Oecologia (2005) 143: 1–10
DOI 10.1007/s00442-004-1788-8

CONCEPTS, REVIEWS, AND SYNTHESES

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Effects of fire on properties of forest soils: a review

Received: 7 November 2003 / Accepted: 19 November 2004 / Published online: 2 February 2005

Abstract Many physical, chemical, mineralogical, and biological soil properties can be affected by forest fires. The effects are chiefly a result of burn severity, which consists of peak temperatures and duration of the fire. Climate, vegetation, and topography of the burnt area control the resilience of the soil system; some fire-induced changes can even be permanent. Low to moderate severity fires, such as most of those prescribed in forest management, promote renovation of the dominant vegetation through elimination of undesired species and transient increase of pH and available nutrients. No irreversible ecosystem change occurs, but the enhancement of hydrophobicity can render the soil less able to soak up water and more prone to erosion. Severe fires, such as wildfires, generally have several negative effects on soil. They cause significant removal of organic matter, deterioration of both structure and porosity, considerable loss of nutrients through volatilisation, ash entrapment in smoke columns, leaching and erosion, and marked alteration of both quantity and specific composition of microbial and soil-dwelling invertebrate communities. However, despite common perceptions, if plants succeed in promptly recolonising the burnt area, the pre-fire level of most properties can be recovered and even enhanced. This work is a review of the up-to-date literature dealing with changes imposed by fires on properties of forest soils. Ecological implications of these changes are described.

Keywords Fire · Forest ecosystems · Forest soils · Soil ecology · Soil properties

Introduction

Fires are often major features of forest disruption and renovation. Widely investigated is their role in altering floristic composition (Cammeraat and Imeson 1999), promoting tree regeneration (Martìnez-Sánchez et al. 1999), enhancing timber production (Vihnanek and Ballard 1988), and conditioning human ecology (Caldararo 2002). Numerous findings on the effects of fire on soil properties are available in the literature. The extent and duration of these effects depend firstly upon fire severity, which, in turn, is controlled by several environmental factors that affect the combustion process, such as amount, nature, and moisture of live and dead fuel, air temperature and humidity, wind speed, and topography of the site. Fire severity consists of two components: intensity and duration. Intensity is the rate at which a fire produces thermal energy. Although heat in moist soil is transported faster and penetrates deeper, latent heat of vapourisation prevents soil temperature from exceeding 95°C until water completely vapourises (Campbell et al. 1994); the temperature then typically rises to 200–300°C (Franklin et al. 1997). In the presence of heavy fuels, 500–700°C is reached at the soil surface (DeBano et al. 1998), but instantaneous values up to 850°C can occasionally be recorded (DeBano 2000). The combination of combustion and heat transfer produces steep temperature gradients in soil. Temperatures at 5 cm in the mineral soil rarely exceed 150°C and often no heating occurs below 20–30 cm (DeBano 2000). The depth-trend of temperatures depends overall on thickness, packing ratio, and moisture of the fuel bed (Hartford and Frandsen 1992; Gillon et al. 1995; Campbell et al. 1995). Duration is perhaps the component of fire severity that results in the greatest belowground damage. In fact, intense but fast moving fires at well fuelled sites do not transfer much heat down to more than few centimetres below the surface. After fires, soil temperatures can remain high for from a few minutes to several days.

Basically, there are two types of forest fires: prescribed (controlled) fires and wildfires. Prescribed burning of naturally accumulated forest floor or slash following tree harvest is a standard practice to reduce fuel levels, with the intention of minimising the extent

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and severity of wildfires or facilitating germination and growth of desired forest species. They are primed when soil is moderately moist, and consequently they show a low severity (Walstad et al. 1990). In contrast, wildfires generally occur in the presence of an abundant and dry fuel load and, thus, are very severe. However, due to the inhomogeneous spatial distribution of severity, naturally burnt soils often appear as chaotic mosaics of areas little affected by the fire alternating with others seriously impacted (Rab 1996).

Soil properties can experience short-term, long-term, or permanent fire-induced changes, depending chiefly on type of property, severity and frequency of fires, and post-fire climatic conditions. A plethora of recent works has investigated what type of modification selected properties of forest soils undergo following fire. A synthesis of this literature is reported here, although quantification of variations could not be attempted because of the wide differences among the various ecosystems and fire severities examined. This review is based on papers satisfying three requirements: (I) papers dealing with natural samples from forest soils not experiencing a tropical type of climate, (ii) papers using standardised protocols for both sampling and analysis, and (iii) papers discussing effects not linked to human interventions to stop fire, such as excavation of firebreak trenches or spraying of retarding foams. For brevity of the treatise, the only features of the studied sites described here are the dominant forest species and the fire intensity (in terms of low, moderate or high).

Discussion

Organic carbon

The most intuitive change soils experience during burning is the loss of organic matter (Table 1). Depending on fire severity, the impact on the organic matter consists of slight distillation (volatilisation of minor constituents), charring, or complete oxidation. Substantial consumption of organic matter begins in the 200–250°C range to complete at around 460°C (Giovannini et al. 1988). Combustion causes reduction or total removal of the forest floor (Simard et al. 2001). The recovery of soil organic matter in the burnt areas starts with the natural or artificial reintroduction of vegetation and generally is fast, thanks to the high net primary productivity of secondary ecological successions. By a statistical analysis of data from 48 observations, Johnson and Curtis (2001) disentangled a positive long-term effect of forest fires on the content of soil organic carbon. In fact, considering bias-corrected 99% confidence intervals, a significant average C increase (+8%) in the A horizon was associated with fires that occurred more than 10 years before. The authors suggested three reasons for this increase: (I) the incorporation in the mineral soil of unburnt residues that, consequently, are more protected from biochemical decomposition, (ii) the transformation of fresh organic materials to more recalcitrant forms, and (iii) the frequent entrance in the burnt areas of N-fixer species, which Johnson and Curtis (2001) themselves demonstrated to be able to enhance significantly soil C sequestration. A further explanation for the long-term fire-induced increase in soil organic C can be the decline of the mineralisation rate, which in a severely burnt *Pinus pinaster* forest Fernández et al. (1999) found lasting at least 2 years.

The effects of fire on soil organic C must be evaluated not only on a quantitative basis but also on a qualitative basis. In a work by Fernández et al. (1997) the top 10 cm of a soil under *Pinus sylvestris* was heated at four different temperatures: 150°, 220°, 350°, and 490°C. While the lowest temperature was totally ineffective and the highest one led to complete oxidation of soil organic matter, the intermediate temperatures imposed major structural changes. At 220°C, where 37% of organic matter was lost, all main C forms pyrolysed, although at different rates: 70–80% for cellulose + hemicelluloses, about 50% for both lignin and water-soluble compounds, 10–25% for lipids. Also the humic fraction was reduced by heating, but less than the non-humic fraction. Guinto et al. (1999) found that under *Eucalyptus pilularis* fire induced a reduction in the O-alkyl C/alkyl-C ratio of soil organic matter, which means a preferential denaturation of carbohydrates relative to waxes and cutins. A relative enrichment of short-chain compounds (<20 C) among lipids after fire was measured by Almendros et al. (1988) in the topsoil of a *P. pinea* forest, while among terpenoids heating caused a marked reduction in pimaric acid and increase in dehydroabietic and secodehydroabietic acids. The relative yield of aromatic C was comparatively higher in the burnt soil where, in contrast, alkanoic diacids decreased drastically compared to pre-fire content. In a mixed *P. ponderosa–Pseudotsuga menziesii* forest, Choromanska and DeLuca (2002) recorded concentrations of soluble sugars augmented by a factor of 4 at 160°C and a factor of 12 at 380°C, probably as a consequence of the lysing of plant and microbial tissues and the distillation of the partially decomposed litter layer. By a multifactorial analysis of GC-pyrograms, Alcaniz et al. (1994) proved that bulk soil organic matter from a *Quercus ilex* forest heated to more than 200°C became richer in benzonitrile, naphthalene and other aromatic hydrocarbons. Fire-induced transformations of humic acids into alkali-insoluble compounds and fulvic acids into acid-insoluble compounds were described by Almendros et al. (1990) in a *Q. rotundifolia* forest. Associated with loss in solubility, humic and fulvic acids suffered structural modifications, chiefly the disruption of the peripheral oxygen-containing aliphatic chains. Moreover, an extractable humic-like fraction formed ex novo from lignin and brown products originated from carbohydrate dehydration. Almendros et al. (1992) found that humic acids and fulvic acids subjected to heating at 350°C for 120 s experience decarboxylation, transformation of aliphatic moieties, and increase of aromatic carbons. Elimination of alkyl structures occurs preferentially after additional

Table 1 Soil properties modifiable by fires

Physical, physico-chemical, and mineralogical properties
Water repellence
Structure stability
Bulk density
pH
Particle-size distribution
Mineralogical assemblage
Colour
Temperature regime
Chemical properties
Quantity of organic matter
Quality of organic matter
Availability of nutrients
Exchangeable capacity
Base saturation
Biological properties
Microbial biomass
Composition of microbial community
Soil-dwelling invertebrates biomass
Composition of soil dwelling invertebrates community

heating, when no further aromatic neoformation is observed. Miltner and Zech (1997) and Ponomarenko and Anderson (2001) suggested fire as a factor of humification, emphasising its capability to entail polymerisation and polycondensation reactions. As in soil the organic matter often is intimately associated to the mineral phase, it is hypothesisable that fire-induced structural modifications of the organic pool depend also on type of mineralogical assemblage. In this regard, Miltner and Zech (1997) demonstrated that the presence of Fe, Al, and Mn oxides accelerates thermal decomposition of polysaccharides and lignin and promotes formation of aromatic rings, while montmorillonite and quartz have almost no effect.

In a recent comprehensive review, González-Pérez et al. (2004) identify the following main effects of fire on soil organic matter: (I) general removal of external oxygen groups that yields materials with comparatively reduced solubility; (ii) reduction of the chain length of alkyl compounds, such as alkanes, fatty acids, and alcohols; (iii) aromatisation of sugars and lipids; (iv) formation of heterocyclic N compounds; (v) macromolecular condensation of humic substances; and (vi) production of an almost unalterable component, the so-called black carbon.

Black carbon originates at temperatures between 250 and 500°C from the incomplete combustion (charring) of woody residues (Baldock and Smernik 2002). In terms of spectroscopic properties, charring results in loss of the O-alkyl and di-O-alkyl structures that dominate wood and a large increase in aromatic C. Associated slight enrichments of phenolic groups and methyl and short-chain alkyl substituents of the aromatic C can be observed, but over 480°C the character of the charred material becomes almost totally aromatic (Schmidt and Noack 2000; Czimczik et al. 2002). At extreme temperatures, graphite (elemental C) represents the final product of charring. Basing on similarities in chemical composition and spectroscopic properties, Haumaier and Zech (1995) hypothesise that charred materials are precursors of humic substances. On the other hand, Knicker et al. (1996) underline the undeniable structural differences between the two types of compounds, which lead me to doubt a direct descent. Contributions of black carbon to total organic carbon higher than 35–40% are not rare in soils frequently experiencing fires (Schmidt et al. 1999; Ponomarenko and Anderson 2001). In a chronosequence of 12 boreal forest soils, Zackrisson et al. (1996) estimated that charred materials ranged from 984 to 2,074 kg ha^{-1}, quantities sufficient to exert important ecological effects. In particular, their sorptive abilities were very effective at reducing phytotoxicity from phenols produced by *Empetrum hermaphroditum*. Sorptive abilities, however, tended to decrease in time, and to disappear after a century. Reduction of the binding action of phenols released by the understorey of *Vaccinium myrtillus* imposed by freshly charred materials exerted important effects on the renovation of a burnt *Betula pendula* forest because of a greater uptake of N and other nutrients by seedlings (Wardle et al. 1998).

Top mineral soil originally contains hydrophobic substances, such as aliphatic hydrocarbons, leached from the organic horizons (Doerr 2000; Horne and McIntosh 2000). The concentration of these substances depends on the type of vegetation and soil characteristics (Scott 2000), but it is generally low beneath 3 cm (Huffman et al. 2001). Heating at 200–250°C induces gasification of the hydrophobic substances (Doerr et al. 1998), which move downwards in response to temperature gradients to condense on aggregates and single grains as a uniform coating (Huffman et al. 2001; Letey 2001). Hydrophobic substances decompose irreversibly at temperatures higher than 280°C (DeBano 2000).

Physical, physico-chemical, and mineralogical properties

A direct effect of moderate fires on physical properties is the creation of a discreet and continuous water-repellent front parallel to the surface that decreases soil permeability (Imeson et al. 1992). In a soil under *P. menziesii* and *P. contorta*, Robichaud (2000) measured fire-induced 40% reduction in hydraulic conductivity, while in a mixed forest dominated by *P. contorta* Everett et al. (1995) verified that water drop penetration in soil increased from instantaneous to longer than 270 s following slash burning. The depth of the water-repellent front is mainly a function of heating but also of soil characteristics, such as moisture and particle-size distribution (Huffman et al. 2001). However, whatever the fire severity and soil features, it rarely exceeds 6–8 cm (Henderson and Golding 1983; Huffman et al. 2001). Often, due to the irregular pattern of fire severities, portions of water-repellent soil alternate horizontally with portions of permeable soil (Imeson et al. 1992; Martin and Moody 2001). Concerning the persistence of the water-repellent soil layer, under *P. ponderosa* and

P. contorta Huffman et al. (2001) recorded a weakening 3 months after fire, but after a further 19 months the layer was still recognisable. Two years after clear-cut and slash burning, Henderson and Golding (1983) did not find any more evidence of the fire-induced water-repellent layer in soils of boreal forests.

Structure stability can be increased by low to moderate fires because of the formation of the hydrophobic film on the external surface of aggregates (Mataix-Solera and Doerr 2004), whilst stability decreases dramatically when, at high temperatures, organic cements are disrupted (Badìa and Martì 2003). In the latter case, however, the surviving aggregates can show a higher stability than the original ones due to formation of cementing oxides (Giovannini and Lucchesi 1997; Ketterings et al. 2000). Bulk density increases as a result of the collapse of the organo-mineral aggregates (Giovannini et al. 1988) and the sealing due to the clogging of soil pores by the ash or the freed clay minerals (Durgin and Vogelsang 1984). It implies a decrease in the water holding capacity of soil (Boyer and Miller 1994; Boix Fayos 1997) and a consequent accentuation of runoff and surface erosion (Martin and Moody 2001). Particle-size distribution is not directly affected by fires (Oswald et al. 1999) but, on steep surfaces, selective removal of the fine fraction through erosion can lead to soil coarsening (Mermut et al. 1997). In terms of erosion, the combustion of vegetation and litter layer, which mitigate the impact of raindrops on soil and break runoff, often is more detrimental than water repellence (Sevink et al. 1989; Scott and van Wyk 1990; Marcos et al. 2000). Shakesby et al. (1993), for example, ascertained that in stands of *E. globulus* and *P. pinaster* burnt 0–2 years before, rainsplash detachment rates were an order of magnitude and soil losses two orders of magnitude higher than in homologous stands burnt 3–4 years before.

Soil pH is inexorably increased by the soil heating as a result of organic acids denaturation. However, significant increases occur only at high temperatures (> 450–500°C), in coincidence of the complete combustion of fuel and the consequent release of bases (Arocena and Opio 2003) that also leads to an enhancement of base saturation (Macadam 1987). Khanna et al. (1994) assessed that the capacity of ash to neutralise soil acidity is well correlated with the sum of the concentrations of K, Ca, and Mg in the ash itself. Analysing a series of soils developed on different lithologies and under *Q. engelmannii*, *P. ponderosa* or mixed conifers, Ulery et al. (1993) found that the topsoil pH could increase as much as three units immediately after burning; this rise was essentially due to the production of K and Na oxides, hydroxides, and carbonates, which did not persist through the wet season. In contrast, the neo-formed calcite was still present 3 years after burning, and maintained moderately alkaline soil pH. Fire-induced increase in pH is negligible in soils buffered by carbonates. Soil exchangeable capacity is decreased by fires (Badìa and Martì 2003), due to loss of a high-density charged fraction such as organic matter (Oswald et al. 1999). The electrical conductivity is ephemerally increased (Naidu and Srivasuki 1994; Hernandez et al. 1997) as a result of the release of inorganic ions from the combusted organic matter.

Generally, the mineralogical assemblage is not altered to any great extent by fires because the first step of disruption of most minerals—dehydroxylation—occurs over 500°C (Tan et al. 1986). Evidence of a fire-induced substantial reduction in 2:1 expanding clays was found by Arocena and Opio (2003) in a *Picea* forest. Here, the amount of kaolinite was not affected by the fire and it testifies that the temperatures experienced by the soil had not exceeded 550°C, the point where the lattice of kaolinite would be destroyed. Ulery et al. (1996) detected mineral alterations in only 1–2% of a burnt forest, just where concentrated fuel such as logs or stumps had burnt for a long time at high temperatures. At these sites, within the upper 1–8 cm, chlorite, chlorite–vermiculite, vermiculite, and hydroxy-interlayered vermiculite had transformed to illite, while kaolinite had been completely decomposed. Under particular conditions, kaolinite can become a poorly crystalline aluminosilicate able to act as cementing agent among particles (Ulery and Graham 1993). Ketterings et al. (2000) ascertained that temperatures exceeding 600°C noticeably reduce gibbsite concentration and convert goethite into ultra-fine maghemite. Maghemite is a magnetic Fe-oxide that forms in soil through thermal transformation (at 300–425°C) of other pedogenic Fe-oxides (Schwertmann and Taylor 1989). Crockford and Willett (2001) found high amounts of this mineral and, consequently, a peak of magnetic susceptibility in the surface layer of soils often experiencing fires; a second peak of magnetic susceptibility detected at depth allowed the authors to recognise a paleosol. Carbonates, the main constituent of calcareous soils, resist temperatures up to 1,000°C (Rabenhorst, 1988) and, thus, rarely undergo fire-induced changes.

Effects of burning on soil colour are evident: in low to moderate fires the ground is covered by a layer of black or grey ash that stays until plant recolonisation modifies the albedo and, thus, the temperature regime of the soil. Ulery and Graham (1993) found that soils of a *Q. engelmannii* and four coniferous forests severely burnt 22 days to 3 years before had a surface layer (1–8 cm thick) redder and with higher Munsell chromas and values than the surrounding unburnt soils. Reddening was due to Fe-oxides transformation and higher values to nearly complete removal of organic matter. Below the reddened layer, a blackened one with a thickness of 1–15 cm formed as result of charring. Because of their refractoriness, charred materials affect the soil colour for a long time (Schmidt et al. 1999). Like changes in the mineralogical assemblage, post-burn colours can represent indicators of fire severity. In this regard, in iron-rich soils Ketterings and Bigham (2000) demonstrated that Munsell hues became more yellow as values and chromas decreased with short-term heating at 300 – 600°C, and that at 600°C reddening did not occur until after 45 min of exposure.

Nitrogen, phosphorus, and other nutrients

Repercussions of fires on the biogeochemical cycle of nutrients have been investigated especially for N and P. The immediate response of soil organic N to heating is a decrement because of some loss through volatilisation (Fisher and Binkley 2000). However, a substantial portion of soil organic N survives low intensity fires, maybe changing its form. Fire-induced transformations of organic N were studied by Knicker et al. (1996) using ^{15}N NMR spectroscopy on ryegrass heated at 350°C for 30 – 90 s. Progressive modification occurs in two well-defined stages: in the early one free amino acids and some NH_2 groups are removed but no substantial disruption of the peptide structure is observed, while in the second stage of burning the amide-N is converted to heterocyclic compounds such as pyrroles, imidazoles and indoles. Moderate to high intensity fires convert most soil organic nitrogen to inorganic forms. In the topsoil of an *E. regnans* forest, Weston and Attiwill (1990) quantified the fire-induced inorganic N increase into three times the original concentration over the first 205 days, however, after 485 days, a return to the pre-fire level was observed. Ammonium (NH_4^+) and nitrate (NO_3^-) are the inorganic forms of nitrogen that originate during the burning. Ammonium is a direct product of the combustion, while nitrate forms from ammonium some weeks or months after fire as a result of biochemical reactions called nitrification (Covington and Sackett 1992). Both NH_4^+ –N and NO_3^- –N are available to the biota, but if not promptly uptaken, they follow quite different destinies: nitrate is soon leached downwards, while ammonium is adsorbed onto the negatively charged surfaces of minerals and organics and, thus, is held by the soil (Mroz et al. 1980). However, unless it is fixed in the interlayer of clay minerals, ammonium tends to transform into nitrate. In a soil under *P. pinaster*, Prieto-Fernandez et al. (1993) studied the effects of a high intensity wildfire on the N status in the surface (0–5 cm) and subsurface (5–10 cm) layers. One month after burning, total inorganic N, which previously was mainly in the form of NH_4^+ –N, was increased in both layers, while NO_3^- –N was increased only in the subsurface layer, perhaps as a consequence of leaching. In a *P. edulis/Juniperus* spp. stand, Covington et al. (1991) found that slash pile burning caused an immediate strong increase (approximately 50-fold) in soil ammonium. Nitrate concentrations were not affected immediately but 1 year after burning they had become dramatically higher than the pre-fire level. Five years after burning the increase of both forms of inorganic nitrogen had disappeared. These findings were substantially confirmed by Covington and Sackett (1992) under *P. ponderosa*. Grogan et al. (2000) assessed that the NH_4^+ pulse generated by a severe wildfire in a mature forest of *P. muricata* was dissipated by the end of the second growing season.

The importance of a prompt plant recolonisation for conservation of soil N in burnt areas is clearly demonstrated by Weston and Attiwill (1996). In the soil solution collected at 10 cm depth in a *Eucalyptus* forest burnt 260 days before, these authors measured average NO_3^- –N concentrations of 60.3 mg l^{-1} in herbicide-treated plots and 24.2 mg l^{-1} in non-treated plots. If the regrowing vegetation includes N-fixer species, a complete recovery of the original pool of organic N may be relatively rapid: for example, in a high-rainfall *E. regnans* forest where several *Acacia* species germinated in large numbers after a severe fire, the recovery occurred within 3 years (Adams and Attiwill 1984). Often, soil organic N at the new steady state can even exceed the pre-fire level (Johnson and Curtis 2001).

Forest fires have not necessarily the same impact on soil P as on N, because losses of P through volatilisation or leaching are small. Nevertheless, the combustion of vegetation and litter causes impressive modifications on biogeochemical cycle of P. Burning converts the organic pool of soil P to orthophosphate (Cade-Menun et al. 2000), the sole form of P available to biota. Furthermore, the peak of P bioavailability being around pH 6.5 (Sharpley 2000), any fire-induced change in soil pH toward neutrality has a positive effect in this regard. Therefore, fires result in an enrichment of available P (Serrasolsas and Khanna 1995), but this enrichment is destined to decline soon. In fact, in acid soils orthophosphate binds to Al, Fe, and Mn oxides through chemisorption, while in neutral or alkaline soils it binds to Ca-minerals or precipitates as discrete Ca-phosphate. The time in which the positive effect of fire in terms of P availability runs short is highly variable, depending on numerous factors. In a *Eucalyptus* forest, 7 months after clearfelling and slash burning, Romanya et al. (1994) found soil concentrations of available P ($NaHCO_3$-extractable) and labile P ($NH_4F + HCl$-extractable) even higher than pre-fire levels. In a *Picea* dominated forest, Macadam (1987) even found that 9 months after slash burning, available P in the upper 30 cm mineral soil had increased by up to 50% and this increase persisted, although somewhat diminished, 21 months after the fire event.

Fire-induced changes to cycles of soil nutrients other than N and P generally are slighter and more ephemeral. The availability of these nutrients generally is increased by the combustion of soil organic matter and the increase is strictly dependent upon type of nutrient, burnt tree species, soil properties, and pathway of leaching processes (Kutiel and Shaviv 1992). A month after a wildfire, available Ca, Mg, and K in the soil of a *Q. rubra–Populus grandidentata* forest were significantly higher than pre-fire levels, but after further 3 months the increases were almost gone (Adams and Boyle 1980). More persistent (some years) fire-induced enhancement of the availability of these three bases were reported by Simard et al. (2001) under *P. mariana*. Tomkins et al. (1991) showed that in the surface soil of a *Eucalyptus* forest changes lasting up to 6 months involved the levels of exchangeable NH_4^+, K^+, and Mg^{2+}, while exchangeable Ca^{2+} required 2 years or more to normalise. Ludwig et al. (1998) ascertained that the relatively soluble magnesian calcite produced by the burning of a *Eucalyptus* forest changed to an insoluble form within 4 years. Concentrations of cations, such as Ca^{2+}, Mg^{2+}, and K^{+}, and the anion SO_4^{2-} increase considerably in the soil solution immediately following burning (Khanna and Raison 1986). Khanna et al. (1994) separate into three classes, on a solubility basis, the nutrients that accumulate in the ash: (I) K, S, and B, easily soluble but with a residual component (~30%) which is not solubilised by further dilution; (ii) Ca, Mg, Si, and Fe, relatively insoluble but at degree related to dilution; and (iii) P, highly insoluble.

The behaviour of micronutrients, such as Fe, Mn, Cu, Zn, B, and Mo, with respect to fire is not well known because specific studies are lacking. However, under *P. pinaster*, Gonzalez Parra et al. (1996) found that both total content and easily reducible (extractable by NH_4-acetate + hydroquinone) forms of Mn increase significantly following fire, thanks to Mn supplied by the ash in the form of amorphous and crystalline oxides, while the exchangeable Mn does not show any variation. Presumably Fe, Cu, and Zn behave similarly to Mn and move downwards very little. Heating can affect nutrient availability indirectly, by modifying the soil microbial community. An enlightening proof of this was provided by Perry et al. (1984). In a coniferous forest dominated by *P. menziesii*, these authors checked the deleterious effect of slash burning on hydroxymate siderophores, high affinity Fe^{3+} chelators released in soil by various microorganisms including mycorrhizal fungi. Siderophores facilitate Fe uptake to both microbial flora and higher plants. The seedlings of *Pseudotsuga* suffered iron deficiency only in burnt areas; evidently, a reduction of siderophore-producing organisms occurred here, because pasteurisation induced similar Fe limitation in unburnt soils.

Horizons not subjected to thermal shock can retain most of the elements leached from above (Soto and Diaz-Fierros 1993). Goh and Phillips (1991) found that in a *Nothofagus* forest 80–90% of Ca^{2+}, Mg^{2+}, and K^{+} released from the litter layer because of a light to moderate fire were retained on negative charges of the intact humus of the underlying soil. As well as being leached out of soil, nutrients can be removed off-site in particulate form by convection in smoke columns during fire or by surface wind transport. Particulate contributions to elemental transfers is higher where combustion is complete, resulting in formation of a highly nutrient enriched, fine, low-density grey ash, rather than where a coarse-sized black ash forms as a result of an incomplete combustion (Raison et al. 1985). In bared steep surfaces, post-fire loss of nutrients is controlled by water erosion (Thomas et al. 1999).

Microorganisms

The immediate effect of fire on soil microorganisms is a reduction of their biomass. In fact, the peak temperatures often considerably exceed those required for killing most living beings (DeBano et al. 1998). In extreme cases, the topsoil can undergo complete sterilisation. Adverse effects on soil biota can be due also to some organic pollutants produced by the combustion processes. In fact, Kim et al. (2003) verified that toxic compounds, such as polychlorinated dibenzo- *p*-dioxins (PCDDs), dibenzofurans (PCDFs), and polynuclear aromatic hydrocarbons (PAHs), are actually released during fire and redistributed on the ground. Heat also indirectly affects survival and recolonisation of soil organisms through reduction and modification of organic substrates, removal of sources of organic residues, buffering and every other eventual change to soil properties (Bissett and Parkinson 1980; Monleon and Cromack 1996). On the other hand, as demonstrated by Wardle et al. (1997) for boreal forests of *P. sylvestris*, continued fire suppression may lead to late secondary succession under which microbial activity declines. This fact can be explained by limitations in microbial activity and organic matter decomposition imposed by excessive concentrations on phenols, not adsorbed and inactivated by freshly charred materials.

In a soil under *Pinus* spp., Prieto-Fernández et al. (1998) assessed that immediately after the occurrence of a wildfire, microbial biomass had almost disappeared in the surface layer (0–5 cm) and reduced by 50% in the immediate subsurface zone (5–10 cm). After 4 years, reductions with respect to pre-fire levels of 60 and 40% for the ratio microbial C/organic C and 70 and 30% for the ratio microbial N/total N were recorded in the top and the subsurface layer, respectively. The addition of cellulose to the burnt soil favoured fungal mycelium development and increased microbial C, but the negative effect of burning was not counteracted completely. In a coniferous stand, up to 12 years were necessary for microbial biomass to return to pre-fire levels (Fritze et al. 1993). In a *P. abies* forest, Pietikainen and Fritze (1995) found that soil basal respiration diminishes after a low-intensity prescribed fire but not proportionally with the reduction in microbial biomass C, evidently because the specific respiration rate (CO_2–C evolved per unit of microbial C) is greater in burnt areas than in the control. In a Mediterranean pine forest, fire reduced the activity of enzymes such as dehydrogenase, urease, alkaline phosphatase, arylsulphatase, and *N*-alpha-benzoyl-*L*-argininamide hydrolysing protease along with both biomass C and basal respiration (Hernandez et al. 1997). Boerner and Brinkman (2003) ascertained that in a soil under *Q. rubra*, the occurrence of prescribed fires decreased the activity of acid phosphatase and β-glucosidase by 15–50% and 5–50%, respectively, but did not modify substantially that of chitinase and even increased that of phenol oxidase by 20–80%.

The impact of fire on biological properties of soil depends strictly on soil moisture. In a mixed *P. ponderosa*–*P. menziesii* forest, for example, Choromanska and DeLuca (2002) showed that soils at three different levels of moisture (−0.03, −1.0, and −1.5 MPa) experienced different fire-induced declines of microbial biomass

C; the highest decline was observed at the moistest condition, maybe as a result of faster heat transmission than in drier soils, being water a better conductor than air.

Burning also alters the specific composition of soil microbial community. Phospholipid fatty acid (PLFA) analysis allowed Bååth et al. (1995) to demonstrate that in burnt coniferous forests fungi were reduced more than bacteria. In soils under *Pinus*, Torres and Honrubia (1997) showed that fire causes a particularly marked decrease in basidiomycetes and a selective enrichment of species typically carbonicolous among ascomycetes. The fragile symbiosis between vascular plants and mycorrhizae generally suffers under heat (Vilariño and Arines 1991). In burnt soils under *P. edulis*/*J. osteosperma*, Klopatek et al. (1988) demonstrated the existence of an inverse relationship between vesicular-arbuscular mycorrhizal fungi colonisation and temperatures reached during fire, with substantial decreases of mycorrhizae (>50%) occurring only over 50–60°C and reductions up to 95% over 90°C. Severe forest fires also modify significantly the specific assemblage of mycorrhizal fungi (Baar et al. 1999).

Choromanska and DeLuca (2001) demonstrated that applying preventive low-intensity fires may predispose the microbial community to the impact of wildfires. In this regard, in *Pinus* forests periodically submitted to controlled fires, Fonturbel et al. (1995) observed that total microflora and single microbial groups, such as heterotrophic bacteria, filamentous fungi, and algae, did not suffer any long-term effect when moderate intensity wildfires occurred.

Soil-dwelling invertebrates

Generally, the direct effects of fire on soil-dwelling invertebrates are less marked than those on microorganisms, due to the higher mobility that enables invertebrates a greater potential to escape heating by burrowing deep into the soil. However, the indirect effects of fire, particularly litter mass reduction, are effective at decreasing drastically both total mass and number of species of soil dwelling invertebrates (Table 2). In a pine forest, where the original nematode community consisted of 26–39% herbivores, 28–40% fungivores, 29% bacteriovores, 3–4% omnivores, and 1% predators, McSorley (1993) noticed that within 6 weeks after controlled burning, total numbers of omnivores and predators were increased, while that of herbivores was the same. The fungivore *Aphelenchoides* declined after burning while *Acrobeloides*, initially the most abundant bacteriovore, increased. Censussing soil

Table 2 Summary of the effects of fires on properties of forest soils

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Physical, physico-chemical, and mineralogical properties

Water repellence: the natural water repellence of soil often increases because of the formation of a continuous water-repellent layer a few cm beneath the surface. It implies limitations in soil permeability and, thus, increased runoff and erosion

Structure stability: complexity decreases as a result of the combustion of organic cements

Bulk density: increases because of the collapse of aggregates and the clogging of voids by the ash and the dispersed clay minerals; as a consequence, soil porosity and permeability decrease

Particle-size distribution: does not change directly, but the increased erosion can remove selectively the fine fraction

pH: in noncalcareous soils increases, although ephemerally, because of the release of the alkaline cations (Ca, Mg, K, Na) bound to the organic matter

Mineralogical assemblage: changes, but only at temperatures higher than 500°C

Colour: darkens, due to charring, and reddens, due to formation of iron oxides

Temperature regime: changes temporarily because of both the disappearing of the vegetable mantle and the darkening of ground (decreased albedo)

Chemical properties

Quantity of organic matter: decreases immediately after fire, but in the long run generally exceeds the pre-fire level

Quality of organic matter: changes remarkably, with a relative enrichment of the fraction more recalcitrant to biochemical attack. This is due to both selective burning of fresh residues (leaves, twigs, etc.), and neoformation of aromatic and highly polymerised (humic-like) compounds. Charred material, an exclusive product of incomplete combustion, shows residence times of centuries or even millennia

Availability of nutrients: increases, often remarkably, but ephemerally

Organic N (unavailable, often almost coinciding with total N) in part volatilises and in part mineralises to ammonium, a form available to biota. Ammonium adsorbs on negative charges of mineral and organic surfaces but, with time, is destined to be biochemically transformed to nitrate, which is leached soon if not taken up by cells. Nitrogen availability lowers to pre-fire levels in a few years

Organic P mineralises to orthophosphate and the loss through volatilisation is negligible; orthophosphate is not leached out of soil but if not promptly taken up, it precipitates as slightly available mineral forms

Calcium, magnesium and potassium often increase remarkably but ephemerally

Exchange capacity: decreases proportionally to the loss of organic matter

Base saturation: increases as a consequence of the prevailing release of bases from the combusting organic matter

Biological properties

Microbial biomass: decreases; the recovery of the pre-fire level depends chiefly on promptness of plant recolonisation

Composition of microbial community: changes as a consequence of the selective effect of fire on some groups of microorganisms and the modification imposed to vegetation; generally, fungi diminish more than bacteria

Soil-dwelling invertebrates biomass: decreases, but less than that of microorganisms thanks to the higher mobility of the invertebrates

Composition of soil-dwelling invertebrates community: changes, the time of recovery of the pre-fire assemblage differs highly among the various phyla nematodes at 99 burnt and unburnt forested sites, Matlack (2001) concluded that, in the long run, fire does not significantly affect the nematode community either in number of individuals or diversity. Wanner and Xylander (2003) found that total biomass and species inventory of testate amoebae in pine forests are considerably reduced by a controlled fire, but within 1 year they return to the original level. Collett et al. (1993) proved that prescribed low intensity fires cause a substantial decrease of earthworm populations under *Eucalyptus* and recovery to control levels requires some years. However, the time of recovery is unpredictable because it is strictly dependent on soil moisture in the post-burn period (Neumann and Tolhurst 1991). Bhadauria et al. (2000) found that in a mixed oak–pine forest, populations of the earthworms *Eutyphoeus nanianus* and *Octolasion tyrtaeum* were reduced remarkably by a wildfire, but proliferated after a few months, perhaps due to the improved soil moisture and nutrient availability that attracted individuals from adjacent unburnt areas.

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