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Fire effects on the persistence of soil organic matter and long-term carbon storage

Adam F. A. Pellegrini[1,2]✉, Jennifer Harden[3], Katerina Georgiou[3,4], Kyle S. Hemes[5], Avni Malhotra[3], Connor J. Nolan[3] and Robert B. Jackson[3,5,6]

One paradigm in biogeochemistry is that frequent disturbance tends to deplete carbon (C) in soil organic matter (SOM) by reducing biomass inputs and promoting losses. However, disturbance by fire has challenged this paradigm because soil C responses to frequent and/or intense fires are highly variable, despite observed declines in biomass inputs. Here, we review recent advances to illustrate that fire-driven changes in decomposition, mediated by altered SOM stability, are an important compensatory process offsetting declines in aboveground biomass pools. Fire alters the stability of SOM by affecting both the physicochemical properties of the SOM and the environmental drivers of decomposition, potentially offsetting C lost via combustion, but the mechanisms affecting the SOM stability differ across ecosystems. Thus, shifting our focus from a top-down view of fire impacting C cycling via changes in plant biomass to a bottom-up view of changes in decomposition may help to elucidate counterintuitive trends in the response of SOM to burning. Given that 70% of global topsoil C is in fire-prone regions, using fire to promote SOM stability may be an important nature-based climate solution to increase C storage.

Soils are the largest pool of organic carbon (C) on land, and they offer both an opportunity and a risk to climate-C feedbacks in the Earth system because of their role in the global C cycle as well as their vulnerability to disturbance[1–3]. Disturbance by fire is often considered to lead to C losses because fires combust plant biomass and organic soil layers and promote erosion and leaching, subsequently reducing inputs to and stimulating losses from soils that can persist for several years after the fire[4–7]. However, fires can cause several transformations within an ecosystem that can offset the immediate C losses and may ultimately stabilize the ecosystem C.

Fire-driven changes to the persistence of soil organic matter (SOM) are potentially important for offsetting combustion-based C losses by reducing decomposition[8]. Despite the extensive research into how fire alters the properties of SOM[9–12], there has been only a limited connection between changes in the SOM properties and the response of soil C fluxes to long-term shifts in fire frequencies and intensities. Most studies that connect fire effects on soil C fluxes with changes in SOM properties focus on the formation of pyrogenic C[8,13–16]. However, pyrogenic C formation can be minimal in ecosystems with a low woody biomass and is just one of the many fire-driven changes relevant to SOM stability: burning can change the soil porosity, aggregate formation, soil hydrophobicity, the potential sorption of organic matter to minerals, microbial biomass and composition, and the soil pH—all of which alter the decomposition of the SOM[10–12]. Here, we review the evidence for how fire affects the factors that determine the SOM stability, and not just the SOM content alone, to better understand how changes in decomposition dynamics influence the response of soil C storage to shifting fire regimes.

Global overlap of fire and soil carbon

The spatial extent of fire combined with the fact that it occurs in most ecosystems worldwide[17–22] make it a relevant ecological process for C cycling at the global scale (Fig. 1a,b). To provide quantitative estimates of the potential impact of fire on SOM, we analysed global maps of ecosystem types, burned area and soil organic C (SOC) (Supplementary Information). By masking the soil C map with the global burned area map, we estimated the potential amount and distribution of soil C in fire-prone regions (Fig. 2a, Table 1 and datasets in the Supplementary Information).

Globally, the topsoils (<30 cm deep) in fire-prone regions store 460.4 PgC, with 19.2 PgC in regions that burn annually. However, the distribution of topsoil C and fire across ecosystems is skewed (Table 1 and Fig. 2b,c). Ecosystems with frequent low-intensity fires, such as savannahs and grasslands, contain 28% of the total soil C but they amounted to 79% of the global burned area. Ecosystems with infrequent high-intensity fires, such as boreal forests, contain 21% of the total soil C but amount to only 1.5% of the global burned area. Tropical and subtropical forests, which generally experience intense deforestation fires, contain 25% of the total soil C and account for 8.5% of the global burned area. Taken together, around 70% of the total global topsoil C (a global total of 653 PgC) is exposed to fire (for perspective, the total C in vegetation globally is estimated to be 450–650 PgC (ref. [23])).

The latitudinal distribution of topsoil C is highest in the northern regions, but when re-scaled to the burned area, soil C exposed to fire is less than the amount in subtropical regions (Fig. 2b,c). This is because the high frequency of fire in the sub-tropics (with fire return intervals of 6–45 years) and the total burned area (3.85 million km^2) compensates for the lower stocks of soil C. Thus, the potential exposure of total soil C pools to fire is skewed towards lower latitude systems.

To understand how these vulnerabilities play out in reality, researchers have drawn on the multitude of wildfires, prescribed fires and fire-manipulation experiments[5,7,24]. A meta-analysis has demonstrated that frequent burning resulted in lower soil C in savannah-grasslands and broadleaf forests, but higher soil C

[1]Department of Plant Sciences, University of Cambridge, Cambridge, UK. [2]Conservation Research Institute, University of Cambridge, Cambridge, UK. [3]Department of Earth System Sciences, Stanford University, Stanford, CA, USA. [4]Lawrence Livermore National Laboratory, Livermore, CA, USA. [5]Woods Institute for the Environment, Stanford University, Stanford, CA, USA. [6]Precourt Institute for Energy, Stanford University, Stanford, CA, USA. ✉e-mail: ap2188@cam.ac.uk

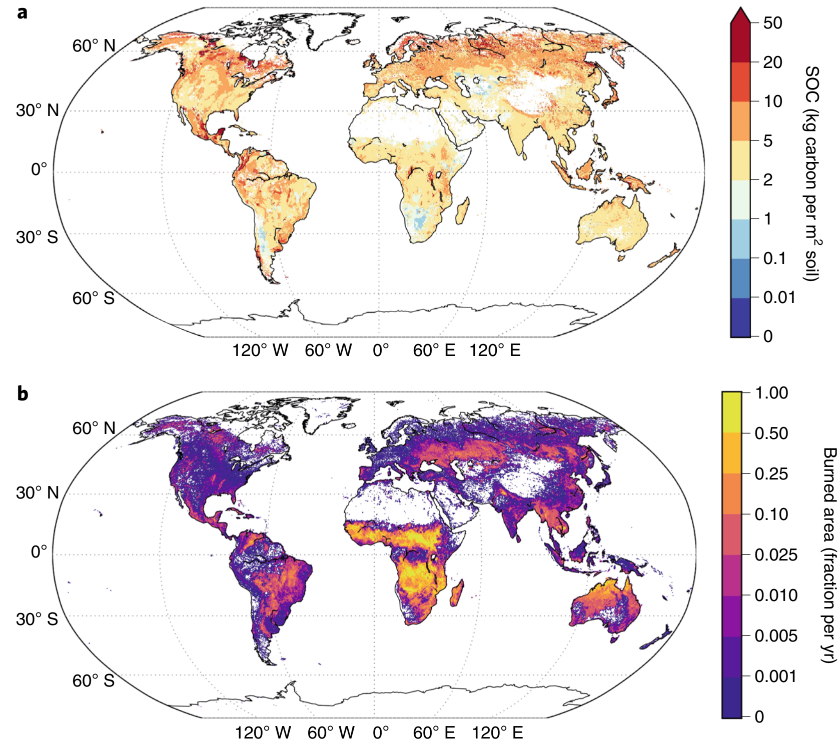

Fig. 1 | Distribution of topsoil organic C and fire frequency across the globe. a, Total SOC in the top 30 cm from ref. [103]. **b**, Burned area (averaged over 1998–2015, Global Fire Emissions Database 4s) expressed as the fraction of a gridcell that burns within a year at 0.25 × 0.25° resolution[104]. The distribution of fire is used to mask the distribution of soil C for all the analyses in the paper.

in warm needleleaf forests relative to unburned plots using measurements in the top 0–20 cm of mineral soils (Table 1)[25–28]. In cooler needleleaf forests, where the SOM is combusted from the thick organic horizon[24,29], C losses from wildfires can be especially large. Although direct heating during fire can be relatively shallow (for example, 4–7 cm in prescribed burns in coniferous forests[30]), repeated burning can change the soil C stocks down to a depth of one metre in some cases[31]. Changes are more generally concentrated in the top 0–20 cm of soils[25–28]. Many studies seeking to explain how fire changes SOM storage have focused on differences in the combustion of biomass and SOM[29,32–34] (that is, inputs into mineral soils); however, the numerous cases where inputs into soils cannot predict SOM storage highlight the limits of traditional input-focused theory and suggest that other factors, such as shifts in decomposition, also matter.

Inputs alone do not always explain soil carbon responses to fire

Despite the near ubiquity of fire changing the aboveground biomass inputs into soils and/or combustion of the organic horizon, the response of total SOM storage is highly variable[5,12,24,35,36]. Part of this variability can be linked to the degree of change in plant biomass inputs to soils, but this is insufficient to explain the cases where (1) biomass declines yet SOM changes very little[37–40], (2) biomass does not change and SOM is lost[41] or (3) SOM increases despite there being either no change or a loss of biomass[25,42,43]. Consequently, a perspective that looks beyond biomass inputs alone as a regulator of SOM may help to understand how fire shifts soil C storage in these different contexts.

There are clear effects of fire on the SOM turnover in ecosystems. For example, in boreal forests, residual SOM after a wildfire was older with longer turnover times based on ^{14}C data (for example, two years after a wildfire, the SOM was over 100 years old, whereas 150 years after a wildfire, the SOM was 60 years old)[44], although these effects tend to decline with soil depth[45]. However, this is not always the case, with turnover not changing after fire in a temperate conifer forest[46]. Changes in the persistence of SOM may thus help to reconcile different fire effects on SOM storage.

As more frequent burning reduces biomass inputs to soils, continued processing of SOM by decomposers could deplete soil C (Fig. 3). If decomposition declines to the same extent as inputs decline, no net change will occur. If fire reduces decomposition more than fire reduces the biomass inputs, then C accumulation may occur (Fig. 3). Numerous examples exist in support of these different responses (presented in Fig. 3). Thus, the net effects of fire on SOM storage are a balance between the changes in decomposition and biomass inputs that manifest over multiple decades and fires.

Processes that contribute to soil organic matter persistence relevant to fire

Here, we focus on the stabilization of SOM, which is defined as a decrease in the long-term SOM loss to oxidation and microbial respiration as well as leaching and erosion[47–50]. Building on previous frameworks[47,51], we evaluate the stability conditions using the following factors. Accessibility, which refers to the spatial location of organics that determine microbial and enzymatic access, such as aggregation. Interactions, which refers to the chemical and physical connections between multiple organics or mineral-organics that alter the degradation or synthesis of new organics. Recalcitrance, which refers to the molecular-level characteristics of the organics that influence microbial and enzymatic degradation. Recalcitrance, interactions and accessibility integrate into stability, which affects the equilibrium pool size in combination with ecosystem characteristics that also impact decomposition.

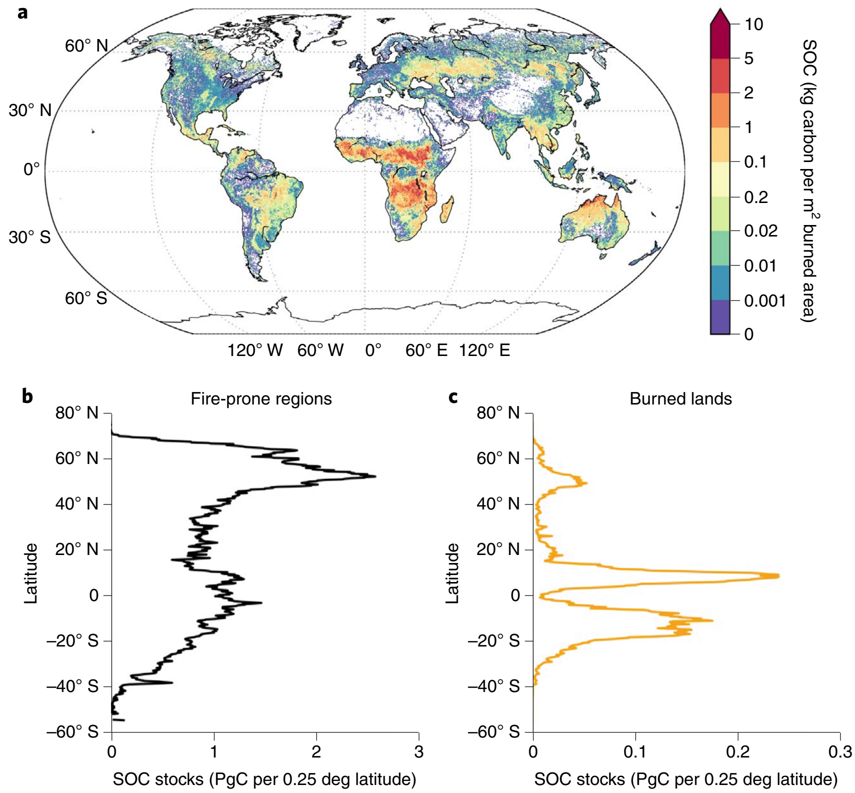

Fig. 2 | SOC stocks vulnerable to fire. a, Topsoil (<30 cm) organic C stocks in a gridcell weighted by the burned area in that cell. The geographic distribution is given in panel (**a**), which only includes gridcells with non-zero burned area (that is, fire-prone regions). The relatively short timescale of global burned area data likely underestimates the extent of fire-prone area, especially at fine spatial scales. **b**,**c**, Latitudinal distributions for SOC stocks in fire-prone regions (**b**; calculated without the burned-area weighting) and in burned lands (**c**; calculated with the burned-area weighting). Datasets are described in the Fig. 1 caption.

Accessibility. Aggregation, as one of the main factors influencing accessibility relevant to fire, plays an important role in stabilizing SOM by decreasing microbial access to substrates, as well as by decreasing the rates of extracellular reactions through hindering the diffusion of reactants and products or by creating anoxic zones[48,52]. Aggregation is driven by a range of underlying physicochemical, microbial and plant processes[53], many of which can be influenced by fire (Fig. 4).

Fire can directly affect aggregation by thermally transforming SOM (Fig. 4)[54]. Physical stabilization can increase when heating promotes the formation of aggregates (for example, aggregate abundance increased from 58% to 84% in unburned versus burned forests[55]), which may stabilize residual SOM after fire. Alternatively, aggregates can be destroyed during a fire[56]; however, this can depend on soil moisture and heating, with faster heating in moister soils leading to greater aggregate disruption and C mineralization in temperate forest soils[57]. Direct heating effects are probably concentrated in the organic and upper mineral horizons[58], with the heating depth depending on the fuel load and mineral content[30]. Thus, the degree and rate of heating are important for predicting how fire changes the aggregation and subsequent C turnover.

Fire can indirectly affect aggregation by modifying the plant-root physiology. In savannahs, fire tends to favour grasses that form symbioses with arbuscular mycorrhizal fungi (Fig. 4), which can increase the aggregate stability[59,60]. Yet multi-decadal fire-manipulation experiments in savannahs have demonstrated that fire can both decrease the colonization of mycorrhiza on grass roots by 50% (ref. [61]) or leave it unchanged[62]. Furthermore, fire tends to select for fine-rooted species[31], and fine roots increase soil binding and aggregate stability[60]. Consequently, fire-driven changes in plant composition may promote aggregation when it stimulates arbuscular mycorrhizal growth and fine-root production.

Fire also impacts accessibility of soil C to microbial decomposition through post-fire erosion and deposition. Erosion and sedimentation rates can increase after wildfires[63], causing a rearrangement of both unburned and burn products of SOM across the landscape. Eroded landscapes can expose deep SOM to microbial access while deposition buries and reduces decomposition[64].

Interactions. Mineral interactions can hinder the ability of microbes to decompose SOM because minerals can adsorb or occlude SOM and immobilize exoenzymes that are responsible for depolymerization reactions[2,48,50,65–67]. These organo–mineral interactions play an important role in SOM preservation, protecting even labile compounds for decades to centuries[50,67], as shown by mineral-associated organic complexes being much older than particulate organic compounds in the same soil[65,66]. The potential for organo–mineral interactions depends on edaphic characteristics, such as the types of clay minerals[68]. For example, soils with high-activity clays (for example, 2:1 phyllosilicates) tend to have higher amounts of mineral-associated organic matter[2]. Hence, fire-driven shifts in the balance of mineral-associated and particulate SOM may have important implications for SOM stability.

Fire can change the stability that arises from association with minerals by thermally modifying the minerals[69] (Fig. 4). For example, intense fires can directly alter the soil texture and mineral composition, with slash-and-burn fires (temperatures > 600 °C) resulting in a coarser soil texture and a reduction in gibbsite and kaolinite concentrations[70]. Hot wildfires caused a collapse of 2:1 phyllosilicates, reducing potential sorption, but the effect was

Table 1 | Distribution of soil C stocks and burned area across the main ecoregions that experience burning

Ecoregion	Area (million km²)	FRI (yr)	FRI s.d. (yr)	Total burned area (million km²)	Soil C in burned area (PgC)	Annual burn soil C (PgC)	Plot soil C (kg m⁻²)	Plot soil C s.d. (kg m⁻²)	Repeated fire effects on mineral soil C
Tropical and subtropical grasslands, savannahs and shrublands	18.4	6	5	3.366	67.7	12.53	3.62	6.33	−30% (−66/+96)
Tropical and subtropical moist broadleaf forests	17.2	45	19	0.375	101.1	1.98	5.82	4.11	−42% (−53/−7)
Deserts and xeric shrublands	11.2	37	21	0.295	34.2	0.83	3.07	2.28	n.d.
Flooded grasslands and savannahs	1.03	7	5	0.166	4.7	0.87	5.15	3.17	n.d.
Temperate grasslands, savannahs and shrublands	8.79	52	33	0.159	44.2	0.82	5.33	3.55	−18% (−42/−1)
Tropical and subtropical dry broadleaf forests	2.83	27	13	0.106	14.7	0.51	5.26	5.09	−29% (−61/+6)
Temperate broadleaf and mixed forests[a]	10.6	121	66	0.088	57.6	0.50	5.56	6.20	−42% (−53/−7)
Boreal forests/taiga[a]	9.89	155	88	0.066	85.9	0.54	8.96	2.52	−39% (−52/−21)
Montane grasslands and shrublands	2.44	37	15	0.065	10.4	0.25	4.37	3.48	n.d.
Mediterranean forests, woodlands and scrub	2.42	107	55	0.024	9.4	0.09	3.70	5.28	−54%
Temperate conifer forests[a]	3.34	187	88	0.018	19.6	0.12	5.81	5.24	−34% (−54/−22)
Tundra	1.43	212	106	0.007	10.9	0.05	8.12	4.88	n.d.
Sum across ecoregions	89.5			4.735	460.4	19.1			

[a]Ecoregions where the direct combustion of SOM is especially large. The methods are described in the Supplementary Information. FRI is the fire return interval, measured in years, within an ecoregion. The total burned area is the sum within an ecoregion. Soil C exposed to fire is the total C (in petagrams carbon (PgC), top 0–30 cm of soil) in gridcells with a non-zero burned area. Annual burn soil C is further weighted by the burned area in the gridcell. Plot soil C is the mean stock of soil C exposed to fire. Repeated fire effects show changes in the mineral soil C between frequently burned/unburned treatments calculated across sites in a meta-analysis (with the range in parentheses)[5,85]. s.d., standard deviation; n.d., not determined. Ecoregions are displayed in Extended Data Fig. 1.

patchy and was estimated to impact ~2% of the total wildfire area[71]. In other cases, wildfires increased the binding between iron and aluminium and organic matter by around twofold[72], promoting SOM stability. Direct measurements of SOM age using ^{14}C data revealed that residual SOM in burned plots had slower turnover times and was older in both free and mineral-associated SOM (rising from 85 to 146 years and 137 to 192 years, respectively)[73,74]. Changes to SOM turnover time decline with depth[74], suggesting the thermal transformation of SOM as the main driver of changes in mineral association. Therefore, fire-driven changes in mineral associations may be important for how fire changes the soil C stability.

Recalcitrance. The chemical recalcitrance of SOM is important for determining the stability of particulate SOM and the potential stabilization via interactions with minerals[47,75,76]. Fire can have direct effects on recalcitrance via the thermally induced heating of organics as partial oxidation and condensation occurs[8], which has been covered extensively in past reviews[12,14,16,69] (Supplementary Information and Fig. 2), and aligns with our stability framework (Fig. 4). First, fire-driven changes in molecular properties include increases in the concentration of lignin and polyphenols in SOM by more than twofold in boreal forests[77,78] and increases in pyrogenic C[14,16,79–81], which tend to persist in the soil relative to other forms of SOM[16,50,80], although the turnover times vary substantially[82]. Discussions of pyrogenic C are extensive[14,16,79–81], so we do not focus on it here.

The temperature of the burn determines the molecular condensation, with temperatures >300 °C resulting in a higher proportion of aromatic structures[83] (Fig. 4). Thermal alteration of organic matter is distinct from decomposition because it removes external oxygen groups rather than enriching carboxyl constituents, resulting in structures with lower potential biodegradation[36]. Depending on the potential physical transport of SOM after fire (erosion discussed in the 'Accessibility' section), the formation of recalcitrant, often hydrophobic, SOM and charcoal is hypothesized to lead to the accumulation of soil C[35].

Decomposer community composition and activity. Fires can cause the direct mortality of bacteria and fungi, thereby reducing decomposition (Fig. 4). A meta-analysis found that fire decreased fungal abundance by 47.6% and microbial abundance by 33.2%, and these effects were most pronounced in high-intensity wildfires. Recovery can take several years to decades. In a boreal forest, burned plots had around one-third of the microbial biomass compared with unburned plots five years after a wildfire[84]. Low-intensity fires can also impact potential decomposition. A survey of four fire-manipulation experiments illustrated that the activity of hydrolytic enzymes that decompose SOM was 187% higher in the unburned versus repeatedly burned plots[85]. Meta-analyses have found that losses of microbial biomass correlate with decreases in decomposition activity[86,87] (Fig. 4). The lower summer albedo in burned areas, especially due to large severe wildfires, can lead to surface warming[88], which could contribute to decomposition changes.

Changes in microbial community composition are important but tend to be context specific. Meta-analyses have shown that fungi tend to be more sensitive to fire, resulting in fire causing a relative reduction in fungi versus bacterial abundance[89,90]. Declines in decomposition after a fire have been attributed to a lower fungal

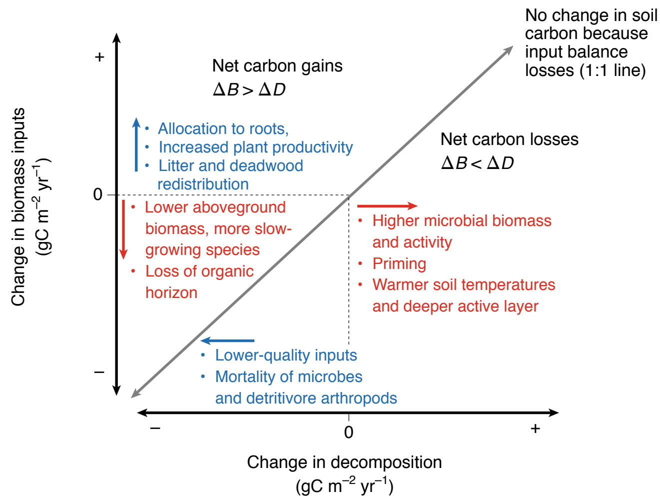

Fig. 3 | The balance between changes in decomposition-based losses of SOM versus biomass-based inputs into SOM may influence fire effects on SOM stocks. Fire effects on biomass-based inputs (*B*) and decomposition-based losses (*D*) can promote SOM gains (blue), including higher root biomass, leaching of ash into soil, reduction in detritivore biomass and formation of aggregates[33,84,90–92,98,105,106]. Fire effects can also promote SOM losses (red) when there are declines in biomass inputs, erosion and leaching of organic matter, priming of microbial activity and warmer soil temperatures[57,85,93,107–113]. The coloured arrows represent the vector direction caused by the processes adjacent to the arrow. Studies providing evidence for these processes are in the Supplementary Information.

abundance based on declines in exoenzymes that are associated with fungi production, such as phenol oxidase[84], as well as correlations between a lower fungal biomass and slower SOM turnover[44], and lower rates of litter mass loss[91]. Changes to detritivore abundance and composition, such as arthropods, can be substantial and last for several years[90,92] especially when the organic horizon is completely removed, potentially reducing decomposition and the transfer of litter into topsoils (Fig. 4).

Fire can also lead to an increase in microbial activity. The mobilization of pyrogenic C and residual ash can prime decomposition[93]. Microbial taxa that are important for C and nutrient cycling can be 'pyrophilic' (for example, *Arthrobacter*), increasing in abundance following a burn[94] (Fig. 4). Thus, changes in the environment, composition and population sizes can either accelerate or decelerate SOM losses, but the causes of directionality are unclear. A more mechanistic understanding of the factors determining how fire changes the composition and activity of decomposer species could be a useful way to improve our understanding of SOM persistence.

Environmental controls on fire-driven changes in decomposition

The potential effects of fire on SOM turnover differ across environmental and ecological conditions. Gradients in the potential sorption of SOM to minerals arising from edaphic variability can shift the dominant forms of SOM from being in particulates to associated with minerals as soils change from having a low to a high stabilization potential[2,48,65]. Changes in stabilization potential shift the potential forms of biomass that are important for soil C pools; when matrix stabilization is high, leaf litter contributes relatively less to soil C than root biomass inputs and exudates[2]. Because fire tends to impact aboveground biomass and organic horizon stocks much more than it does belowground biomass, and the heating depths in soils decline with a greater mineral content because minerals dissipate heat[30] (Fig. 4 and references therein), we expect fire effects on SOM turnover to be lower in soils with a high stabilization potential. In soils with a low stabilization potential, particulate SOM is the dominant form and tends to be derived from leaf litter and is more prone to decomposition. Given that fire readily combusts leaf

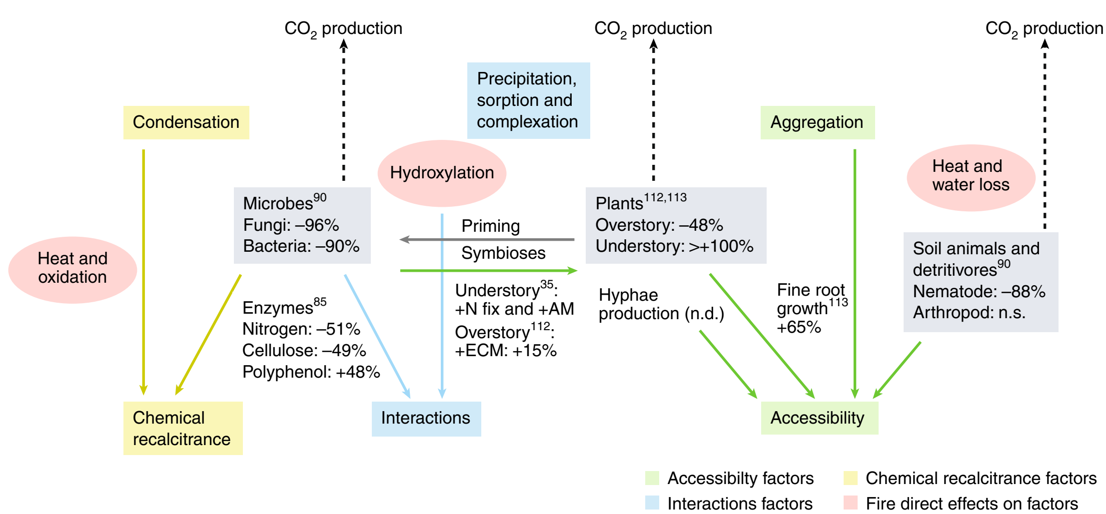

Fig. 4 | Factors that influence SOM stability that may be influenced by fire. The boxes filled with colours denote the three main factors that influence the stability: chemical recalcitrance (yellow), interactions (blue) and accessibility (green). There are direct fire effects (shown in red ovals) and indirect effects. These indirect effects are outlined in the three grey-shaded boxes (for microbes, plants and soil animals/detritivores) that trigger downstream changes. The percentage change shows a comparison between burned plots and unburned plots in a site for a process (for example, −96% for fungi is a 96% decrease in fungal abundance) taken from either field surveys or meta-analyses[35,85,86,90,114,115]. Heating and oxidation impact chemical recalcitrance[12,14,16,69,116], such as increases in pyrogenic C[14,16,79–81,116] and the promotion of aromatic structures under high heating[9,69,83]. Heating effects on aggregation[54,55] and water repellency[58] can depend on the rate of heating and the soil moisture[57]. Heating effects on interactions such as thermal modification change the potential to sorb SOM[69,70], which can depend on the underlying soil chemistry[72]. Fire-driven turnover in plant symbiont strategies can encourage aggregate formation[59,60,117] or priming[112]. N fix, nitrogen-fixing plants; AM, arbuscular mycorrhizal fungi; ECM, ectomycorrhizal fungi; n.d., not quantified (to our knowledge). n.s., not significant.

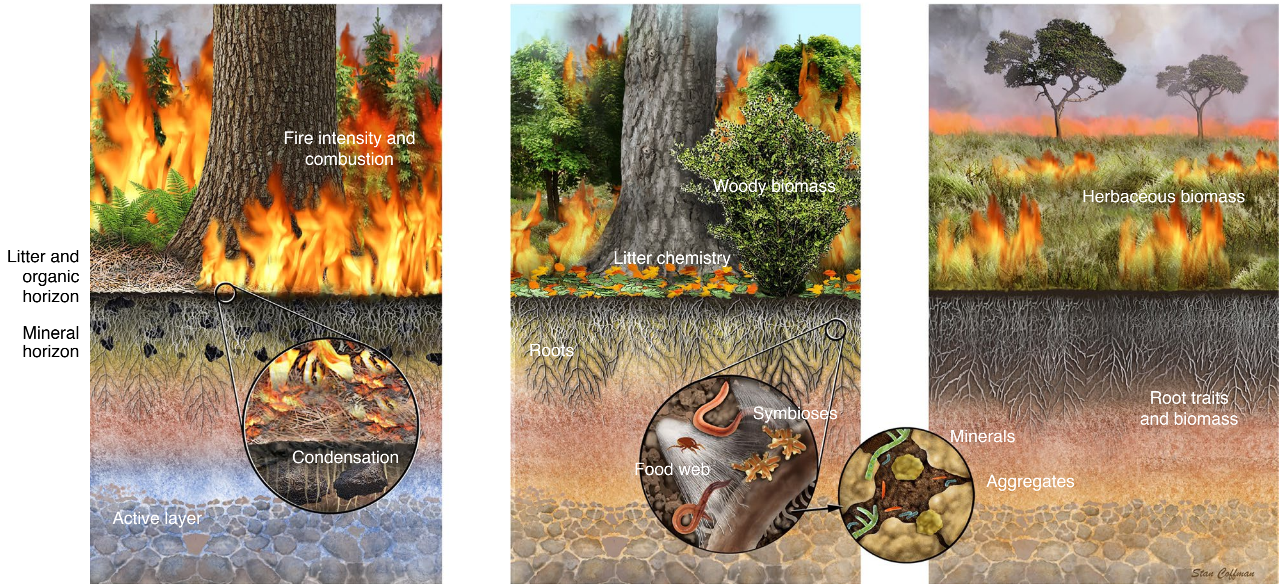

Fig. 5 | Stabilization dynamics differ across ecosystems and can inform fire management for nature-based climate solutions. Intense wildfires in boreal and permafrost systems (left) can result in high soil-burn severities by combusting SOM and increasing the active layer depth[7,45,78,118]. Low-intensity fires in these systems can increase chemical recalcitrance and lower the fuel biomass, limiting decomposition and wildfire severity[84,110,119–122,103,104]. Low-intensity fires in temperate deciduous and mixed forests (middle) can enrich aromatics and the soil pH, change aggregation, promote productive understory species and change the detritivore activity[37,81]. In savannahs (right), fire can be used to promote grasses and fine-rooted species, which can increase aggregate formation and interactions between organic matter and minerals, especially in areas with clay-rich soils and deep-rooted grasses[31,59,60,117]. Further empirical evidence for these processes is presented in the Supplementary Information. Figure adapted from an illustration by Stan Coffman.

litter and kills detritivores that are responsible for litter decomposition (Figs. 3 and 4 and references therein), fire should have larger effects on the turnover of SOM in areas with a low matrix stabilization potential.

Another important axis of variability is the relative storage of SOM in organic versus mineral horizons. In boreal forest soils with a prominent organic horizon, combustion of SOM by fire is a large loss pathway and takes decades to centuries to re-accumulate[7,29,95]. In severe forest wildfires, losses also arise because plant mortality and soil structure destruction enable erosion and leaching[96]. Generally, losses of physical integrity are greater in areas that have experienced higher severity fires and are on steep slopes or exposed to large precipitation events[96]. Organic horizon decomposition is regulated by factors such as the litter quality, microbial community composition and microbial biomass rather than mineral–organic interactions[97]. Given that fire can thermally alter the litter quality, reduce the microbial biomass and activity and shift key decomposer taxa such as fungi (Fig. 4 and references therein), fire is likely to have large effects on SOM turnover. By contrast, fire effects on SOM turnover in mineral soils will be lower and will probably depend on how much SOM is sorbed to the minerals[98] and how plant-root traits impact the SOM stability[59,60] to name but a few factors.

Fire management implications for nature-based climate solutions

Fire management strategies can impact the persistence of SOM and thus be used to promote C storage. For example, nature-based climate solutions aim to avoid ecosystem emissions and enhance C sequestration in both plants and soils[99,100]. While fire is generally treated as a liability to long-term C storage[101], our literature review reveals a number of ways that fire could enhance the SOM stability (Fig. 5). Evidence from the literature suggests that prescribed burns could potentially lead to more stable SOM stocks via two mechanisms: limiting the fuel amount to reduce the wildfire severity[102] and thus the combustion-based losses of SOM during high-intensity wildfires; and/or by increasing the C stability in soil by impacting the accessibility, interactions, recalcitrance and decomposer communities. The relative importance of these different processes differs across environments, requiring the implementation of management strategies that depend on the ecological context.

In ecosystems with deep organic horizons, such as boreal forests and peatlands, the utility and feasibility of prescribed burning to manage SOM losses via greater stabilization is less clear. Low-intensity burning has been shown to reduce the decomposition of residual SOM, but this has received mixed support. Furthermore, it is unclear if lower decomposition offsets combustion-based losses (Fig. 5 and references therein). By contrast, prescribed burns can reduce wildfire severity by limiting fuel accumulation[102]. Consequently, using prescribed fire to reduce wildfire severity and the direct combustion of the organic horizon is likely to be an impactful way to protect standing stocks from future losses. In temperate forests dominated by broadleaf or mixed tree species, there is a distinct litter layer, a high root biomass and a potential variability in symbioses between plants and microbes. Studies suggest that low-intensity fire can be used to promote deep-rooted species and productive understory species as well as SOM condensation while minimizing combustion of the organic horizon (Fig. 5 and references therein). In savannah-grasslands, combustion of an organic horizon is relatively non-existent, but fire-driven changes in root traits and symbiont strategy can impact the accessibility and interactions components of stability (Fig. 4). Consequently, focusing on fire management in areas with high clay soils could bring the largest potential gains with a reduction in burning (Fig. 5).

Conclusions

Our review illustrates that fire has a multitude of impacts on SOM stability and that the effect of fire on the decomposition of SOM could be important for understanding the long-term changes in soil C storage and fluxes, which often cannot be reconciled by the changes in inputs alone. Consequently, a realignment of our focus on biomass combustion to belowground processes like decomposition is needed if we are to reconcile the diversity of ways that fire affects soil C storage, and by doing so potentially reveal novel uses of fire to mitigate climate change when it increases the SOM stability.

Data availability

All data are available from the databases cited in the text or as supplements online to the various meta-analyses.

Code availability

All analyses were conducted using standard code packages.

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Author contributions

A.F.A.P. conceived of the project and led the writing of the text. All other authors provided substantial conceptual input and contributed to the writing. K.G. and A.F.A.P. produced Figs. 1 and 2, and 3 and 4, respectively, and jointly produced Table 1.

Competing interests

The authors declare no competing interests.

Additional information

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Correspondence should be addressed to Adam F. A. Pellegrini.

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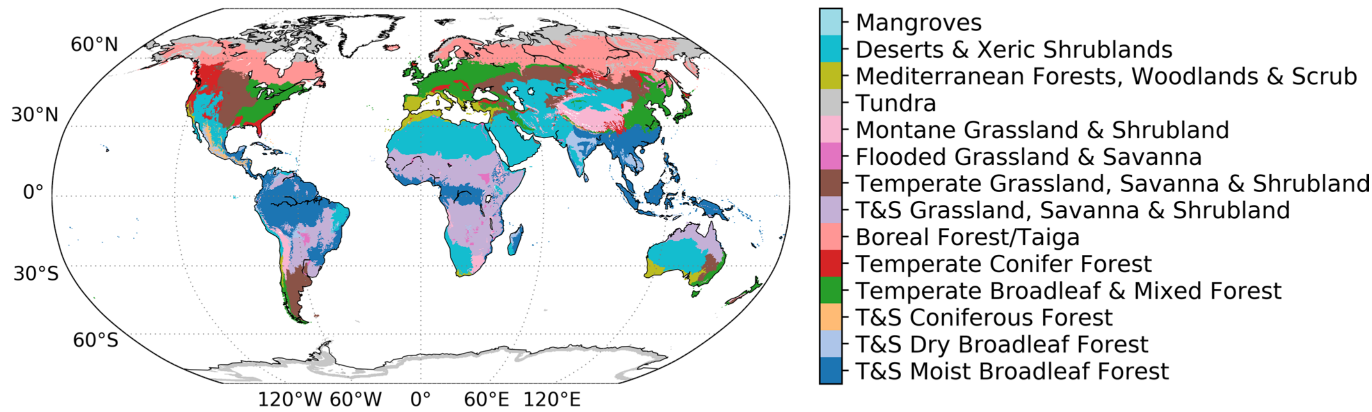

Extended Data Fig. 1 | Ecoregion distribution. Distribution of ecoregions used in the calculations for Table 1.