STATE of KNOWLEDGE of SOIL BIODIVERSITY

Status, challenges and potentialities

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Report
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CHAPTER 4
THREATS TO SOIL BIODIVERSITY - GLOBAL AND REGIONAL TRENDS

4.1 | INTRODUCTION

The vast diversity and the important role of soil biodiversity in ecosystem functioning and ecosystem service delivery can be deeply affected by human activities as well as by natural disasters, though the latter may also be influenced by human-induced changes (for example, deforestation or road building causing landslides). Most threats to soil biodiversity and function are directly related to human activities and associated with land use cover, management and change. These include deforestation, urbanization, agricultural intensification, loss of soil organic matter/carbon, soil compaction, surface sealing, soil acidification, nutrient imbalance, contamination, salinization, sodification, land degradation, fire, erosion and landslides (Figure 4.1.1).

Land-clearing is a major global threat to soil ecosystem services. This threat has many disguises depending on the specific characteristics of the world's ecoregions. For example, deforestation to make way for food and fibre production systems can lead to massive erosion and nutrient depletion in ecoregions characterized by mountainous terrain with steep hillsides and major precipitation events such as in Latin America and New Zealand.

Land clearing for agricultural intensification in large, low relief ecoregions in Australia (for example tropical and subtropical grasslands) can lead to wind erosion or to acidification. Besides losing tree cover, some agricultural practices result in rapid loss of soil C and microbial biomass, particularly of soil fungi. Simplified and often highly dynamic communities suffer from increasing rates of erosion and leaching due to reduced capacities to absorb water and mineral nutrients therein. Monocultures commonly result in proliferation of above-ground and below-ground pests and pathogens, which require introduction of pesticides in intensively managed fields. These have variable and largely unpredictable effects on natural soil biota. Agricultural practices also reduce soil nutrient concentrations, which requires fertilization. Excess fertilizer applications typically reduce the abundance of mutualistic soil biota, which enables increase in pathogenic microbiota

(Wall *et al.*, 2015). Modern intensive agriculture demands a continuous and constant trade-off between provisioning and regulating/supporting services. Productivity aims to increase the rate of provisioning services to the detriment of regulating services; however, when regulating and supporting ecosystem services are disrupted, food production is seriously affected, the result being a vicious downward spiral (FAO, 2011).

Global trends in urbanization and infrastructure-building bury much of the land surface under concrete, strongly reducing biodiversity and ecosystem services. The same applies to mining and waste dump areas. Two other threats – climate change and invasive species – are usually indirectly associated with human activities, but have become increasingly worrisome in the last couple of decades. The effects of changes in climate and invasive species are poorly understood for much of the soil biota. Accumulating evidence suggests that global change effects may largely differ by taxonomic and functional groups, specific factors and their combinations.

The last decade has shown that extreme climatic events, such as drought and floods, are the aspect of climate change that may be most relevant, overriding gradual shifts in temperature and precipitation. Orgiazzi *et al.* (2016) also discuss variations among groups of soil organisms in responding to these environmental change drivers. For example, soil acidification may affect soil microorganisms more strongly than meso- and macrofauna (Orgiazzi *et al.*, 2016), while for other threats, such as land-use intensification, larger soil fauna may be more strongly affected than microorganisms (Gossner *et al.*, 2016; Phillips *et al.*, 2019a). This points to the fact that the effects of land use intensification and climate change on soil biodiversity are organism-dependent (George *et al.*, 2019), and therefore that detailed information on multiple soil species and traits is urgently needed to better understand and predict threats to the different facets of soil biodiversity (Pey *et al.*, 2014; Salmon *et al.*, 2014).

The level of impact to soil biodiversity and function is not the same for all types of threats and for all regions of the world, and the effects of global change on soil biodiversity may be direct or indirect – via altered vegetation and nutrient availability. Importantly, climate change and land use intensification drivers are not completely independent of one another and thus co-occur. For instance, soil erosion is a process that is particularly relevant in disturbed ecosystems like agricultural lands, with approximately 80 percent of the Earth's agricultural lands experiencing significant levels of soil erosion (Orgiazzi *et al.*, 2016; Borelli *et al.*, 2017; Sartori *et al.*, 2019).

Furthermore, intensively used agricultural lands that are treated with soil tillage often also receive high levels of mineral fertilizers and pesticides. Land degradation is often related to other drivers like overgrazing by livestock and/or intensive agricultural use. This means that greater efforts are needed to understand the multiple direct (such as intensive land use) and indirect (such as climate change) anthropogenic impacts (Veresoglou *et al.*, 2015; Orgiazzi *et al.*, 2016) on soil biodiversity. Another important implication is that threats to soil biodiversity do not only co-occur but can have additive, interactive or synergistic effects (Thakur *et al.*, 2018), reducing soil biodiversity to even lower levels

than what we would expect to find based on single driver studies. For example, effects of increasing temperature were shown to be minor under ambient water conditions, but detrimental for soil biological activity under drought (Thakur *et al.*, 2018).

Important interactions among several of the individual threats listed above and the combination of factors may synergistically affect soil biota and its functioning. For example, plants under drought stress may be more vulnerable to invasive pathogens and pests. At the same time, altered climatic conditions may promote invasion of co-introduced microbial species. Fragmentation of natural communities may reduce migration of both macro- and microorganisms, with increasing risks of extinction. Many of these combined effects may be unpredictable, because of our poor knowledge of ecophysiology and functioning of key soil organisms. Taken together, it is likely that the combined global change factors reduce biodiversity of native species, which is partly compensated for by increasing spread of cosmopolitan species. The combined global change effects are predicted to be context-dependent (that is, they differ by biome, organism group and relative effect on dominant vegetation or its shift).

Unfortunately, the level of knowledge of the impacts of these threats on soil biodiversity and function are highly variable, depending on the threat and the region, as well as the target biota (macro-, meso- or microfauna, microbes). Notably, despite slowly accumulating evidence for the ubiquity of significant interactive effects of environmental change drivers (Eisenhauer *et al.*, 2012a; Thakur *et al.*, 2018, 2019), there are currently almost no mechanistic understanding nor well-informed predictions of interactive impacts of multiple drivers (Borelli *et al.*, 2018; Thakur *et al.*, 2019) on soil diversity and consequences for ecosystem functions.

Despite the mounting scientific evidence warning about major threats to soil biodiversity and function in response to climate change and land use intensification, soil biodiversity has been omitted from many global biodiversity assessments and conservation actions (Cameron *et al.*, 2019; Eisenhauer and Guerra 2019), and understanding of global patterns of soil biodiversity remains limited (Delgado-Baquerizo *et al.*, 2018; Cameron *et al.*, 2019; Crowther *et al.*, 2019; Phillips *et al.*, 2019b; van den Hoogen *et al.*, 2019).

In the following sections, specialists from around the world tackle each of these threats and their potential impacts on soil biodiversity and ecosystem functions, highlighting knowledge gaps to address in future research.

Figure 4.1.1 (next pages) | **Major anthropogenic threats to soil biodiversity**

The major threats to soil biodiversity are caused by human-induced changes and the negative impacts can be amplified by the synergistic and additive effects that might occur among such threats.

Deforestation
drivers and effects on soils

- Land use change
- Loss of SOM and nutrients.
- Changes in soil physical properties.
- Disruption of suitable habitat.
- Changes in pH.

Impacts on soil biodiversity ⚠

- Loss of specialist species and increase in generalist taxa.
- Decrease in predator species.
- Reduced soil and functional diversity.
- Recovery could take decades.

Agricultural intensification
drivers and effects on soils

- Greater use of external inputs (pesticides, fertilizers) and more soil disturbance.
- Greater risk of soil erosion, contamination, land degradation, compaction and salinization.
- Alteration of hydrological and biogeochemical cycles.
- Disturbance of soil structure.
- Loss of SOM.

Impacts on soil biodiversity ⚠

- Decrease in soil biodiversity.
- Smaller and less complex belowground food webs.
- Recovery of soil communities may take years or decades.
- Less efficient and functional soil food webs.
- Loss of soil carbon and nutrients through leaching.

Nutrient imbalances
drivers and effects on soils

Cl P Mg
Zn Na Cu
K Mo S
B Mn N Fe Si
Ca

- Change in the availability of essential nutrients.
- Excessive use of mineral fertilizers.

Impacts on soil biodiversity ⚠

- Reduces the growth capacity of soil microorganisms.
- Reduces nutrient flow through the soil food web.
- Alteration of the nutritional content of primary producers and litter inputs.
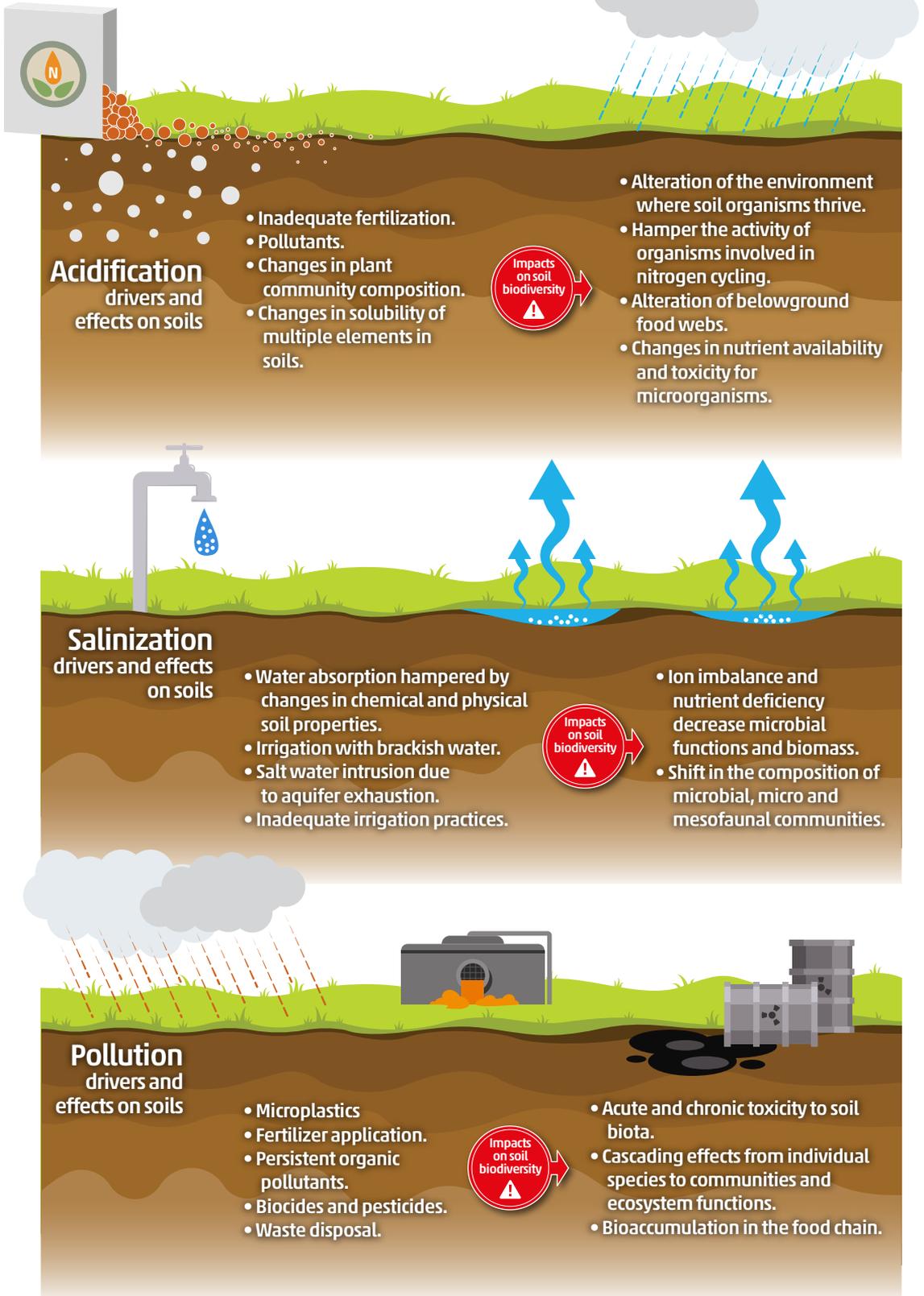
Acidification
drivers and effects on soils

- Inadequate fertilization.
- Pollutants.
- Changes in plant community composition.
- Changes in solubility of multiple elements in soils.

Impacts on soil biodiversity ⚠

- Alteration of the environment where soil organisms thrive.
- Hamper the activity of organisms involved in nitrogen cycling.
- Alteration of belowground food webs.
- Changes in nutrient availability and toxicity for microorganisms.

Salinization
drivers and effects on soils

- Water absorption hampered by changes in chemical and physical soil properties.
- Irrigation with brackish water.
- Salt water intrusion due to aquifer exhaustion.
- Inadequate irrigation practices.

Impacts on soil biodiversity ⚠

- Ion imbalance and nutrient deficiency decrease microbial functions and biomass.
- Shift in the composition of microbial, micro and mesofaunal communities.

Pollution
drivers and effects on soils

- Microplastics
- Fertilizer application.
- Persistent organic pollutants.
- Biocides and pesticides.
- Waste disposal.

Impacts on soil biodiversity ⚠

- Acute and chronic toxicity to soil biota.
- Cascading effects from individual species to communities and ecosystem functions.
- Bioaccumulation in the food chain.
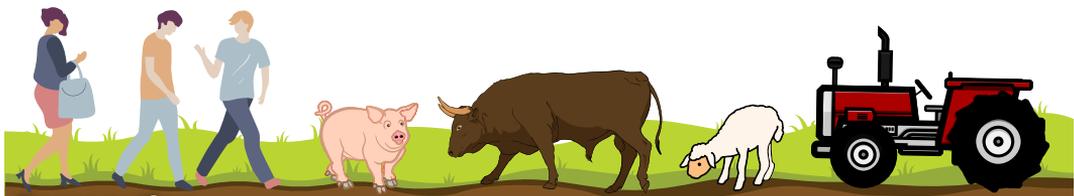
Compaction
drivers and effects on soils

- Decreases macropore volume.
- Increases resistance to root penetration.
- Reduces water infiltration and increases runoff.
- Affects oxygen, and CO_2 fluxes as well as redox potential.

Impacts on soil biodiversity ⚠

- Loss of habitat and pore spaces for soil biota.
- Affects faunal activity.
- Decrease in faunal biomass and population density.

Urbanization
drivers and effects on soils

- Soil sealing, increasing water runoff and reducing infiltration.
- Pollution.
- Topsoil removal or replacement, and addition of anthropogenic materials.

Impacts on soil biodiversity ⚠

- Reduced habitat for soil biota, and increased spatial heterogeneity and fragmentation.
- Alteration in soil communities and food web dynamics.
- Drastic alteration of the environment where soil organisms live.
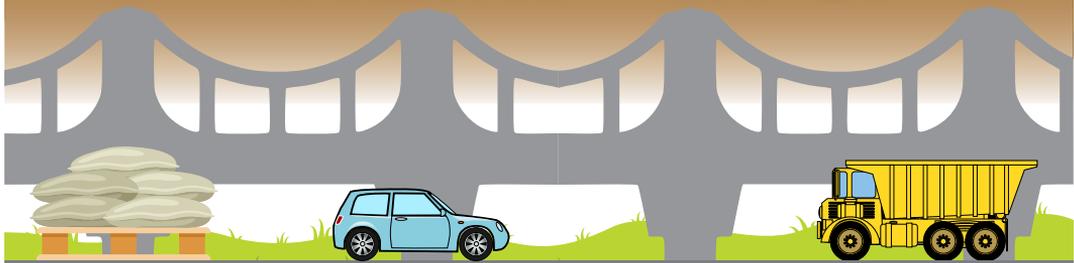
Surface sealing
drivers and effects on soils

- Increases water runoff and reduces water infiltration.
- Changes nutrient and carbon cycling.
- Affects climate and microclimate regulation.
- Building of roads and other permanent infrastructures.

Impacts on soil biodiversity ⚠

- Loss of habitats for soil organisms.

Fire
drivers and effects on soils

- Wildfires.
- Anthropogen burning for land clearing.
- Removal of topsoil organic matter.

Impacts on soil biodiversity ⚠

- Severe damage to soil biodiversity in the topsoil.
- Recolonization, with shift from bacteria-driven towards fungi-driven community.
- Decrease in soil protist and invertebrate abundance, biomass and diversity.
- Very slow recovery of macroinvertebrate diversity and functional structure (decades).

Erosion and landslides
drivers and effects on soils

- Detachment, transport and deposition of soil particles by water or wind.
- Loss of organic matter and changes in soil physical and chemical properties.
- Creation of degraded and enriched depositional environments.

Impacts on soil biodiversity ⚠

- Inhabitants of upper soil layers may be eliminated or displaced.
- Loss of habitat and decrese in its quality for soil biota.
- Spread of pests and pathogens.
- Reduced soil biodiversity and functioning.
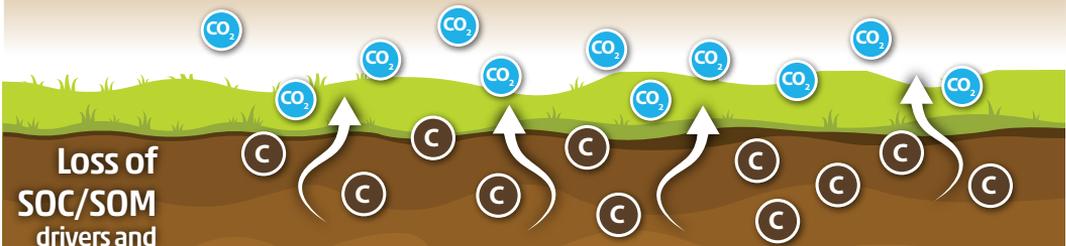
Loss of SOC/SOM
drivers and effects on soils

- Decrease in:
  - Formation and stabilization of aggregates.
  - Cation exchange capacity.
  - Water infiltration and retention.
  - Soil fertility and C sequestration.

Impacts on soil biodiversity ⚠

- Lower microbial biomass and diversity (especially in extreme environments).
- Decreased resources to belowground food webs.

4.2 | THREATS TO SOIL BIODIVERSITY

4.2.1 | DEFORESTATION

Forest ecosystems cover roughly 30 percent of the Earth's land surface and contain highly diverse and poorly studied soil communities. These systems are increasingly under threat, with over 1.3 million square kilometres lost in the last three decades (World Bank, 2016). The negative environmental impacts of deforestation are most evident in the tropics, where the majority of future deforestation is anticipated (Laurance *et al.*, 2014). In Amazonia, the largest intact tropical forest in the world (Lapola *et al.*, 2014), about 17 percent of the rainforest has been destroyed over the past 50 years, with recent losses again on the rise (INPE, 2017). Deforestation often involves the removal of plant biomass through logging of high-value wood trees and slashing and burning of low-value trees prior to consolidation into cattle ranching operations or mechanized agriculture with highly disturbed soils. This results in loss of soil organic matter and nutrients and changes to soil physical properties that disrupt resource supply and habitat suitability to a variety of soil organisms (Neill *et al.*, 1997; Garcia-Montiel *et al.*, 2000; Cerri *et al.*, 2004; Smith *et al.*, 2016). Consequently, deforestation can dramatically alter the structure of soil communities (Crowther *et al.*, 2014), commonly through the loss of specialist species (Mueller *et al.*, 2016), which in turn leads to decreased functional diversity and functional homogenization (Clavel *et al.*, 2011; Nordén *et al.*, 2013).

The increased prevalence of generalist taxa is a consistent response to deforestation across broad taxonomic groups. However, recent studies in tropical rainforests have shown that responses of soil biodiversity to deforestation can be remarkably different from those of above-ground plants and animals. For example, the rapid invasion of a single peregrine earthworm species following deforestation and pasture establishment in Amazonia can enhance earthworm abundance and biomass while decreasing species richness (Barros *et al.*, 2002, 2004). In Central Amazonia, deforestation and the establishment of pastures leads to a dramatic fall in the diversity of ecosystem engineering taxa, with approximately 70 percent of the original taxa disappearing and being replaced by large populations of invaders such as the earthworm *Pontoscolex corethrurus*, a species that can cause profound changes to soil structure and functioning (Barros *et al.*, 2004; Chauvel *et al.*, 1999). Logging of old growth Bornean forest has been shown to reduce termite abundance and diversity, with studies indicating a reduction of 65 percent in termite species richness following forest disturbance (Donovan *et al.*, 2007) and broad effects over all termite functional groups (Luke *et al.*, 2014). These impacts, together with climate change, can have important implications for ecosystem function and resistance to drought, since termites are key regulators of decomposition, nutrient heterogeneity and moisture retention (Ashton *et al.*, 2019).

Most tropical rainforest soils are naturally acidic, and often receive large quantities of lime following deforestation to neutralize pH, especially with the establishment of more intensive cropping systems. However, large shifts in pH impose stress to native microorganisms, affecting their growth (Fierer and Jackson 2006; de Carvalho *et al.*, 2016). This process results in the loss of endemic species of soil microbial decomposers and homogenization of soil communities after conversion of tropical rainforests to pastures and croplands, altering C sequestration and element cycling, and reducing ecosystem resilience to disturbance (Rodrigues *et al.*, 2013).

The abundance and biomass of soil predators such as spiders and predatory insects consistently decreases following deforestation (Franco *et al.*, 2019), indicating that the conversion of forests to arable land affects key organisms involved in population regulation and may favour a few groups that can tolerate disturbance (Franco *et al.*, 2016; Rousseau *et al.*, 2013). These benefited organisms are often plant pests that can harm crops or existing forest. For example, Silva *et al.*, 2008, showed increased populations of plant-parasitic nematodes following forest conversion to pasture in the Brazilian Amazonia.

A recent meta-analysis focused on Amazonian deforestation reported that the abundance, biomass, richness and diversity of soil fauna and microbes are all reduced following deforestation, with greater losses in wetter Amazonian regions and sites with acidic soils (Franco *et al.*, 2019). No evidence of soil biodiversity recovery was found in converted areas over time; biodiversity losses were still evident up to 30 years after forest conversion to arable land. However, limited geographic coverage, omission of micro and mesofauna, and low taxonomic resolution reported in most studies impede our ability to make more specific predictions of deforestation responses and associated management recommendations (Franco *et al.*, 2019).

A cross-biome study in North America showed that the conversion from forest to pasture has consistent directional effects on microbial community composition and catabolic profiles relevant to ecosystem function. Both bacterial and fungal biomass decreased in response to land-use conversion, and although the diversity of both groups increased, the effect size was moderated by soil texture with lesser effects observed on fine-textured soils (Crowther *et al.*, 2014).

Finally, not only deforestation, but also most forms of within-forest degradation (such as wildfires and selective logging) can have pronounced impacts on biodiversity (Gibson *et al.*, 2011). Recent research shows that soil biodiversity and related ecosystem processes may be lost after even very-low, reduced-impact logging intensities (de Carvalho *et al.*, 2016; França *et al.*, 2017). With logging operations rapidly expanding across public lands and more frequent severe dry seasons increasing the prevalence of wildfires in tropical forests, the question of how these within-forest disturbances in intact primary forests affect soil species and their functions emerges as an important research priority for conserving soil biodiversity.

4.2.2 | URBANIZATION

Around the world, urbanized environments – those dominated by residential, commercial and industrial land uses, including cities, towns, villages and suburban and exurban landscapes – continue to expand in conjunction with growing urban human populations (UN, 2019). The initial process of urbanization significantly alters soils and their biodiversity in many ways, especially through removal and replacement of topsoil, compaction, sealing (paving) and addition of anthropogenic materials (Marcotullio *et al.*, 2008; Pickett and Cadenasso, 2009). Within urbanized environments, pollution, landscape management, invasive species and the urban heat island effect, among other variables, further directly and indirectly affect soil properties, including those in remnant native habitat patches that have become surrounded by urban land uses. The multiple interacting and long-term outcomes of urbanization can be perceived as threats to soil biodiversity because urban environmental conditions may degrade soil communities through reduction and loss of populations and shifting communities in ways that affect food web dynamics and ecosystem processes. In turn, soil-derived urban ecosystem services are often negatively affected (Pavao-Zuckerman and Pouyat, 2017). Because such services are critical to supporting the well-being of urban residents, a focus on urban soil biodiversity must become an integral part of global and local efforts to support Sustainable Development Goal (SDG) 11: creating a more sustainable future for cities and other urban communities.

Unfortunately, knowledge about urban soil biodiversity needed to guide sustainable planning and management of urbanized environments is woefully underdeveloped. In a recent review, Guilland *et al.* (2018) identified approximately 100 scientific articles (since 1990) that focused on urban soil organisms and their functional aspects, about half of which focused on arthropods. Even if this review underestimates the amount of relevant research, it does suggest an overall scarcity of basic research about urban soil biodiversity. In particular, there are few, if any, studies that have examined patterns in one place before and after urbanization or how diverse urban variables interact to shape soil communities. Many studies about urban soils examine physicochemical conditions and biogeochemical processes without also investigating the biota. In this context, it is not currently possible to provide robust, generalized conclusions and predictions about how urbanization impacts soil biodiversity patterns, especially at a global scale because of the geographical imbalance of research: 88 percent of studies have been in Europe and North America, with 7 percent, 4 percent, 1 percent and 0 percent from Asia, Australia, South America and Africa, respectively (Guilland *et al.*, 2018). Thus, in addition to increasing the total amount of research about urban soil biodiversity, a major challenge is to increase the breadth of examined biomes and regions. This is a critical need given that many of the fastest growing urban areas and human populations, and thus most pressing concerns about urban sustainable development, are in regions for which nearly nothing is known about urban soil biodiversity (UN, 2019).

Though it remains limited, research advances about urban soil biodiversity over the past three decades have led to a few emerging foundational principles. A key insight is that many urban soils, despite their potentially degraded quality, are inhabited by abundant and diverse organisms from across all taxonomic and functional groups (see Chapter 2), sometimes at levels similar to or greater than other land uses including agriculture (Ramirez *et al.*, 2014; Joimel *et al.*, 2017) (Figure 4.2.2.1). Important drivers of population and community patterns are the environmental conditions created by human management of above-ground habitat structure (for example, plants, detritus layers, impervious surfaces), which influences organic matter inputs and physicochemical conditions such as soil temperature, moisture and pH (Byrne, 2007). Diverse combinations of management goals and activities by many managers across urbanized environments help generate high levels of spatial habitat heterogeneity (alongside background environmental conditions including the underlying native soil template) which likely influences biodiversity patterns (Ossola and Livesley, 2016). This heterogeneity is associated with high habitat fragmentation due to many small, isolated soil patches created by impervious surfaces (roads, buildings); such landscape structural patterns interact with other factors (including pollutants) to create unique conditions that determine which organism can colonize and persist in which patches (Reese *et al.*, 2015). The overall nature and strength of this "urban filtering process" (*sensu* Aronson *et al.*, 2016) for determining the structure and dynamics of soil communities across diverse urban land covers is not well characterized which prevents robust assessment of the degree to which urbanization threatens soil biodiversity from local through global scales. This is also hindered by the lack of studies examining community structure with lower levels of taxonomic resolution (genera, species), especially for protozoans and animals. It is, however, safe to assume that not all soil species are able to pass through the filter such that urban soil biodiversity is degraded to some degree as compared to native communities. Regarding this, which specific soil organisms may need targeted conservation attention is unknown for many places around the world. Future research should aim to investigate how ecological filtering of key functional groups (for example, soil structure formers or population regulators) in different urban conditions affects food web dynamics, ecosystem processes and the associated ecosystem services desired for a specific location.

Given the already large percentage of people that live in urbanized environments and predicted continuation of growth in urban human populations worldwide (UN, 2019), our overall ignorance about urban soil biodiversity may be a bigger threat than urbanization itself. Without more knowledge about how soil organisms are "filtered" by diverse urban variables, we cannot know how to more sustainably plan and manage current or future urbanized environments in ways that conserve and restore crucial soil-based ecosystem services. Indeed, urban soil restoration represents a major opportunity for providing solutions to help urbanized communities reach SDG 11 (Byrne, 2020). To support this, major investments in basic urban soil biodiversity research, including how urban biota contribute to ecosystem services and human health (Li *et al.*, 2018), are urgently needed, especially in tropical biomes and developing countries. Policies and

urban planning that integrate the sustainable management and restoration of soils are rare but also needed for reducing urban threats to soil biodiversity (da Silva *et al.*, 2018). On a rapidly urbanizing planet, the well-being of humanity depends in large part on how well we can quickly improve our knowledge, appreciation and management of urban soil biodiversity.
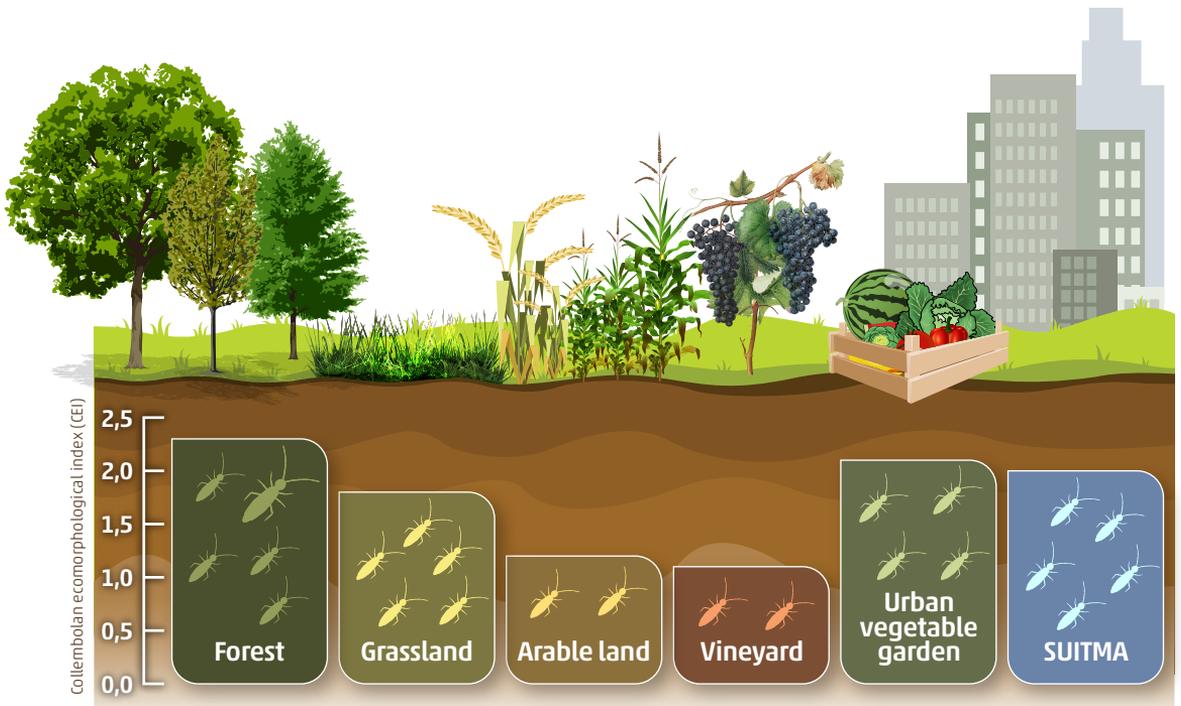
Figure 4.2.2.1 | Collembolan Ecomorphological Index

Soil invertebrates such as microarthropods, including Collembola and Acari, are not just considered as biological indicators of soil quality but also as bioindicators of anthropisation including urbanization and contamination. Indices are useful tools to compare soil biological quality. The higher the Collembolan Ecomorphological Index (CEI), the greater is the abundance of microarthropods adapted to their habitat in the soil. The CEI shows that microarthropod communities are more constrained in agricultural ecosystems compared to urban and forest ecosystems. SUITMA: soils of urban, industrial, traffic, mining and military areas. Adapted from Joimel *et al.* 2017.

4.2.3 | AGRICULTURAL INTENSIFICATION

Agricultural intensification is defined by the Food and Agriculture Organization of the United Nations (FAO) as the "increase in agricultural production per unit of inputs." Related land use management includes among other practices simplified cropping systems (monocultures and few varieties), use of heavy machinery, high input of chemicals such as fertilizers and pesticides, soil tillage and slash and burning. All these practices are driving

forces that pose a range of threats to soil organisms and soil functions. In fact, arable lands, which cover extensive terrestrial areas, have been identified as ecosystems where soil organisms and functions are most threatened (Orgiazzi *et al.*, 2016a).

Agricultural intensification is placing tremendous pressure on ecosystems, leading to large-scale ecosystem degradation and loss of productivity in the long term (Tilman *et al.*, 2001; Vitousek *et al.*, 2009). For example, conversion of natural ecosystems to agricultural lands has resulted in substantial environmental costs, including land degradation, increased emissions of greenhouse gases, decreased organic matter in soils, loss of biodiversity and alterations of biogeochemical and hydrological cycles (Balmford *et al.*, 2005). Modern agriculture thus faces great challenges not only in terms of meeting the food, fibre and fuel demands of an ever-increasing human population, but also in mitigating environmental costs, particularly in the context of inappropriate management practices, a changing environment and growing competition for land, water and energy (Chen *et al.*, 2014). Understanding the mechanisms that control the extent to which soil properties and biological communities change following the conversion of natural to agricultural systems and management practices is of paramount importance to comprehend the consequences of land use changes for soil functions and agricultural productivity (Sala *et al.*, 2000).

Agricultural management practices act and interact with each other in different ways, and affect the soil ecosystem to different degrees and to different extents. In general, they alter soil environmental properties and disturb the soil structure, leading to loss of Soil Organic Matter (SOM; see the following section), degrading micro-habitats that are important to many soil organisms. As the application of agricultural management practices is frequent, biological processes are constantly disrupted and the soil ecosystem is not allowed to recover. The magnitude of effects of specific agricultural treatments on the soil ecosystem depends on their level of intensity, application frequency, timing and extent (Snapp *et al.*, 2010; Roger-Estrade *et al.*, 2010). Agricultural intensification may impact soil organism abundance, biomass, community structure, species richness, species diversity, functional diversity and distribution, and effects of the same disturbance are not equal for all organisms. Relatively larger soil animals and those at higher trophic levels such as earthworms, mites, Collembolans and predatory nematodes are usually more affected (Postma-Blaauw *et al.*, 2010; de Vries *et al.*, 2013; Tsiafouli *et al.*, 2015).

Negative impacts of agricultural intensification have consequences on the specific functions that soil animals perform, including soil structure formation and ecosystem engineering, population regulation by predation, and feeding on fungal hyphae. Considering the entire soil food web, intensive agriculture reduces the biomass and number of functional groups, thus decreasing the links (interactions) between them. Moreover, within the functional groups intensive agriculture reduces species richness, Shannon diversity and taxonomic distinctness (Tsiafouli *et al.*, 2015). Smaller and less complex food webs may negatively impact on ecosystem functioning, with important implications for the services ecosystems provide. For example, a shift from "slow" fungal-based to "fast" bacterial-based soil food webs (Thiele-Bruhn *et al.*, 2012; de Vries *et al.*,

2013), leads to losses of C and N from soil in the form of gases. Furthermore, there may be a decline in the resistance and resilience of food webs to environmental stressors, such as drought (De Vries *et al.*, 2012). Adopting sustainable agricultural practices might lead to recovery of biological communities, but recovery might take years or even decades depending on organisms (de Groot *et al.*, 2016).

The threats posed by agricultural intensification are often multiplied due to the interactive effect of other threats. For example, losses of carbon, soil structure and soil biodiversity can reduce an ecosystem's ability to sequester carbon (Wiesmeier *et al.*, 2019). Changes in these soil properties also decrease water infiltration capacity, root penetration and access to nutrients for plants. In concert, all of these changes increase the risk of soil erosion, land degradation, compaction and salinization, thus reducing agricultural productivity which threatens the achievements of Sustainable Development Goals, particularly SDG 2 (zero hunger). The excessive use of fertilizers and pesticides affects the quality of water (Foster and Custodio, 2019) posing several threats to other ecosystems, and also poses direct threats to animal and human health. Ecosystem services provided by beneficial crop-associated organisms, such as regulation of pest and diseases (Tamburini *et al.*, 2016) and pollination, are also reduced (Bretagnolle and Gaba, 2015). Monocultures and the use of few varieties reduces local variety traits and this above-ground loss is likely coupled to loss of soil biodiversity, though the magnitude of these impacts is still uncertain.

Several syntheses and meta analyses have been conducted to evaluate how agricultural intensification affects soil organisms. Examples include analyses of nitrogen (N) additions on soil microbial biomass (Treseder, 2008), of nutrient inputs on mycorrhizal abundance (Treseder, 2004), and of agricultural intensification on soil biodiversity (de Graaff *et al.*, 2019). Results from these analyses indicate that agricultural intensification can significantly alter soil biodiversity, with negative impacts of synthetic N fertilization on microbial biomass, arbuscular mycorrhizal fungal (AMF) and faunal diversity, and a reduction in soil faunal and bacterial diversity with tillage (Treseder, 2008; de Graaff *et al.*, 2019). Results also indicate that soil biodiversity may be enhanced by agricultural practices if agricultural management practices promote soil organic matter (SOM) accumulation and retention (de Graaff *et al.*, 2019), highlighting the importance of implementing sustainable agricultural management practices to promote soil health.

Agricultural intensification can negatively impact ecosystem functioning through its effect on soil microbial properties. For example, a meta-analysis showed that an N fertilization-induced reduction in microbial biomass also affected ecosystem carbon (C) fluxes by reducing carbon dioxide emissions (Treseder, 2008). Reductions in AMF abundance (Treseder, 2004) and diversity (de Graaff *et al.*, 2019) following agricultural intensification are likely to significantly impact ecosystem functioning, because AMF are crucial to plant nutrient acquisition, plant production, and C transfer from the atmosphere to soil (Smith and Read, 1997). While many studies have quantified the effects of agricultural management practices on ecosystem functioning, fewer have linked changes in soil organism diversity directly to changes in ecosystem functioning.

However, agricultural intensification impacts on functional microbial diversity have been evaluated by community-level physiological profiling of heterotrophic bacterial or fungal assemblages (Zak *et al.*, 1994; Lupwayi *et al.*, 2017). A recent meta-analysis of these studies found that microbial functional diversity significantly increased if N fertilizer inputs promoted soil carbon retention (de Graaff *et al.*, 2019). We caution that methods employed to study functional diversity use standardized incubation conditions that are likely not be optimal for all soil communities and may bias results when extracted communities rather than whole soils are measured (Chapman *et al.*, 2007). Future research exploring a more direct link between soil biodiversity and ecosystem functioning will improve our understanding of agricultural intensification impacts on biodiversity and ecosystem functioning.

Given the dearth of studies that directly link changes in soil biodiversity and ecosystem functioning following agricultural intensification, some have synthesized data from studies that experimentally manipulated changes in soil biodiversity and measured consequences for ecosystem functioning (de Graaff *et al.*, 2015; Kardol *et al.*, 2016; Nielsen *et al.*, 2011). These biodiversity manipulation studies indicate that changes in soil biodiversity affect ecosystem process rates (de Graaff *et al.*, 2015; Nielsen *et al.*, 2011), but the manipulations used in these studies tend to exaggerate biodiversity losses and possibly overestimate consequences for ecosystem functioning relative to measured biodiversity losses from agricultural intensification. Finally, while many studies focus on the impact of biodiversity loss within a trophic group on ecosystem functioning, others have shown that loss of interactions among species can supersede these effects (Valiente-Banuet, 2014), thus highlighting the importance of understanding the soil food web for the sustainable provisioning of ecosystem services. Advances in analytical techniques (for example, meta-genomics) to identify soil organisms and link their structure to their function, coupled with an increase in soil biodiversity manipulation experiments that manipulate diversity within and across energy channels, trophic groups, functional groups, taxa and genetic differences should help solidify links among agricultural intensification, soil biodiversity and ecosystem functioning.

4.2.4 | LOSS OF SOIL ORGANIC MATTER AND SOIL ORGANIC CARBON

Soil organic carbon (SOC), as a main resource for soil organisms, affects several soil functions, including the support of biodiversity (Wiesmeier *et al.*, 2019). There is evidence from global analyses that soils with higher SOC harbour larger microbial biomass (Maestre *et al.*, 2015; Crowther *et al.*, 2019; Wiesmeier *et al.*, 2019), and SOC also appears as one of the main drivers of soil microbial diversity at the global scale (Delgado-Baquerizo *et al.*, 2016), with a generally positive effect of SOC content

on microbial diversity (Fierer and Jackson 2006; Maestre *et al.*, 2015), particularly in extreme environments with low plant productivity such as polar (Siciliano *et al.*, 2014) and dryland regions (Maestre *et al.*, 2015). The pattern, however, differs between taxonomic groups (Tecon and Or, 2017). Global patterns for the distribution of diversity in soils are poorly understood, so that the factors driving them are difficult to understand (Decaëns, 2010). For some groups, patterns of diversity respond to a latitudinal gradient that can be partly explained by organic matter variation (Decaëns, 2010; Caruso *et al.*, 2019 for Oribatid mites). Global-scale patterns of diversity for other groups are not explained by SOC content (Nielsen *et al.*, 2014 for nematodes) or productivity gradient (Decaëns *et al.*, 2010). Earthworm diversity, for instance, was not related to SOM/SOC content across Europe (Rutgers *et al.*, 2016) and the world (Phillips *et al.*, 2019), although at the local scale it may be important (Hendrix *et al.*, 1992).

Nonetheless, the generally positive relationship between soil C stock and soil biodiversity suggests that soil carbon loss is a threat to soil biodiversity. In this regard, Orgiazzi *et al.* (2016a) identified SOC decline as a major threat to both soil microbial and fauna biodiversity. But the underlying causes may be different, as the main drivers of SOC loss, land use change and climate change (see below) also directly impact soil biodiversity. For instance, soil biodiversity was higher in agricultural soils than in carbon-rich northern forests (Griffiths *et al.*, 2016), but the main factor explaining biodiversity was pH, and low pH soils tend to have higher carbon content. Several authors also highlight the importance of soil carbon quality in addition to quantity for below-ground diversity on a global scale (Crowther *et al.*, 2019). For instance, Szoboszlay *et al.* (2017) found evidence of associations between particular SOC fractions (especially particulate organic matter) and specific bacterial taxa across a large range of European soil with various land uses. In their study, SOC content explained 5 percent of the variation in bacterial diversity, while SOC quality explained 22 percent (Box 4.2.4.1). SOC loss results in a decline of several soil functions, including soil fertility and C sequestration (maintaining and increasing SOC storage in soil is crucial in climate change mitigation), and SOC loss is an important indicator of soil degradation (Lal 2015; Lorenz *et al.*, 2019).

There are global maps of soil carbon available (FAO and ITPS, 2017), but there are currently no global maps of carbon loss directly, and data available on SOC dynamics is unbalanced (Jandl *et al.*, 2014), even though carbon loss is highly related to land use change, particularly conversion of natural environments into agricultural or pastoral use. Mapping carbon loss can therefore be done quite reliably by mapping land use change. Climate change also threatens soil carbon, and has been mapped; this is discussed elsewhere in this report. A map of threats to soil carbon could be made by overlaying these maps. This would require a way of calculating the effect of these factors. There are many ways of doing this, from simple statistical models to IPPC methods and dynamic simulation models. The most commonly used soil carbon models, CENTURY (Parton *et al.*, 1993) and RothC (Jenkinson *et al.,* 1990) as well as the IPCC method (IPCC, 2006) have been set up in a dynamic simulation tool linked to GIS maps (Easter *et al.*, 2007). There are also earth system models such as LPJ (Sitch *et al.*, 2003) and CLM (Oleson

et al., 2010) that can simulate the land surface on a global grid. The large uncertainties caused both by model structure and input data means that results from models must be interpreted and used cautiously. However, models do provide a consistent way of simulating SOC as a function of soil texture, climate and land use. This can give indications about where the risk of loss is high, and what management decisions can minimize the risk.

Overall, the effects of SOC loss on soil biodiversity are globally poorly understood, because (i) data on soil biodiversity and the patterns of distribution at large scale are insufficient (especially for soil fauna), (ii) the biological mechanisms involved in how SOC affects soil biodiversity are poorly known, and (iii) many of the other threats to soil biodiversity also lead to changes in SOC content, especially land use and climate change. Furthermore, although we address the role of SOC/SOM content for soil biodiversity, several studies suggest that shifts in SOC/SOM quality and heterogeneity is more important and should be considered for a comprehensive understanding of SOC/SOM impacts on soil biodiversity. This would also require a better understanding of which aspects of SOM/SOC quality are important for soil organisms, and how to assess this. Finally, long-term experiments designed to investigate these effects are needed, to disentangle the effect of soil organic matter directly from those of other factors causing SOC loss in addition to affecting soil biodiversity directly. Especially more data from non-agricultural systems are needed.

Soil organic carbon (SOC) and its different fractions affect soil microbial diversity across different land-use types. The figure shows associations among different SOC fractions - such as particulate organic matter (POM) - and specific bacterial taxa, highlighting the importance of soil carbon quality – set of indicators that allow to establishing how easily SOM can be mineralized- over the total carbon quantity for belowground microbial diversity. POM represents a substrate and a microhabitat for soil microbial communities strongly influencing bacterial community structure. Isolated SOC fractions with different functional traits and turnover rates (POM included) explained 22% of the variation in the soil bacterial community in contrast with total SOC that explained 5%.

Circles represent Operational Taxonomic Units (OTUs), and hexagons represent SOC fractions. The size of the circles is indicative of microbial abundance, and different colors show their taxonomic classification. Green edges indicate positive and red edges mean negative associations. SC-rSOC: particle size between 0.45 and 63 µm, oxidizable; DOC: dissolved organic carbon, particle size <0.45 µm; POM: particulate organic matter, particle size >63 µm, low density, SA: sand and stable aggregates, particle size >63 µm, high density). Szoboszlay *et al.*, 2017
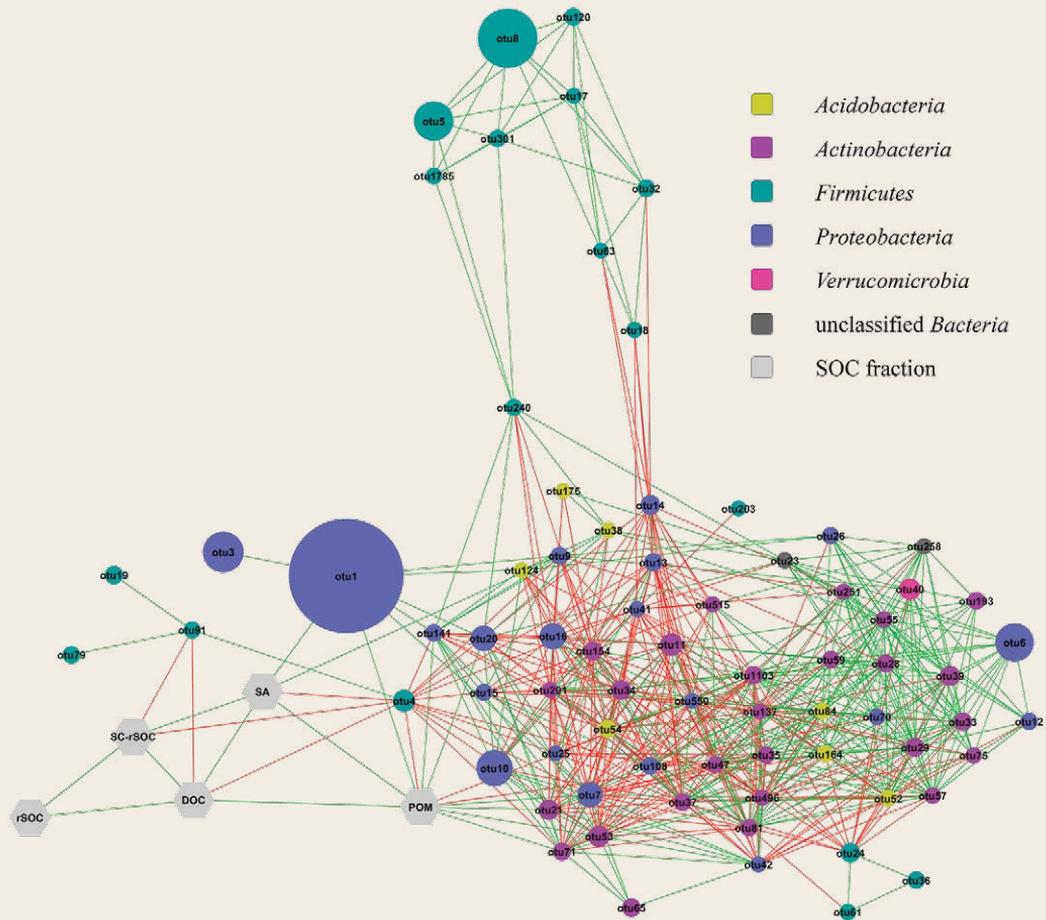
4.2.5 | SOIL COMPACTION AND SEALING

Compaction is a soil physical degradation process affecting soils in agricultural and urban areas. Soil compaction is associated with loss of crop productivity but can also affect grasslands and tree plantations. This degradation process can occur at the soil surface or at the subsurface affecting root elongation and water and air exchanges. According to the *Status of the World's Soil Resources* (FAO and ITPS, 2015) the status of soil compaction around the world is highly variable according to each world region and varies from fair to poor, whilst the trend in most regions is classified as deteriorating.

Soil compaction decreases the volume of macropores and, consequently, alters soil structure, penetration resistance, soil pore distribution and bulk density. As a result of this decrease, the proportion of water and air volumes are modified, affecting oxygen and carbon dioxide concentrations as well as redox potentials (Figure 4.2.5.1). These changes affect faunal activity and cause a decrease in biomass and population density (abundance; Beylich *et al.*, 2010). Increase in penetration resistance and bulk density affect the burrowing action of macrofauna, especially of burrowing species such as earthworms. They also impair the action of ecosystem engineers by reducing available habitats and access to water and oxygen. The reduction in macropore volume and its consequences on other soil physical attributes also affects the habitable space for mesofauna. Soil compaction and soil biodiversity are interdependent, compaction affects soil biodiversity, but soil organisms can counteract compaction. Ecosystem engineers can counteract the effects of soil compaction and contribute to the regeneration of compacted zones with time (Turbé *et al.*, 2011).

Soil microbial activity and biomass are also affected by soil compaction. The effects of soil compaction on soil microorganisms and microbial processes are complex and depend on many factors (Nawaz *et al.*, 2013). Changes in water and air volumes, waterlogging and redox potential affect microbial processes which, for example, cause changes in carbon gas effluxes (CO_2 and CH_4) and net nitrogen mineralization processes..

Soil biodiversity and soil fauna may be more affected by soil compaction than plant growth and plant yield. Nevertheless, many threshold values of soil compaction can be found for soil physical processes affecting plant growth but are non-existent for soil biota and biological processes. The increasing need of soil protection and protection of soil functions demands the need for the identification and development of threshold values related to soil organisms and biological processes (Beylich *et al.*, 2010).

The migration of rural populations to urban environments during the last two centuries has led to the growth and expansion of cities worldwide, impacting the landscape and soil resources. Urbanization has caused an increase in soil sealing, which is defined as the permanent covering of the soil surface by an impermeable material impeding changes between above-ground and below-ground environments (Turbé *et al.*, 2011). Soil sealing can be considered as total soil loss, permanently affecting many soil functions related to water production and regulation, food production, biodiversity and climate regulation.

With the exception of the South West Pacific and Southern African regions, the status of soil sealing in the other regions varies from fair to very poor and the main trend is classified as deteriorating (FAO and ITPS, 2015). The South West Pacific and Southern African regions are classified as good but with a deteriorating trend.

Natural soil sealing occurs by soil crusting, which impedes soil infiltration, but most sealing occurs due to anthropogenic activities related to urbanization. Sealed soils are considered non-functional, reducing infiltration and increasing runoff, decreasing organic matter input, and isolating the soil from the above-ground environment. Consequently, water and gaseous exchanges are affected, as well as nutrient cycling related to organic matter dynamics. Soil biota can survive with residual water and organic matter after recent sealing, but when exhausted, bacteria can enter an inactive state and soil fauna may disperse or die off (Turbé *et al.*, 2011). Reduction in soil carbon and nitrogen contents, soil respiration, changes in soil physico-chemical attributes and enzymatic activity negatively influence microbial activity. Most sealed soils have their topsoil removed, causing a reduction in soil organic matter, increasing moisture stress, and creating an alkaline environment and poor ventilation affecting soil biota and their activity (Piotrowska *et al.*, 2015).

Fragmentation of native ecosystems and the implementation of green areas with non-native species cause impacts on soil organisms and their activity (Scalengle *et al.*, 2009). Sealed ecosystems under cities are replaced by pavement and concrete infrastructures, isolating small to medium areas with native ecosystems and/or green areas with exotic species, such as parks, affecting above and below-ground biodiversity connectivity (see the earlier section on urbanization). The use of concrete and asphalt pavements to seal the soil elevates the soil temperature, which exerts pressure on soil biota and biological processes. Overall, soil sealing affects the hydrological cycle, nutrient and carbon cycling, climate, and microclimate regulation, resulting in the loss of habitats for soil organisms, soil biodiversity and all services and functions with the exception of the capacity to support infrastructure (FAO and ITPS, 2015).

Undisturbed aggregates

Soil compaction

Aggregate

Water film →

← Air
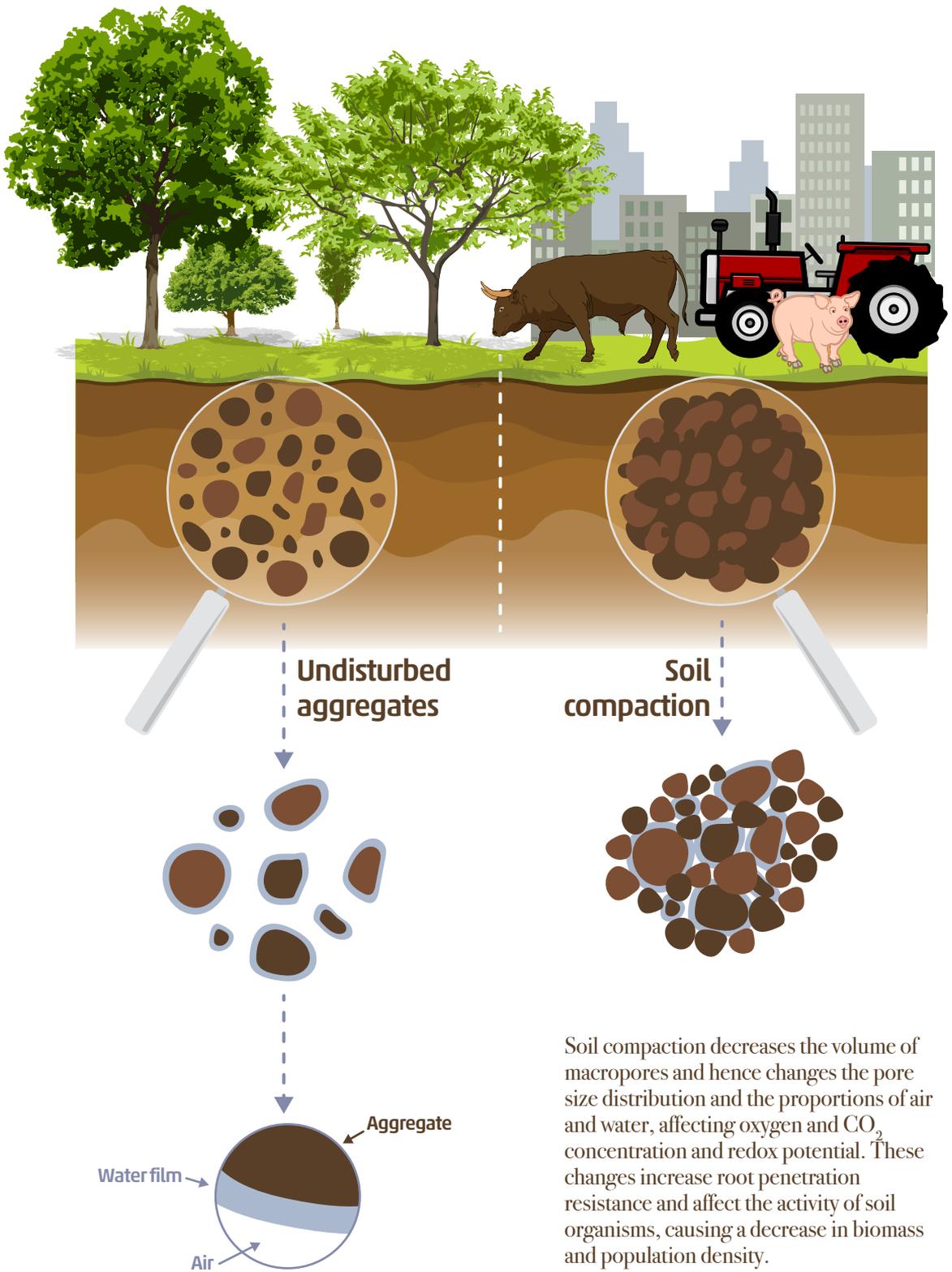
Soil compaction decreases the volume of macropores and hence changes the pore size distribution and the proportions of air and water, affecting oxygen and CO_2 concentration and redox potential. These changes increase root penetration resistance and affect the activity of soil organisms, causing a decrease in biomass and population density.

Figure 4.2.5.1 | Soil compaction

Compaction and sealing adverse impacts on soil.

4.2.6 | SOIL ACIDIFICATION AND NUTRIENT IMBALANCES

Soil acidification

The acidification of ecosystems is a natural process that is driven by the metabolic activity of soil organisms and plants through the ecosystem succession and that is also linked to the build-up of soil organic matter (Delgado-Baquerizo *et al.*, 2019). This natural acidification drives changes in the biodiversity and abundance of soil communities across decades, centuries, millennia or even millions of years of ecosystem development (Delgado-Baquerizo *et al.*, 2019). The resulting acidophylic or acid-tolerant species present in these soils are adapted to these conditions but may still be threatened by pH changes in the soil, be they positive (for example, liming to raise pH) or negative (such as acidification due to acid rain). This process is, however, very different from the acidification of ecosystems that has been historically linked with the emission and further deposition of pollutants since the beginning of the industrial revolution and, later on, the green revolution (Bobbink *et al.*, 2010; Greaver *et al.*, 2012; Tian and Niu, 2015).

The industrial revolution started a period in which massive amounts of sulphur- and nitrogen-rich compounds derived from the burning of fossil fuels were emitted to the atmosphere, and were deposited back to the soil as sulphuric and nitric acids dissolved in rainwater, or acid rain (Fowler *et al.*, 2013). This impacted the vitality and structure of forests across very wide areas (Menz and Seip, 2004), with unknown but potentially negative consequences for the abundance and biodiversity of soil organisms that thrive in these soils (Lv *et al.*, 2014). Due to abatement policies, the amount of oxidized nitrogen, and particularly sulphur, that is emitted to the atmosphere has now been reduced in many world regions, particularly in Europe and in the United States of America, although it remains as a problem in areas of China and India, which are also amongst the less studied areas in terms of soil biodiversity (Lv *et al.*, 2014; Menz and Seip, 2004). The emission and further deposition of reduced nitrogen compounds derived from agricultural practices and three-way catalytic converters is still a threat to ecosystems in many regions of the world and is also a main agent of acidification, given the ability of NH_4^+ to release protons in the soil solution (Forest *et al.*, 2013). Other human-induced causes of soil acidification include acid mining for the extraction of minerals.

The effects of soil acidification on soil organisms can be direct, via alterations of the physicochemical environment in which the soil organisms thrive. For example, many bacterial taxa are known to be highly selective for the soil pH range in which they can grow, which is typically associated with the importance of pH for the regulation of their metabolic activity (Fierer *et al.*, 2009; Lauber *et al.*, 2009). The greatest abundance and diversity of active bacteria is typically found in soils with pH around 7 (Lauber *et al.*, 2009; Ochoa-Hueso *et al.*, 2018). Moreover, acidification is known to hamper the activity of soil microorganisms involved in N transformations, such as mineralization of organic N and biological N_2 fixation, while low soil pH promotes the production

of N_2O, a potent greenhouse gas, during nitrification and denitrification (Granli and Bøckman, 1994), with potential consequences for the global climate. Indirect effects of soil acidification on soil organisms can operate by different mechanisms: the first are changes in plant community composition, which may result in the complete alteration of soil communities which very typically depend on plant litter inputs and rhizodeposits as the main carbon source supporting brown food webs; thus changes in plant community structure can cascade through the whole soil food web (that is, across trophic levels) by altering the abundance, composition and activity of those soil organisms that are at the base of the food web. A second set of mechamisms involves changes in soil pH, which determine changes in the solubility of multiple elements in soil, including trace elements, many of which are typically needed in low concentrations but that are toxic at high concentrations (Stevens *et al.*, 2011). Soil acidification results in the leaching of base cations, particularly in poorly buffered soils, making them unavailable for soil organisms, including microbes (Velthof *et al.*, 2011). Acidification may thus lead to deficiencies of nutrients such as phosphorus, calcium, magnesium and molybdenum, and a release of toxic compounds, including aluminum, iron, manganese and heavy metals that are immobile at higher pH values.

A global meta-analysis showed that the decrease in soil pH in response to the addition of mineral N was more evident in grasslands, whereas boreal forests were more resilient to the N-induced soil acidification (Tian and Niu, 2015). This suggests that the consequences of acidification for soil organisms may also be more important in acidic and poorly buffered soils, such as those from many natural and semi-natural grasslands in the United Kingdom of Great Britain and Northern Ireland and in central Europe, which have seen a dramatic loss of their plant biodiversity in the last 150 years (Stevens *et al.*, 2004), with likely mirroring consequences for the abundance and biodiversity of below-ground communities. In agroecosystems, acidification considerably reduces soil fertility, affecting microbial transformations, which may ultimately cause depression of crop growth, and yields (Marschner, 1995; Bolan, Bolan *et al.*, 2003).

The effects of acidification on soil organisms are, however, often very difficult to separate from those of the direct effects of the agents of acidification. For example, a study carried out in a calcareous semi-arid shrubland in central Spain (pH ~7.5) showed that mineral nitrogen additions up to 20 kg N/ha annually increased the abundance of soil microarthropods, a response attributed to an incipient soil acidification (Ochoa-Hueso *et al.*, 2014). Beyond that load, the addition of nitrogen resulted in a decrease in soil faunal abundance, attributed to the negative effects of excessive nitrogen, particularly to high concentrations of ammonium, which is known to be toxic for many soil organisms. This response was driven by collembolans, the most abundant group in those soils.

Nutrient imbalances

Ecosystem productivity is co-limited by the availability of N and P at the global scale (Elser *et al.*, 2007). Thus, changes in the absolute and relative availability of key essential nutrients such as N and P can greatly affect soil biodiversity and their functioning (Elser *et al.*, 2018). Nutrient imbalances occur when one or more essential nutrients are in short supply in relation to other essential nutrients. This situation is now widespread in soils worldwide due to the above mentioned increase in the availability of mineral N and P from polluting sources (atmospheric N deposition, runoff water) and in agroecosystems due to excessive use of mineral fertilizers. This increase in the availability of essential nutrients has consequences for plant growth and microbial decomposition that can cascade to more complex effects on soil food webs. This is because energy and nutrient imbalances (typically C:N, C:P or N:P, but also N:K and so on) between consumers and their resources strongly constrain nutrient cycling and limit consumer reproduction and growth (Andersen *et al.*, 2004; Frost *et al.*, 2005; Person *et al.*, 2010). The effects of nutrient imbalance on soil organisms and food webs can be direct (in the case of soil organisms such as bacteria and fungi that take up their nutrients directly from the soil solution) or indirect, via alterations of the nutritional content of primary producers and their litter inputs, with cascading consequences for both green and brown food webs.

The ecological stoichiometry theory has been applied to study the balance of energy and chemical elements such as C, N and P in ecological interactions (Sterner and Elser, 2002). This theory can help us to better understand trophic interactions by analyzing the imbalances in the relative supplies of key elements between organisms and their resources, yet the mechanisms that control elemental stoichiometry in different taxonomic groups and the effects of nutrient supply imbalances are not yet clear. It has been observed that the concentration of major elements in the soil can be high or low, without this altering the natural ecosystem functioning. However, what is really important is the degree of C:N:P nutritional imbalance that affects biodiversity, causing a cascade of unknown ecosystem effects. The main mechanism by which nutritional imbalance affects soil biodiversity is associated with the growth capacity of organisms. Elser *et al.* (1996) proposed the growth rate hypothesis that postulates that cellular stoichiometry varies according to growth rate due to increased allocation to P-rich ribosomal RNA to support rapid growth. To date, this hypothesis has been a powerful tool for understanding variation in biomass C:N:P ratios in microbes and small consumers, important components of soil biodiversity (Elser *et al.*, 2018) and the base of the food webs.

Additionally, C:N:P ratios for different organisms have been proposed (Table 4.2.6.1) as a reference of nutritional needs and immobilization capacities of organisms in different ecosystems (Redfield, 1958; Elser *et al.*, 2000; Cleveland and Lipzin, 2007; Zhang and Elser, 2017). Deviations from this elemental stoichiometry reflect a nutritional imbalance and therefore a greater energy investment to acquire the limiting nutrient; this energy investment may not necessarily be feasible for all microorganisms and small consumers, thereby causing a decrease in soil biodiversity.

Table 4.2.6.1

C:N:P ratios for different organisms

Organism	C:N:P	Reference
Ocean phytoplankton	106:16:1	Redfield (1958)
Terrestrial plants	968:28:1	Elser *et al.* (2000)
Aquatic plants	307:30:1	Elser *et al.* (2000)
Earthworm	127:26.5:1	Marichal *et al.* (2011)
Soil bacterial biomass	60:7:1	Cleveland and Lipzin (2007)
Fungal biomass	250:16:1	Zhang and Elser (2017)

In soil, organic matter represents the main energy and nutrient input via decomposition (solubilization, depolymerization and mineralization). Downing and McCauley (1992) established that P limitations occur when N:P exceeds ~30. In a study carried out in a calcareous semiarid grassland in the Chihuahuan desert in northern Mexico, Hernández Becerra *et al.* (2016) found that land use change (grassland to alfalfa crop) modified soil microbial N:P stoichiometry from 5.3 in grassland to 33.2 in alfalfa crop, increasing soil acidification (pH from 9 to 7) and reducing bacterial diversity (12 and 9 phyla respectively). Interestingly, they found that there were no OTUs shared between the agricultural plot and the native grassland, which may indicate a change not only in taxonomic diversity but also in functional diversity, associated with nutrient imbalance. However, further research is needed to better understand how to reduce nutrient imbalance in human managed ecosystems.

4.2.7 | POLLUTION

A recent study identified environmental pollution as the largest cause of premature death in the world, killing more people than AIDS, malaria, and tuberculosis combined, and accounting for one in four deaths in the poorest countries (Landrigan *et al.*, 2018). Chemical pollutants are also further known to affect wildlife species and ecological communities including those in the soil. This can lead to local- or regional-scale losses in biodiversity that can be explicitly linked through evidence to impacts on ecosystem functions and associated services (Cardinale *et al.*, 2012; Hayes *et al.*, 2018). These real impacts shape the public debate on the use and safety of chemicals, fueling concerns even in those cases where limited impacts may actually exist.

Recognition of the effects of chemical contaminants on ecosystems underpins a desire to improve the chemical condition of our environment, encouraging the mitigation of some but not all of these effects. Landmark policies on abatement of acid rain, nutrient management strategies, control on the use of certain persistent organic pollutants (POPs), biocides and pesticides and improvement in wastewater treatment, as well as

economic and cultural shifts affecting industrial sectors, such as energy production, transportation and metal processing, have changed the type and amount of chemicals entering the soil environment, often reducing loads and mitigating impacts. The aim of these risk-based policies is to restrict inputs to levels below those expected to cause biological effects on soil species populations and humans.

By preserving structure, the assumption is that function will be protected, especially for those ecosystem processes for which there is recognized functional redundancy.

Despite management effort, legacies of past activities or unmanaged chemical use and release, poor governance and gaps in knowledge lead to chemicals still being released into the soil environment. Progressive advances in chemical analysis methods for soils allow increasingly accurate measurements of soil contaminants supporting monitoring applications. Monitoring programs and the development of concepts such as 'pesticidovigilance' – the practice of monitoring the effects of pesticides after approval for use – are developing, placing greater emphasis towards post-approval assessment to allow regulatory decisions to be refined, and to make the trade-offs between environmental costs and intended food security more explicit (Milner and Boyd, 2017).

To date the majority of large-scale regulatory contaminant monitoring programs are designed to assess the chemical status of water bodies. The initiatives are directed to support major policies such as the Water Framework Directive in Europe. Nutrients, pesticides and trace elements are the primary focus. With few national frameworks for soil protection published and implemented, large-scale contaminant monitoring of soil status at the national scale is limited to programs in only a few countries (Gardi *et al.,* 2013; Hassanin *et al.*, 2005; Spurgeon *et al.*, 2008). Most research instead remains the domain of academic groups conducting small pilot programs. There are literally thousands of such studies describing concentration of macronutrients, trace elements, pesticides, plant protection products and biocides, other persistent organic pollutants (POPs), industrial chemicals, pharmaceuticals, veterinary medicines, nanomaterials and recently plastics in soils collected from individual sites, transects and local and regional surveys. If compiled together with any associated georeferenced and other meta-data, these studies would present a fantastic resource by which to study the characteristic of soil contaminant loads across biomes, continents, countries, landscapes and land-uses across the world. However, lack of consensus on terminology and analytical methodologies make such a task a non-feasible activity. 4.2.7.1 shows that chemicals that reach soils can come from a number of sources.

Box 4.2.7.1 | Main Sources of contaminants that impact soil biodiversity

Direct inputs to agricultural land occur as a result of the widespread use of pesticides and fertilizers. The complexity of such inputs is rising. For example, in the United Kingdom of Great Britain and Northern Ireland there has been approximately a 50 percent rise in the average number of pesticide active ingredients applied to arable crops (from 11 in 2000 to 17 in 2015; FERA). Fertilizers are also well known to have resulted in the eutrophication of soils leading to changes in plant, invertebrate and microbial community structure (Rowe *et al.*, 2014; Stevens *et al.,* 2010). The use of alternative sources of fertilizers, such as manure or sewage sludge can also be a source of soil contaminants, such as trace elements and emerging contaminants like pharmaceuticals, veterinary drugs and antibiotics and plastics, if their quality is not controlled.

Diffuse inputs of contaminants include polycyclic aromatic hydrocarbons (PAHs), trace elements (such as zinc) and also emerging contaminants, such as rare earth elements to soils, especially in urban areas. Fossil fuel burning from domestic heating, cooking, vehicle emissions, and tire and brake wear are also important sources of these types of contaminants.

Raw material extraction can result in the wide-scale pollution of soils surrounding mining sites. Mining is responsible for soil pollution in many countries. Coal, oil, metal and ore extraction all lead to the production of waste materials (slags and waste and drilling fluids) that transfer contaminants including hydrocarbons and trace elements to soil. Even though the impacts of mining have been known since the first pollution investigations, prevention practices are still poorly managed in many countries, resulting in soil contamination and pollution.

Industrial and transport activities result in the direct release of contaminants to soil. For example, the United Kingdom of Great Britain and Northern Ireland harbours a significant legacy of contamination resulting from past industrial activities. An audit report estimated that there are 325 000 contaminated land sites in the United Kingdom of Great Britain and Northern Ireland (House of Commons Environmental Audit Report). This level would be typical for a post-industrial country of similar size and population density.

There is good evidence to suggest that deposition of some major contaminants, such as trace elements and organic contaminants associated with combustion products (for example, polycyclic aromatic hydrocarbons, PAHs) have significantly declined in developed areas (such as Europe) (Cape *et al.,* 2003; Dore *et al.*, 2014). However, these are increasing in rapidly developing economies in Asia, South America and Africa (Yu *et al.*, 2017).

Chemicals present in waste streams can also reach land. More than 10 million tonnes of biosolids, composts and digestates are applied to the United Kingdom of Great Britain and Northern Ireland land each year. These wastes can contain established contaminants, such as PAHs and polychlorinated biphenyls (PCBs). However increasingly, the presence of newer POPs, such as polybrominated diphenyl ether (PBDEs), fluorinated organics (PFOS, PFOA), chlorinated naphthalenes, veterinary medicines, antibiotics, human pharmaceuticals and nanomaterials in these wastes is being recognized.

With large areas of the world involved in armed conflicts in the present and recent past, chemicals associated with military applications, such as energetic compounds (explosives and propellants), trace elements and oil products occur as contaminants in soil, where they can have impacts on soil species and processes (Kuperman *et al.*, 2017, 2018).

Building on recent observations of the widespread occurrence of plastics in the marine environment, similar studies in soils have also found the widespread presence of large- and small-scale (microplastic and nanoplastics) plastic debris in terrestrial ecosystems (Horton *et al.,* 2017).

To improve our knowledge of the contamination status of soils and the impacts on species and processes, progress is needed in the characterization of contaminant presence and concentration and in understanding the resulting effects in realistic exposure scenarios (for example, long-term exposure to mixtures for interacting species). Progress in analytical methods means that the detection of ultra-low concentrations of inorganic and organic contaminants in soil is now possible. What is less well understood is, first, how available these contaminants are for uptake by organisms, and secondly whether

exposure to these concentrations over extended time as mixtures has any impacts. To assess bioavailable exposure, significant progress has been made in understanding real bioavailability. Concepts such as the biotic ligand model as applied for trace elements (Thakali *et al.*, 2006a; Thakali *et al.*, 2006b) and chemical activity as applied for organic contaminants (Mayer and Holmstrup, 2008; Schmidt *et al.*, 2013) have advanced the ability to understand pollution exposure in specific soil chemistry contexts (such as pH and organic carbon content).

Contaminants represent only one aspect of the anthropogenic pressure to which ecosystem may be exposed. As many of these pressures can lead to changes in ecosystem structure and function, it can be difficult to attribute those effects of contaminant perturbations to those caused by other stressors. The effect of some contaminants is known based on the reasons for their application; examples include nematocides, molluscicides and carbamate insecticides for earthworm control on golf courses that target specific invertebrate groups. Other established examples of pressures include the widely reported impacts of copper fungicides on soil fungal and invertebrate communities in vineyards (Hayes *et al.*, 2018) and the effect of energetic materials on soil communities. Thus, a study by Kuperman *et al.* (2014) showed that, although overall soil microarthropod or nematode communities abundance was not affected by exposure to polynitramine EM CL-20 (China Lake compound 20), greater sensitivities were found for predatory mesostigmatid mites and predatory nematodes. Similar greater sensitivity of predatory nematodes to chemical exposures compared with other trophic groups of the nematode community was observed in studies with copper and p-Nitrophenol by Parmelee *et al.* (1993). In the same microcosm assay, total microarthropod numbers were reduced by 50 percent in the 30 mg/kg 2,4,6-trinitrotoluene (TNT) treatment compared with numbers in control oak-beech forest silt loam soil (Parmelee *et al.*, 1993). Adverse effects of exposure to EM were also determined in single-species toxicity tests with earthworms (Simini *et al.*, 2003), enchytraeid worms (Kuperman *et al.*, 2006, 2013) and springtails (Phillips *et al.*, 2015). These cases provide an illustration of the cascade of possible effects from individual species to communities and functions. Measurements of functional endpoint have shown that the activities of soil microorganisms, which are critical to terrestrial biogeochemical cycles and are key to sustaining the functioning of the terrestrial ecosystems, were inhibited in EM-polluted soils. Basal respiration was the more sensitive endpoint for assessing the effects of nitroaromatic compounds on microbial activity in a sandy loam soil, whereas substrate-induced respiration and microbial biomass were more sensitive endpoints for assessing the effects of nitroglycerin (Kuperman *et al.*, 2017). Litter decomposition was inhibited in soil polluted with dinitrotoluene or nitroglycerin (Kuperman *et al.*, 2018).

Understanding of the effects of long-term exposure on species is still hindered by the lack of methods to assess soil community change over extended time-scales. In ecotoxicology, short-term single species testing still dominates. Further, even with our increasing knowledge, the lessons learned from the cases where contaminants have been found to impact species have taught us that unexpected effects can and do occur. The development of the ecosystem services approach provides an opportunity to establish protection goals that are more explicit in their aims, such as species or local population conservation for

certain species, or a broader protection of ecosystem function. Targeting of assessment to vulnerable species and landscapes can prioritize areas and critical receptors for protection, allowing efforts to be better tailored to meet specific goals.

Soil contaminants are of high concern to the public; as a result, their sustainable management is a high-profile and demanding task. What challenges regulators is that contaminants are found ubiquitously in soils in all biomes and regions. These contaminants may, however, have no effect if they do not reach a threshold for toxicity. Defining these thresholds and making them robust for variation in soil types, mixtures of soil contaminants, extended exposure time and for multiple interacting soil species remains a challenge. Risk assessment tools allow having a balance between different exposure scenarios and the ecotoxicological costs. This balance needs to be continuously scrutinized as evidence changes and tools develop.

It is also important to note that the same approaches to the management of soil pollution are not applied worldwide. Recent years have seen the large-scale transfer of heavy industry and an associated outsourcing of pollution to the newly industrialising economies. In these countries, the issues being faced with poor air quality and pollution of soil and water now mirror those faced by the industrialized world over the twentieth century. To meet growing demands for access to cleaner air, water and land, these nations can draw on the policies and practices applied in countries that have already gone through the process of improving pollution management and control.

The implementation of chemical management policies is recognized within the United Nations SDG 12: ensuring sustainable consumption and production patterns. However, rather than follow an incremental path, there is an opportunity to short-circuit the process of policy development by taking note of the lessons of past failures. By working to promote the idea that new opportunities for industrial innovation take place within an environment where both the costs and benefits are assessed, the negative impacts of pollution can be limited. This can help to mitigate the negative environmental impacts of chemicals, while gaining benefits from their use and creating an environment in which there is better understanding and advocacy of chemicals, based on an acceptance that occurrence is not risk, but in which real and recognized impacts can be identified and mitigated.

4.2.8 | SALINIZATION AND SODIFICATION

Soil salinization is a term used for the accumulation of salts in soils at a level that negatively impacts agricultural productivity, environmental health and economic welfare (Rengasamy, 2006). Generally, a soil is described as saline if the electrical conductivity measured in a saturated soil paste (ECe) is higher than 4 dS/m at 25°C (US Laboratory Staff, 1954), while 8-16 dS/m and levels >32 dS/m correspond to medium and hypersaline environments, respectively (Brouwer *et al.*, 1985). The USDA (1954) classification of salt-affected soils is shown in Table 4.2.8.1

Table 4.2.8.1

The USDA classification (USDA, 1954) of salt affected soils

Characteristics	Saline soils	Saline-sodic* soils	Sodic* soils
Electrical conductivity (EC_e) (dSm^{-1} at 25°C)	> 4	>4	<4
Exchangeable Sodium Percent (ESP)	<15	>15	>15

* Sodic soils are also alkaline if their pH is over 8.5

The Food and Agriculture Organization of the United Nations (FAO) estimates that globally over 830 M hectares of arable land are affected by salinization (Szabolcs, 1989; Martinez-Beltran and Manzur, 2005), corresponding to about 10 percent of the globe's arable land (Szabolcs, 1989). Salinization affects up to 3 M hectares of land in Europe, the 17 western states of the United States of America, more than 5 percent of the land in Africa, about a fifth of the arable land of western Asia, and 30 percent of the Australian land area (Chhabra, 1996; Rengasamy, 2006; Ladeiro, 2012), making it a world-wide environmental challenge. Of the global threats that collectively compromize about 10 hectares of arable land per minute (Griggs *et al.*, 2013), salinization contributes about 30 percent (Buringh, 1978). The distribution of salt-affected soils in the world is shown in Table 4.2.8.2.

Table 4.2.8.2

Worldwide distribution of salt-affected areas (Million ha) (Metternicht and Zinck, 2003)

Area	Saline Soils	Sodic Soils	Total	Percentage
Australia	17.6	340.0	357.6	38.4
Asia	194.7	121.9	316.5	33.9
America	77.6	69.3	146.9	15.8
Africa	53.5	26.9	80.4	8.6
Europe	7.8	22.9	30.8	3.3
World	351.2	581.0	932.2	100

Salt-affected soils are an important ecological entity in the landscape of any arid and semi-arid region in the world, and these naturally-occurring saline soils have an ecological value, as a habitat for hallophytic plant, animal and microbial communities. For

instance, microbial communities that inhabit ecosystems of naturally saline conditions are structured to function well at high salt concentrations, thus maintaining both high growth rates and other ecosystem processes (Rath *et al.*, 2019). Composition of microbial communities and their abundance are significantly different in saline and normal soils. However, bacterial diversity systematically decreases with an increase in soil salinity (Rath *et al.*, 2019), suggesting that ecosystem function provisioning grows less resilient at high salinities.

Salt accumulation in the surface soil is often found in agricultural areas in arid and semi-arid regions, where it is caused by irrigation with brackish or saline water in poorly drained soils (Allison, 1964) (Figure 4.2.8.1). In areas with shallow groundwater, evaporation can also lead to higher salt concentration in the soil surface layer (Rengasamy, 2006). In addition, soil salinization can be the result of changes in vegetation cover that alter ecosystem water balances. Saltwater intrusion from marine environments is also an important cause of soil salinization (Chandrajith *et al.*, 2014) that has resulted in the salinization of 53 percent of coastal regions in Bangladesh (Haque, 2006).

Figure 4.2.8.1 | Salinization

Salt efflorescences in an irrigated wheat field in Chaplanay in Kandahar Province, Afghanistan. The main effect of excessive salt content in the soil solution is the increase of osmotic pressure that impedes the absorption of water by

Soil salinization has direct impacts on plants and has subsequently been a research priority for crops for decades (Ayers and Westcot, 1976; Chhabra, 1996; Stevens and Partington, 2013). For instance, salt exposure is known to reduce crop yield under both greenhouse and field conditions in barley (Pal *et al.*, 1984; Richards *et al.*, 1987), wheat (Richards, 1983; Bajwa *et al.*, 1986), cotton (Meloni *et al.*, 2001; Soomro *et al.*, 2001), sugar cane (Choudhary *et al.*, 2004), rice (Bajwa *et al.*, 1986), maize (Bajwa *et al.*, 1986) and sugar beet (Ghoulam *et al.*, 2002). Crops and cultivars differ in their tolerance to salinity, and this is also modulated by environmental and soil factors. Furthermore, indirect consequences of salinization are ion imbalance and nutrient deficiency (Marschner, 1995), further aggravating the negative effects on plant productivity. Although crop resistance to salt exposure is a promising development (see for example, Bennett *et al.*, 2013), overall plant productivity will be impeded by salinization. Salinity not only adversely affects agricultural production but also influences the naturally occurring plant community assembly. These salt-affected lands are either devoid of any vegetation or have very meagre cover. High salt concentration in soil results in high osmotic potential, which affects the metabolic processes of vegetation. In these areas biosaline agriculture, which involves salt-tolerant conventional and non-conventional tree, shrub and herbaceous crops. has emerged as an alternative option. A restricted number of highly salt-tolerant plant species such as *Prosopis juliflora* (Sw.) DC, *Salvadora persica* L., *S. oleoides*, *Acacia nilotica* (L.) Willd., *Capparis decidua* (Forssk.) Edgew., *C. sepiaria* L., *Ziziphus nummularia* Aubrev., *Clerodendrum phlomidis* L. and *Maytenus emerginatus* were reported in alkali soils. Some herbaceous species *Desmostachya bipinnata* (L.) Stapf, *Sporobolus marginatus* Hochst., *Cynodon dactylon* (L.) Pers., *Chloris virgata* Sw., *Trianthema triquetra* Willd., *Suaeda fruticosa* Forssk. and *Kochia indica* Wight are prominent, particularly during the rainy season (Dagar *et al.*, 2001).

While the influence of salinity on plants has received much attention, less is known about the effects on soil microorganisms. Soil microorganisms are negatively affected by high salt concentrations, which are reflected in decreased microbial functions such as respiration and growth after salt exposure (Setia *et al.*, 2011; Rath *et al.*, 2015). However, microorganisms can counteract some of the negative effects of salinity through physiological adaptations. Organisms can adapt their physiology through the synthesis of osmolytes (Kakumanu *et al.*, 2014; Turk *et al.*, 2007) and changes in the composition of cell membranes (Turk *et al.*, 2007; Zhang *et al.*, 2008). In addition to physiological responses of the resident community, selection for more salt-tolerant species can lead to a shift in the taxonomic composition (Rath *et al.*, 2018). The changes in both community composition together with physiological adaptations manifest as an increased community salt tolerance in response to salt exposure (Rath and Rousk, 2015). As community salt tolerance increases, microbial process rates that were inhibited in response to acute salt exposure could, at least partially, recover. Fungi and bacteria are reported to be differently affected by salt exposure (Rath *et al.*, 2016; Kamble *et al.*, 2014). Generally, fungi are considered to be more resistant to short-term exposure to salinity (Rath *et al.*, 2016). However, it is unclear whether an increased resistance to short-term exposure

would indeed translate to a shift towards a more fungal-dominated system, as both increasing fungal (Kamble *et al.*, 2014; Wichern *et al.*, 2006) and increasing bacterial dominance (Sardinha *et al.*, 2003; Pankhurst *et al.*, 2001) in response to high soil salinity have been reported. While bacteria and fungi fulfill similar roles as decomposers of organic matter, they differ in the range of substrates they can decompose. Fungal and bacterial biomass also differs in their chemical composition (Six *et al.*, 2006) and nutrient content (Mouginot *et al.*, 2014; Stickland and Rousk 2010). Thus, shifts in the relative contribution of fungi and bacteria to decomposition in response to salinity could have implications for C and nutrient dynamics in soil (Strickland and Rousk, 2010; Schmidt *et al.*, 2011). Given these physiological constraints, microbial communities and subsequently ecosystem function are affected by salt in saline soil ecosystems.

Most general microbial metrics systematically decrease in more saline soils, including respiration (Rath and Rousk, 2015), soil enzyme activities (Batra and Manna, 1997) and microbial biomass (Batra and Manna, 1997; Wichern *et al.*, 2006; but also see Rath and Rousk, 2015). It is still debated whether soil bacteria and fungi are differently affected by soil salinity (Rath and Rousk, 2015), with reports of both higher fungal sensitivity (see for example Wichern *et al.*, 2006), and higher fungal resistance to salinity (see for example Rath *et al.*, 2016). However, when combined with plant litter, it was reported that fungal growth was maximum when bacterial growth was inhibited by the highest salinity, and fungal growth was lowest when the bacterial growth rate peaked at intermediate salt levels, which shows a competitive interaction between bacteria and fungi (Rath *et al.*, 2019a). Additionally, incorporation of an easily available and decomposable source of energy will improve the ability of microbes to withstand salinity (Mavi and Marschner, 2013) and reduce the negative effect of salinity on soil microbes. Microbes belonging to phylum Proteobacteria, Spirochaetes, Tenericutes, WS3 Plantomycetes, Bacteroidetes, Halobacteria, Nitriliruptoria, Gammaproteobacteria and Alphaproteobacteria were found more in saline soils (Canfora *et al.*, 2014; Rath *et al.*, 2019; Zhang *et al.*, 2019).

4.2.9 | FIRE

Wildfires are catastrophic events that occur in most biomes of the world. Burning results in dramatic changes in both functional structure and species composition of terrestrial ecosystems both above and below ground; this is unavoidably linked to shifts in their functioning and provision of key ecosystem services (Niklasson and Granström, 2000). The United Nations clearly recognized fires among the key threats to global biosphere sustainability (United Nations Forum on Forests, 2007). Although the total area burned each year has been decreasing due to the ongoing campaign to prevent burning for agricultural purposes and forest protection, around 340 million hectares are still damaged by fire every year (Willis, 2017). Moreover, fire frequency has remained stable or even increased, and up to one-sixth of the entire area of certain biomes like tropical

savannas and grasslands are annually burned, while boreal forests are burned every 15-25 years. This means that soils in large areas are subject to drastic thermal and toxic effects, which may result in severe damage to below-ground biota (Zaitsev *et al.*, 2016).

However, there are contrasting opinions about the real impact of fires on below-ground ecosystems and their functionality (Pressler *et al.*, 2019). On one side, numerous studies demonstrated that fires reduce soil biodiversity and biomass of below-ground organisms (see reviews by Zaitsev *et al.*, 2016 and Pressler *et al.*, 2019). At the same time, certain soils demonstrate remarkable resilience against burning due to various mechanisms like presence of microrefugia and patchiness (Gongalskym and Zaitsev, 2016). However, there is a general consensus that burning has negative implications on soil functions and ecosystem services such as organic matter mobilization and immobilization safeguarded by soil organisms (de Vries *et al.*, 2013).

With the exception of very few organisms, such as the so-called pyrophilous animals, most invertebrates reduce their abundance and biomass immediately after a fire event. Gongalsky *et al.* (2012) showed that organisms living deeper in the soil have higher survival rates after fire than surface-dwelling species. Sterilization effects of the fire on the topsoil is strongly modulated by fire intensity and may result in the almost complete extinction of soil bacteria and fungi in the topsoil immediately after burning. During recolonization, which is rather quick, due to large volumes of unburnt dead wood after the fire (Bastias *et al.*, 2006), there is an overall shift from bacteria-driven towards fungi-driven community with the associated distribution of organic matter flow in detrital food webs. Soil protist and microinvertebrate abundance and biomass are significantly decreased (by 25 percent on average) after single fires and in repeatedly burnt ecosystems rarely achieve 50 percent of the values typical for control (unburnt) sites. Similar reduction levels affect the taxonomic diversity of these groups. Due to their higher mobility macroinvertebrate abundances recover after a relatively short time (a few years) and are mainly limited by microhabitat availability in the burned sites. However, recent assessments clearly demonstrated that in boreal forest ecosystems, recovery of macroinvertebrate diversity and functional structure may require up to 75 years depending on the fire intensity and ecoregion, with longer times associated with higher latitudes (Zaitsev *et al.*, 2016). Overall, current research suggests that in the first years after a fire event soil biodiversity may decrease two-fold and may never recover to initial levels if fires are repeated (Zaitsev *et al.*, 2016).

Therefore, under such conditions, overall functioning of detrital food webs may be considerably reduced. However, there are certain compensatory mechanisms ensuring surprising functional resilience of below-ground communities after burning. Detrital food web modelling showed that soil protists, enchytraeids and associated micropredators form the most sensitive channels of element and energy flow below ground (Zaitsev *et al.*, 2017). Recent studies also showed remarkable stability of microbial community functioning and associated greenhouse gas emission levels due to compensatory mechanisms of overall microbial activity driven by changing physical and chemical soil properties and mobilization of additional carbon and nitrogen sources (Goncharov *et al.*, 2020).

The current level of knowledge about fire effects on soil biota brings us to a conclusion that in the short run, burning strongly reduces below-ground biomass and functioning. Both, however, may recover within a few years after burning. There are multiple and complex mechanisms behind the functional resilience of soil ecosystems and consequent sustainability of ecosystem services provision in fire-prone ecosystems. However, soil biodiversity requires more time for recovery and may never return to an initial level in case of repeated fires. Thus, multiple fires of predominantly anthropogenic nature, especially in agricultural lands, grasslands and some forest ecoregions, represent one of the greatest threats to below-ground biodiversity and stability of soil ecosystems.

4.2.10 | EROSION AND LANDSLIDES

Erosion involves the detachment, transport and deposition of soil particles through water or wind. Erosion, the natural process that has shaped the earth's landscape, is now one of the main drivers of degradation of the upper layers of the soil due to its acceleration by anthropogenic activities (such as agriculture, deforestation and soil sealing) (Figure 4.2.10.1).

Landslides are gravitational movements of a mass of rock, earth or debris down a slope. Landslides displace great volumes of soil, and can be triggered by natural processes (for example, heavy or prolonged rainfall, earthquakes, volcanic eruptions, rapid snowmelt and permafrost thawing), but the likelihood of their occurrence is magnified by human actions (such as slope excavation and deforestation).

At global scale, approximately 36 Pg (1 x 10^{15} g) of soil are estimated to be eroded each year by water (Borrelli *et al.*, 2017). Soil loss due to wind erosion in arable land has been estimated at about 2 Gt (1 x 10^{12} g) (FAO and ITPS, 2015). There are no global estimates of yearly soil losses due to landslides as they are very variable in volume (from a few cubic metres to several cubic kilometres), depending on the area and depth of the layers involved. Over the last years, climate changes have significantly affected the frequency of erosive and landslide occurrences. As the possibility of extreme climate events has increased, so has the concern about their negative impact on soil biodiversity (Orgiazzi and Panagos, 2018).

Erosion creates both degraded-eroded and enriched depositional environments. Following landslides, the upper parts of the affected slopes are usually stripped of soil, while fallen soil and mineral masses are accumulated and mixed at the foot of the slopes. In both cases, the impact on soil organisms can be direct or indirect. As a direct effect, the inhabitants of the upper soil layers may be eliminated or displaced far away from their original environment.

Water erosion comprises several processes (such as water splash and sheet, rill and gully formation) that selectively affect different soil species. Landslide effects on soil organisms may be considered similar to those caused by water erosion. Splash detaches soil particles and degrades soil microstructure (due to microaggregate disruption and soil pore clogging), worsening the quantity and quality of habitable soil microhabitats. Habitat destruction affects microorganisms and micro-invertebrates living near the soil surface, while endogeous meso- and macro-invertebrates (such as earthworms) are vulnerable to clogging of the burrows they excavate (Baxter *et al.*, 2013). Sheet, rill and gully erosion result in mid- and long-distance transport of great quantities of soil and associated soil biota from degraded-eroded to depositional sites. In many cases, the depositional sites are favoured by the arrival of nutrients and biological propagules, but soilborne plant pests, such as plant-pathogenic nematodes (Chabrier and Quénéhervé, 2008), can also be spread.

The predisposition of various taxa to be transported by wind depends on their size, abundance and location in the vertical soil profile. Billions of tonnes of desert dust, made of very fine soil particles and astonishing quantities of soil organisms, are transported yearly across continents (Griffin, 2007). In the case of the microscopic and very abundant soil prokaryotes (that is, bacteria and archaea), every year up to 4.3×10^{11} per square metre of prokaryotic cells may be wind-blown from the upper layer of forest, 1.6×10^{12} square metres per year from pasture, and 1.9×10^{12} square metres per year from arable soils (adapted from Torsvik *et al.*, 2002). A rich variety of soil animals has been found in the aeroplankton, including nematodes, rotifers, collembolans, tardigrades and mites (Nkem *et al.*, 2006; Ptatscheck, 2018). Effects of wind erosion on transport and dispersal of soil organisms has been studied in particular for soil nematodes, whose eggs and larvae can be transported as far as 40 kilometres from their origin (Carroll and Viglierchio, 1981). Among soil micro-arthropods, collembolans and mites are the most abundant. Collembolans have been found in air samples taken at a height of 3 000 m (Glick, 1939) and wind dispersal is proposed as a significant dispersal route for both epigeic and endogeic springtails and oribatid mites (Querner *et al.*, 2013; Lehmitz *et al.*, 2011).

Soil and soil organism mobilization by wind has net negative effect on the biodiversity and functioning of the eroded soils worldwide. That vulnerability is particularly relevant in some ecosystems, such as drylands, where plant cover is very scarce and soil life is carbon-limited. Drylands are protected against erosion by soil biological crusts formed by bacteria (cyanobacteria and heterotrophic), algae, mosses, liverworts, fungi and lichens (Maestre *et al.*, 2011). These biological structures are crucial not only for stabilization of the underlying soil, but also the regulation of water cycles and provision of nutrients to the rest of the below-ground communities. Biological crusts are extremely vulnerable to physical disturbance by human activities (such as trampling, cattle raising and off-road vehicles), to the point that their disruption is associated to growing desertification (Pointing and Belnap, 2012)

Overall, soil microorganisms and microscopic eukaryotes show a very high vulnerability to wind and water transportation (Finlay, 2002). This phenomenon, along with the ability of several soil organisms to survive many weeks in seawater and freshwater (Coulson *et al.*, 2002), suggest that wind and water erosion may be a key vector in long-distance colonization of soils and are at the basis of the ubiquitous geographic distribution of many soil organisms (Figure 4.2.10.2).

Erosion and landslide events can lead to changes in soil physico-chemical properties, which indirectly affect living communities (Baxter *et al.*, 2013). In fertile soils, most of the organic matter is close to the soil surface in the form of decaying plant litter, or associated to fine (silt and clay) soil particles (Plante *et al.*, 2006). Both wind/water erosion and landslide events remove the fine organic particles in the soil, leaving behind large particles and stones (Lal, 2001). Erosion of the topsoil layer (0-10 cm) significantly decreases soil organic matter to the point that soil removed by either wind or water erosion is 1.3 to 5.0 times richer in organic matter than the soil left behind (Pimentel and Kounang, 1998). Since across biomes as well as at local scales, soil microbial and faunal abundance positively correlate with soil organic matter content (Schnürer *et al.*, 1985; Fierer *et al.*, 2009), erosion is bound to lessen soil biota biomass.

Effects of landslides and erosion on organisms living in deeper soil layers are less clear. There is some evidence that soil impoverishment (reduction of soil organic carbon inputs) results in reduced below-ground food webs as they are predominantly regulated by bottom-up forces (Moore and de Ruiter, 2000). However, there is no conclusive proof of the correlation between carbon content and soil faunal and/or microbial composition, which might be more dependent on other soil properties (such as soil pH and texture) (Cole *et al.*, 2005; Fierer and Jackson, 2006; Johnson *et al.*, 2003).

Both the richness and the abundance of soil organisms may have different impacts on soil susceptibility to erosion. For instance, the dense network of mycorrhizal fungi present in a grassland may reduce the amount of soil loss by rain or windstorms (Burri *et al.*, 2013). On the other hand, at the opposite size extreme, the extensive excavation activity of some mammals (such as moles and pocket gophers) may weaken soil structure and thus accelerate erosive processes (Reichman and Seabloom, 2002). Other groups of organisms have less definitive effects on soil loss. For instance, the burrowing activity of earthworms can reduce erosion by favouring water infiltration (Shuster *et al.*, 2002). Simultaneously, surface cast production by some earthworm species may accelerate erosion, as this material can be easily transported by water or wind (Shipitalo and Protz, 1987). Therefore, the quantity of material loss in a living soil is different from that in an inert one. Only a few applied studies, mainly targeting plants (Allen *et al.*, 2016; Berendse *et al.*, 2015), have analyzed the effects that soil organisms have on erosion, and have estimated the potential amount of material eroded/preserved due to the presence of soil organisms.

Large-scale studies of soil erosion go through the application of models that permit estimation of the amount of soil loss by combining different factors (such as soil physico-chemical properties, rainfall erosivity and land management). So far, none of these models has included a biological factor accounting for the diversity of organisms living in the soils. Nevertheless, the possibility to include an "earthworm factor", taking into account both the abundance and richness of this group of soil organisms, has been recently proposed (Orgiazzi and Panagos, 2018). Therefore, the incorporation of a biological component is feasible, upon the availability of distribution data of soil biodiversity at large scales.

In recent years, the importance of soil organisms in shaping large-scale processes, such as climate regulation through impact on the carbon cycle, has been recognized (Luo *et al.*, 2016). The derived models benefited from the integration of biological factors, as confirmed by successive validations through ground data collection (Wieder *et al.*, 2015). A similar path is desirable for soil erosion and biodiversity, in order to ensure more accurate estimates of soil loss.
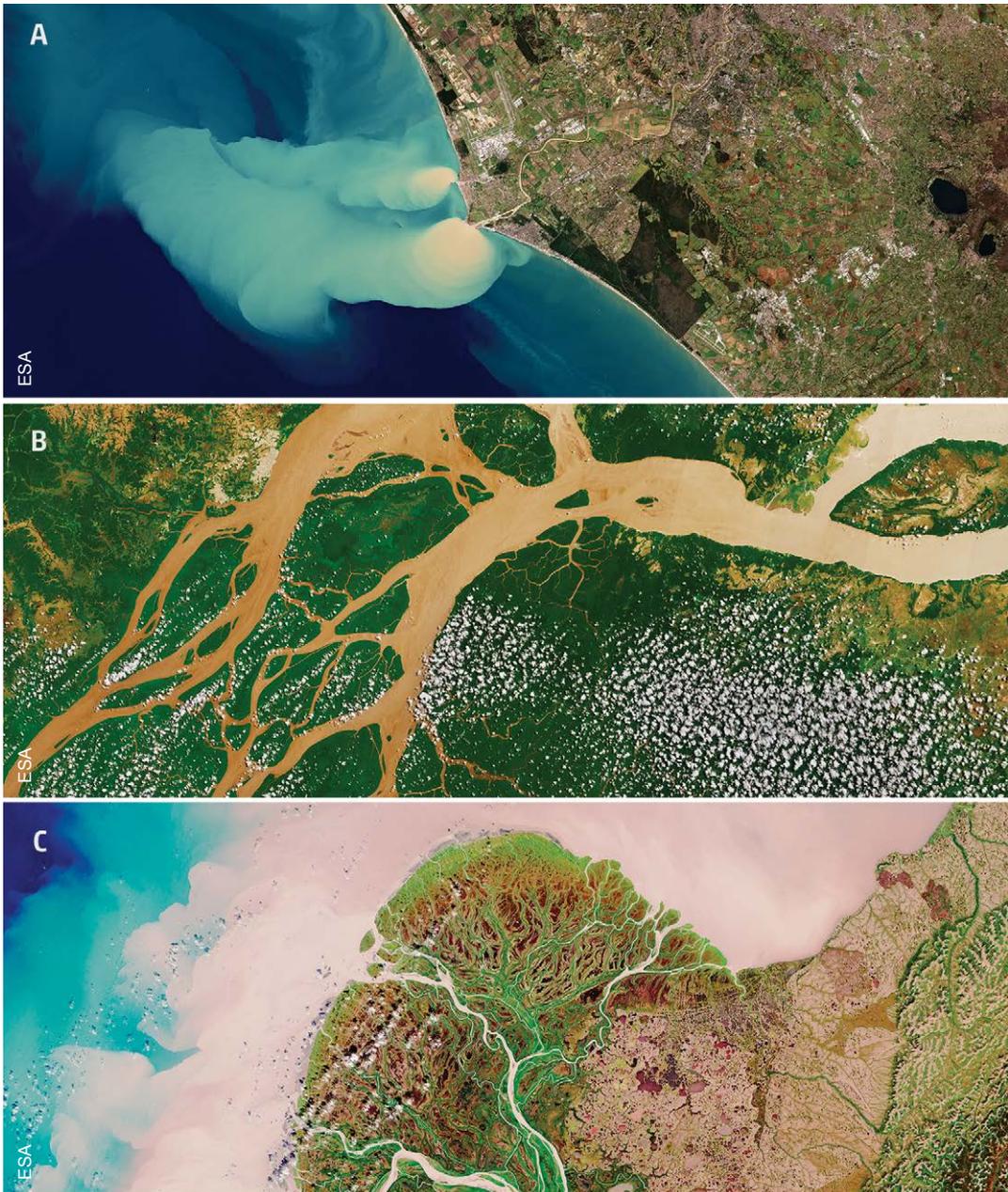
Figure 4.2.10.1 | Satellite images of natural and anthropogenic-enhanced water erosion

A. True-colour image captured three days after heavy rainfall in Rome and the surrounding area of Lazio, Italy. The image shows sediment gushing into the Tyrrhenian Sea, part of the Mediterranean Sea. Image captured on February 2019. **B.** Satellite image of northern Brazil showing the sediment-laden water that appears brown as it flows from the lower left to the open ocean in the upper right. Image captured on August 2017. **C.** Image of Yukon Delta in the US state of Alaska showing how the river branches off into numerous channels flowing to the sea. The sandy colour of these channels and of the coastal water illustrates how much sediment the river carries to the sea at this time of year. Image recorded on August, 2017.
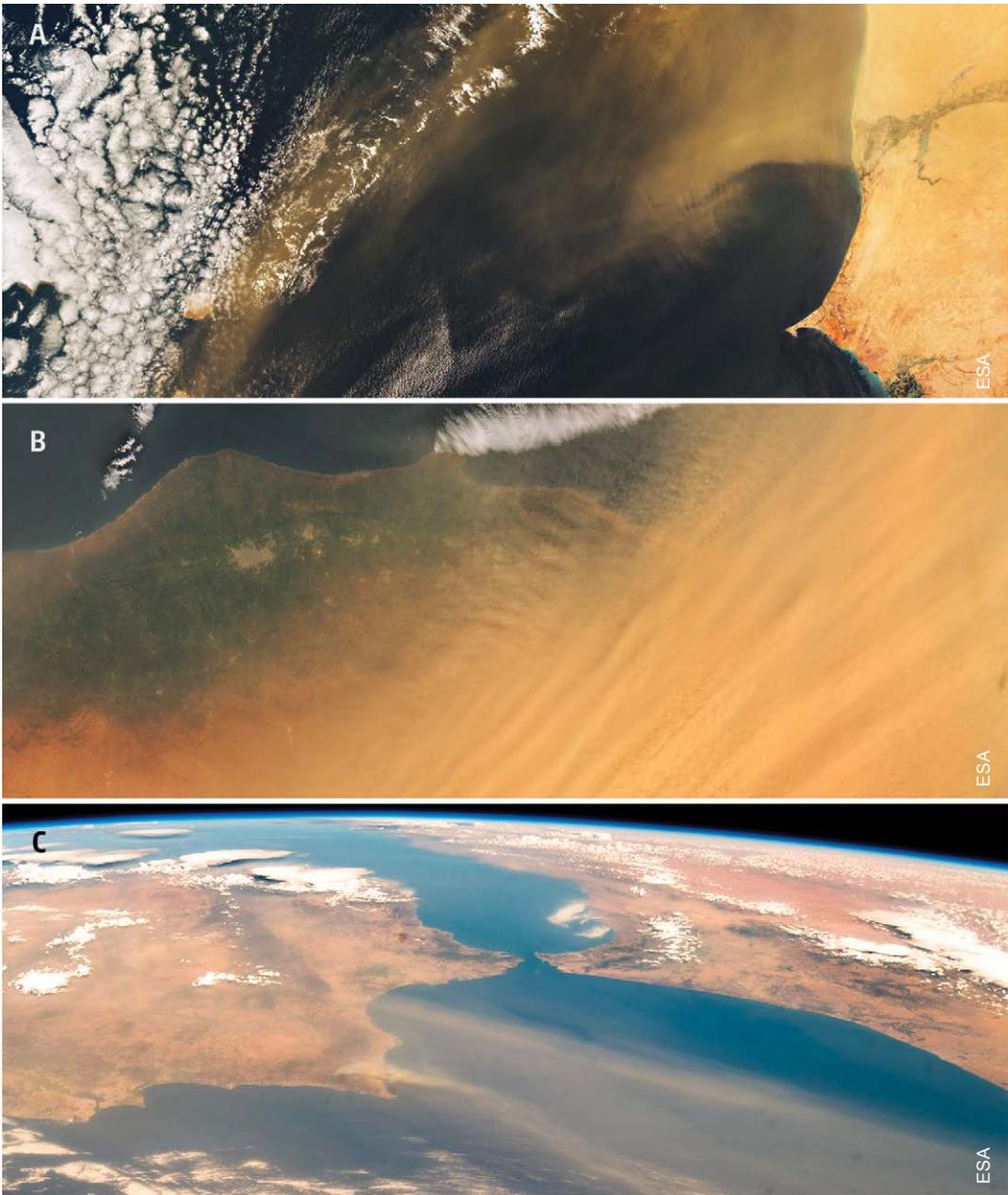
Figure 4.2.10.2 | Satellite images of soil erosion

A. Satellite images of the Atlantic Ocean and the Cape Verde archipelago peeking out from under the clouds, 570 km off the west coast of Senegal and Mauritania, seen on the right of the image. The dust and sand coming mainly from the Sahara and Sahel region are being carried by the wind towards Cabo Verde from Africa. Image captured on May, 2018. **B.** Dust, carried by the wind from Desert storms in North Africa was blown northwards across the Mediterranean Sea causing snow in eastern Europe to turn orange. Dust reached as far afield as Greece, Romania, Bulgaria and Russia. Image captured on March, 2018. **C.** This image of Portugal and Spain was captured by ESA astronaut Alexander Gerst who commented "it looks like a mixture of dust, sand and smoke" International Space Station on August 2018.

4.2.11 | CLIMATE CHANGE

Climate change is associated with uniformly rising CO_2 levels and, in most ecosystems, increased temperature and water limitation. Generally, elevated CO_2 initially enhances photosynthesis, which aggravates limitation of macro- or micronutrients in soil and intensifies plant-microbe competition for soil resources. Elevated CO_2 may thus alter the balance of mutualistic and free-living groups or favour certain mutualists over others (Terrer *et al.*, 2016). Elevated temperature *per se* may promote growth and biodiversity of most soil communities, especially in cold ecosystems. However, increased fluctuations of temperature seasonally and annually are likely to enhance stress from water limitation, especially when coupled with reduced rainfall or longer dry periods. Reduced water availability may directly reduce overall soil biomass and biodiversity (Maestre *et al.*, 2015; Bahram *et al.*, 2018). Long drought periods render the native ecosystems more vulnerable to natural or human-induced burning, which may have devastating effects on ecosystems not adapted to wildfire. Combined, altered temperature and precipitation patterns and fire regimes shift biomes, potentially with enormous changes in plant growth forms and vegetation types. These changes in vegetation may result in cascading effects on all soil biota that largely determine the soil functional potential, including nutrient cycling. In particular, loss of tree cover due to drought stress, pest outbreak or intensive fire results in excessive soil drying and decline of soil organic material and fungal biomass. Moderate nitrogen pollution may act as a fertilizer to counteract soil nutrient limitation, whereas heavy pollution may alter the balance among taxonomic and functional groups and ecosystem nutrient cycling. Pollution of nitrogen and sulphur acidify soil, which favours saprotrophic groups.

Climate change has different implications for different areas of the globe, and therefore the potential impacts of climate change will take a number of different forms; each of these, in turn, will have different potential effects on soil biodiversity. For example, global models of climate variously predict, for different geographic regions, that future conditions will be hotter, drier, wetter, or have more frequent droughts and/or extreme temperature, wind, precipitation events, and so on (IPCC, 2014). Since these climatic changes will drive changes in vegetation type and community composition (mediated by the associated disturbances such as drought, flooding, wind or fire), there is clear potential for major alterations to ecosystem-defining processes like organic matter accumulation, decomposition and cycling, which are known to influence the diversity and composition of soil biota (Coyle, *et al.*, 2017; Coleman *et al.*, 2018). In any case, there will almost certainly be large geographic expanses consisting of novel combinations of climate, vegetation and soil (Hobbs *et al.*, 2009), and there is general consensus that climate-driven changes will result in major losses of biodiversity for above-ground biota (Bellard *et al.*, 2012).

Soil is a remarkably rich reservoir for biodiversity, and the mechanisms by which this diversity arises are still not fully understood. Nevertheless, soil microbes (bacteria, fungi, archaea) are responsible for many critical ecosystem processes which humans (and indeed

the entire terrestrial biosphere) rely upon. Recent advances in understanding the diversity of soil bacteria suggest that a number of co-varying factors can explain global patterns of diversity (Bickel *et al.*, 2019), and chief among these are temperature and climatic water content (a composite of soil water-holding capacity, number of consecutive dry days and potential evapotranspiration or PET). Climatic water content thus relates strongly to the heterogeneity of niche space produced by differences in water-filled pore space and the connectivity of such habitats, which in turn can influence soil pH, also known to correlate with bacterial diversity (Lauber *et al.*, 2009). Thus, changes in factors influencing climatic water content (consecutive dry days, PET), may be reasonably expected to also change soil bacterial diversity.

Similar to bacterial diversity, soil fungal biodiversity is still in the process of being documented and is not particularly well known at the global scale. However, there is evidence that many fungal communities are dominated by a few taxa world-wide, and that these taxa possess genes that are related to stress resistance (Edigi *et al.*, 2019). This is consistent with findings from a mesocosm study performed in peat soils from Canada that became dominated by a few taxa from the Ascomycota and Basidiomycota after 18 months of warming treatments (Asemaninejad *et al.*, 2018). Climate change responses of fungal communities and biodiversity are likely to be biome-specific, however. Other workers in Mediterranean-type ecosystems found that fungal diversity increased in certain parts of the landscape with warming treatments (Birnbaum *et al.*, 2019), so it is difficult to generalize about how fungal diversity may respond to climate change. The implications of these findings are that fungi, from a functional (and perhaps even biodiversity) standpoint, may be particularly well suited to adapt to changing climate conditions, but this may be dependent upon ecosystem type. Indeed, microbial diversity was unchanged when soils from dryland ecosystems around the globe were subjected to warming treatments, although microbial community composition shifted in response to the treatments (Delgado-Baquerizo *et al.*, 2017). However, the capacity of soil microbial communities to buffer ecosystem responses to climate change has not been evaluated for other global biomes.

Faunal components of the soil biota are also globally diverse, and most groups are still not well characterized in terms of their biodiversity, including the nematodes and microarthropods, although molecular approaches are producing better estimates of genetic diversity on the global scale (for nematodes, for example, see Nielsen *et al.* 2014). These advances notwithstanding, most studies that deal with the effects of climate change on the soil mesofauna address faunal responses in terms of communities, assemblages, functional groups or trophic groups. As such, studies addressing species-level biodiversity responses to climate change treatments are relatively rare, but a few trends are noteworthy. For example, Caruso *et al.* (2019) found that oribatid mite diversity generally increased along a south to north gradient in the United Kingdom of Great Britain and Northern Ireland, and that precipitation and soil organic matter were positively (but not strongly) associated with this increased diversity. Considering that both precipitation and soil organic matter content are expected to be influenced by

changing climate, it is possible to infer that mite diversity could reasonably be expected to respond to changes in these conditions.

Macroinvertebrates, many of which are ecosystem engineers (capable of modifying habitat to create niche space for other taxa), are also likely to be responsive to climate changes, as their distribution and abundance are frequently well predicted by general patterns of temperature, moisture and vegetative cover. Local earthworm species richness, abundance and biomass, for instance, were positively related to precipitation at the global level, so that any climate change involving changes in precipitation level of frequency may have important impacts on earthworm communities and their potential for ecosystem service delivery (Philips *et al.*, 2019). Termites, for example, are most diverse in tropical rainforest systems, and their diversity decreases as ecosystems become more arid (Bourguibignon *et al.*, 2017). Thus, if climate change causes drier conditions to predominate, for example, in tropical dry forests where termite diversity is highest at elevations where rainfall is more abundant (Casalla and Korb, 2019), then negative effects on termite diversity can be expected. On the other hand, there is some evidence that the presence of ecosystem engineering macroinvertebrates may provide some buffering of ecosystems to climate changes. For example, Ashton *et al.*, (2019) found that termites were associated with greater soil nutrient heterogeneity (and soil moisture and decomposition rates) during a drought, relative to soils where termites were experimentally excluded, suggesting that when termites were present, these soils experienced less relative change when conditions were drier, with likely positive results for the biodiversity of other soil organisms present at the sites. Similar patterns have been observed for another group of ecosystem engineers, where the presence of earthworms in simulated agricultural soils reduced the negative effects of warming on total below-ground meso- and macrofauna taxon richness (Siebert *et al.*, 2019).

In light of other anthropogenic threats to soil biological diversity (such as land cover change, agricultural intensification and atmospheric deposition of pollutants), which have been relatively well-documented (Coyle *et al.*, 2017), the expected responses of soil biodiversity to climate change are barely known and difficult to predict. It is clear that climate-change-driven impacts on soil biodiversity will be context dependent, and will be strongly influenced by the starting condition (that is, by what vegetation and/or ecosystem type is present), and by the degree and direction of climate change. Focusing on conservation of the diversity of particular groups of soil invertebrates, specifically the ecosystem engineers, may have cascading positive effects on pools of soil biodiversity among other soil flora and fauna.

4.2.12 | INVASIVE SPECIES

Terrestrial invasive species can arise from any level of biological organization ranging from viruses and microbes (bacteria and fungi) to plants, invertebrates and mammals, and each type of invasive species has the capacity to alter soil biological diversity either directly (for example, through competitive displacement), or indirectly (for example, through changes in vegetation composition and/or habitat modification). Introductions of non-native species have been ongoing for centuries, if not millennia, around the globe, and the frequency at which new introductions are made is also increasing (Simberloff *et al.*, 2013). Worse, once established, there appears to be a synergistic relationship between climate change, specifically warming, and the ability of introduced species to increase their ranges in the invaded area (Walther *et al.*, 2009).

Globalization, especially global trade, increases intentional and unintentional introduction of new plant, animal and microbial species into new environments. Import of potted seedlings or saplings is of particular concern, because of undeliberate co-introduction of thousands of microbial species. Although only 10 percent of introduced plant species become naturalized and 1 percent become invasive (Gallagher *et al.*, 2014), these figures may be much higher for microbial species that usually have larger distribution range and ecological amplitude.

It is well recognized that invasive trees, such as species of pines, eucalypts and wattles, may transform entire ecosystems, partly via their recalcitrant litter, stimulation of burning and activities of root-associated microorganisms. Symbiotic biota may further facilitate the invasion process and switch from their intimate plant partners to local potential host trees and invading native soil communities. Introduced pathogens and perhaps endophytes may find new naïve hosts and become serious pathogens of native plants in the new environment. Furthermore, these antagonistic microorganisms may occasionally hybridize with local pathogens and evolve pathogenicity in new hosts. Introduction of animal-associated pathogens pose similar threats to those of plants.

Microbial introductions

Exotic microbes can have profound, ecosystem-changing consequences when pathogens from one continent are introduced to another continent where potential hosts have no inherent resistance. For example, in Australia, more than half of all the putative species from the genus *Phytophthora* sampled in a recent continental-scale sampling were thought to have been introduced, and in fact the two most widespread species were non-native (Burgess *et al.*, 2019). These findings have wide-ranging implications, as *Phytophthera* is a relatively well-known group of fungal pathogens which can have important economic (agricultural and forestry) as well as ecological consequences. Another example of microbial introduction is the case of the parasitic fungus *Cryphonectria parasitica*, known as chestnut blight, which resulted in the total removal a dominant canopy tree (*Castanea dentata*, American chestnut) in North American forests. The loss of chestnut from forests of eastern North America resulted in major changes in

forest composition, with clear changes in the abundance of species with different chemical composition of leaf litter (for example, *Rhododendron maxima*). Considering that studies of soil fauna have shown that "single-tree influences" can be observed for the distribution of earthworm species in eastern North America (Boettcher and Kalisz, 1991), and also for eastern Europe (Hobbie *et al.*, 2006), as well as for broader measures of soil and litter biodiversity including for example mites, nematodes and spiders (Mueller *et al.*, 2016), it is clear that any microbial pathogen that significantly impacts the occurrence of a canopy dominant tree species could have cascading effects on soil biodiversity.

Plant introductions

Plant invasions have strong potential to influence soil biological diversity through various mechanisms. Plant invasions can alter the overall plant community, often resulting in a monoculture stand consisting of only the invasive species (both in overstory and understory plant communities), which changes the diversity of below-ground plant roots and root exudates, which in turn may influence the diversity of organisms that rely on roots and exudates as food or habitat resources. Thus, invasive plants can influence soil biological diversity at nearly all levels, ranging from microbial to vertebrate animal diversity. One example of soil microbial diversity impacts is from Canada where Bugiel *et al.* (2019) found a negative impact of dog-strangling vine (*Vincetoxicum rossicum*) invasion on soil bacterial composition and diversity as measured by variation in terminal restriction fragment length polymorphism data relative to uninvaded sites. Interestingly, not all plant invasions have negative effects on soil microbial species richness, and in some cases (albeit rarely encountered in scientific reports) plant invasions can actually increase species richness of mycorrhizal fungi as demonstrated for Hawaiian subtropical forest ecosystems (Gomes *et al.*, 2018).

Invasive plants can also exert impacts at the meso- and microfauna level, as observed in central Europe (Slovakia), where invasion of the herbaceous knotweed, *Reynoutria japonica* (syn. *Fallopia japonica*) negatively affected soil nematode species richness compared to uninvaded soils (Čerevková *et al.*, 2019). Similarly, in southern Europe (in the Tuscan Appenines), invasion of native oak-dominated stands by black locust trees (*Robinia pseudoacacia*) was associated with decreases in richness in nematodes and microarthropods, as well as decreased plant species richness (Lazzaro *et al.*, 2018). It is notable that mesofauna responses to invasive plants, as with microbes mentioned above, are not always negative. In the Guangdong Province of China, invasion of abandoned farmland by *Artemisia artemisiifolia* was accompanied by an increase in abundance of soil fauna, but this may be partly related to the relatively degraded condition of these populations following agricultural use of the soils (Qin *et al.*, 2018).

Above ground, monocultures of invasive plants can change the timing, chemistry and decomposability of leaf litter inputs into soil ecosystems, and this too has potential to influence the below-ground biotic community. For example, Lobe *et al.* (2014) found that invasion of riparian forests in the state of Georgia in the United States of America by Chinese privet (*Ligustrum sinense*) altered surface soil pH, and that this apparently

favoured populations of non-native European earthworms, but that when the invasive plant was removed, soil pH trended back toward those observed in uninvaded forests, and the abundance of native earthworms rebounded. Recovery of wetland soil invertebrates negatively impacted by non-native plants was also shown in ecosystems in Kwa-Zulu National Park in South Africa when aggressive efforts to remove the invasive plants were undertaken (Eckert *et al.*, 2019). These case studies suggest that although plant invasions have capacity to negatively affect soil biodiversity, active management of invasive plant populations can result in cascading benefits to soil biota.

Soil invertebrate invasions

In large areas of the globe, non-native soil invertebrates have been introduced. Depending on the ecosystem invaded, these organisms can have dramatic negative impacts on native plants, microbial communities and other soil animals. Perhaps the best-studied group of invasive soil animals are the earthworms, and this is owing to their relatively large size and their ability to act as ecosystem engineers; thus their large impacts on soil ecosystems where they invade (Hendrix *et al.*, 2008). Invasive European lumbricid earthworms were reported to reduce species richness and diversity of litter- and soil-dwelling microarthropods (specifically oribatid mites) by 50 to 75 percent in forests of the Allegheny Plateau in the eastern United States of America (Burke *et al.*, 2011). In a study in the southeastern United States of America, Snyder *et al.* (2011) found that millipede species richness was negatively impacted by an invasive Asian earthworm species (*Amynthas agrestis*), and they attributed this to the rapid consumption of forest floor leaf litter by the earthworm. Overall, managing earthworm invasions is quite difficult, and the best approach is thought to be prevention of introductions, as removing invasive earthworms after their populations become well established may be very cost and labour intensive (Callaham *et al.*, 2006).

4.3 | REGIONAL STATUS OF THREATS TO SOIL BIODIVERSITY

There are important regional differences in the importance and role of threats to soil biodiversity and functioning, depending on various abiotic and human factors such as climate, extent of industrialization, area in different types of native vegetation and anthropogenic land uses (especially urbanization, agriculture and forestry), and level of protection of soil resources, among others. These differences are explored in the following sections, in six of the main world regions (using the classification of FAO, 2015): Asia, South West Pacific, Latin America and Caribbean, North America (excluding Mexico), Europe and sub-Saharan Africa. Eurasia, North Africa and the Near East received no input from specialists, so they were excluded from this analysis.

In each region, the impact of thirteen threats (deforestation, urbanization, agricultural intensification, loss of SOM and SOC, soil compaction and sealing, soil acidification and nutrient imbalance, contamination, salinization and sodification, land degradation, fire, erosion and landslides, climate change and invasive species) were evaluated per ecoregion, following the ecoregions used by Orgiazzi *et al.*, (2016b) in the *Global Soil Biodiversity Atlas*, modified from the original maps of the World Wildlife Fund ecoregions of the world (Olson *et al.*, 2001). Hence, ten ecoregions, separated mainly by their relationship to vegetation types but also generally associated with specific types of climates and often soils, were evaluated: tropical and subtropical forest; tropical and subtropical grassland savanna and shrubland; temperate grassland, savanna and shrubland; montane grassland and shrubland; Mediterranean forest, woodland and shrubland; temperate broadleaf and mixed forest; temperate and boreal coniferous forest; tundra; desert and dry shrubland (Figure 4.3.1; Orgiazzi *et al.*, 2016).

For each region of the world, and for each ecoregion therein, the current status and future trends of each threat, as well as the potential impact of the threats on soil biodiversity and function and the present knowledge level (such as the extent of available literature and number of studies) were obtained from expert opinion and consensus among various experts from each world region. For Europe, the exercise was further performed using GIS-based quantitative data, following methods based on Orgiazzi *et al.* (2016a), but for the rest of the world, only expert opinion was used. From the list of thirteen threats, the most important ones were singled out: those that presented high or fair level of geographic spread within the ecoregion, an increasing trend over time, and a high or fair level of impact on soil biodiversity.

Ecoregions of the world

■ Tropical and Subtropical Moist Broadleaf Forests	■ Boreal Forests/Taiga	■ Tundra
■ Tropical and Subtropical Dry Broadleaf Forests	■ Tropical and subtropical grasslands, savannas, and shrublands	■ Mediterranean Forests, Woodlands, and Scrub
■ Tropical and Subtropical Coniferous Forests	■ Temperate Grasslands, Savannas, and Shrublands	■ Deserts and Xeric Shrublands
■ Temperate Broadleaf and Mixed Forests	■ Flooded Grasslands and Savannas	■ Mangroves
■ Temperate Coniferous Forests	■ Montane Grasslands and Shrublands	■ Lakes
		□ Rock and Ice

Figure 4.3.1 | Terrestrial Ecoregions of the World used for the assessment of regional threats

Source: World Wildlife Fund - US

4.3.1 | SUB-SAHARAN AFRICA

Sub-Saharan Africa (SSA) is diverse in terms of relief, climate, lithology, soils and agricultural systems. A combination of some of these has been used to stratify the region into agro-ecological zones (Fischer *et al.*, 2002; Otte and Chilonda, 2002; Global Harvest Choice, 2010), including the sub-humid zone, the humid zone, the highland zone and the arid and semi-arid zones. These represent the same basic zones as the ecoregions listed below.

Tropical and subtropical grassland

The sub-humid zone occupies 22 percent of SSA, mainly in southern and central Africa. The zone receives 1 000 mm to 1 500 mm of rain annually. This zone is very diverse in terms of climate, soils and land use. It is mainly covered by Luvisols, Cambisols Ferralsols, and Acrisols which are developed from parent material that is strongly weathered, and the levels of plant nutrients as well as the clay fraction are low. Their natural vegetation cover consists of medium height or low woodland with understory shrubs and a ground cover of medium to tall, perennial, grasses. Agriculture is the mainstay of the communities that cultivate both food and cash crops, including cassava, yams, maize, fruits, vegetables, rice, millet, groundnut, cowpeas and cotton. From these crops, products such as cottonseed cakes and the residues of the crops are available as feed for livestock. In some areas of this zone farmers grow soybean and leguminous forage crops. The use of fire and deforestation, and the associated loss of soil organic carbon and compaction with changes in land use and urbanization, as well as erosion and landslides and invasive species, were selected as the most important threats to soil biodiversity and function in this ecoregion (Table 4.3.1.1).

Tropical and subtropical forest

The humid zone occupies 19 percent of SSA, mostly in central and west Africa at low latitudes north and south of the equator and receives more than 1 500 mm of rainfall annually. Soils in this zone include Ferralsols, Acrisols and Luvisols, the last of which are commonly encountered at the forest-savannah boundary. Vegetation consists of rainforest and derived savannas with natural vegetation dominated by tall, closed forest which may be evergreen or semi-deciduous and which is often floristically rich. The herbaceous vegetation often contains large amounts of the major nutrients. The soils are strongly weathered and hence have high levels of iron and aluminum oxides and low levels of phosphorus. The organic matter content is therefore generally low and the soils are fragile and easily degraded when the vegetative cover is lost. Deforestation and agricultural intensification may lead to significant decreases in soil biodiversity and are the main threats to soil biodiversity and function in this ecoregion (Table 4.3.1.1).

Montane grassland and shrubland

The highland zone represents 5 percent of SSA's land area, most of which is in eastern Africa and half in Ethiopia. This zone includes areas above 1 500 m altitude that have a mean daily temperature of less than 20°C. The main highland areas are in Ethiopia, Kenya, Uganda, Rwanda, Burundi, the eastern Congo, Tanzania, Angola and Lesotho. The highland areas vary in climate, topography, soils and land use with topography varying from gently rolling hills to deeply incised valleys and steep slopes. Soils are sometimes deep and fertile Andisols and Nitosols, but shallow soils of inherently low fertility are widespread. In many mountain grassland areas, soils only have a very shallow surface horizon that is fertile. Cultivating these so-called shallow low-fertility soils forms a surface crust which reduces water infiltration, resulting in high runoff causing soil erosion, and unless soil conservation measures are taken and soils are sufficiently covered with vegetation, overland flow removes large amounts of topsoil, carrying with it soil organic carbon and its associated biodiversity. The zone receives bimodal rainfall (more than 1 000 mm annually) and there are two growing seasons. Livestock rearing is widespread: farmers grow fodder, and animal traction is of increasing importance. Population pressure is encouraging crop–livestock integration, for which the cool highlands have high potential. Threats to biodiversity in highlands and humid and sub-humid zones include deforestation due to rising population, overgrazing and burning of above-ground cover leading to soil erosion, and loss of plant species with potential negative effect on rhizosphere biodiversity (Table 4.3.1.1).These areas are also particularly threatened by climate change.

Desert and dry shrubland

The arid and semi-arid zones occupy 54 percent of the land area of SSA, most of which is in West and East Africa. Rainfall is low and extremely variable ranging from 500 mm to 1 000 mm annually. Due to high temperatures and evapotranspiration rates, these are mostly associated with Arenosols and Cambisols, sandy and loamy sandy soils poor in plant nutrients and with low water-holding capacity. Vegetation cover consists of short annual grasses, legumes, scattered shrubs and trees. The main livelihood activity of the communities living in the drier zone is keeping livestock including sheep, goats, cattle and camels that browse the herbage and shrubs and move from place to place in search of fodder. Where rainfall is higher and more reliable in the semi-arid zone, there is better vegetation cover of open low-tree grassland and a relatively healthy environment for humans and livestock. Cropping and crop–livestock systems dominate these areas and farmers commonly grow millet, sorghum, groundnut, maize and cowpeas. Threats to soil biodiversity in this ecoregion include wind and water erosion, loss of soil organic matter and soil nutrients, salinization and sodification and waterlogging in low areas (Table 4.3.1.1).

Table 4.3.1.1 | Threats to soil biodiversity in Sub-Saharan African Ecoregions

The main threats to soil biodiversity and function, and the level of scientific knowledge of the impacts of these threats on soil biodiversity in each of the ecoregions of sub-Saharan Africa (SSA)

Ecoregion	Main threats	Knowledge level
Tropical and subtropical forest	Deforestation	High
	Agricultural intensification	High
	Loss of SOM and SOC	Fair
	Erosion and landslides	Fair
Tropical and subtropical grassland	Deforestation	High
	Loss of SOM and SOC	Fair
	Soil compaction and sealing	Low
	Fire	Low
	Erosion and landslides	Low
Montane grassland and shrubland	Deforestation	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Erosion and landslides	Low
	Climate change	Low
Desert and dry shrubland	Loss of SOM and SOC	Low
	Salinization and sodification	Low
	Land degradation	Low
	Fire	Low
	Erosion and landslides	Low
	Climate change	Low

4.3.2 | ASIA

The current status of soil biodiversity varies immensely within Asia. Deforestation for traditional agriculture has caused huge losses of soil biodiversity in the distant past and conventional high-input agriculture, urbanization and contamination in the recent past (Table 4.3.2.1). Climate change is a new threat, particularly affecting highlands and coastal areas. Coexisting with the threats is the opportunity of conserving and restoring biodiversity by organic farming. Knowledge on impacts of the threats suffers from several knowledge gaps: (i) coverage of selected socio-ecological scenarios, taxa and functional groups; (ii) lack of long-term monitoring following a common protocol such as long-term soil fertility experiments and permanent forest plots revealing soil biodiversity/ecosystem function relationships; and (iii) lack of analysis of interaction among different threats.

Tropical and subtropical forests

Deforestation is a threat more to taxonomic diversity of earthworms, termites and mycorrhiza than to bacteria, saprophytic fungi and nematodes, and it reduces functional diversity of all groups of soil organisms (Wong *et al.*, 2016; Kerfahi *et al.*, 2016; Kimber and Eggleton, 2018). Further, detrimental impacts are more pronounced in primary

forests than in secondary forests, and when natural forests are replaced with shifting agriculture with short cycles (4 to 8 years) in the humid tropics and settled annual crop systems with high agrochemical inputs/low organic inputs in the dry tropics (Bagyaraj *et al.*, 2015; Yimyam *et al.*, 2016; Bhadauria, 2016). Though the deforestation rate in Asia is twofold higher than the global average (0.1 percent), countries differ in the magnitude of impact of the threat of deforestation. The threat is not so high in Bhutan, China, India, Laos and Vietnam, where reforestation/afforestation rates have exceeded deforestation rates for the past 10 to 15 years. In Indonesia and Malaysia natural forests are, by and large, replaced by oil palm/rubber plantations, a change that reduces the diversity and abundance of macro-invertebrates (Mumme *et al.*, 2015) but not necessarily of bacteria, fungi and nematodes (Kerfahi *et al.*, 2016). Much of tropical Asia has islands of primary forests conserved for cultural/religious purposes (Lyngwi and Joshi, 2015) and equally structurally complex and species-rich home gardens (Mohan Kumar, 2016). Intensive use of agrochemicals invariably correlate with loss of soil organic carbon, and salinization in many situations is a persistent threat in alluvial plains, but its impact is declining with increasing adoption of organic farming, rotation of crops with positive below-ground interactions and integrated nutrient/pest management (Venkateswarlu, 2016). Rapid urbanization coupled with industrialization and contamination in developing countries and changing precipitation patterns are potential current threats that have not been properly analyzed in the available studies.

Temperate broadleaf and mixed forests

Deforestation, agricultural intensification and loss of soil organic matter are not as extensive in temperate forests as in tropical and subtropical regions. Deforestation reduces invertebrate abundance and diversity (Ma and Yin, 2019) with decreases in termite (Thakur, 2016) and earthworm populations (Bhadauria *et al.*, 2016). Crustaceans (for example, Talitridae and *Ligidium japonicum*) are less likely to persist in the absence of *Cryptomeria japonica* forests in Japan (Ohta *et al.*, 2015). Loss of inland forests results in a greater magnitude of decline in generic richness of nematodes than coastal forests (Kitagami *et al.*, 2018). The region is witnessing agricultural intensification in accessible areas and abandonment in remote areas. Bacterial and fungal communities (Zhang *et al.*, 2018) appear to be more resilient to intensification, abandonment and deforestation/ forest degradation than are macrofauna (Bhadauria *et al.*, 2012) and mesofauna (Miura *et al.*, 2008). Urbanization has caused soil biodiversity loss over large areas, but urban parks can conserve substantial biodiversity (Song *et al.*, 2015; Wang *et al.*, 2018). Increase in salinity in coastal areas due to sea level rise/climate change will reduce diversity of both macro- and microfauna (Wu *et al.*, 2015).

Desert and dry shrublands

This ecoregion suffered extensive degradation of natural vegetation in the past and is inherently poor in soil biodiversity. At present loss of natural vegetation is balanced by afforestation and, with the development of irrigation facilities, degraded lands are being increasingly restored (artificial oases). Agrochemical inputs are quite low and

intensification of organic production is leading to recovery in soil biodiversity along with vegetation cover and enhancement of soil carbon stocks. Studies tracking changes in biodiversity following deforestation are lacking but some insights can be seen from tree planting. Planting of the shrub *Haloxylon ammodendron* increased the diversity of predators, collembola and fungi, decreased the diversity of insect herbivores and oribatida and did not change bacterial diversity, while *Populus gunsuensis* planting increased the diversity of virtually all groups in north-west China (Li *et al.*, 2018). A switch over to wheat-maize intercropping from wheat and maize monocultures is leading to more efficient biocontrol (Liu *et al.*, 2018). In India, mixed planting of leguminous and non-leguminous trees resulted in an 8 to 65-fold increase depending on season and taxon (Tripathi *et al.*, 2009) and organic cowpea/maize/Lucerne mixed cropping a 7-26 percent increase in collembola, oribatid mites, nematodes, actinomycetes and fungi populations (Roy *et al.*, 2012). The ecoregion is also facing increasing urbanization and climate change but studies evaluating them are lacking.

Table 4.3.2.1 | Threats to soil biodiversity in Asian Ecoregions

The main threats to soil biodiversity and function, and the level of scientific knowledge of the impacts of these threats on soil biodiversity in each of the ecoregions of Asia

Ecoregion	Main threats	Knowledge level
Tropical and subtropical forest	Deforestation	High
	Agricultural intensification	High
	Loss of SOM and SOC	Fair
Tropical and subtropical grassland	Deforestation	High
	Loss of SOM and SOC	Fair
	Soil compaction and sealing	Low
	Fire	Low
	Erosion and landslides	Low
Montane grassland and shrubland	Deforestation	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
Desert and dry shrubland	Loss of SOM and SOC	Low
	Salinization and Sodification	Low
	Land Degradation	Low
	Fire	Low
	Erosion and landslides	Low
	Climate change	Low
Temperate broadleaf and mixed forest	Urbanization	Low
	Agricultural intensification	Fair
	Loss of SOM and SOC	Fair
Temperate grassland	Climate change	Low

Montane grasslands and shrublands

This ecoregion suffered extensive degradation because of overgrazing in the past, and while currently degradation is outweighed by restoration, the region is facing perhaps the highest rate of global warming and expansion of woody cover. Studies in this region are

few and confined largely to China and Mongolia. Degradation has impacted beta diversity more than alpha diversity and composition more than richness of bacterial communities. Prevalence of many populations associated with human diseases is a major threat to both livestock and humans (Zhou *et al.*, 2019). Tree (*Pinus tabuliformis*) planting reduced diversity of fungi but had insignificant impact on bacterial diversity and favoured ECM fungi at the expense of decline in biotrophic fungi (Wang *et al.*, 2019). Tree planting alone thus may not be an effective way of restoring soil biodiversity. Climate change may have dramatic impacts on soil biodiversity and its functions evident from earthworm (*Pheretima aspergillum*) invasion and soil organic carbon depletion around Zoige peatlands in China (Wu *et al.*, 2017). Threats of urbanization and agrochemical based intensification are considered quite low.

4.3.3 | EUROPE

Within the European Union (EU), pressures on soils that affect their function have been highlighted by the EU Soil Thematic Strategy (EC, 2006). The Strategy, adopted by all Institutions and Member States explained why further action is needed to ensure a high level of soil protection. To this end, the European Commission highlighted soil erosion, organic matter decline, compaction, salinization and landslides as issues that should be reduced, together with preventing further contamination and limiting or mitigating the effects of sealing, for instance by rehabilitating brownfield sites. Reviews of threats to soil for Europe are provided every five years by the European Environment Agency as part of the State of the Environment Report (SOER), the JRC's State of Soil in Europe (JRC, 2010) and the Status of the World's Soil Resources (FAO and ITPS 2015).

There are several studies assessing soil threats (such as erosion, compaction, pollution, land degradation) at the European level, but indicators related to soil biodiversity are rarely measured at an appropriate scale or resolution. The main sources available are The European Atlas of Soil Biodiversity (Jeffery *et al.*, 2010) and the Global Soil Biodiversity Atlas (Orgiazzi *et al.*, 2016b) that contain maps with rather coarse information on soil biodiversity. Rutgers *et al.* (2019) predicted soil biodiversity at the scale of Europe, using data for soil biological (earthworms and bacteria) and chemical (pH, soil organic matter and nutrient content) attributes in a soil biodiversity model. Aksoy *et al.* (2017) also made an assessment of soil biodiversity potential in Europe, showing that the main threats to soil biodiversity are soil degradation, land use management and human practices, climate change, chemical pollution as well as genetically modified organisms (GMOs) and invasive species. Gardi *et al.* (2013) and Orgiazzi *et al.* (2016a) further specified habitat fragmentation, intensive human exploitation, soil organic matter decline, soil compaction, soil erosion, soil sealing and soil salinization as important threats. European soils are a widely used resource, submitted to a number of relatively well identified threats (ENVASSO, 2008), and therefore soil biodiversity can be threatened by all previously

mentioned processes in Europe. The potential importance of several of these threats to soil biodiversity as defined by a group of experts are presented in Figure 4.3.3.1.

Agricultural land management is one of the most significant anthropogenic activities that greatly alters soil characteristics, including physical, chemical and biological properties (Jangid *et al.*, 2008; Garcia-Orenes *et al.*, 2010). This is particularly relevant in Mediterranean environments, that take up a great part of southern Europe, where unsuitable land management together with climatic constraints (scarce and irregular rainfall and frequent drought periods) can contribute to increased rates of erosion and other soil degradation processes of agricultural land (Caravaca *et al.*, 2002). These conditions can lead to a loss in soil fertility and a reduction in the abundance and diversity of soil organisms. More than 45 percent of Europe's land is used for agricultural production (EUROSTAT, 2019) and 12.7 percent of European arable lands have soil loss >5 t/ha annually requiring protection. Panagos *et al.* (2015) estimated the mean soil loss rate in European Union as 2.46 t/ha annually and the monetary loss for agriculture in Europe due to soil erosion is about 1.25 billion Euros per year (Panagos *et al.*, 2017). Hence, agricultural management in Europe is one of the most important threats to soil biodiversity (Figure 4.3.3.1 and Table 4.3.3.1).
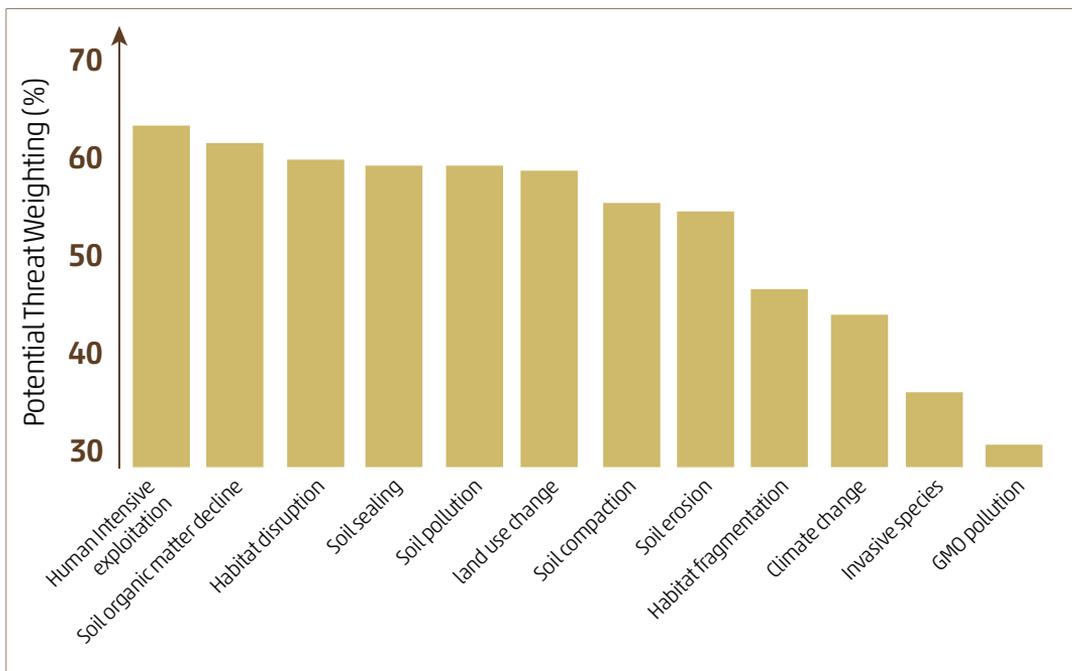
Figure 4.3.3.1 | Importance of threats to soil biodiversity in Europe

The potential threat weighting given by specialists to a selection of soil threats to soil biodiversity in Europe (after Jefferey *et al.*, 2010).

Salinization in Europe affects an estimated area of several millions of hectares (4 dS m^{-1} is the threshold to define saline soils), and has consequences not only for crop productivity, but also for soil organisms (Jeffery *et al.*, 2010).

The loss of soil organic matter, is especially relevant in Mediterranean soils with semi-arid climate (Novara *et al.*, 2011; Laudicina *et al.*, 2015). According to Turbe *et al.*, (2010) the largest emissions of CO_2 from soils are resulting from land use change (for example, from grassland to agricultural fields) and the related drainage of organic soils in Europe. However, precise future estimations are difficult to extract from the literature, given the number of uncertainties, including the dynamic trends in land-use change in Europe. Given the political importance of the management of soils for carbon storage, some recent works have estimated the potential for agricultural soils to sequester more carbon through changes in management, and this has been recently considered in the context of different biological strategies for C sequestration (Woodward, 2009).

As a consequence of human practices many land areas of Europe have suffered forest fires. After the fires, the post-fire management of the burned soil can be a key to promote the recovery of soil biodiversity recovery. Different studies have shown that there are several post-fire management actions, such as salvage logging, that is a common practice in most fire-affected areas in Europe, that can retard soil biodiversity recovery compared with other types of management (Garcia-Orenes *et al.*, 2017; Pereg *et al.*, 2018).

Climate change is probably one of the main environmental problems facing the world, causing major known and unknown effects on all ecosystems in our planet. In this sense, there is an important knowledge gap about the impact of the storage and release of greenhouse emissions on soil biodiversity (Vries and Griffiths, 2018). Several studies carried out in the 1990s found that plant growth and below-ground allocation of C, particularly of rhizodeposits, increased under elevated CO_2 levels, which had consequences for microbial biomass and respiration rates (Zak *et al.*, 1993; Newton *et al.*, 1995). These authors hypothesized that the proportion of fungi would increase under elevated CO_2 because of increased plant litter production. It has long been recognized that soil moisture, that it has been influenced by climate change is an important driver of the composition and activity of soil communities, and the first studies to assess the effect of fluctuations in soil moisture on soil communities did not do so from a global climate change perspective.

Several studies reported since the 1990s have shown context dependent effects, highlighting the need to understand the role of how different soil and vegetation types drive soil biodiversity response to climate change. While it might be hard to see consistent patterns, some generalities are starting to appear. In particular, microbial groups and bacteria taxa that are associated with oligotrophic or K-strategist life-history strategies seem to be consistently increasing in abundance under drought and increasing temperatures, while they decrease in response to elevated CO_2. In contrast, under pulse disturbances such as drought followed by rewetting, the more copiotrophic or r-strategist groups, with high maximum growth rates, are able to rapidly regain their abundance (De Vries and Shade, 2013).

Biological invasions and introduced exotic species are becoming a problem in the invaded areas because they develop excessive abundance over native species. An overview

of invasive species in Europe can be found on the DAISIE European Invasive Alien Species Gateway (http://www.europealiens. org), where the current estimate is that approximately 11 000 species are invasive in Europe. For the EU, knowledge on the distribution of regulated invasive alien species is collected through the European Alien Species Information Network. The potential impact of exotic invasive species can be particularly worrying in rare ecosystems and there is a critical knowledge gap concerning impacts of invasive species on soil biodiversity. Considering the immense biodiversity of organisms present in one gram of soil, it is irrelevant to simply describe how invasive species influence the total numbers of soil organism species. It is more insightful to consider what sort of species exotic invaders influence and what the functions of those species are. In this sense The European Atlas of Soil Biodiversity discusses different concrete examples of the effect of exotic species on soil biodiversity.

Table 4.3.3.1 | Threats to soil biodiversity in European Ecoregions

The main threats to soil biodiversity and function in each of the ecoregions of Europe

Ecoregion	Main threats
Temperate and boreal coniferous forest	Urbanization Contamination Land Degradation Invasive species
Temperate broadleaf and mixed forest	Deforestation Urbanization Contamination Land degradation Invasive species
Temperate grassland	Contamination Land Degradation
Mediterranean forest, woodland and shrubland	Urbanization Contamination Land Degradation
Desert and dry shrubland	Deforestation
Tundra	Contamination
Boreal Forests/Taiga	Deforestation

4.3.4 | LATIN AMERICA AND THE CARIBBEAN

Temperate and boreal coniferous forest

In Latin America and the Caribbean (LAC) these ecoregions are located in Argentina and Chile in a narrow region along the Andes, ranging between the 37th and the 52nd parallels south, and in Mexico they are known as the "bosques de Oyamel y de Pinus". In the former countries, the weather is temperate to cold humid, with rainy winters.

Coniferous forest are dominated *Araucaria araucana*, *Austrocedrus chilensis* and *Fitzroya cupressoides*. In general, these species constitute mixed forests with species from the family Nothofagaceae. These trees have small, needle- or scale-like, acidic leaves. The main threat to soil biodiversity in this region are deforestation and fires, which cause great changes in vegetation composition in big areas (Table 4.3.4.1). Furthermore, agricultural intensification and urbanization has increased in the past years and are therefore important threats to soil biodiversity in coniferous forests.

Temperate broadleaf and mixed forest

These forests are located in LAC in part of Argentina, Chile and Mexico. In Argentina and Chile, temperate broadleaf forests occur in the Chaco and Pampean provinces (Chacoan dominion) and in the Maule and Valdivian Forest provinces (Andean Region) (Arana, 2017). In Mexico, they occur in the mountain regions of the Sierra Madre Oriental and Occidental, and consist mainly of pine-oak forests. The weather is temperate with distinct warm and cool seasons and variable precipitation. The input of C to the soil from the forest (as leaf litter or dead wood) allows high abundances of decomposers. The main threats to soil biodiversity in these forests in LAC are related to very high rates of (historical and current) deforestation, in part due to wood extraction and the expansion of agricultural frontiers that are leading now to the agricultural intensification. These activities usually co-occur with fire, which is another of the main threats to soil biodiversity. The concomitant loss of SOC and soil compaction are also important threats to soil biodiversity. As a consequence, a great reduction in surface covered with broadleaf forests has occurred.

Temperate grassland

The main temperate grasslands in LAC are located in the Pampas of Argentina, a wide plain with more than 52 M ha in extension. Rains range from 1200 mm in the northeast to 400 mm in the southwest and are concentrated mainly from late spring to early autumn, with dry winters. Annual mean temperature is about 16 °C with warm summers and severe winters. Temperate grasslands support high levels of soil microbial and faunal diversity but the impact of current threats is reaching worrisome levels. Many of the grasslands have been replaced with annual crops, strongly reducing the original grassland cover. High rates of deforestation have occurred to enable the expansion of cropland frontier. This phenomenon, together with agricultural intensification associated with soybean monocropping without rotations and high rates of pesticide use, have caused important losses of SOM, soil compaction and sealing and soil contamination. All these processes are threatening soil biodiversity and jeopardizing soil ecosystem processes relying on soil biology.

Montane grassland and shrubland

High altitude grassland and shrubland biomes are mainly associated to the Andes Mountains in South America, which include the ecosystems known as Páramo (Ecuador, Columbia, Peru, Venezuela), Puna (Argentina, Chile, Bolivia and Peru), Estepa (in

southern Argentina and Chile). In North and Central America (Mexico, Guatemala), they are known as Zacatonales. Unique climatic conditions occurring at high altitude, including elevated radiation and extreme temperature fluctuation, among others, provide unique niches resulting in a high degree of endemism. The main threats of these ecosystems are climate change, erosion and landslides, land degradation, loss of SOC and contamination, all of which affect not only soil biodiversity but also their ecosystem functioning and services. The latter is of particular interest for these ecosystems, as soil microbiota adapted to harsh environmental conditions important for biotechnology, human health and agricultural adaptation to climate change, among others.

Tropical and subtropical grasslands, savannas and shrublands

Tropical and subtropical grasslands, savannas and shrublands tend to receive from 900 to 1 500 mm yearly rainfall, have prolonged dry seasons conditions, and often frequent burning, that do not allow development of extensive tree cover. They include the large area of Cerrado in Brazil and Bolivia, as well as the Paraguayan and Argentinean Chaco, grasslands in the Pampa and Llanos of Venezuela and Colombia, and seasonally flooded regions along the Pacific and Gulf coasts of Mexico. They are well known for their complexity of habitats and unusually high levels of endemism and beta diversity. Many of these regions have been extensively occupied for agricultural and pastoral uses, as well as for mining, greatly reducing original cover of this ecoregion, as well as impacting above and below-ground biodiversity. The greatest threats to soil biodiversity and function in this ecoregion are agricultural intensification and associated erosion, invasive species and climate change.

Mediterranean forest, woodland and shrubland

Mediterranean forest, woodland and shrubland occur in part of Mexico (Baja California Peninsula) and central Chile, in areas under climatic conditions characterized by rainfall and droughts concentrated during winter and summer, respectively. Due to geographical isolation, these regions have a remarkable level of biodiversity and endemism, and as such are considered hotspots for global biodiversity. Mediterranean ecosystems in LAC are highly susceptible to human interventions such as urbanization and agricultural intensification. Moreover, the effects of climate change, land burning and erosion processes also represent important pressures to soil biodiversity in these ecosystems, at all trophic levels.

Tropical and subtropical moist and dry forests

Tropical and subtropical moist and dry forests are characterized by low variability in annual temperature and can be found from Argentina to Mexico, but with the vast majority of remaining forest in the Amazonian basin. Both forests tend to receive 1500 mm or more rain per year but moist forests have mostly semi-evergreen and evergreen deciduous tree species, while dry forests tend to have a prolonged dry season and mostly deciduous trees that lose their leaves in the dry season. Both forests are well known for their high alpha and beta biodiversity, and may be home to half of all species on the planet. They are

highly sensitive to burning and deforestation, mainly for agriculture and pastoral land use, which have already greatly reduced original forest cover in LAC, impacting local and regional climate patterns, and having major impacts on both above and below-ground biodiversity both locally and regionally.

Table 4.3.4.1

The main threats to soil biodiversity and function, and the level of scientific knowledge of the impacts of these threats on soil biodiversity in each of the ecoregions of Latin America and the Caribbean

Ecoregion	Main threats	Knowledge level
Tropical and subtropical forest	Deforestation	Fair
	Agricultural intensification	Fair
	Contamination	High
	Fire	Fair
	Erosion and landslides	Fair
	Climate change	Fair
Tropical and subtropical grassland	Deforestation	Fair
	Agricultural intensification	Fair
	Loss of SOM and SOC	Fair
	Soil compaction and sealing	Low
	Contamination	Fair
	Fire	Fair
	Erosion and landslides	Low
	Climate change	Low
	Invasive species	Low
Mediterranean forest, woodland and shrubland	Deforestation	Low
	Urbanization	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Land Degradation	Low
	Fire	Low
	Erosion and landslides	Low
	Climate change	Low
	Invasive species	Low
Montane grassland and shrubland	Loss of SOM and SOC	Low
	Land Degradation	Low
	Erosion and landslides	Low
	Climate change	Low
Desert and dry shrubland	Contamination	Low
	Land Degradation	Low
	Erosion and landslides	Low
	Climate change	Low
Temperate and boreal coniferous forest	Deforestation	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Fire	Low

Temperate broadleaf and mixed forest	Deforestation	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Soil compaction and sealing	Low
	Fire	Low
	Erosion and landslides	Low
Temperate grassland	Deforestation	Fair
	Agricultural intensification	Fair
	Loss of SOM and SOC	Fair
	Soil compaction and sealing	Fair

Desert and dry shrubland

These ecoregions are present in part of Mexico, northern Venezuela, northeast Brazil, central Argentina and along the Pacific coast in Peru and northern Chile. Due to natural climatic conditions of these habitats, they have rather low population densities as compared to other ecosystems; however, environmental pressures due to human intensification is significantly growing. The main threats to soil biodiversity to these habitats in LAC are climate change, land degradation, pollution, salinization and sodification and erosion and landslides. Although low or lack of rainfall is a distinctive feature of these ecoregions, counterintuitively, water soil erosion could be of special significance in particular zones of the hyper-arid Peruvian and the Atacama Desert in Chile, where intense rains from the "invierno altiplánico" occurring at high altitude or derived from the El Niño events originate considerable floods that dramatically affect bare and dry soils. Thus, soil microorganisms adapted to these environments such as cyanobacteria, lichens and a range of extremophiles, among others, are highly affected.

4.3.5 | NORTH AMERICA

Currently, threats to soil biodiversity associated with the direct and indirect effects of climate change represent the largest threat to North American soils (Table 4.3.5.1).

Boreal Forests

Boreal forests are the most widespread ecoregion in North America and because the rates of deforestation have remained relatively unchanged over the past several decades (Alig *et al.*, 2003), threats to soils are largely due to climate change, although land conversion remains an important factor (Dyk *et al.*, 2015). Changes in fire intensity and frequency have resulted in rapid and recent net carbon loss from soils (Walker *et al.*, 2019). Such losses compound the effects of climate change by increasing the susceptibility of young forests to fire and further soil carbon loss. Additional risks include gas and oil expansion, which includes urbanization, land conversion and contamination (Yeung *et al.*, 2019).

Broadleaf forests

Change in vegetation cover through insect defoliation is a major concern, particularly for bark beetles in both eastern deciduous and western coniferous forests. The combined effect of these agents is expected to grow due to changes in historical climate no longer limiting establishment of invasive species (Potter *et al.*, 2018). Fire in these systems is increasing, particularly for continental forests, which have had steadily increasing fire incidence since 2000 (Potter, 2018).

Grasslands

Because most of the land in the temperate grassland regions is privately owned, conservation of these systems is challenging. More than 80 percent of North American temperate grasslands have been converted to agriculture since European settlement (Glasser *et al.*, 2012). Current threats to grassland soils include agricultural intensification, including a resurgence of monocropping (Wang *et al.*, 2019) and increased biocide use (FAOSTAT, http://www.fao.org/faostat/). Grassland soils outside of agriculture are threatened by increasing land conversion chiefly due to urbanization and oil and gas development (Schaeffer and DeLong, 2019).

Mediterranean shrublands

North American chaparral, although one of the smallest ecoregions in North America, is a biodiversity hotspot. Chaparal soils are increasingly threatened by shorter fire intervals associated with climate change (Sypard *et al.*, 2019). Short fire cycles lead to conversion of shrublands by invasive herbaceous plants (Park *et al.*, 2018). Increasing land conversion due to urbanization is also a growing concern as the region overlaps with one of the most densely populated areas of the continent.

Deserts

The deserts of North America comprise both warm and cool deserts and both are affected by altered fire regimes and subsequent encroachment of invasive species. More frequent and severe fires, accompanied with altered moisture regimes means that native plants are being replaced both by woody vegetation (Juniper in the Great Basin; Davies and Bates, 2017), or invasive grasses (cheat grass).

Tundra

The biggest threat to soil biodiversity in the tundra is the loss of soil organic carbon due to climate change (Plaza *et al.*, 2019). This region has experienced the most rapid warming, leading to earlier phenology, warmer soil temperatures and changes in vegetation (Myers-Smith *et al.*, 2019). Even with conservative increases in global air temperature, increases in soil temperature will lead to accelerated losses of SOM (Biksaboom *et al.*, 2019).

Table 4.3.5.1

The main threats to soil biodiversity and function, and the level of scientific knowledge of the impacts of these threats on soil biodiversity in each of the ecoregions of Canada and the United States of America

Ecoregion	Main threats	Knowledge level
Temperate and boreal coniferous forest	Fire	High
	Climate change	High
	Invasive species	Fair
Temperate broadleaf and mixed forest	Climate change	Low
	Invasive species	Fair
Temperate grassland	Urbanization	High
	Agricultural intensification	High
	Salinization and sodification	High
	Erosion and landslides	Fair
	Climate change	Low
	Invasive species	Fair
Mediterranean forest, woodland and shrubland	Urbanization	High
	Fire	High
	Climate change	High
	Invasive species	High
Desert and dry shrubland	Deforestation	Low
	Urbanization	Low
	Agricultural intensification	Low
	Salinization and sodification	Low
	Land Degradation	Low
	Fire	Low
	Erosion and landslides	Low
	Climate change	Low
	Invasive species	High
Tundra	Loss of SOM and SOC	High
	Climate change	Fair
	Invasive species	None

4.3.6 | SOUTH WEST PACIFIC

The South West Pacific region includes the 22 island nations of the Pacific, New Zealand and Australia (*Status of the World's Resources, Main Report*; Chapter 15: Regional assessment of soil change in the Southwest Pacific). The soils of this region are diverse and cover a wide breadth of latitudes and altitudes. These soils include highly weathered soils in humid tropical areas and continental Australia and relatively young volcanic soils of Indonesia, Papua New Guinea, New Caledonia, Norfolk Island and New Zealand. The main ecoregions and associated threats are listed in Table 4.3.6.1 Some of these threats are interactive as noted.

Agricultural intensification was mentioned in all the ecoregions as a major threat to soil biodiversity. Invasive species was also considered important in all but one ecoregion, and deforestation was considered a major threat in all the forested ecoregions (Table 4.3.6.1). The reduction in threats associated with land clearing can be promoted by the protection of remaining forests (old growth), the replanting of new forests and the ongoing assessment of growth stage (SOFR, 2018). Old growth forests have significant habitat, nature conservation and aesthetic value and contribute to C storage and water production. According to SOFR (2018), there has been no national survey of old growth forests since 1995-2000 when it was estimated that they made up 5M ha of a total survey area of 19M ha (excluding Northern Australia; Qld and NT). Australia claims that it has met target 11 of the Aichi Biodiversity targets about preserving Australia's native forest (SOFR, 2018). Partial information is available for 60 percent of Australia's forest dwelling vertebrate fauna and vascular plants, but the report (SOFR, 2018) indicates that there are 'no comprehensive lists of invertebrate fauna, non-vascular flora (including algae, liverworts and mosses, fungi and lichens) or microorganisms that occur in forests, even though these species play key roles in ecological processes'.

Table 4.3.6.1

The main threats to soil biodiversity and function, and the level of scientific knowledge of the impacts of these threats on soil biodiversity in each of the ecoregions of the South Western Pacific

Ecoregion	Main threats	Knowledge level
Tropical and subtropical forests	Deforestation	Low
	Agricultural intensification	Low
	Fire	None
	Erosion	None
	Climate change	Low
	Invasive species	Low
Tropical and subtropical grasslands	Deforestation	None
	Urbanization	None
	Agricultural intensification	None
	Fire	None
	Erosion and landslides	None
	Climate change	None
	Invasive species	Low
Temperate broadleaf and mixed forest	Deforestation	Low
	Urbanization	None
	Agricultural intensification	None
	Loss of SOM and SOC	Low
	Land Degradation	None
	Invasive species	Low
Temperate grasslands	Agricultural intensification	None
	Loss of SOM and SOC	Low
	Land Degradation	Low
	Invasive species	Low

Mediterranean Forest, Woodland and Shrubland	Deforestation	Low
	Urbanization	Low
	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Soil acidification	Low
	Land Degradation	None
	Fire	None
	Invasive species	Low
Montane grassland and shrubland	Agricultural intensification	Low
	Loss of SOM and SOC	Low
	Invasive species	None
Desert and dry shrubland	Agricultural intensification	None
	Loss of SOM and SOC	Low
	Fire	Low

Tropical and subtropical forests

These forests occur in large, discontinuous patches on the equatorial belt and between the Tropics of Cancer and Capricorn made up of small fragmented coastal areas in Queensland and patches on Lord Howe, Norfolk Islands, the North and South Cook Islands, New Caledonia, Vanuatu, Fiji, Tonga, Samoa and associated small island chains (Somerville *et al.*, 2017). Characterized by low variability in annual temperature and high levels of rainfall (>200 centimetres annually) these forests are dominated by semi-evergreen and evergreen deciduous trees and predominantly eucalyptus and acacia species (SOFR, 2018). Land clearing (deforestation) for agriculture is considered the most important threat. More than 80 percent of the 1.2 million ha cleared in Australia between 1991 and 1995 was on the coastal periphery in Queensland (Bradshaw, 2012) and while National Parks protect the high diversity of plants and animals in this small ecoregion, the quality of the biodiversity has been significantly impacted by land-clearing for agriculture and mining, introduced pests and diseases and other anthropogenic sources (Sommerville *et al.*, 2018). Significant modification of the surrounding tropical and subtropical grassland ecoregion associated with land clearing of the Burdekin River catchment for cereal and sugar cane production pose significant threats to the quality of this ecoregion and its soil biodiversity. Related to these threats are the interactive threats of erosion and landslides and fire, exacerbated by climate change.

Tropical and subtropical grasslands

This region is described by rainfall levels between 90-150 centimetres per year. In Australia, this is one of the four dominant ecoregions, covering more than 30 percent of the continent and occurring mainly in Queensland (approximately 70 percent), NT (approximately 50 percent) and WA (approximately 20 percent). Although northern NSW coastal regions and an extensive area in SE Qld was cleared between 1950-1980 (Mackenzie *et al.*, 2017), land clearing related to agricultural intensification mainly for grazing and pasture land-uses in Queensland with some cropping and urbanization proceeds at an accelerated rate post 1980 (Bradshaw, 2012; Mackenzie *et al.*, 2017).

One year of increased land-clearing in Queensland has already removed many more trees than will be planted during the entire AUD 50 million Australian Government 20-million trees program. Further, under 'Caring for our Country' and Biodiversity Fund grants, tree planting to restore habitat across Australia since 2013 was just over 42, 000 ha while 296 000 ha was cleared in Queensland alone in 2013-2014 (Australian Government, 2017b). Land clearing for agricultural intensification has been associated with a significant risk of hillslope erosion particularly impacting marine water quality (MacKenzie *et al.*, 2017). A closer examination of this ecoregion by NRM regions within it identifies the Burdekin NRM region having the highest hillslope erosion rates in Australia (Teng *et al.*, 2016), Cape York with emerging erosion with land development (Olley *et al.*, 2013), and Qld Mackay-Whitsunday and Qld SW both having moderate to high rates of erosion due to large areas of bare soil attributed to cropping for sugarcane, overgrazing and the Millennial drought (MacKenzie *et al.*, 2017). Adding to this, vegetation clearance for new banana enterprises in the southern Cape York NRM region in the last decade was identified as a significant factor in water erosion of soil (SoE 2016, 2018). The impact of sediment movement primarily as the result of erosion of 31.1 million hectares of land and over 100 000 kilometres of streambank on the poor quality of the Great Barrier Reef catchments has received considerable attention (Schaffelke *et al.*, 2017). This has been attributed to erosion and resultant loss of ground cover and the adoption of best management practices that exclude cattle from gullies and maintaining ground cover to promote 'healthy soil are needed' (https://www.reefplan.qld.gov.au/science-and-research/the-scientific-consensus-statement). Native vegetation clearance and increased grazing intensity across large parts of Northern Australia related to the arrival of drought tolerant African cattle breeds in the 1980s, and increased density of watering points were identified as causal factors of gully erosion and a major source of excess sediment in streams and estuaries (SOE, 2016; Teng *et al.*, 2016; Bartley *et al.*, 2014). Despite the availability of management recommendations to adopt grazing and cropping best practices to improve soil conditions (erosion acidity and carbon) and the quality of ecosystem services (Cork *et al.*, 2012) there is little evidence of wider adoption in susceptible areas of this ecoregion.

Climate change has also been associated with higher frequency of intense bushfires in the ecoregion (SOFR, 2018). It has been claimed that climate change is increasing the intensity of extreme weather events in Queensland – drought, bushfires, heatwaves, floods and cyclones. Currently 65 percent of Queensland is drought declared and parts of the state's west and south have been drought-affected for more than six years (https://www.climatecouncil.org.au/wp-content/uploads/2019/08/qld-report-climate-council.pdf).

Temperate broadleaf and mixed forest

This ecoregion has a moderate climate and high rainfall that gives rise to unique eucalyptus forests and open woodlands and extends across Tasmania (approximately 100 percent), Victoria (approximately 60 percent), NSW (approximately 30 percent) southern Queensland (approximately 10 percent) and eastern SA (approximately

<1 percent). It has served as a refuge for numerous plant and animal species when drier conditions prevail over most of the continent, resulting in a remarkably diverse spectrum of organisms with high levels of regional and local endemism. It includes major urban centres or capital cities of Melbourne, Sydney, Brisbane and Hobart and is therefore significantly impacted by human activities such as urbanization and land-use intensification. Land clearing of forests is historical and most of the biodiversity is preserved in National Parks (e.g.; South-west, Boodera, Wollomi and Blue Mountains being the largest) with 40 percent of Tasmania protected by national parks or state reserves. This ecoregion has significant coastal frontage with the highest level of soil loss by water erosion (MacKenzie *et al.*, 2017). In western and central Tasmania and the steep forested areas along the Great Dividing Range (mainland) greater than 25 tonnes of soil/ha/year are lost (Teng *et al.*, 2016). This erosion has been discounted due to the steepness of slopes. From this erosion data however, the greatest percentage nutrient loss for total N, Total P and SOC occurs in these coastal regions (MacKenzie *et al.*, 2017). The soil acidification risk is high particularly where agricultural activities occur in higher rainfall areas and this threatens long-term agricultural viability especially in the SW and NE regions of Victoria and SE regions of NSW where the gross value of production is highest (MacKenzie *et al.*, 2017). The pH of soils especially on coastal fringes are highly acid with pH <4.8 and further inland between 4.8 and 5.5 in others. Soil carbon stocks are declining under current land-use in Tasmania and the Northern reaches of NSW and into Southern Queensland and in other regions it remains steady (MacKenzie *et al.*, 2017).

Temperate grasslands

This ecoregion has cooler and wider annual temperatures and extends from northern Victoria (approximately 20 percent) into NSW (approximately 55 percent) and southern Queensland (approximately 25 percent). These regions are devoid of trees, except for riparian or gallery forests associated with streams and rivers. Positioned between temperate forests and the arid interior of Australia, the southeast Australian temperate savannas span a broad north-south swatch across Queensland, New South Wales and Victoria. Australia's most significant river system, the Murray Darling river system catchment occurs mainly in this region. A combination of drought (attributed to climate change) and water mismanagement has been reported as the cause of mass fish kills reported in December 2019 (Australian Academy of Science, 2019). Most of this ecoregion supports agricultural enterprises such as sheep breeding and grazing and wheat cropping and only small fragments of the original eucalypt vegetation remains. Soil pH data is between 4.8 and >5.5 and soil acidification risk is considered low to medium despite there being insufficient data for modelling and soil Carbon stocks are declining under current management (MacKenzie *et al.*, 2017).

Mediterranean Forest, Woodland and Shrubland

Mediterranean ecoregions are characterized by hot and dry summers, while winters tend to be cool and moist. In Australia, this ecoregion is significant in the southern

states of Victoria (approximately 30 percent), central and western NSW (approximately 20 percent), southern SA (approximately 25 percent) and south eastern WA (approximately 25 percent) and represents the most significant cereal production regions in Australia. In the south eastern corner of WA, there is one of the world's 34 internationally recognized biodiversity hotspots with more than 1500 endemic species of plants and only 30 percent of its original habitat (Williams *et al.*, 2011). In SA, the significant wine growing regions (such as The Barossa Valley, McLarenVale). Alongside this hotspot there is significant and expanding intensive cereal cropping region with widespread soil acidification, particularly in the subsoil which threatens long term business viability in these regions if left untreated. In regions of SA, soil acidification risk is also high with pH_{Ca} values of <4.8 in much of the region. Land clearing is largely historical (between 1920-1950; SOFR, 2018) in the SE WA region however since the 1980s this region is extending on the northern and southern margins. All areas in this region have the capacity to store carbon through best crop management strategies except in central Victoria (MacKenzie *et al.*, 2017).

Montane grassland and shrubland (mostly New Zealand)

This ecoregion includes high elevation (montane and alpine) grasslands and shrublands. In Australia montane grassland and shrublands are restricted to the montane regions of south-eastern Australia above 1300 metres. This region occupies less than 3 percent of the Australian landmass and straddles the borders of the Australian Capital Territory, Victoria and New South Wales on the Australian mainland, as well as a significant element in Tasmania. There is very little soils data available for this region however it falls within the zone of highly acid soils where the pH_{Ca} is < 4.8 (MacKenzie *et al.*, 2017). In contrast, this represents a major ecoregion in New Zealand, especially the central region of the south island.

Desert and dry shrubland

These ecoregions vary greatly in the amount of annual rainfall they receive; generally, however, evaporation exceeds rainfall in these ecoregions. In Australia, they represents the largest region covering more than 70 percent of SA, 65 percent of WA, 50 percent of the NT, 25 percent and 10 percent in western central Qld and north western NSW respectively. Land clearing due to mining and exotic weed species incursions are the greatest threats (NT Landcare, 2019) together with climate change and drought and associated wind erosion and fire. According to MacKenzie *et al.* (2017), the bare soil index over a 16-year period is high and the region is prone to significant wind erosion events. The bare soil index developed in Australia to identify the risk of erosion by wind, calculates the proportion of each year when bare ground is equal to or greater than 50 percent. For this ecoregion, a significant area shows fractional cover (or <50 percent) for the entire year. The influence of drought on vegetation cover coupled with the effects of management such as the reduction in stock as dry weather persists, are key in reducing this threat of soil loss. As this ecoregion is not considered agricultural, there is little data on soil conditions.

4.4 | GLOBAL SYNTHESIS

Using the information provided on the main threats in the ten ecoregions present in the six world regions (Table of threats for Sub-Saharan Africa, South West Pacific, Latin America and Caribbean, North Africa and Near East, North America and Europe; FAO and ITPS, 2015), a summary table of the main threats common to these ecoregions was produced (Table 4.4.1). This exercise showed that the most widespread threat to soil biodiversity in the world was the loss of SOM and SOC, and that this could be associated with other threats such as deforestation and agricultural intensification (both linked with land use change) and with climate change (particularly in tundra). This clearly shows the importance of sustainable management and conservation practices, to maintain this resource in soils, which represents one of the bases for the soil's food webs. Deforestation and agricultural intensification were also major threats worldwide, being important in tropical and temperate broadleaf and mixed forests and temperate and montane grasslands and boreal forests/taiga, although the level of available information on the topic was highly variable, depending on the particular world regions where these ecoregions occur. The ecoregions with the highest number of threats were the deserts and dry shrublands, the tropical and subtropical grasslands, and the temperate broadleaf and mixed forests. Invasive species also represented an important threat, particularly in Mediterranean and temperate forests and tundra.

Table 4.4.1 | Threats to soil biodiversity in global Ecoregions

The main threats to soil biodiversity and function in the world's ecoregions

Ecoregion	Main threats
Tropical and subtropical forest	Deforestation Agricultural intensification
Tropical and subtropical grassland	Deforestation Loss of SOM and SOC Soil compaction and sealing Fire Erosion and landslides
Mediterranean forest, woodland and shrubland	Urbanization Land degradation Fire Invasive species
Montane grassland and shrubland	Agricultural intensification Loss of SOM and SOC
Desert and dry shrubland	Loss of SOM and SOC Salinization and sodification Land degradation Fire Erosion and landslides Climate change

	Deforestation
Temperate broadleaf and mixed forest	Urbanization
	Agricultural intensification
	Loss of SOM and SOC
	Invasive species
Temperate grassland	Agricultural intensification
Temperate and boreal coniferous forest	Fire
	Invasive species
	Loss of SOM and SOC
Tundra	Climate change
	Invasive species
Boreal Forests/Taiga	Deforestation

The only other available diagnosis of the extent of various threats to soil biodiversity was published in the Global Atlas on Soil Biodiversity (Orgiazzi *et al.*, 2016) and also used expert opinion, but this did not include all of the threats listed here. The resulting consensus map of global threats (Figure 4.4.1) was produced using data on: 1) loss of above-ground biodiversity (plant species loss) as a proxy of land use change (such as deforestation); 2) nitrogen fertilizer application (as a proxy for pollution and nutrient overloading); 3) cropland cover and cattle density (as a proxy for agricultural intensification and associated soil compaction); 4) fire density (as a proxy for risk of fires); 5) water and wind erosion vulnerability indices, to assess soil erosion risks; 6) desertification vulnerability index; and 7) global aridity index (as a proxy for climate change).

Threats to Soil Biodiversity
- No Data
- Very Low
- Low
- Moderate
- High
- Very High
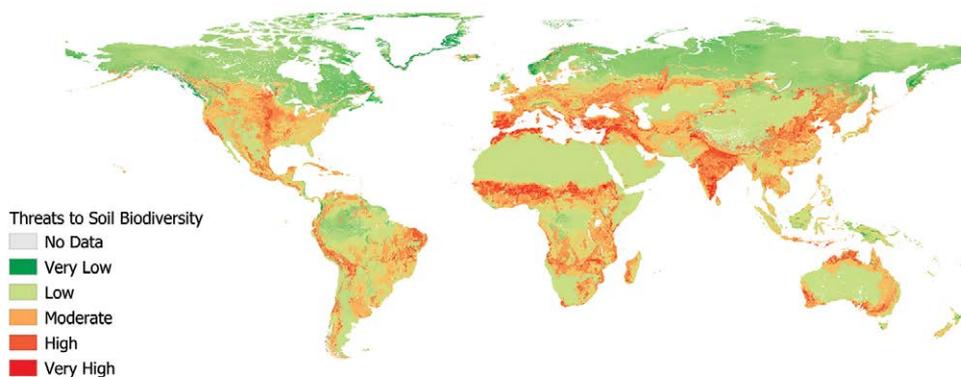
Figure 4.4.1 | Estimated levels of current potential threats to soil biodiversity worldwide

Source: "Global Soil Biodiversity Maps" associated to the Global Soil Biodiversity Atlas, European Soil Data Centre, Joint Research Centre of the European Commission. June 2016.

This map (Figure 4.4.1) should be considered as a first attempt to locate sites with important potential threats to soil biodiversity at a global scale. However, its interpretation should be made with caution, given that the actual extent of the threats to soil biodiversity can be assessed only if we know what is present in these soils; and the expert assessment done for the world's regions (above) showed that there are many places

for which very little data or information is available. Clearly, further efforts are needed, taking into consideration various other sources of data that may have become available since then, and using data on other important threats identified in Table 4.4.1. Means of overcoming, or better considering, potential interactions among various threats is also an important issue to take into account in future attempts to better address threats to soil biodiversity worldwide.

Many environmental variables (such as temperature and land cover) can now be mapped and monitored for change relatively easily, using data collected by remote sensing (satellites). However, these still do not provide direct information on the state of the organisms present (diversity, populations). These must be derived from case studies performed throughout the world in the different ecoregions and include a range of taxa, with distinct functions in soils, so that the risk to soil biodiversity and function can be better assessed.

Finally, even if threats can be mapped with their extent of impacts on soil physical integrity and chemical quality as done by the FAO and ITPS (2015) in the *State of the World's Soil Resources* for various drivers (erosion, loss of SOM, soil nutrient depletion, contamination and pollution, soil acidification, salinization and sodification, waterlogging, compaction, crusting and sealing), syntheses of available data on impacts of these to soil biota (as many potential representative groups/taxa as possible) and support to obtain missing data are needed in order to produce accurate maps that reflect the true potential impacts of these threats on soil life worldwide. So far, these were generally found to be absent for most of the world regions, although efforts to map some more well-known taxa such as fungi, earthworms and nematodes worldwide have been made (Tedersoo *et al.*, 2014; Phillips *et al.*, 2019; van den Hoogen *et al.*, 2019), and these could be used as surrogates for the whole soil biota, to help produce more realistic models of impacts of threats on overall soil biodiversity. Nonetheless, even these still show important limitations in available data, particularly for tropical regions, which may limit application of the models where an important part of the world's soil biodiversity may be residing.

As globalization connects markets worldwide, source (producing countries) and sink (consuming countries) relationships may have important consequences to soil biodiversity that need further consideration. Countries with large areas devoted to intensive agriculture, or in the process of major intensification efforts to feed global markets, mostly dominated by commodities (such as sugar and soybean) may be experiencing especially important negative effects on soil biodiversity that need better attention. In addition, invasive species may become increasingly widespread as transport between countries and continents is stimulated by growing markets and improved by countries with emerging economies. These are just a few of a number of important issues that need to be addressed, particularly in international fora and agreements on climate change (IPCC) sustainable development, conservation of biodiversity (CBD) and ecosystem services (IPBES).

GLOBAL SOIL PARTNERSHIP

The Global Soil Partnership (GSP) is a globally recognized mechanism established in 2012. Our mission is to position soils in the Global Agenda through collective action. Our key objectives are to promote Sustainable Soil Management (SSM) and improve soil governance to guarantee healthy and productive soils, and support the provision of essential ecosystem services towards food security and improved nutrition, climate change adaptation and mitigation, and sustainable development.