & Guber 2017, Wohl et al. 2017, Wiesmeier et al. 2019). Refining the soil C stock estimates provided here by characterizing different C densities for different floodplain areas over their entire extent (e.g., permanent open water, frequently flooded low-lying wetland areas and infrequently flooded forested areas) would be a valuable next step for managing the C stored in this ecosystem. Understanding interactions between flood frequency and fire in floodplains with managed flows would further help inform opportunities for C sequestration enhancement as a potential new nature-based measure to help combat climate change.

We highlight the importance of landscape heterogeneity, underpinned by flooding frequency, as a driver of floodplain soil C and N concentrations and C:N ratios in fire-impacted areas. Proportional cover of open water and wet vegetation, representing ecosystem flood frequency and vegetation cover, are evidently related to soil C storage to a greater degree than disturbance from both prescribed burning and wildfires. We suggest that more frequently inundated floodplain areas store more C, even after

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**Table 3.** Estimated carbon pool and annual carbon uptake across the 66 000 ha Barmah-Millewa Forest (BMF).

| | Amount per unit area | Amount whole BMF | Source |
|---|---|---|---|
| ***Carbon pool (stocks)*** | | | |
| Soil 0–20 cm | 59.52 Mg C $ha^{-1}$ | 3.93 Tg C | This study |
| Soil 21–30 cm | 14.87 Mg C $ha^{-1}$ | 0.98 Tg C | This study |
| Coarse woody debris | 63 Mg C $ha^{-1}$ | 4.16 Tg C | Robertson et al. (1999) |
| Leaf litter | 5 Mg C $ha^{-1}$ | 0.33 Tg C | Robertson et al. (1999) |
| ***Annual carbon uptake (sequestration)*** | | | |
| Soil sequestration | 13.21 Mg C $ha^{-1}$ $yr^{-1}$ | 0.87 Tg C $yr^{-1}$ | Thoms et al. (1999), this study |
| Primary productivity – river red gum (*Eucalyptus camaldulensis*) | 6 Mg C $ha^{-1}$ $yr^{-1}$ | 0.39 Tg C $yr^{-1}$ | Robertson et al. (1999) |
| Primary productivity – aquatic macrophytes | 25 Mg C $ha^{-1}$ $yr^{-1}$ | 1.65 Tg C $yr^{-1}$ | Robertson et al. (1999) |

disturbance, than less frequently flooded areas, demonstrating the potential for water management to help buffer against C losses from fire. Our data also highlight the value of large red gum floodplains as significant stores of C in this region. Optimizing management of fire and flooding together will probably deliver the best outcomes for long-term floodplain C sequestration, enhancing the climate mitigation potential of these large C sinks.

## Supplementary material

To view supplementary material for this article, please visit https://doi.org/10.1017/S0376892924000213.

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