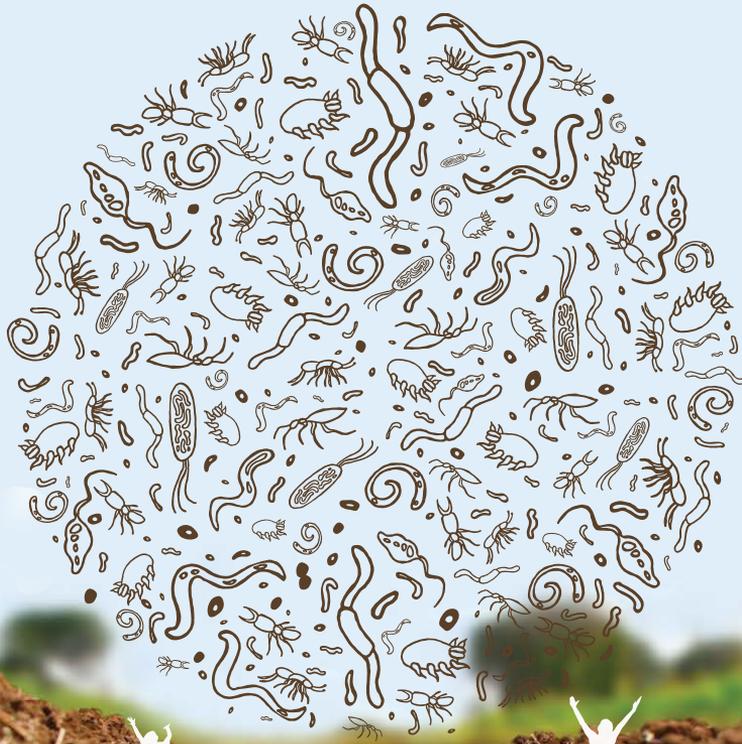




Food and Agriculture
Organization of the
United Nations

Report
2020

STATE of KNOWLEDGE of SOIL BIODIVERSITY



Status, challenges and potentialities



Convention on
Biological Diversity



European
Commission



GLOBAL
SOIL BIODIVERSITY
INITIATIVE

STATE of KNOWLEDGE of SOIL BIODIVERSITY

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CHAPTER 3

CONTRIBUTIONS OF SOIL BIODIVERSITY TO ECOSYSTEM FUNCTIONS AND SERVICES

3.1 | DEFINING SOIL FUNCTIONS

The definition of soil function and the determination of its boundaries is not an easy task. Various disciplines provide different perspectives; some focus only on the outcome, ignoring the real producers of these functions (see below). Moreover, the terms “functions” and “services” are also often confounded. Functions refer to natural-biological processes, while services refer to those functions that contribute to human well-being. Banwart *et al.* (2019) thoroughly described soil functions as “flows and transformations of mass, energy, and genetic information that connect soil to the wider critical zone, transmitting the impacts of human activity at the land surface and providing a control point for beneficial human intervention.” In simple words, the soil functional outcome is a result of interactions among physical, chemical, biological and human factors.

Soil functioning is principally defined by the parent material, the climate, and the topography, as these specify the physical and chemical environment and determine the conditions for living organisms (Dominati *et al.*, 2010; van Leeuwen *et al.*, 2019; O’Connor *et al.*, 2009). While the limits are set by external factors, the ability of soil to function and to support life is provided by soil organisms interacting in complex food-webs (Figure 3.1.1.1). One essential, complex ecosystem process that encompasses several soil functions and is supported by the whole soil food web is decomposition. Soil organisms are directly or indirectly (that is, through one or more trophic levels) involved in the decomposition of dead organic matter for covering their needs in energy, carbon and nutrients (de Ruiter *et al.*, 1994), but there are also other sources of carbon and nutrients deriving from roots and rhizodeposits/root exudates (Haichar *et al.*, 2014) that feed soil organisms involved in the soil food web.

Through their biological activity, soil organisms change their micro-environment. They transform complex chemical forms into simple molecules or compounds that can be absorbed by plants, providing feedback to plant productivity and re-growth. They structure soil by changing conditions of aeration and drainage and they control populations of other organisms, thus regulating above-ground biodiversity. Although their biological activities operate at small scales, their activities aggregate at larger scales, resulting in major functional outputs (Kibblewhite *et al.*, 2008) such as: a) carbon transformation, b) nutrient cycling, c) formation of soil structure and d) biodiversity regulation (biocontrol). These biodiversity-based soil functions define soil health, sustain soil ecosystem processes, influence above-ground diversity and contribute to climate regulation, and at the same time underpin regulating, supporting and provisioning ecosystem services that are essential for human well-being (Hedlund *et al.*, 2004; Lavelle *et al.*, 2006; Jeffery *et al.*, 2010; Bardgett and van der Putten, 2014; Orgiazzi *et al.*, 2016).

As a basis for understanding soil biodiversity and its role in ecosystem functioning, soil organisms have been approached from a functional diversity perspective in the context of the soil food web. Thus, soil organisms interacting in the soil food web are assigned to functional groups, based on characteristics such as their feeding source, their form, size, taxon and trophic level.

Each of these functional groups may contribute to one or more soil functions, but there is also high functional redundancy, (that is, a variety of soil organisms having the same functional ability). Decomposer organisms such as bacteria and fungi facilitate chemical breakdown of substrates and play a significant role in the functions of carbon transformation, soil structure formation and nutrient transformation. In the case of the latter, nitrogen-fixing bacteria and mycorrhizal fungi are significant players. Organisms that feed on microbes (for example, bacterivorous and fungivorous nematodes) regulate the populations of bacteria and fungi, and affect key ecosystem functions, such as nutrient transformation, indirectly. Together with organisms at higher trophic levels (for example, predatory nematodes, oribatid mites), members of the soil food web play a significant role as biological regulators. Organisms that feed directly on dead organic matter (detritivores) as well as other macro-faunal organisms (such as isopoda, earthworms) also affect several soil functions, and specifically soil structure formation, and are considered ecosystem engineers.

That said, human activities and interventions play a critical role on the outcome of soil functioning. Human-induced changes, such as the intensification of land use, lead to the modification of soil structure and abiotic properties as well as to tremendous changes in the structure, composition and diversity of the soil food web. Soil health (Kibblewhite *et al.*, 2008), as well as related Sustainable Development Goals (SDGs), depend on the maintenance of the four major biodiversity-based soil functions (carbon transformation, nutrient cycling, formation of soil structure, and biodiversity regulation). Unfortunately, these functions are recognized as being under threat (CEC, 2006; Gardi *et al.*, 2013). To overcome these obstacles and to sustain soil functions at specific levels, knowledge on

how soil food webs respond to specific management and restoration regimes under the perspective of global climate change is essential. To this end, it is crucial to focus research on better understanding the links among biodiversity attributes and soil functions and ecosystem services (see for example de Vries *et al.*, 2013), and among abiotic properties, soil organisms and climate (Bhusal *et al.*, 2015; Orgiazzi and Panagos, 2018), and to develop efficient monitoring tools and maps by up scaling the bio-indication potential to the scales that are important for management decisions (Stone *et al.*, 2016; van Leeuwen *et al.*, 2017; van den Hoogen *et al.*, 2019).



Table 3.1 | Examples of organisms, their main soil functions and ecosystem services, and gaps and opportunities

Organisms	Main Soil Ecosystem Functions	Ecosystem Services	Gaps and Opportunities
Megafauna (e.g. moles, beavers, armadillos)	Bioturbators Soil, organic matter and sediment redistribution to greater depths		
Macrofauna (e.g. earthworms, ants, termites, millipedes, insects)	Ecosystem engineers Fragment, rip, and tear organic matter, providing smaller pieces for decay by organisms; mineralization of organic matter; Bioturbators Moving and mixing soil, increasing water permeability and soil aeration	Supporting services: <i>Soil formation</i> <i>Nutrient cycling</i>	Lack of data and limited knowledge on: Organisms and their functional roles Interaction between organisms and their communities, as well as with terrestrial biodiversity (including crops) Distribution of soil organisms globally Better understanding of how biodiversity loss by anthropogenic activities might affect stability of food webs and ecosystem functioning Better understanding of the impact of climate change on soil communities
	Bioremediation <i>Eisenia fetida</i> earthworms can accumulate cadmium and some other pollutants like polycyclic aromatic compounds (PAHs)		
Mesofauna (e.g. mites, springtails)	Decomposers All mesofauna modify the fine-scaled structure of the soil through their feeding activities. Foodweb stabilizers Serving as predators, fungivores and preys to different communities	Regulating services: <i>Climate regulation</i> <i>Disease and pest regulation</i> <i>Water regulation (water availability, including regulating extremes – drought and flood)</i> <i>Remediation</i> <i>Pollination</i>	Improvement of biological control practices for pests and pathogens Opportunities: Undescribed microorganisms may have a wide range of metabolic capabilities influencing ecosystem services Application of soil microorganisms for specific industrial, agricultural or environmental purposes
Microfauna (e.g. tardigrades, nematodes)	Decomposers Recycle nutrients and increase nutrient availability for primary production Nitrogen fixers arbuscular mycorrhizal fungi (AMF) biologically fix atmospheric N ₂		
Microbes (e.g. virus, bacteria and archaea, fungi)	Bioremediation: Break down, removing, altering, immobilizing, or detoxifying various chemicals and physical wastes from the environment like PAHs (see chapter 5) Decomposers: Weathering minerals; Carbon transformation by decomposition of organic matter and storage, nutrient cycling by transforming inorganic compounds into forms usable by plants, regulate soil structure and pathogens	Provisioning services: <i>Food</i> <i>Freshwater</i> <i>Fuel</i> <i>Fibre</i> <i>Biochemicals</i> <i>Genetic resources</i>	
	Gas producers: Methanogenic archaea transfer of C, N ₂ , N ₂ O, CH ₄ denitrification		
	Nitrogen fixers: Rhizobia bacteria found on legume roots helping to increase nitrogen uptake.		

3.1.1 | THE SOIL FOOD WEB APPROACH

The soil food web approach provides a way to describe and quantify the soil biodiversity and its role in soil ecosystem functioning. The food web approach occupies a central position in community ecology. The strengths of food web interactions (that is, those between resource species and consumer species) affect the distribution and abundance of organisms in fundamental ways, since the success of populations is largely a function of benefits from the acquisition of food and losses from predation. In this way, food webs provide a way to analyze the dynamics and persistence of soil biodiversity. The food web approach also enables the analysis of the relationship between the structure of the soil biological community and soil ecosystem processing, as the food web interactions represent flows of matter, energy and information. For this reason, the soil food web approach contributes to two central aspects of soil biodiversity: its preservation; and its role in local, regional and global cycling of materials, energy and nutrients.

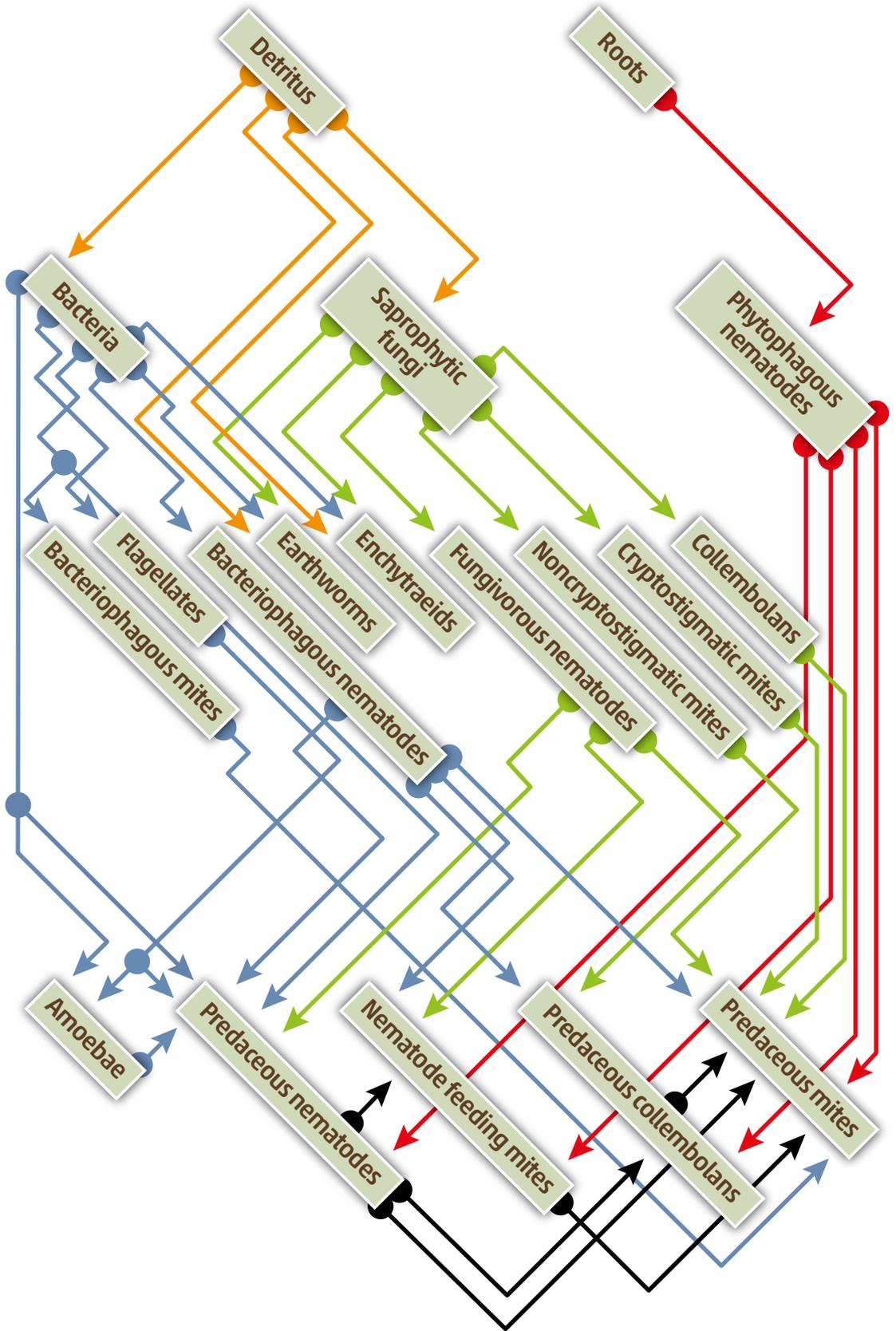
The *Soil Food Web* described soil biodiversity in the form of an ecological network. Figure 3.1.1.1 gives a schematic representation of a food web (de Ruiter *et al.*, 1994). Boxes denote the presence of a functional group. Functional groups aggregate species that have similar diets and life-history characteristics, such as growth and death rates and energetic efficiencies. Arrows denote feeding rates pointing to the consumers. The first trophic level in soil food webs consists of microorganisms; that is, bacteria and fungi, decomposing soil organic matter, and herbivorous nematodes feeding on roots. At the higher trophic levels, we see a large variety of faunal organisms: protozoa (amoebae, flagellates, ciliates), nematodes (bacterivores, fungivores, omnivores, herbivores and predators), mites (bacterivores, fungivores, predators), collembola (fungivores and predators), enchytraeids and earthworms.

The soil food web diagram also reflects the large variety in soil organisms in connection to the large heterogeneity in soils (Moore and Hunt, 1988). The red arrows in the soil food web diagram form the so-called bacterial channel. In this channel, the fluxes of material all originate from bacterial productivity. The pink arrows form the fungal channel and originate from fungal productivity. The green arrows form the root channel, originating from root productivity. The black arrows form linkages among the different channels. The red channel are mostly aquatic organisms such as bacteria, protozoa, and nematodes, living in the water film around soil particles. The fungal channel are mostly terrestrial organisms – fungi, mites, and insects. Exceptions are the bacterivorous mites in the bacterial channel and the fungivorous nematodes in the fungal channel. These energy channels show that the soil organisms form one biological community, while different components of the community can be organized in relatively separated and independent ways; that is, in soil food web compartments.

Soil food webs provide an excellent model for studying the biological mechanisms underlying soil ecosystem functioning. Soil food webs are directly responsible for soil processes that originate from the decomposition of soil organic matter and the food consumed and processed in the other trophic interaction in the soil food web (de Ruiter *et al.*, 1994). Together, the organisms in the soil food web are directly responsible for soil ecosystem processes, such as carbon sequestration and nutrient cycling. Given the sizeable amounts of materials they process, soil food webs are responsible for major components in the global cycling of materials, energy and nutrients (Wolters *et al.*, 2000, Griffiths *et al.*, 2000). Furthermore, soil food webs play a key role in terrestrial systems in their ability to provide services in the form of food productivity (Brussaard *et al.*, 2007) and the conservation of above-ground biological diversity (Hooper *et al.*, 2000; de Deyn *et al.*, 2003).

Figure 3.1.1.1 (next page) | Diagram of a soil food web

Boxes denote the presence of a functional group. Functional groups are aggregations of species on the basis of similar diets and life-history traits. Arrows denote feeding rates pointing to the consumers. The colour of the arrows refer to 'energy channels', which are defined on the basis of the primary resource of food chains within the food web. **Red** arrows: root channel, **orange** arrows: detritus channel, **blue** arrows: bacterial channel, **green**: fungal channel, **black**: energy channels linked by omnivorous and polyphagous organisms



3.1.2 | THE MULTIFUNCTIONALITY OF SOIL BIODIVERSITY

It has been long debated whether soil biodiversity is superfluous insofar as many species are redundant. Currently evidence is increasing that there is less functional redundancy than previously supposed. For example, mycorrhizal fungi form symbiotic associations with the majority of plant roots and help plants to acquire limiting nutrients such as phosphorus and nitrogen (van der Heijden *et al.*, 2015). Nitrogen-fixing bacteria associate with legume roots and fix large amounts of nitrogen that are of pivotal importance for plant productivity in a wide range of ecosystems across the world. Other soil organisms decompose organic matter (Heemsbergen *et al.*, 2004; Lindahl *et al.*, 2006), contribute to soil carbon stabilization, soil aggregate formation and soil structure (Rillig and Mummy 2006), cause disease (van der Putten *et al.*, 1993) which can enhance plant species diversity in nature (van der Putten *et al.*, 2013), or suppress disease (Mendes *et al.*, 2013). Moreover, soil biota break down contaminants such as pesticides (Fenner *et al.*, 2013), produce antibiotics (Nesme *et al.*, 2014), clean water that percolates through the soil profile, and prevent leaching of nutrients into ground and drinking water (Bender and van der Heijden, 2016). Thus, it is important to consider that soil organisms play multiple roles in ecosystems and influence multiple ecosystem functions (multifunctionality).

Experiments carried out in microcosms (van der Heijden *et al.*, 1998; Philippot *et al.*, 2013; Wagg *et al.*, 2014; Rillig Semchenko *et al.*, 2018) and at global observational scales (Jing *et al.*, 2015; Delgado-Baquerizo *et al.*, 2016) revealed that soil biodiversity and soil microbial diversity promotes or is directly linked to ecosystem multifunctionality and functioning. These observations mirror results obtained in plant communities that revealed that plant diversity enhances ecosystem sustainability (Tilman *et al.*, 1996) and multifunctionality (Hector and Bagchi 2008). A recent study showed that grassland microcosms with poorly developed soil microbial networks and reduced soil microbial richness had the lowest multifunctionality (Wagg *et al.*, 2019). This was due to fewer taxa present that support the same function (redundancy) and lower diversity in the abundance of taxa that support different functions (reduced functional uniqueness; Wagg *et al.*, 2019). This indicates that high soil biodiversity provides an insurance that is required to sustain multiple functions (*sensu* Yachi and Loreau, 1999). Interestingly, an increasing number of studies also show that cross-kingdom interactions between various groups of soil biota (for example between bacteria and fungi or between different guilds of soil arthropods) and soil food web structure and composition are of pivotal importance for plant health (Van der Putten *et al.*, 2006; Duran *et al.*, 2018), decomposition (Heemsbergen *et al.*, 2004), or overall ecosystem functioning (Wagg *et al.*, 2019). Microbial decomposition is traditionally supposed to be the result of non-specialized microbes. However, recent work shows considerable specialization among decomposers (Veen *et al.*, 2017), making decomposers less functionally redundant than was previously supposed.

Soil biota are involved in multiple functions simultaneously, and thus functional redundancy is likely to fade as more functions or environmental conditions are considered. For example, in a mesocosm study, extreme drought or extreme rainfall conditions revealed that part of the microbial community is more drought-tolerant, while the other part is more tolerant to extreme moisture (Meisner *et al.*, 2018). Most likely, when a larger variety of environmental conditions are eventually examined, there will be less functional redundancy in the soil community than originally assumed. Also, the loss of soil bacteria has been shown to negatively affect the nitrogen cycle (Philippot *et al.*, 2013), of which some aspects are only carried out by few species (Swift *et al.*, 1998).

To understand how changes in soil biodiversity affect ecosystem functioning, it is therefore important to consider not only whether the total number of taxa present relates to a function, but how the reduction in the number of species that support a single function relates to the loss of multiple functions simultaneously. In addition, the influence of an individual species on an ecosystem function is not independent of other species present and is a result of a myriad of positive and negative, direct and indirect associations among the different species that as a whole drive ecosystem functioning. In conclusion, there is increasing evidence that soil biodiversity plays a pivotal role in the functioning of the Earth's ecosystems. However, precise mechanisms and general patterns are still unclear and there is poor knowledge about how to promote and use soil biodiversity for the development of sustainable agricultural ecosystems.

3.2 | DEFINING SOIL ECOSYSTEM SERVICES

Ecosystems are living systems, which interact with one another and their surrounding environment. They provide benefits, or services, to the world. Ecosystem services, then, are the multitude of benefits that nature provides to society. Ecosystem services are classified into supporting, provisioning, regulating, and cultural ecosystem services. Despite an estimated value of USD 125 trillion (Costanza *et al.*, 2014), these assets are not adequately accounted for in either political or economic policies, which means there is insufficient investment in their protection and management.

Biodiversity losses can negatively affect the supply of ecosystem services. Even if soil organism biomass is very low compared with the mineral fraction, the activity is crucial for a functioning soil because the soil organisms support many fundamental processes and enhance key ecosystem provision services, such as food and fibres, water quality, biodiversity conservation, and ecosystem supporting services such as nutrient cycling and soil structure formation (Kibblewhite *et al.*, 2008; Pulleman *et al.*, 2012; El Mujtar *et al.*, 2019). Biodiversity, as with ecosystem services, must be protected and sustainably managed (FAO, 2019). The links from soil and soil biodiversity to ecosystem services are not well described despite the underlying significance for soil organisms, soil functions and soil biodiversity. A conceptual scheme of the relationship between soil biodiversity and ecosystem services is provided in Figure 3.2.1.

Soil biodiversity is generally under pressure due to some threats such as intensification of agriculture, which has a negative impact on several provisioning and supporting ecosystem services (Tsiafouli *et al.*, 2017). Bender *et al.* (2016) highlighted that agriculture has the potential to restore the sustainability of agricultural ecosystems by stimulating soil life and regulating ecosystem processes. Despite this collective understanding, however, soil biodiversity still receives little recognition in agricultural management strategies and in support of ecosystem services. Forest ecosystems also benefit from below-ground soil biodiversity for the vast majority of interactions studied among provisioning, regulating, supporting and cultural ecosystem services (Bakker *et al.*, 2019).

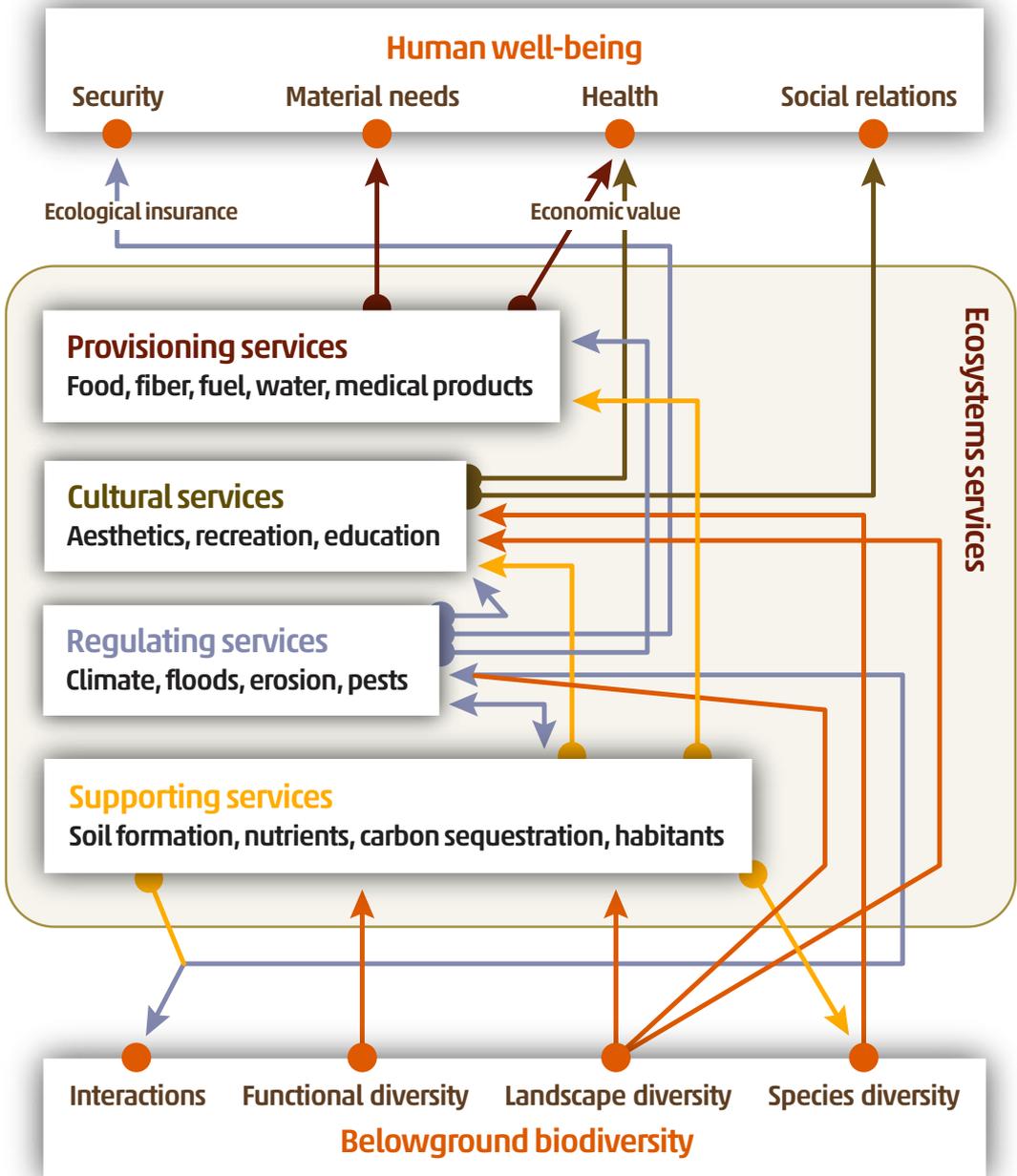


Figure 3.2.1 | Relationship between soil biodiversity and ecosystem services

A conceptual scheme of the relationship between soil biodiversity, ecosystem services and human wellbeing (Bakker *et al.*, 2019, modified from Scholes *et al.*, 2010).



3.3 | SOIL BIODIVERSITY SUPPORTS THE *SUSTAINABLE DEVELOPMENT GOALS*

Soil biodiversity is integral to sustaining all life on Earth, especially humanity. With the adoption of the 2030 Agenda for Sustainable Development, United Nations Member States have agreed to work toward accomplishing 17 ambitious Sustainable Development Goals (SDGs) globally to improve and sustain life for people and the natural world on which we depend. In this chapter, we explore how soil biodiversity plays a key role in meeting many of these SDGs (Figure 3.3.1).

Although the SDGs do not refer directly to soil, the strong connection between soil biodiversity and the SDGs is clear. Many SDGs such as food security, water scarcity, climate change, biodiversity loss and health threats are closely linked to or dependent on soil biodiversity (Table 4.0.1, Keesstra *et al.*, 2016). Deterioration of relationships between humans and soil has resulted in unsustainable management of agricultural, forest and urban ecosystems, leading to environmental degradation and major societal consequences. Soil cannot be untangled from its biodiversity, though these intimate interconnections are not always recognized

Soil biodiversity supports human health and well-being (SDG 3) through regulation of many disease-causing organisms as well as playing a central role in agricultural production, supporting SDG 2 (no hunger) as well as agricultural livelihoods contributing to SDG 1 (no poverty), SDG 5 (gender equality), SDG 8 (decent work and economic growth) and SDG 12 (responsible consumption and production). Life in the soil mediates water flow and uses excess nutrients and pollution, advancing SDG 6 (clean water and sanitation) and SDG 14 (life below water). Carbon cycling from soil biological activity, both in terms of C sequestration and greenhouse gas emissions, are central to regulating climate (SDG 13). The soil contains arguably the most diverse terrestrial communities on the planet, and is home to more than 25 percent of global biological diversity. Moreover, it supports most life above ground via increasingly well understood above- and below-ground linkages (SDG 15). Evidence supports these vital connections between soil biodiversity and the SDGs, and emerging knowledge is shaping actions we can take for a more sustainable future. Soil organisms play many important roles in managing invasive species. Soil biodiversity can help avoid, reduce, and reverse land degradation, sustaining and improving habitat for people and other life on Earth. Long taken for granted, soil biodiversity should be embraced as part of the urgent need to develop a more sustainable future for all.

Flood regulation



Climate regulation



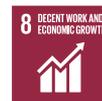
Foundation for human infrastructure



Provision of construction materials



Provision of food, fibre and fuel



Habitat for organism



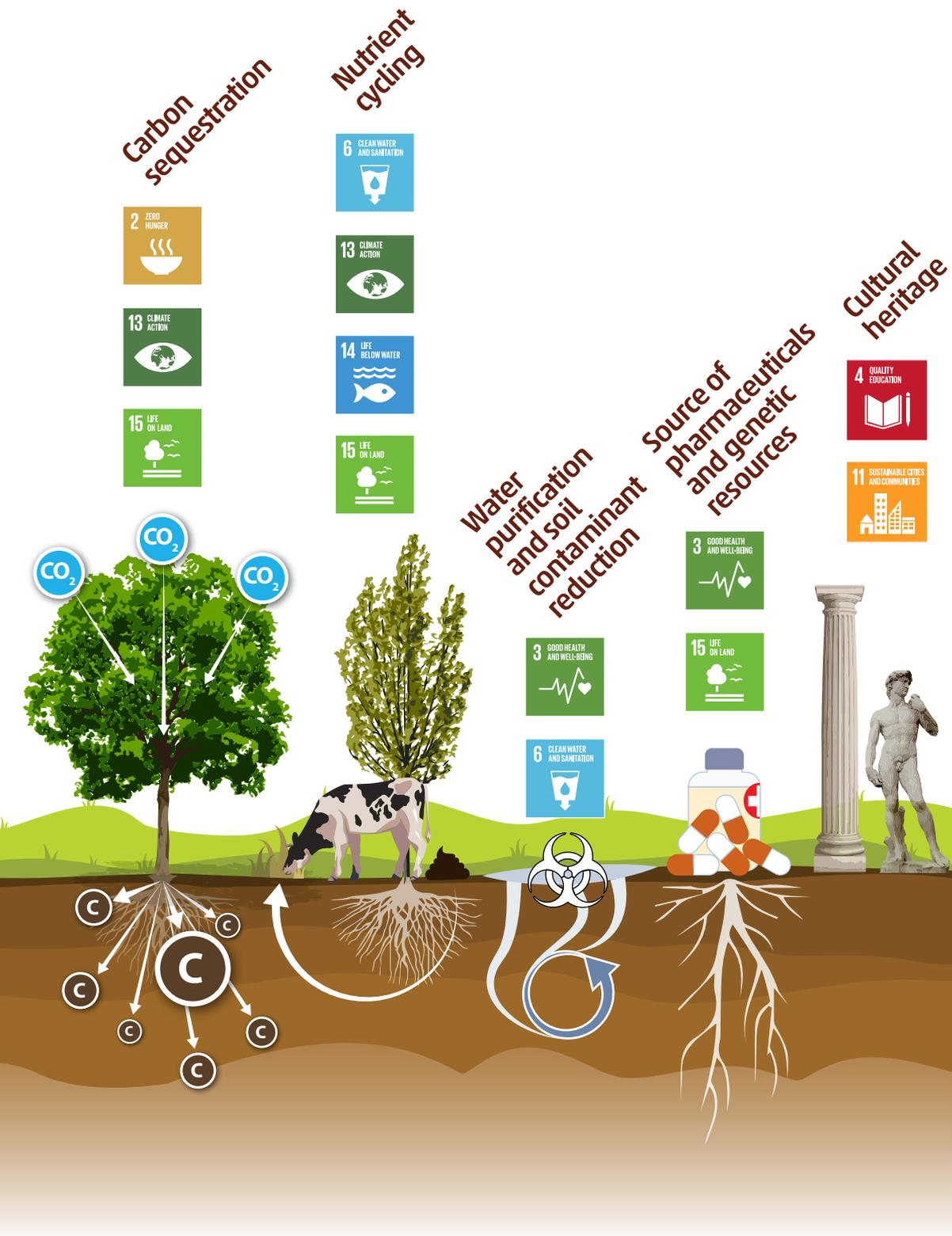


Figure 3.3.1 | Healthy soils, a prerequisite to achieve the SDGs

A healthy soil is capable of providing most ecosystem services and therefore achieving compliance with SDGs and human well-being.

Box 3.3.1 | Soil Biodiversity and the SDGs

Food Production (SDG 2): Achieving global food security is one of the greatest challenges of our time (Burns *et al.*, 2010; Lobell *et al.*, 2008). The mismatch between the rate of human population growth and global food production has long been recognized (Malthus 1798). We need to double global food production in the coming decades, on less land and using less inputs. The majority of the world's human population eats a soil-grown, plant-based diet. Even where diets are rich in animal products, livestock rely on soil-grown plants (fish being a notable exception). The quantity and nutritional quality of crops is very much a product of the soils in which they grow. The link between crop production and soil quality is well established. For example, that 30 percent of the world's human population is affected by Zn deficiency (Alloway 2008) is not surprising, given that the soils of almost half of the world's cereal growing regions are low in Zn (Cakmak, 2002; Graham and Welch, 1997). Taken together, it is clear that efforts focused on achieving food security cannot ignore the link between soil and food.

Water quality (SDGs 6, 14): Nutrient exports from agricultural systems are a large contributor to water quality issues in many parts of the world. For example, reports of up to 160 kg of nitrogen (N) and up to 30 kg of phosphorus (P) per hectare can be lost via leaching and surface runoff (Herzog *et al.*, 2008; Sims *et al.*, 1998). When nutrients reach waterways, they can contaminate groundwater and can also cause eutrophication which can lead to algal blooms and the loss of terrestrial and aquatic biodiversity (Carstensen *et al.*, 2014). Soil biota play an important role in regulating the movement of water into and through soil as well as cycling of nutrients in the soil and water. This can impact their risk of nutrients being lost via leaching. Similarly, some soil microbes (such as mycorrhizas) play an important role in helping plants to interact with such nutrients, thereby reducing the risk of nutrient leaching (Cavagnaro *et al.*, 2015). Wider use of agricultural practices that leverage soil biota to manage nutrient availability and loss could contribute significant progress toward water quality goals

Air quality (SDGs 3, 13): Soil plays an important role in air quality and soil greenhouse gas emissions. When agricultural soils are tilled, the increased oxygen can spur biological activity and respiration of carbon dioxide that can contribute to global warming. Moreover, certain soil microbes under anaerobic conditions (such as flooded or very wet soils) can transform nitrate into the potent greenhouse gas nitrous oxide. Similarly, other soil microbes can release methane from soil, which also contributes to global warming. Soil microbes have also been reported to help purify air (Khaksar *et al.*, 2016). It is also worth noting that soil microbes and soil fauna can also help to bind soil particles together and improve soil structure in some situations. In doing so they can reduce the risk of wind erosion, thereby helping to reduce levels of dust in the air we breathe.

Soil health (SDGs 2 and 3): The prevention of soil pollution could reduce soil degradation, increase food security, contribute substantially to the adaptation and mitigation of climate change, and contribute to the avoidance of conflict and migration. By taking immediate actions against soil pollution we can thereby contribute to the achievement of almost all the SDGs, with a significant impact on goals 2 and 3.

Climate mitigation (SDG 13) and adaptation: Soil, and the biota it supports, have a central role to play in climate mitigation, and agricultural soils play a key role because they cover such a large portion of the earth. In addition to reducing greenhouse gas emissions from soil (see air quality above), the soil has the potential to sequester large amounts of carbon. Agricultural practices such as reducing tillage and maximising plant cover can improve soil biological activity and C sequestration. Not only should we mitigate climate change, we must also adapt to it. Soil-provided ecosystem services can help to buffer systems against (resistance) and rebound from (resilience) external perturbations (Jackson *et al.*, 2003).

Affordable and clean energy (SDG 7): Biofuel can be produced using plant-based feedstocks, using plants grown expressly for this purpose or grown for other purposes and waste materials used in energy production systems. For example, waste materials, such as those from forestry, food processing, wine production, and many others can be processed in various ways (such as pyrolysis, gasification, liquefaction) to extract the energy they contain. The production of such materials for the most part has their genesis in the soil (for example, crop residues). Moreover, many of these processes produce bioproducts that can potentially be used as a soil amendment (often rich in carbon) and (potentially) nutrients (Dreake *et al.*, 2016). Furthermore, such materials can provide habitat for soil organisms.

Above-ground biodiversity/biodiversity loss (SDG 15): It is increasingly well understood that above- and below-ground communities are closely linked, and that a change in one can affect the other. For example, a reduction in below-ground diversity can reduce above-ground plant diversity (van der Heijden *et al.*, 1998). Similarly, changes in above-ground vegetation can alter below-ground communities. For example, in a recent meta-analysis it was found that by reducing soil tillage and planting a cover-crop, the formation of beneficial mycorrhizal associations (symbiosis between plant roots and soil fungi that improve plant nutrient acquisition)

was improved (Bowles *et al.*, 2017). Soil fauna including nematodes, collembola, and mites have been shown to increase grassland succession and plant diversity (De Deyn *et al.*, 2003).

Gender Equality (SDG 5) and Decent Work and Economic Growth (SDG 8): Women are important members of farming communities around the world. Men and women relate to land differently, and their unique perspectives are driven by varying roles, responsibilities, access to resources and control. Understanding these roles, along with power relations in land management, is a primary requirement to achieving effective outcomes when addressing sustainable agricultural development (Chotte *et al.*, 2019). Efforts to mainstream gender as proposed by the United Nations Convention to Combat Desertification Gender Action Plan (GAP) as well as recommendations from the United Nations Framework Convention on Climate Change (UNFCCC), the United Nations Convention on Biological Diversity (UNCBD), UN Women, the International Union for Conservation of Nature (IUCN), and the Convention on the Elimination of All Forms of Discrimination against Women (CEDAW) among others convey the importance of gender equality and gender inclusive action in supporting agricultural production and protecting land from degradation. The numerous recommendations promoting gender equality and human rights and empowerment of women and girls in environmental governance have been proposed by the United Nations Environment Assembly (UNEA, 2019). Women form a major part of agricultural development (UNCCD, 2017b) with traditional knowledge and skills in farming being closely tied to the maintenance and improvement of land productivity (UNCCD - Global Land Outlook: Gender-responsive land Degradation Neutrality, 2017). Women's contributions can include knowledge and respect for soil organisms and their role in supporting farming practices. These vital roles of women need to be understood and addressed, on the one hand to enable communities to support women as farmers and as leaders, and on the other hand to ensure that men and women benefit equally, and that inequality is not perpetuated. It is important to enable more equal access to natural resources and to facilitate women in becoming active users and managers of natural resources (Okpara *et al.*, 2019), goals that are interlinked with SDG 5 and SDG 8.

3.4 | PROVISIONING ECOSYSTEM SERVICES

Provisioning ecosystem services are produced within agriculture, forestry and fisheries and include food, fodder, fibres and wood, fresh water, raw materials, and genetic medicinal and ornamental resources (Adhikari and Hartemink, 2016). The significance and support of soil biodiversity of these provisioning ecosystem services will be the focus in this section.

3.4.1 | NUTRIENT CYCLING

Nutrients are chemical elements that are used by plants and other organisms for their growth. Soils are the major terrestrial reservoir of organic carbon and nutrients such as nitrogen and phosphorus. The soil nutrient pool includes macronutrients that plants need in a large quantity (such as nitrogen, phosphorus, and potassium) and micronutrients, which are needed in smaller quantities (iron, manganese, boron, molybdenum, copper, zinc, chlorine, nickel, sulfur, and cobalt). Nitrogen is a critical nutrient for plant growth, since it is part of the chlorophyll molecule, and plants require nitrogen for amino acid and protein production. Phosphorus is present in DNA and is important in photosynthesis,

respiration, healthy root systems and seed development. Potassium is used in photosynthesis, protein synthesis, regulation of water usage, and disease resistance. Micronutrients are required as cofactors for enzyme activity. Many scientific studies have focused on the ratio C:N:P for quantifying the fluxes of elements between compartments and then highlighting the functioning of a given ecosystem (for example, Zechmeister-Boltenstern *et al.*, 2015). A deficiency of one of the nutrients may result in decreased plant quality and/or productivity. As a consequence, nutrient deficiency can induce the reduction of overall biodiversity since plants underpin above-ground and below-ground food webs.

Nutrient cycling is the transformation of nutrients contained in minerals from the original bedrock and in dead biomass into simple molecules or compounds that are assimilable by plants and other organisms. For instance, nitrogen inputs are naturally made through soil organic matter (SOM) transformation by decomposer organisms and biological fixation from the atmosphere through symbiosis between bacteria and plants, especially legumes. Organic matter also provides natural chelates that maintain micronutrients in forms that plants can uptake. All soil organisms contribute to nutrient cycling through a number of physical (fragmentation of litter, bioturbation and transport of soil particles) and chemical (organic matter mineralization, transformation of molecules by a large number of enzymes) processes. Plant uptake of nutrients stimulates plant growth, in turn increasing the amount of plant-based inputs back into the soil. In total, 80 to 90 percent of primary production enters the below-ground system via detritus (Bardgett, 2005; Coleman *et al.*, 2017). Dead plant material (litter, woody debris, roots), animal excrements and carrion make up the majority of potential sources of nutrients for plants, soils and their biodiversity. Soils also receive inputs of nutrients from human activities. For instance, agricultural soils can receive large inputs of fertilizer (mineral and/or organic) K, N or P; N and S can be indirectly added at the soil surface of acid rain while NO_x and SO_x can enter soils through from the burning of fossil fuels.

Turnover processes and nutrient movement (mineralization, immobilization) depend on the climate, physical properties (texture, structure) and chemical parameters (pH, SOM, cation exchange capacity) of soils, plant cover and plant chemical composition, and taxonomical and functional diversities of soil animals and microorganisms (Colman and Schimel, 2013; Coleman *et al.*, 2017). There are still many gaps in our understanding of the mechanisms of the process of decomposition and its regulation. Nutrient cycling can be altered by soil moisture content and temperature, since these factors directly impact microbial biological activity. In this context, global warming can modify nutrient cycling such as N pools and fluxes, leading to higher N losses from soil because of the increase of net N mineralization (Bai *et al.*, 2013). Furthermore, soil degradation is one of the four major global challenges currently facing humanity and has been shown to impact the interrelated biogeochemical cycles of C, N, and P (Quinton *et al.*, 2010; Berhe *et al.*, 2018). Urbanization can also have an impact on nutrient cycling through indirect effects (such as increase of temperature, increase of N deposition because of traffic) and direct effects through management practices that alter bio-physico-chemical soil parameters

(Lorenz and Lal, 2009; Bittman *et al.*, 2019). Finally, non-native plant invasions can also alter nutrient cycling (Jo *et al.*, 2017). Long-term field experiments might help to follow potential acclimation of biodiversity and thus evolution in nutrient cycling functioning in the face of human disturbance. (Figure 3.4.1.1).

According to the studied spatial scale, excess of nutrients can induce toxicity for plants, but also alteration of global nutrient cycling (Lavelle *et al.*, 2005). For instance, in N cycling, plants and microbes take up nitrate produced by transformation of SOM, but a significant amount may also be lost from soil by leaching or runoff, especially in the case of excess fertilizer. This leads to eutrophication of soils, ground waters and streams. Systems of crop-livestock production are the largest cause of human alteration of the global N and P cycles (Bouwman *et al.*, 2013), since it is common for farmers to apply nutrients in excess to make it more available to crop plants. (Figure 3.4.1.1). Global studies show that P deficits covered 29 percent of the global cropland area, and 71 percent had P surpluses. This nutrient overloading causes not only environmental damage but also a financial loss. Supporting soil users in any interventions to reduce nutrient overuse is a first necessary step.

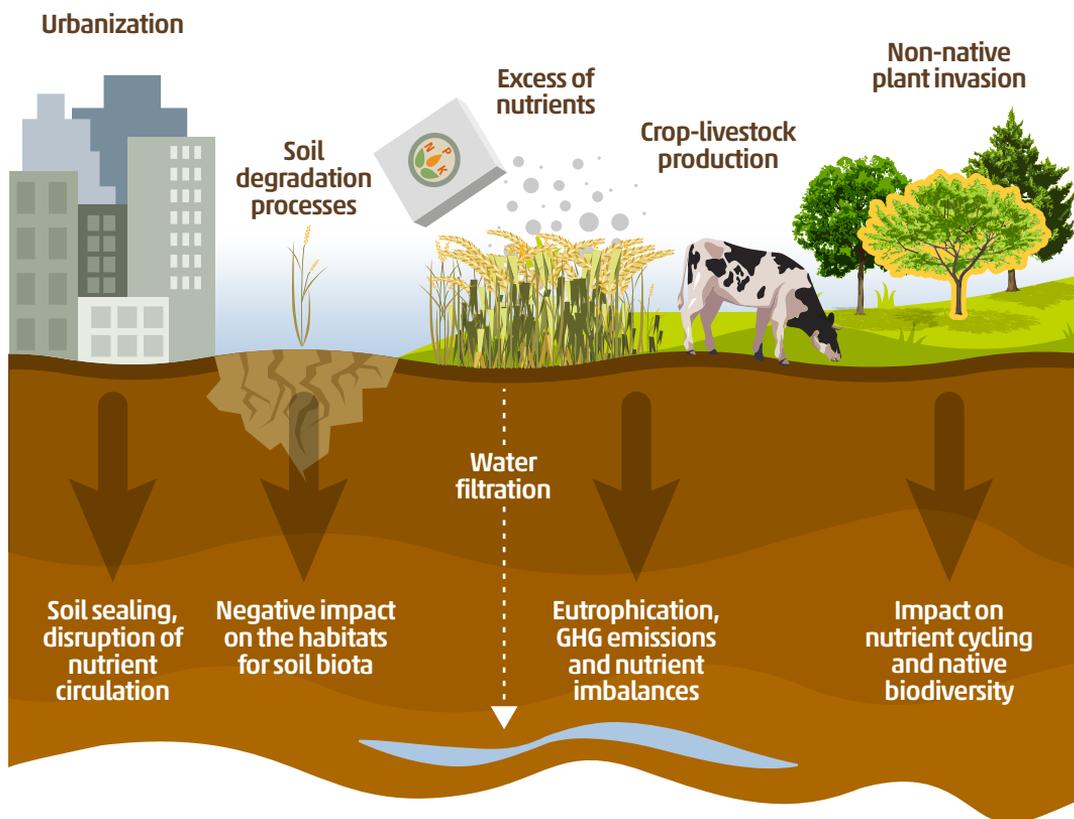


Figure 3.4.1.1 | Main drivers for the alteration of global cycles

Main drivers for the alteration of global cycles and their side effects on the environment.

3.4.2 | FOOD PRODUCTION

Achieving global food security is one of the greatest challenges of our time (Burns *et al.*, 2010, Lobell *et al.*, 2008). The mismatch between the rate of human population growth and global food production has long been recognized (Malthus, 1798). We need to ensure food production to meet future demands, on less land, using fewer inputs and safeguarding ecosystems. Achieving this goal in a time of significant environmental change makes the challenge all the more significant. Food security by its very definition demands that enough food of sufficient quality be produced. We need enough food to meet the energy and nutritional requirements of the global human population (SDG 2) while ensuring environmental sustainability.

The majority of the world's human population eats a soil-grown, plant-based diet. Even where diets are rich in animal products, livestock rely on soil-grown plants (fish being a notable exception). The quantity and nutritional quality of crops is very much a product of the soils in which they grow. The link between crop production and soil quality is well established. For example, that 30 percent of the world's human population is affected by Zn deficiency (Alloway, 2008) is not surprising, given that the soils of almost half of the world's cereal growing regions are low in Zn (Cakmak, 2002; Graham and Welch, 1997). Taken together, it is clear that efforts focused on achieving food security cannot ignore the link between soil and food.

Agriculture comes in many shapes and forms, ranging from high-input industrial-scale agriculture to subsistence farming. In some cultures, soil invertebrates are an important direct source of protein (Paoletti *et al.*, 2000). Most of the world's agriculturists are subsistence farmers, where the food farmers eat is the food they produce. Whereas industrialised large-scale agriculture depends heavily on complex resources (such as inputs of water, nutrients, and pesticides) and distribution and supply chain and networks, subsistence farmers may not have ready or reliable access to supply chains and markets. They lack the access to resources that can be used to boost production (such as mineral fertilizer) or to protect crops (such as pesticides), or to participate in markets.

The lack of access to industrial inputs (such as synthetic fertilizers and pesticides) means subsistence farmers are organic farmers by default (Cardoso and Kuyper, 2006). As such they rely heavily on soil biota and the ecosystem services they provide to support production. Similarly, soil biota play an important role in high-input agricultural systems. For example, soil organisms play a key role in nutrient cycling, including the transformation of nutrients into forms that are more or less available to plants (ammonium versus nitrate), more readily leached into waterways (nitrate), or converted into greenhouse gasses (nitrous oxide). Soil biota also play a key role in the cycling of C in soils, including increasing soil C which can help mitigate climate change, while improving soil structure and hence water retention and thereby reduce risk of soil erosion. Further, soil biota can form beneficial associations with plants that can symbiotically fix N (rhizobia and legumes) and take up and deliver nutrients including P, Zn and N to plants

(mycorrhizas). Soil biota can also be significant pests and pathogens that cause significant crop losses. Taken together, soil biota provide important ecosystem services that every form of agriculture relies on, or is affected by, in differing ways and to different extents.

The development of high-yielding crop varieties and access to pesticides and synthetic fertilizers has resulted in substantial increases in global food production following the Green Revolution (Khush, 1999). However, if we are to achieve global food security in a sustainable manner, much more needs to be done (Lynch, 2009). Aside from the fact that many of the world's farmers do not have ready access to synthetic fertilizers, there are also increasing concerns around access to easily mined sources of some mineral fertilizers (such as P) and the embedded energy in others (such as N). This has significant implications for large-scale industrial agricultural systems that rely on such inputs. Consequently, attention is turning to alternative sources of nutrients, such as compost, manure and crop residues (Cavagnaro, 2015). In addition to providing nutrients, organic materials can also increase soil C, thereby providing other benefits. While such sources of nutrients may be more readily available to subsistence farmers (and indeed are currently in use), they rely on soil biota to make the nutrients in them available to plants (via mineralization; Ng *et al.*, 2014). Moreover, this reliance on soil biological processes to drive and regulate nutrient supply to plants can make organic amendments less predictable compared to synthetic fertilizers.

Plant growth-promoting bacteria (PGPB), phyto-bacteria found in the soil that stimulate plant growth and improve soil and plant health, were described over one hundred years ago (Lugtenberg and Kamilova, 2009; Backer *et al.*, 2018; Olanrewaju *et al.*, 2017). A subset of the PGPB are the plant growth-promoting rhizobacteria (PGPR) which colonize the plant's roots. PGPR have increased nutrient uptake, provided biocontrol, and improved plant health. Likewise, drought- and saline-tolerant plants have been linked to microbes that provide the plants with ability to tolerate these abiotic stresses. When these microbes were added to susceptible cultivars and then exposed to these stresses, the plant biomass and yield increased compared to the non-inoculated stressed plants (Naylor and Coleman-Derr, 2018; Kearl *et al.*, 2019).

3.4.3 | RAW MATERIALS

Production of raw materials from biomass, especially wood, fibres and biofuels, are the major provisioning services of soil following food production. Fibres are used for such products as textiles and rope, while forestry has been a major provider of raw material for thousands of years for heating, building materials and even decoration. Within the last decades, wood, agricultural crops and agricultural waste have increasingly been used for biofuel production. It is anticipated that the use of raw materials for biofuel production will increase in the near future to meet the demands of reduced GHG emissions

(Hoekman and Broch 2018; Qin *et al.*, 2018). Hence, these raw materials are very important to produce carbon neutral energy.

The production of the majority of raw materials has the same or even more severe environmental issues as does agriculture for food production with use of agrochemicals, such as increasing risk of erosion and reducing soil carbon. This is due to the increase in monoculture and removal of all organic matter from the agricultural field (Hoekman and Broch, 2018).

Various climate mitigation initiatives of green energy (supporting SDG 13) by using raw plant and wood material for biofuel potentially change the land and soil management. This was described by Mishra *et al.* (2019), who evaluated the ecosystem services in alternative landscape scenarios. Land use systems that integrate woody vegetation with livestock and/or crops represent high nature and culture value (HNCV) agroforestry, have higher above-ground biodiversity (Moreno *et al.*, 2018), reduce erosion risk and nutrient leaching, and have high production of raw materials. Crops (including trees) that are grown for bioenergy production often have a longer growth time, which generally results in increased soil carbon content. Thus, HNCV agroforestry will probably benefit soil biodiversity.

3.4.4 | CLEAN WATER

Essential for agriculture and critical for the survival of people, water quality and availability are crucial for numerous SDGs, from SDG 6 (clean water and sanitation) and SDG 14 (life below water) to SDG 2 (zero hunger) and SDG 3 (good health and well-being) for its essential role in agriculture and food security. However, the pollution of water and the increasing frequency and severity of droughts and floods resulting from climate change, unsustainable management of natural resources and inadequate sanitation constitute major threats to our human societies and to biodiversity in general.

The influence of soil biodiversity on water dynamics and quality is often complex and varies with the environment. The influence of microorganisms is usually indirect and results from their impact on soil organic matter dynamics, which in turn affects soil aggregation and soil porosity dynamics as well as the composition of the soil solution (such as the amount of dissolved organic carbon and minerals). An exception to the rule is, however, biological crusts produced by cyanobacteria and mosses that control water dynamics in arid environments (Ram and Aaron, 2007). Conversely, larger soil organisms have a direct impact on soil structure through their ability to move inside the soil, then impacting a large number of ecological functions, including those that control water infiltration, diffusion and retention, as well as the susceptibility of soil to resist to wind and water erosion. Among soil organisms, animals from the soil macrofauna group (that is, soil invertebrates larger than 2 mm) such as earthworms, termites and ants are

considered to play important roles in controlling soil structure dynamics (Bottinelli *et al.*, 2015). They are commonly named soil “bioturbators” and “engineers” because of their large population and activities in temperate and tropical ecosystems (Lavelle *et al.*, 1997; Jouquet *et al.*, 2006). Their foraging and burrowing activities, as well as their ability, in the case of social insects, to create nest structures with specific soil properties in soil, has an important impact on soil structure dynamics and hence on water dynamics and quality in soil. Although less widespread, other soil macro-invertebrates can also significantly impact soil structure and water dynamics because of their role in alteration of porosity, soil surface microrelief, soil aggregation and bulk density – for example, beetles’ larvae (Brown *et al.*, 2010; Cheik *et al.*, 2019) and millipedes (Toyota *et al.*, 2006; Fujimaki *et al.*, 2010).

The sensitivity of ants to environmental disturbance, combined with their great functional importance as ecosystem engineers, make them not only powerful monitoring and assessment tools in restoration programs but also key actors in ecological engineering (Bulot *et al.*, 2014). The size of the entrance of ant nests as well as their architecture also have been shown as impacting bulk density (Cammeraat *et al.*, 2008) and water infiltration (Eldridge *et al.*, 1993); indeed, large nests decrease bulk densities and increase water infiltration. At a smaller spatial scale, soil mesofauna (that is, soil invertebrates smaller than 2 mm) can actively make microtunnels in the soil, and they play a role in soil structure alteration through litter comminution, casting and other mechanisms of microdisintegration. Their interaction with microorganisms can also indirectly influence soil structure. However, in comparison with earthworms, termites and ants, the effects of these smaller organisms remains poorly known.

Soil macrofauna can influence soil hydrological properties at different scales of observation and through antagonistic processes. At a small scale, any changes in clay and soil organic matter contents, as well as in soil porosity, are likely to influence water-holding capacity. At a medium scale, the production of a dense network of foraging galleries connected to the soil surface usually improves water infiltration. This network can occupy a significant volume of soil in some situations (Capowiez *et al.*, 1998; Mando *et al.*, 1999; Buhl *et al.*, 2004; Perna *et al.*, 2008), and obviously these large macropores are of primary importance in the regulation of water infiltration, the diffusion of solutes and therefore water quality (Ehlers, 1975; Nkem *et al.*, 2000; Léonard and Rajot, 2001; Cammeraat *et al.*, 2002; Dominguez *et al.*, 2004; Zehe *et al.*, 2010). However, the translocation of soil on the ground in the form of earthworm casts, termite and ant sheeting or mounds can either increase soil roughness and favour water infiltration, or rapidly generate structural crusts that foster water runoff and soil detachment (Jouquet *et al.*, 2012; Bargués Tobella *et al.*, 2014). The degradation of mound nests by rain or organisms (for example, bears and pangolins breaking termite nests for feeding purposes) can also form seals that locally reduce water infiltration (Traoré *et al.*, 2019). Therefore, the final impact of soil fauna on soil hydrological properties results from the balance between processes favouring soil infiltrability (such as the formation of open galleries on the soil surface) and those fostering the formation of impermeable erosion crusts and the detachment of soil aggregates. Finally, at a larger scale, the concentration of nutrients

and the presence of specific vegetation on mounds produced and maintained by soil fauna activity can also have an impact on the hydrological characteristics of watersheds in certain situations. They can contribute to better water infiltration, preferential flow and groundwater recharge (Ackerman *et al.*, 2007; Bargués Tobella *et al.*, 2014). Recent studies also suggest that termite mounds may increase the robustness or resilience of African dryland ecosystems against water shortage and desertification (Bonachela *et al.*, 2015).

Compared to the abundant literature focusing on the impact of soil macrofauna on soil structure, the effect of soil organisms on water quality remains poorly studied. Soil bioturbation and the creation of interconnected soil macropores by soil macrofauna can increase nutrient leaching below the root zone. This has been observed mainly with earthworms and using short-term experiments and lysimeters, where the addition of earthworms significantly increased the amount of soil leachate and the leaching of NH_4^+ , NO_3^- and dissolved organic nitrogen in cultivated environments (Subler *et al.*, 1997; Dominguez, 2004; Jouquet *et al.*, 2010). Bioturbation by earthworms can also impact the mobility and leaching of heavy metals from casts (Udovic *et al.*, 2007), but in an experiment carried out in controlled conditions, Amossé *et al.* (2013) did not find any significant impact on the amount of bacteria and viruses in water. These examples are, however, to be considered with caution because the impact of soil fauna is often site- and species-specific, because interactions between organisms have never been taken into account, and because these studies were carried out in controlled or semi-controlled conditions, making it difficult to generalize from them.

Two examples of the utilization of soil biodiversity for improving ecosystem functioning or for the rehabilitation of degraded lands can be found in (i) the intensification and externalization of earthworm activity and (ii) the stimulation of termite activity in the field. The intensification and externalization of earthworm activity using epigeic earthworms is at the core of the production of vermicompost and the treatment of liquid manure. In the last decades, vermitechology has been applied to the management of various types of wastes and sludge, to convert them into vermicompost for increasing land fertility. Although this practice is gaining in popularity, surprisingly few studies have questioned their impact on water quality and availability. Studies carried out in northern Viet Nam showed that the leaching of mineral nutrients and then the pollution of water can be significantly reduced if mineral nutrients or compost are replaced by vermicompost (Jouquet *et al.*, 2011; Doan *et al.*, 2013). Similarly, vermifiltration is an efficient and low-cost technology for processing organically polluted water and reducing the transfer of minerals and pathogens to the aquatic system (Li *et al.*, 2008; Morand *et al.*, 2011).

In arid or semi-arid environments, bioturbation is mainly carried out by termites. The ability of termites to promote water infiltration in crusted soils as part of soil rehabilitation and vegetation cover regeneration has been strikingly demonstrated in Africa (Kaiser *et al.*, 2017), Asia (Pardeshi and Prusty, 2010) and Australia (Dawes, 2010). In these studies, the application of mulch or organic matter on or into the soil, as in the case of the agricultural and forestry “zai” systems (see Roose *et al.*, 1999 for a description of the Zai

agricultural practice), triggered termite activity which then created burrows through the crusted soil surface. Increase in termite activity can therefore result in an enhancement in water infiltration by a factor of 1.5 to 25, an increase in water retention and a reduction in the bulk soil density of the upper soil layer (Cheik, 2019).

Since the capacity of soil biodiversity to enhance the water quality and quantity is indirect, research studies are scarce and the awareness of this capacity is low. Better understanding of the functional roles of soil biodiversity could help stakeholders to remediate soils by considering soil organisms in their sustainable ecosystem restoration plan. Moreover, it is obvious that the interactions between microbial and faunal communities and between below-ground and above-ground compartment should be considered in ecological restoration context. In this vein, long-term study cases should be carried out *in situ* to help scientists and stakeholders to go forward together and find indicators of restoration success. There is also the need to create a global database of management projects to connect the initiatives around the world. In parallel, transdisciplinary approaches, such as putting together ecologists and economists, might increase awareness. Finally, one hot topic to focus on could be emerging pollutants such as plastics and microplastics and their interaction with soil biota, in order to find a solution to fight this issue by considering both soil and water.

3.4.5 | SOIL BIODIVERSITY AND HUMAN HEALTH

Ever since the nineteenth-century microbiologist Louis Pasteur discovered that soil can harbour *Bacillus anthracis*, the bacterium that causes Anthrax, scientists have recognized that life in the soil is a source of human disease (Thorkildson *et al.*, 2016). From the perspective of biomedicine, *risk* and *disease* continue to be the primary focus of studies of the link between soil and human health (Jeffrey *et al.*, 2011). But recently, a countervailing narrative has emerged: far from making us sick, most soil creatures help protect and maintain our health (Wall *et al.*, 2015).

As discussed elsewhere in this report, soil biodiversity helps prevent erosion, filter and conserve water, and break down environmental pollutants. It also plays a role in capturing and storing atmospheric carbon—which might help fight climate change. Even if this were all soil organisms did, they would clearly be central to our well-being and survival on this planet. But emerging research suggests that soil biodiversity has a more direct impact on our health by boosting the nutrient content of our food, protecting us from foodborne illness, and modulating our immune response. This section focuses on these three associations and discusses how greater collaboration between health science and agriculture is needed to better understand these links, and to protect soil and human health.

Soil biodiversity, nutrient cycling, and nutrition

The phytobiome – a region surrounding the roots of plants comprised of non-living structures, plants, and micro- and macro-fauna – influences plant yield and nutrition and, by extension, human health and nutrition (Leach *et al.*, 2017). Plants secrete compounds that feed nearby organisms, and those organisms have a dynamic exchange with the plant. This relationship enables plants to capture essential minerals and to manufacture various chemicals, such as antioxidants, that protect them from pests and other stressors (Garcia *et al.*, 2018). When we consume these plants, these antioxidants benefit us by stimulating our immune system, regulating our hormones, and slowing the growth of human cancer cells (Manach *et al.*, 2004).

The abundance and profile of microorganisms can vary across plant habitats and plant genotypes, but one consistent finding is that biodiversity within the phytobiome hastens plant growth, increases plant yield, and increases plant nutrient density. Complementing these findings, several studies show that organic farming methods (which are known to promote soil biodiversity) produce plants with a higher concentration and variety of antioxidants (Reganold *et al.*, 2010; Barański *et al.*, 2014; Ren *et al.*, 2017). While there is much to discover in the phytobiome, this complex zone of plant-organism-soil interaction has the potential to enhance food security and improve human nutrition.

Soil biodiversity and food safety

Soil biodiversity helps mitigate the risk of foodborne illness by boosting plant defenses against opportunistic infections. For example, *Listeria monocytogenes* is found in low concentration in many agricultural soils, but its pathogenicity depends on the richness and diversity of soil microbial communities (Vivant *et al.*, 2013; it also depends on soil type, pH and other soil factors). In one study, bacteria spread rapidly in autoclaved soil but was suppressed in soil containing an abundance of other species. Other research shows that small groups of rare species might have a disproportionately large effect on counteracting pathogenic soil organisms (Cernava *et al.*, 2019). In general, plants surrounded by greater diversity of microbes in their root structure have greater fitness and resilience against pests and pathogens (Johnson *et al.*, 2013). Understanding and promoting these soil dynamics could help protect humans from foodborne illness.

Soil biodiversity and immune modulation

In the early 1900s, scientists began identifying antibiotic substances in soil that could fight specific microbial infections or more generally modulate human immune response. Since then, a wide range of therapeutic drugs and vaccines have been derived from (or inspired by) substances produced by soil organisms. For example, amphotericin, the principal systemic medication used to treat invasive fungal infections, was first recovered from a strain of bacteria growing in the soil of a riverbed in Venezuela. Bleomycin, also originally identified in soil, is used as chemotherapy for a variety of common cancers (Vitorino *et al.*, 2017).

While most biopharmaceutical research is focused on identifying unique microbes (or their by-products) that can be developed into biotherapeutics, new technologies that make it possible to study the metagenome – or collective genome – in an environmental sample have sparked an interest in exploring how complex microbial communities in soil and other indoor and outdoor environments influence human immune and nervous response via the skin, gut, and lungs. The route of exposure and the profile of microbes vary from study to study, but species diversity consistently emerges as an important health mediator. This may be due to the fact that we co-evolved with a constellation of environmental microbes (often referred to as “old friends”) and they continue to communicate with our internal physiology. (Langgartner *et al.*, 2019).

A series of studies focused on farm children in Europe and the United States of America illustrates this phenomenon. Researchers observed that children raised on traditional dairy farms (farms that use little fossil fuel or chemical inputs) have lower rates of allergic disease than children raised on conventional farms. Findings suggest that messages from organisms found in soil and on farm animals “program” the immune system and determine how farm children respond to allergens later in life. This programming likely starts *in utero* and continues through the first few years of life (von Mutius, 2016; Stein *et al.*, 2016; von Mutius, 2018).

Separate evidence suggests that environmental microbial inputs from soil and other niches can bolster resilience in humans and modify the risk for neurodevelopmental and psychiatric pathology. This line of inquiry is in its infancy but suggests that early exposure to a diverse collection of soil microorganisms might help prevent chronic inflammatory diseases including allergy, asthma, autoimmune diseases, inflammatory bowel disease, and depression (Lowry *et al.*, 2016).

Transdisciplinary collaboration is needed to protect human health

Soil biodiversity plays a central role in preserving human health through a range of pathways including water purification, climate stabilization, nutrient and food security, and immune regulation. Additionally, the dramatic global rise in food-related chronic diseases – including obesity, diabetes, stroke and heart disease – can largely be traced back to crops produced using soil-degrading agricultural practices (Franck *et al.*, 2013; Mattei *et al.*, 2015; Smith *et al.*, 2015). Such practices include encroaching on wild space, deforestation, contaminating and diverting water supplies, monocropping, excessive tilling and the overapplication of pesticides, herbicides and fertilizers. Ongoing efforts to do true cost accounting have increased awareness about the direct and indirect health costs generated by these agricultural practices and the cost savings that would accrue from regenerating soil and promoting human health from the soil up (<http://www.fao.org/nr/sustainability/full-cost-accounting/en/>). By most estimates, these savings (measured in years of disability and health care expenditures) could be enormous (IPES Food report, 2017).

Despite this evidence of a strong link between soil health and human health, it is rare for health institutions – international, national, or regional – to take an active and visible role in protecting soil. The World Health Organization, for example, has issued dozens of white papers on the dangers of soilborne illness (with a particular focus on helminthiasis) but only one statement encouraging Good Agricultural Practices (GAP) and other activities that build soil biodiversity (WHO guidelines, 2003). In the United States of America, the National Institute of Food and Agriculture (NIFA) awards nutrition research grants through their Agriculture and Food Research Initiative (AFRI), but these rarely attract human health researchers. A few non-profits and academic centres, including Health Care Without Harm, the Planetary Health Alliance and One Health Initiative, attempt to bridge the artificial divide that exists between soil health and human health, but these groups have limited resources and influence. We need more collaboration between the health and agricultural sectors to protect soil and to address mounting health crises related to climate change, soil loss, and industrial farming.

Health institutions urgently need to educate their members about the role soil plays in human health, and about activities (including those outlined elsewhere in this report) that preserve soil or contribute to its destruction (Dwivedi *et al.*, 2017). They need to participate in transdisciplinary research and innovation that builds and preserves soil structure and biodiversity. And most importantly, the health sector needs to leverage its immense influence and financial resources to shape policies, public health education programs, land use planning, and procurement practices that create healthy soil and that protect this rapidly disappearing health resource.

Environmental exposures are responsible for 70 percent of morbidity and mortality world-wide. Allergies, chronic lung diseases and other conditions caused or triggered by environmental factors are attributable to a subject's genetic responsiveness to distinct environmental stressors. Thus, it is obvious that environmental quality is a key factor driving human health, which has been acknowledged in SDG 3 (ensure healthy lives and promote well-being for all at all ages). Well-known examples of such complex interactions include emerging infectious diseases, food safety, the selection of antimicrobial resistant pathogens and the large number of pollutants from the environment, which strongly impact our health (Xi *et al.*, 2017). This is in fact not a new finding but has been already proposed almost 2 500 years ago by Hippocrates. He claimed as one of the first that our health is triggered by the kind of land where we are living (Brevik and Sauer, 2015).

Nowadays it is well accepted, both by scientists and decision makers and by some members of the general public, that our health will be more and more difficult to maintain on a polluted planet with diminishing resources (Destoumieux-Garzón *et al.*, 2018): a document provided in 2017 as a result of a joint meeting of the European Union ministers for environment and health, as well as the World Health Organization stated that “environmental factors that could be avoided and/or eliminated cause 1.4 million deaths per year in the EU” (Landrigan *et al.*, 2018). The authors postulated that “public

authority shares the common responsibility for safeguarding the global environment and for promoting and protecting human health for all environmental hazards across generations and in all policies.” This has been summarized as the “One Health Concept”, which is an important contribution to the discussion of how to improve and sustain our health on a planet that will soon host 10 billion people. As one of the first consequences, “One Health” has been implemented in the framework of ecosystem services provided by soils (Keith *et al.*, 2016). In addition, the new focus on interactions between humans and their environment as a trigger for health and disease also stimulated new research efforts in medicine to unravel fundamental processes of immunity, repair and regeneration, tolerance and disease development. It is obvious that a consequent implementation of the “One Health Concept” into medicine requires a very strong personalized component, taking individual exposure levels with different environmental factors into account as well as individual genetic disposition. Based on such data, strategies for a general improvement of environments should be developed in combination with new personalized strategies for therapy of diseases caused by losses in environmental quality. Some important examples of how the abiotic and biotic soil factors can influence human health are given in the following.

Soil microbes for human, plant and animal health

A handful of soil may contain between 10 and 100 million microorganisms, belonging to more than 5 000 different taxa (Ramirez *et al.*, 2015) and providing a wealth of metabolites with many applications. Due to the immense potential that soil microorganisms hold for pharmaceutical industries, bioremediation, and agriculture, characterizing and preserving soil biodiversity are essential in order to facilitate the discovery of new drugs and therapeutic measures for treating and controlling diseases. The object of the present subsection is to present examples of scientific observations illustrating the impact of soil microbes on human health and on disease prevention. Once transferred to the public discourse, these observations and other similar findings could coalesce into a solid argument in favour of soil biodiversity preservation.

An important contribution of soil microorganisms to human and animal health is the provision of antibiotics. An antibiotic is a low molecular weight product from a microorganism, with the capacity to kill or inhibit the growth of other, susceptible, microorganisms. As antimicrobial agents, antibiotics are used widely to treat infectious diseases (Aminov, 2010; Crofts *et al.*, 2017). Another potential use of soil microorganisms in health care stems from viruses known as bacteriophages, which are parasitic non-cellular agents that can kill their bacterial host, and therefore are being investigated as a cure for bacterial infections (Burrowes *et al.*, 2011; Frampton *et al.*, 2012; Jones *et al.*, 2007). Harboured an immense diversity of viruses, soils can be considered as a promising reservoir of bacteriophages useful to phage therapy. While this subsection mainly deals with antibiotic-producing microorganisms and bacteriophage with potential for phage therapy, it should be mentioned here that soils also serve as reservoirs of fermentative bacteria and fungi useful to food processing, and of health-sustaining components of the gut microbiota (on this last point, see Blum *et al.*, 2019).

Soil microorganisms for disease control

Soil microorganisms have been contributing to protect humans and animals against infectious diseases for thousands of years, particularly through antibiotic production. Although industrial-scale production of antibiotics started only after the 1940s, human exposure to antibiotics has a very long history. Ancient human populations could have been exposed to antibiotics through a number of mechanisms: (i) application of soil on skin; (ii) traditional remedies (composed of soil and herbal products); (iii) ingestion of material contaminated with soil or antibiotic-producing microorganisms through diet; (iv) direct ingestion of soil for medicinal or spiritual purposes (referred to as ‘*geophagy*’).

For instance, the presence of tetracycline has been detected in human skeletal remains from ancient Sudanese Nubia that date back to 350–550 CE (Nelson *et al.*, 2010) and in skeletal remains from the Dakhel Oasis, Egypt, that date back to the late Roman period (36 BCE to 400 CE) (Cook *et al.*, 1989). Scientists suggest that lack of traces of bone infections in skeletal remains from the ancient Sudanese Nubian population and the Dakhel Oasis indicate that exposure to antibiotics may have provided some protection against infectious diseases in ancient civilizations (Cook *et al.*, 1989; Nelson *et al.*, 2010).

In another example, the presence of antibiotic-producing actinomycetes was reported in Red Soil from Jordan, which was used for thousands of years for treating skin infections (Falkinham *et al.*, 2009; Aminov, 2010). In a study conducted with Jordan Red Soil, *Staphylococcus aureus* and *Micrococcus luteus*, bacteria that cause skin infections, were applied to sterilized and unsterilized soil and a bactericidal effect on these pathogens was observed with unsterilized soil only (Falkinham *et al.*, 2009). In the same study, a number of antibiotic-producing *Lysobacter*, actinomycetes and *Bacillus* species effective against *M. luteus*, *S. aureus*, *Escherichia coli*, *Mycobacterium smegmatis*, *Saccharomyces cerevisiae* and *Aspergillus niger* were isolated from the soil. Guo *et al.* (2015) reported that acidic Red Soil of China hosts various groups of bacteria producing secondary metabolites with antimicrobial properties.

In addition to antibiotics, other biotic factors such as protozoan or myxobacterial predation, lytic microorganisms and bacteriophages, as well as antimicrobial abiotic components such as clay minerals and ions of zinc, iron, copper, silver and other metals, may also provide disease control traits to soils used in traditional medicine (Otto *et al.*, 2013).

The modern antibiotic era

Prior to the discovery of antibiotics, infectious disease control depended on inorganic or synthetic antimicrobial agents such as inorganic mercury salts, copper salts, organo-arsenic derivatives of Atoxyl (Salvarsan and Neosalvarsan) and sulfonamidochrysoidine (Prontosil), with many side effects and poor efficacy (Aminov, 2010). The production of antibiotic compounds by *Pseudomonas aeruginosa*, a bacterium frequently reported in soils, was known by the late 1890s, when Emmerich and Low obtained an antibiotic called pyocyanase from extracts of this bacterium. It was used in hospital for treatment

of bacterial diseases, but was eventually abandoned owing to inconsistent results and to side effects. In 1929, Alexander Fleming discovered the antibiotic activity of *Penicillium*, a fungus commonly found in soils. It was not until 1940 that the technique for mass production of penicillin was developed by Howard Florey and Ernst Chain. Penicillin is earmarked as the first antibiotic successfully trialed and mass produced. Its therapeutic use launched the modern antibiotic era. As the techniques developed by Fleming to identify antibiotic-producing microorganisms came into general use, a wide variety of antibiotics was discovered in the following years. Currently, nearly 80 percent of all antimicrobials in use are of microbial origin, and soils are the primary source of bacteria and fungi producing new molecules potentially useful to human, animal and plant disease control (Grossbard, 1952; van Elsas *et al.*, 2008; Falkinham *et al.*, 2009; Aminov 2010; Guo *et al.*, 2015; Rogozhin *et al.*, 2018).

Antibiotic discovery was at its peak from the 1950s to the 1970s, and all the classes of antibiotics we know of today were discovered during this period. Since then, the rate of new discoveries has declined (Aminov, 2010; Crofts *et al.*, 2017). Antibiotics are currently used in prevention and control of infectious diseases in human healthcare, animal husbandry, veterinary care, cropping (mainly fruit, vegetable and flower production), and postharvest management of perishables (McManus *et al.*, 2002). Lack of policies and of proper regulations have resulted in mismanagement and misuse of antibiotics in the above-mentioned sectors (McManus *et al.*, 2002; O'Neill 2016), with antibiotic residues being released in soil and water environments.

The consequences of widespread antibiotic use

In 1945, Alexander Fleming cautioned about potential resistance development to penicillin (Aminov, 2010). It is now commonly accepted that all microorganisms will eventually develop resistance to any antibiotic. The widespread use and misuse of antibiotics maintains high selection pressure for antibiotic resistance in a variety of environments (including human and animal gut, soil, waterbodies and others). The spread of antibiotic resistance has become one of the major challenges facing human and animal health care, with acquisition of multidrug resistant traits by aggressive human pathogens being of concern (Crofts *et al.*, 2010). According to the Food and Agriculture Organization of the United Nations (FAO), antimicrobial resistance by disease-causing microorganisms is responsible for 7 million human deaths annually on a global scale (O'Neill 2016). O'Neill (2016) emphasizes “that by 2050, 10 million lives a year and a cumulative USD 100 trillion of economic output are at risk due to the rise of drug-resistant infections if we do not find proactive solutions now to slow down the rise of drug resistance.”

The soil is the recipient of a spectrum of antimicrobial resistance determinants (namely residues and genes) that we release in the environment. Some soil-dwelling heterotrophic bacteria belonging to orders *Burkholderiales*, *Pseudomonadales* and *Actinomycetales* use antibiotics as a carbon source (Dantas *et al.*, 2008), implying that their activity might decrease the antibiotic residue concentrations in soil environments.

Phylogenetic evolution, rather than horizontal gene transfer, is thought to determine the genetic structure of the “soil resistome” (van Goethem *et al.*, 2018). Hence, the genetic flow of antibiotic resistance genes would be largely vertical in the soil, and as such would be uncoupled from the mostly horizontal flow occurring in clinical settings (Forsberg *et al.*, 2014). This observation, associated with the ubiquity of antibiotic resistance in diverse environments, with the genetic relatedness of the soil and of the human gut resistomes (Nesme and Simonet, 2015) and with the capacity of gut bacteria such as *Escherichia coli* to survive for extensive periods in soils (Joergensen and Wichern, 2018), suggests that biodiversity and antibiotic resistance are two independent characteristics of the soil microbiome, influencing one another. The effect of antibiotic resistance determinants on soil biodiversity has been investigated (Martinez 2009; van Goethem *et al.*, 2018), and several studies have emphasized the importance of antibiotic resistance genes for bacterial ecophysiology at the ecosystem level (see the review by Nesme and Simonet, 2015). Reciprocally, diverse populations of soil organisms have been postulated to limit pathogen numbers (Brevik *et al.*, 2018; Wall *et al.*, 2015), and the same may be true for survival of antibiotic-resistant bacteria passing from the gut to the soil.

Opportunities and knowledge gaps

Until recently, discovery of antibiotic-producing microorganisms depended on culture-based technologies. However, the high-resolution power of metagenomics led researchers to take full measure of the depth and breadth of soil biodiversity and to realize that over 90 percent of soil microorganisms, including many of ecological importance, escaped detection by culture-based techniques (van Elsas *et al.*, 2008; Xin *et al.*, 2015). A comparison of culture-based and metagenomic characterization of soil fungi from a dry, mixed evergreen forest indicated that the fungi commonly occurring in culture, such as *Aspergillus*, *Penicillium* and *Trichoderma*, formed a lesser percentage of the detected fungal community when the diversity was analyzed using metagenomics, as compared to a culture-based approach (Dandeniya and Attanayake, 2016). Through culture-based methods, at most ten different fungal types were distinguished based on colony morphology, whereas the metagenomic analysis indicated a hundred-times greater diversity. *Pestalotiopsis*, *Cladosporium*, and *Chaetomium*, genera known to include antibiotic producers, were among the fungi reported in these forest soils (Dandeniya and Attanayake, 2016). Many other studies yielded similar observations, showcasing the soil as a virtually untapped source of antibiotic-producing microorganisms.

Soils from extreme environments appear particularly promising in this respect. Extreme conditions require unique metabolic adaptations. Rogozhin *et al.* (2018) isolated a new antibacterial and antitumour substance from a strain of the fungus *Emericellopsis alkalina* obtained from an alkaline soil. Working with soils with pH values ranging from 2.6 to 6.6, Guo *et al.* (2015) identified a diverse group of acidophilic actinobacteria producing a wide range of novel antimicrobial compounds.

In addition to promising discoveries of new antibiotics from highly diverse soil microbial communities, soil biodiversity holds other potentialities for medical practice. While

it remains largely unexplored, the biodiversity of soil viruses offers the promise of bacteriophage therapy for alternative treatment of bacterial infections in humans and plants (Burrowes *et al.*, 2011; Frampton *et al.*, 2012; Jones *et al.*, 2007). Reported viral abundance in soils ranges from 2.2×10^3 gdw⁻¹ (grams per dry weight) in desert sands from Saudi Arabia to 5.8×10^9 gdw⁻¹ in forest soil from eastern Virginia in the United States of America, with the majority of soil viruses being bacteriophages (defined as viruses infecting bacterial hosts). Distinct viral communities exist in different soils, with most communities being dominated by tailed bacterial viruses (Williamson *et al.*, 2017). For the control and prevention of infections caused by *Pseudomonas aeruginosa* (such as pneumonia, bloodstream infections, urinary tract infections and surgical site infections) such tailed bacteriophages isolated from environmental water, hospital wastewater and sewage wastewater have been used (Pires *et al.*, 2015). Soil is a largely underexplored, yet promising, environment for isolating bacteriophages for phage therapy.

Soil microorganisms contribute to the healthy living of humans, animals and plants. Several soil bacteria and fungi are being used traditionally in the production of soy sauce, cheese, wine and other fermented food and beverages. Lactic acid bacteria that could potentially be used to produce heavy metal (cadmium and lead) -absorbing probiotic products have been discovered from mud and sludge samples (Bhakta *et al.*, 2012). Soils provide habitats for a variety of lactic acid bacteria belonging to *Lactobacillus*, *Lactococcus* and other genera (Chen *et al.*, 2005), opening the possibility that probiotic bacteria useful in food fermentation or other processes be isolated from soils.

Preserving the richness of soil biodiversity will keep the gates open for healthy living.

3.5 | REGULATING ECOSYSTEM SERVICES

Regulating ecosystem services are services by which ecosystems act as regulators of the natural processes that make life possible (such as clean air and water or waste management). They are often invisible and hence underappreciated. When they are damaged, the resulting losses can be substantial, and the services can be very difficult to restore. Soil biodiversity is essential for the continuous function of the regulating services.

3.5.1 | CLIMATE REGULATION

The regulating service of soil on climate is mainly focused on GHG emission and uptake (that is, emission of carbon dioxide, methane and nitrous oxide) and uptake

or sequestration of carbon, and is important for SDG 13 (climate action). For climate regulating services, soil biodiversity has an important impact. With the ambitions of reducing GHG emissions to combat climate change, this ecosystem service has come into focus as a key function to achieve the Paris Agreement goals. Agriculture, forestry and natural environments all have significant impacts in this regard.

Globally, agroecosystems contribute 10 to 12 percent of all direct anthropogenic GHG emissions each year, with an estimated 38 percent resulting from soil N₂O emissions and 11 percent from CH₄ in rice cultivation (Smith *et al.*, 2008, 2014). These emissions come largely as a result of intensification that relies heavily on agrochemicals (fertilizers, herbicides and pesticides) and has consequently reduced functional group diversity in soils, particularly for larger soil biota (Postma-Blaauw *et al.*, 2010). In addition, the practice of recirculating organic waste and manure to the field also increases emissions of GHG. This has reduced the biological capacity for soils to self-regulate, leaving them vulnerable to changing environmental conditions (Brussaard *et al.*, 2007; Tsiafouli *et al.*, 2015). While these issues have been long recognized (Black and Okwakol, 1997; Giller *et al.*, 1997), specific attempts to design agricultural practices around sustainably managing soil biodiversity are relatively recent (Bender *et al.*, 2016). However, early indications suggest that this soil ecological engineering can create more resilient (Ciancio and Gamboni, 2017; Dubey *et al.*, 2019), adaptive (Birgé *et al.*, 2016) and resource-efficient (Leff *et al.*, 2016; Bowles *et al.*, 2017) agroecosystems that in turn reduce associated GHG emissions (Pagano *et al.*, 2017). Indeed, organic farming approaches that limit/exclude agrochemical use can reduce soil N₂O emissions by up to 40 percent (Skinner *et al.*, 2019). Similarly, new approaches in rice cultivation indicate that targeting cultivars to increase root porosity can increase methanotroph abundance (Jiang *et al.*, 2017), and application of purple non-sulphur bacteria can increase competition with methanogens for substrate availability (Kantachote *et al.*, 2016); each can significantly reduce CH₄ emissions relative to conventional practices. Current initiatives in agricultural practices are investigated with the aim of reducing climate negative impact of agriculture, especially focusing on reducing GHG emissions (Olesen *et al.*, 2018). These initiatives include adding nitrification inhibitors to manure and chemical fertilizers to reduce nitrous oxide emissions, control storage of manure and modifying barn conditions to decrease methane emission. (3.5.1.2) These initiatives are all dependent on well-functioning soil biota.

In addition to direct effects of climate change on soil organisms and biodiversity, indirect effects through above-ground vegetation changes could also threaten soil biodiversity. For example, there is accumulating evidence that warmer and drier conditions will benefit vascular plants at the expense of the peat-forming vegetation such as *Sphagnum* mosses (see for example Dieleman *et al.*, 2015). A recent study (Juan-Ovejero *et al.*, 2019) has shown that, under this scenario, mites in detriment of the more hydrophilic species (for example, enchytraeids dipteran larvae and many collembolans) will increasingly dominate soil communities, due to their different physiological adaptations to water stress. Similar shifts have also been observed in fungal communities, from ericoid mycorrhiza and

arbuscular mycorrhiza under shrubs and grasses to ECM in forests (Hagedorn *et al.*, 2019). These functional shifts in soil biota communities are likely to exacerbate C losses from these systems through mining for nutrients.

The report produced by the Convention on Biological Diversity (CBD Technical Series no. 10) concluded that functionally diverse communities with high biodiversity would be able to better adapt to climate change than impoverished ones. However, in these C rich systems, where a low number of species performs soil functions and hence where ecosystem stability is low, designing activities that aim to conserve these key functional groups should be a high priority. This urgent activity, along with preventing habitat loss, fragmentation and degradation, will increase resilience of these vulnerable ecosystems and in turn will result in a greater likelihood of adapting and mitigating to climate change. Indeed, the first proposed target to achieve SDG 13 on climate action is to strengthen resilience and adaptive capacity to climate-related hazards and natural disasters in all countries.

◇ **Box 3.5.1.1 | Climate Change and Soil Biodiversity** ◇

Climate change and the conservation of biodiversity are recognized globally as being a common concern of humankind (UNFCCC, CBD). The ways we manage our land and soil, and its biodiversity, have the potential to have a profound influence at a local, regional and global scale. The relationship between climate change and soil biodiversity is complex. We are only beginning to scratch the surface of understanding the connections, feedbacks, threats and opportunities posed by the interactions. Nevertheless, science points to certain key messages that have fundamental implications for policymakers: climate change must be addressed as a matter of urgency; above ground biodiversity is declining at a staggering rate; and land management practices, and climate change itself, can disrupt crucial plant-soil biodiversity-carbon feedbacks. The science is clear that whether natural or working lands, undisturbed or restored systems with perennial vegetation and zero tillage management typically have greater ecosystem-level carbon and soil biodiversity, and resilience to climate change, than their more disturbed and degraded counterparts.

There is also increasing acknowledgment that addressing climate change is inextricably linked with critical issues of food security, secure water supplies, poverty eradication and sustainable livelihoods, and requires transformational change (Campbell *et al.*, 2018; Ripple *et al.*, 2019). The IPCC *Special Report on Climate Change and Land* points to the dire consequences of continuing with a business-as-usual approach (IPCC, 2019). However, by acknowledging the interconnections, and reconnecting land and food systems with environmental, social and cultural dimensions that take into account soil biodiversity, we can identify ways to address climate change, with synergistic impacts across a range of SDGs, especially SDG 1 (no poverty), SDG 2 (zero hunger), SDG 3 (good health and well-being), SDG 6 (clean water and sanitation), SDG 11 (sustainable cities and communities), SDG 12 (responsible consumption and production patterns), SDG 15 (life on land, including Land Degradation Neutrality (LDN)) and with other global agendas supporting these objectives (IPCC 2019; Lal *et al.*, 2018).

SDG 13, the UNFCCC and the Paris Agreement

The urgency of the need to take climate action is recognized in SDG 13, which calls on states “to take urgent action to combat climate change and its impacts” and includes the target to integrate climate change measures into national policies, strategies and planning. It acknowledges the United Nations Framework Convention on Climate Change (UNFCCC) as the primary international, intergovernmental forum for negotiating the global response to climate change. The UNFCCC, and the Paris Agreement concluded under it, require Parties to provide national inventory reports of anthropogenic GHG emissions by sources and removals by sinks; to communicate their nationally determined contributions (NDCs) to the global temperature reduction goal, as well as national adaptation plans (NAPs); and to report on progress with them. NDCs and NAPs can include commitments that

aim to harness and increase the contribution of soil biodiversity to climate mitigation and adaptation, and may also contribute to fulfilling other international commitments, for example, under the Convention on Biological Diversity (CBD), Convention to Combat Desertification (UNCCD) and Ramsar Convention on Wetlands (Ramsar Convention).

Link between soil health, soil biodiversity and mitigating and adapting to climate change

The importance of adaptation in many areas cannot be overstated, and the Paris Agreement set out a new global adaptation goal (Paris Agreement, Article 7). Fostering diverse communities of soil organisms to maintain soil health and support vegetation is crucial to climate adaptation and resilience. Increasing soil organic matter (SOM) is a triple win for climate change adaptation, mitigation and food security (Sommer, 2014). Soil health is enhanced by SOC storage because of the many co-benefits of soil organic matter with ecosystem properties such as soil drainage, nutrient retention, compaction mitigation. Whether natural or productive, virtually all ecosystems are managed (whether through protection/restoration or intensive cultivation). Native ecosystems, grazing lands, agroforests,

and croplands may all be protected or managed in ways that maintain or improve soil health. In productive systems, adaptive soil health management systems are based on four principles: 1) continuous living roots, 2) minimize disturbance, 3) maximize soil cover, and 4) maximize diversity, which are supported by an array of management practices to help build soil organic matter (Manter *et al.*, 2018). The multiple benefits of increased SOM underscore the objectives of the 4 per 1000 Soils for Food Security and Climate initiative launched by France in 2015 at the UNFCCC COP21 with an aspiration to increase global soil organic matter stocks by 4 per 1000 (or 0.4 percent) per year to compensate for the global emissions of greenhouse gases by anthropogenic sources. Regardless of whether that target is realistic, increases in SOM as a result of the initiative will benefit resilience and adaptation to climate change and food security. Healthy soils unify productive and natural systems by focusing on maintaining an intact, diverse below-ground ecosystem that ensures the flow of carbon, nutrients, and energy necessary for resilient landscapes (Kibblewhite *et al.*, 2008). As a terrestrial system accrues soil organic matter, both soil health and soil organic carbon increase and a multitude of co-benefits emerge.

Future plans, challenges and gaps

While the potential for synergies from harnessing soil biodiversity for climate action with other global agendas is enormous, transformative policies and action and enabling conditions are necessary for success. Existing nature protection policies need to be fully implemented, and there is a pressing need for support and security of tenure for those at the front line of protecting, restoring and caring for land and soil and conserving its biodiversity, including indigenous peoples and local communities. There is also a need for education and awareness raising, capacity strengthening and full engagement across the public and private sector towards overcoming financial barriers and implementing natural capital accounting and payment for ecosystem services. Fulfilment of, and increases in, existing international climate finance commitments is essential, as is an increase in the allocation of funding for the implementation of natural climate solutions, which can provide cost effective, low risk, mitigation with multiple co-benefits including water filtration, flood buffering, soil health, above- and below-ground biodiversity habitat as well as adaptation and resilience (Griscom *et al.*, 2017; IPCC 2019). Ongoing work in the context of the Koronivia Joint Work on Agriculture (KJWA or Koronivia process) that was initiated at UNFCCC COP 23 in 2017, and which operates as a platform for agriculture related discussions under the UNFCCC, requires broad participation and interdisciplinarity to achieve transformative outcomes. A promising option in the framework of KJWA for mitigate GHGs emission could be achieved by supporting the implementation of actions on the ground through RECSOIL (Recarbonization of Global Soils), which constitutes an implementation tool for scaling up SOC-centred Sustainable Soil Management (SSM), based on collaborative efforts under the Global Soil Partnership (GSP). The main priorities of RECSOIL and associated multiple benefits are: a) to prevent further SOC losses from carbon-rich soils (peatlands, black soils and permafrost) and, where feasible (in agricultural and degraded soils), to increase SOC stocks; b) to enhance farmer income by increasing soil productivity; c) to contribute to improved food security and nutrition; and d) to mitigate climate change through NDCs of Parties within the framework of the UNFCCC.

The complexity of GHG accounting in the land sector, and lack of harmonization in use of terminology between different initiatives and across different disciplines in the land, climate change, and biodiversity arenas is a challenge.

Our knowledge of how SOM is formed and stabilized is changing rapidly. New technologies and studies highlight key interactions among plant carbon allocation, soil biota, and microaggregates and soil structure. A deeper understanding of the ecological controls on soil carbon is needed if we are to create models of carbon balance under changing patterns of climate, land use, and other factors (with a view to more accurately identifying geographic areas with high potential for increased and long term sink capacity, as well as those at high risk of becoming net sources of GHG emissions) and to calculate carbon budgets, and assess the impacts and potentialities of land use practices and bioengineering techniques and innovations to maintain and increase levels of SOC. In particular, more knowledge is required about SOC equilibriums and saturation points in mineral soils, and the influence of vegetation, and relevance of depth.

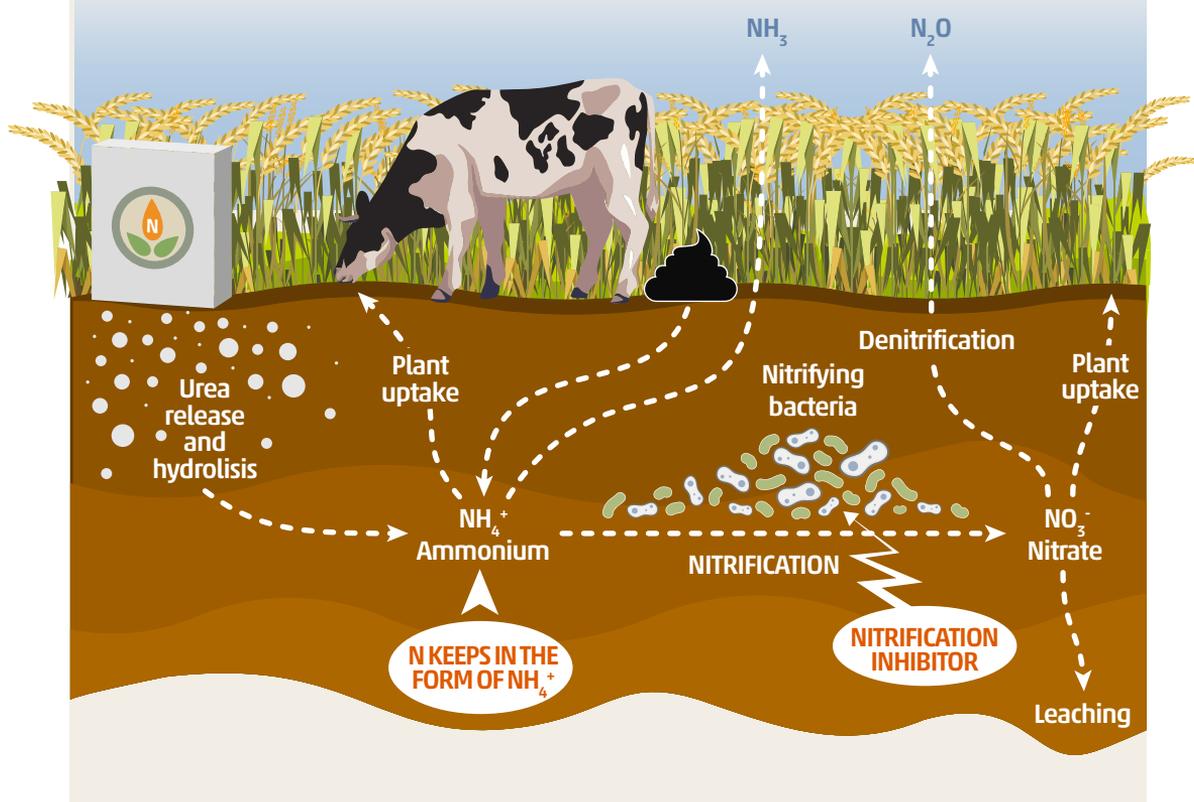
Understanding the critical role that soil biodiversity plays in carbon sequestration and stabilization of SOM, and the implications of a changing climate, is an important aspect of this work. Lack of comparable definitions and of global data regarding tillage or crop rotations is a challenge (Jackson *et al.*, 2017). Soil scientists and data technologists are working to address gaps in the science and data and to map soil types, biogeographical and land use areas, and SOC stocks and baselines, and to provide relevant guidelines (Vargas-Rojas *et al.*, 2019). Accessible methods for measuring changes in SOC levels are also important (Smith *et al.*, 2019). A good example of this is the development of the Global Soil Organic Carbon Map (GSOCmap) (FAO and ITPS, 2018). This map allows the estimation of SOC stock from 0 to 30 cm, and it is the first global SOC assessment which was produced through a bottom-up approach where countries developed their capacities and stepped up efforts to compile or collect all available soil information at the national level. Using the GSOCmap as input, the GSP is currently developing the GSOCseq, another global data generation initiative that is intended to provide updated information on the potential for carbon sequestration in the world's soils. The general procedures to develop this map will be through a bottom-up approach (based on best available data), collaboration with country experts and implementation of widely used and validated soil organic matter simulation models.

Seeking to take into account fuller (CO_2 , CH_4 and N_2O) GHG balances in research, and further work on soil biodiversity and climate change related aspects of N and P cycles, and on the role of black soils, could add significant value. Limiting factors and tradeoffs must also be identified and addressed (Zomer *et al.*, 2017; Campbell *et al.*, 2018; IPCC 2019). Accelerating ambition and action on climate change is critical. The sink capacity of land must not be seen as an alternative to an ambitious emissions reduction strategy (Fee, 2019), and soils can be expected to have reduced capacity to act as sinks for carbon sequestration at higher temperatures. Adopting value chain strategies, such as dietary change and sustainable sourcing, alongside land-based strategies as outlined by the IPCC (IPCC, 2019) could also have positive impacts for soil biodiversity. Recognizing the value, opportunities and risks posed by terrestrial and coastal soils and mainstreaming these critical GHG sinks and reservoirs, and their vital soil biodiversity communities, into decision making in all sectors is fundamental; we need to protect, restore and fund.

Box 3.5.1.2 | Nitrification inhibitors

An adequate supply of nitrogen (N) is essential to maintain/increase crop yields. Plant roots take up N from the soil solution as nitrate (NO_3^-) and ammonium (NH_4^+) soluble ions. The production of NO_3^- occurs with the oxidation of NH_4^+ through the nitrification process carried out predominantly by autotrophic soil bacteria (i.e. Nitrosomonas and Nitrobacter). An excessive supply of N through fertilizers increases the rate of NO_3^- formation, which is highly prone to leaching causing water contamination and nitrous oxide emissions (N_2O), a potent greenhouse gas.

Nitrification inhibitors are compounds that hinder the activity of the Nitrosomonas bacteria, in the nitrification process. This action boosts the efficiency of the use of N fertilizers by slowing down the transformation from NH_4^+ to NO_3^- . This helps to keep N in the form of NH_4^+ reducing potential contamination and greenhouse gas emissions, by a gradual release of N without affecting the supply of N to crops. However, N fertilizer additions in alkaline pH conditions, low soil moisture content, and high temperatures promote the volatilization of N in the form of NH_3 .



3.5.2 | SOIL CARBON CYCLES

Soils comprise the largest carbon stocks on earth, with an estimated total of at least 1 500 Gt carbon (C) (Crowther *et al.*, 2019). Contrary to above-ground biomass patterns, the largest below-ground carbon stocks are found in cold or waterlogged conditions (Crowther *et al.*, 2019). The living component of soils is dominated by fungi, totalling a global biomass of approximately 12 Gt C, followed by bacteria (7 Gt C), soil animals (2 Gt C), and archaea (0.5 Gt C) (Bar-On *et al.*, 2018). Whereas above-ground metazoan biomass distribution is shaped primarily by climatic conditions, the distribution of below-ground microbial biomass is governed primarily by edaphic characteristics (Crowther *et al.*, 2019). Globally, fungal and bacterial biomass are generally higher in regions with high soil organic matter (SOM) contents and lower pH (Bahram *et al.*, 2018). These SOM stocks are among the primary drivers of the accumulation of microbial biomass (Xu *et al.*, 2013) and high abundances of nematodes (van den Hoogen *et al.*, 2019) in the high latitudes.

Soils across the globe contain highly variable soil organic matter (SOM) levels, ranging from very low percentages in desert soils to extremely high percentages in boreal forest soils. SOM is mainly derived from above-ground plant and animal inputs (mainly at soil surface layers), and below-ground plant, animal and microbial inputs; a small fraction forms from abiotic sources. In SOM, soil organic carbon (SOC) is a major constituent, followed by organic nitrogen, phosphorus and sulphur. However, much of SOM is biologically inaccessible as a basal resource for soil organisms, and while other SOM components are considered accessible, they vary in degrees of biodegradability (often termed labile and recalcitrant) that largely depends on the relative elemental concentrations (C:N) and the nature of the carbon compounds. (Figure 3.5.2.1). The organic carbon in the soil occurs in the form of cellular (macro) molecules such as polysaccharides, phenolics, organic and amino acids.

The transformations of carbon in the soil all boil down to cycling between abiotic (mineral) and biotic (organic) soil carbon pools. The key forms of mineral carbon in the soil are carbon dioxide (CO₂ - the final product of carbon oxidation processes) and methane (CH₄ - the final product of reductive processes). The aforementioned mineral compounds both constitute greenhouse gases (GHGs) and are produced and released to the atmosphere under particular conditions. For instance, CO₂ can be removed (fixed back into organic carbon), whereas CH₄ can be back-oxidized in the soil microbiome. Thus, depending on the conditions soils can either function as a net CO₂ source or as a net sink. Similarly, local oxygen conditions will determine the fate of CH₄ in the system. Proper assessments of these processes are key to our understanding of the global impact of these soil functions. Below, we discuss the role of the soil microbiome in the key soil carbon cycling processes.

The role of the soil microbiome in the soil carbon cycle

In a broad sense, the carbon in soil is recycled within a microbe-driven soil food web. A key process is the respiration of carbonaceous substances with molecular oxygen serving as the terminal electron acceptor; this major process returns C to the atmosphere as CO₂. Microorganisms are thus central players in the transformation of plant and animal residues and are also key reservoirs of organic C in soil. As described above, there is another large reservoir of SOM, which can – to some extent – be degraded, and for another part may condense over time to form humus. Much of the latter is typically protected from microbial degradation, but it can be made more available by tillage (ploughing). In undisturbed soils, though, the organic C may accumulate into humus.

CO₂ fixation - Some soil bacteria (and archaea) fix CO₂ autotrophically, and so up to 5 percent of respired CO₂ may be re-fixed. The Calvin-Benson-Bassham (CBB) cycle, responsible for photosynthesis in green plants as well as microorganisms, is the predominant pathway in this process, with RubisCO (ribulose-1,5-bisphosphate carboxylase/oxygenase) being the key enzyme involved. In addition, phototrophs of the *Proteobacteria*, *Chlorobi*, *Chloroflexi*, *Firmicutes*, *Acidobacteria* and *Gemmatimonadetes* phyla also contribute fixed C to the soil carbon reservoir. These bacteria carry the RubisCO large-subunit gene *cbbL* and fix CO₂ during autotrophic growth. Another, often overlooked, process is carbon monoxide (CO) fixation. CO is present in soil in relatively low concentrations compared to CO₂. Up to 20 percent of it can be oxidized by *Proteobacteria* and Actinomycetes that possess the CO dehydrogenase gene *coxL* (Hirsch, 2019). Thus both CO₂ fixation and CO oxidation play important roles in the soil C cycle.

Methane production and oxidation - Methane is generated in soil in a process called methanogenesis which occurs in wet, carbon-rich soils. Anaerobic conditions in such systems spur processes like fermentation by bacteria and anaerobic respiration by specific methanogenic archaea (Hirsch, 2019). A common mechanism uses CO₂ as a substrate, whereas other mechanisms use methylated compounds or organic acids such as like acetate. Wetlands, including rice paddies, have been estimated to generate around 160 Tg CH₄/y (Hirsch, 2019), which outweighs the production by ruminant animals (100 Tg CH₄/y). Conversely, methanotrophic bacteria in soil are estimated to be able to re-oxidize 30 Tg of CH₄/y (Hirsch, 2019). These methanotrophs (members of the Alpha- and Gammaproteobacteria) produce the key enzyme methane monooxygenase. Methanogenesis and methane oxidation are mechanisms of key relevance to global warming. In fact, CH₄ emission rates are found to rise globally due to global warming and thawing permafrost.

Other anaerobic processes - Both denitrification and sulphate reduction can be involved as electron capture processes in organic-rich soils in anaerobic (wet) conditions. Denitrification is a main cause of GHG (N₂O) emissions. Denitrification is beneficial for facultative anaerobes, as they can switch to this process in waterlogged, anoxic soil conditions when organic matter is abundant. Fungi that contain a bacterial-type nitrite

reductase gene in their mitochondria can also denitrify. The *nosZ* gene is important for the conversion of N_2O to N_2 . Thus, next to managing drainage, organic matter and fertilizer inputs into soil, maintaining conditions that promote microorganisms with a functional *nosZ* gene is thought to be important for reducing GHG emissions (Hirsch, 2019).

With respect to sulphate reduction, soil bacteria and archaea from several phyla (including *Deltaproteobacteria*, *Firmicutes* and Nitrospirae) can oxidize carbon compounds via this pathway, including the reduction of sulfate (SO_4^{2-}) to sulphide (S^{2-}) via sulfite (SO_3^{2-}) in a dissimilatory reaction. Typically, this process occurs in waterlogged soils (including rice paddies) and sediments and results in the familiar “rotten egg” odour when such soils are disturbed.

Outlook - Carbon is either fixed or released from soils, depending the activity of the soil microbiomes and driven by abiotic conditions such as water content, temperature, oxygen level and pH. Moreover, soil type and soil management regime have strong effects. For instance, in agricultural practice, the large-scale removal of plant material (grazing by herbivores, agricultural practices) may reduce soil carbon. The grazing, in conjunction with the activity of the soil biota, may also affect carbon release at the roots. With respect to the diverse forms of permafrost, so-called pleistocene-aged permafrost appears to be a major contributor, as it has high OM content, yet low abundance of methanogens. Thus, thawing is predicted to result in high CO_2 release yet low CH_4 . Over the long run, the respective fluxes will strongly depend on the soil conditions, with respect to water (soils becoming wetter or dryer), plant cover and disturbance regime (Waldrop and Creamer, 2019). Extreme factors (such as temperature) may affect soil function (Hirsch, 2019). A better understanding of the organisms involved is important as global climate change coupled with the needs of an ever-increasing population place pressure on soil functions and agricultural sustainability. Carbon cycling in soil is clearly carried by suites of bacteria, archaea, fungi, protists, micro- and meso-fauna. Their relative involvement depends on soil type and conditions. Thus, from the large diversity of microorganisms in soil there is a capacity for selection of appropriately adapted microbiomes as the environment changes.

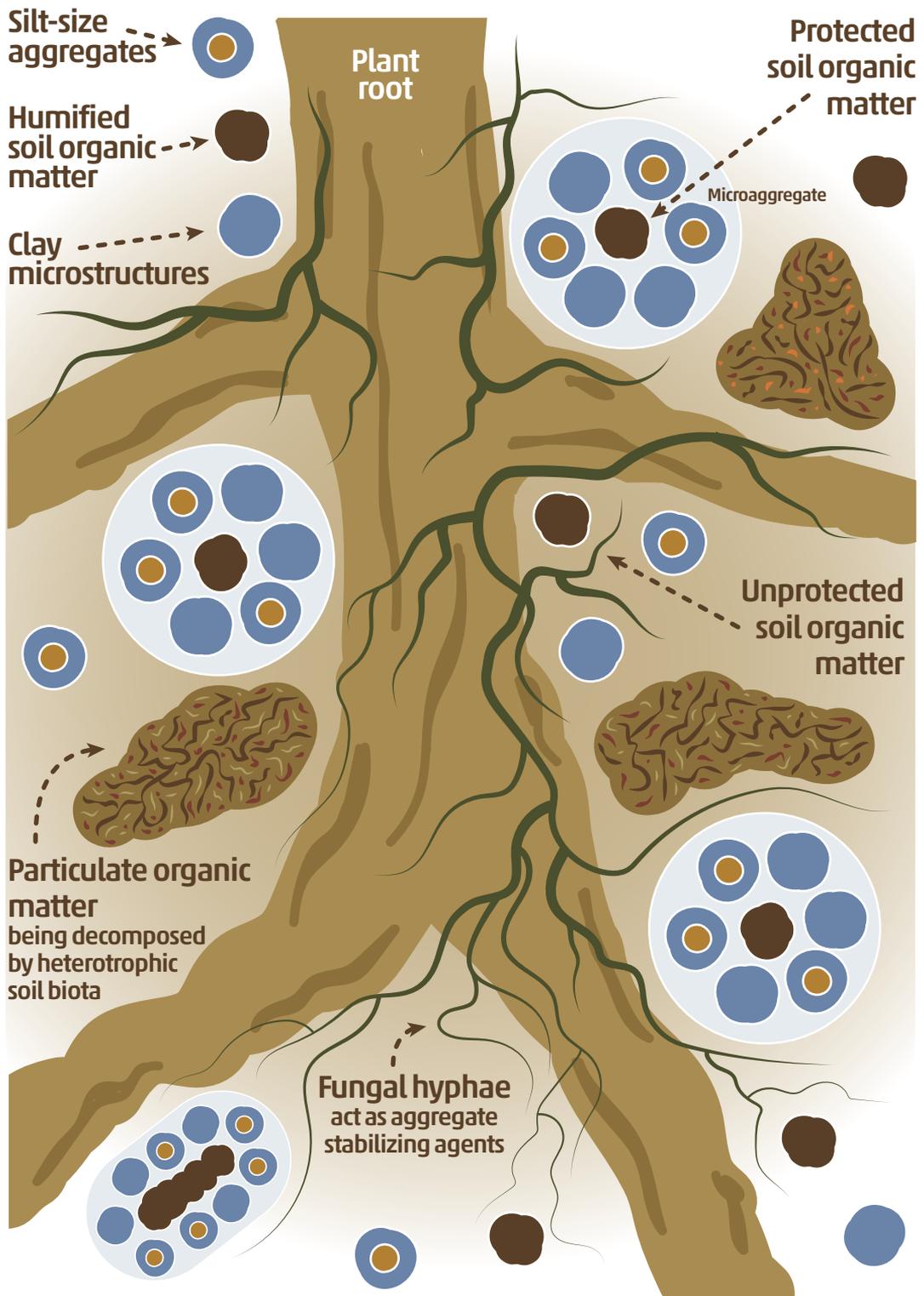


Figure 3.5.2.1 | Role of soil biodiversity in aggregate stability and carbon storage

Most soils contain 1 to 6 percent of organic matter, however not all this amount can be used by heterotrophic soil organisms. SOM accessibility to biota depends upon the physical protection that microaggregates provide from microbial attack, the biochemical composition and consequent resistance to microbial decomposition, and the chemical bonds to organo-mineral particles

3.5.3 | CARBON SEQUESTRATION AND STORAGE

Soil has a tremendous potential for regulating the atmospheric carbon content by sequestering carbon and thus mitigating climate change for the benefit of SDG 13 (climate action). Effective biodiversity conservation and management can lead to higher levels of carbon sequestration and hence climate change mitigation. Very few studies have focused on the role of specific soil organisms and biodiversity on C sequestration and GHG mitigation, with the exception of some studies reporting that specific microorganisms have specific control of many atmospheric trace gasses (Conrad, 1996).

The global changes in temperature and rainfall patterns have a great impact on soil biodiversity abundance, distribution patterns and activities, with strong implications on soil organic matter decomposition and the carbon balance (see for example Crowther, 2017). According to the Status of the World's Soil Resources report (FAO and ITPS, 2015), SOC loss is the second largest global threat to soil functions. Global estimates indicate that soil organic carbon is being lost at a rate equivalent to 10 to 20 percent of total global carbon dioxide emissions (Olivier *et al.*, 2015).

Since organic matter decays faster with increasing temperatures, the areas surrounding the pan-Arctic region (including tundra, boreal forests and peatlands ecosystems) are of special concern because they store vast amounts of carbon that can be released into the atmosphere (~1700 Pg total C; Parish *et al.*, 2008; Tarnocai *et al.*, 2009; Joosten, 2015). Furthermore, carbon accumulation in these areas occurs relatively slowly (~21 g C m⁻² yr⁻¹; Chaudhary *et al.*, 2017) due to harsh environmental conditions limiting soil biological activities (such as waterlogging conditions, acidity and cold temperatures).

In these systems, soil invertebrate communities are dominated by small sized organisms, collectively known as “mesofauna” (that is, with body diameter < 2 mm and including enchytraeid worms, mites and collembolans). In particular, enchytraeid populations are very sensitive to increasing temperatures and frequency of droughts, leading to important losses of non-resilient species and changes in the vertical distribution of more adaptable species (Briones *et al.*, 2007a). Such changes have important implications for C storage, as a result of an increase in the decomposition rates at deeper soil layers, and the potential release of “previously locked carbon” (Briones *et al.*, 2007b, 2010). Consequently, peatlands drainage and fires are responsible for almost 10 percent of greenhouse gas emissions from the Agriculture, Forestry and Other Land Use sector (AFOLU) according to FAO (2015). Therefore, they are likely to continue to be a hotspot of GHG emissions of global importance (Tuniello *et al.*, 2016).

In the case of C sequestration, it is widely accepted that high fungal to bacteria ratios are usually associated with higher carbon sequestration (Briones *et al.*, 2014; Malik *et al.*, 2016), a typical condition of tundra and boreal regions. In particular, it has been shown that AFM fungi promote soil carbon storage through a positive effect on soil aggregation (Wilson *et al.*, 2009), but also via increased fungal grazing by collembolans (Duhamel *et al.*, 2013). However, using global data sets, Averill *et al.* (2014) showed

that soil in ecosystems dominated by ectomycorrhizal and ericoid mycorrhizal-associated plants contains 70 percent more carbon per unit nitrogen than soil in ecosystems dominated by AM-associated plants. In tropical and temperate areas, ecosystem engineers (earthworms, ants and termites) are the main promoters of C stabilization by creating biogenic structures (casts, galleries, nests and mounds) that form organo-mineral associations (Vidal *et al.*, 2016). Similarly, other soil animals such as mites producing slowly decomposing excrements may contribute to stable, carbon-storing humus (Hågvar, 2016).

Due to the variation in spatial scales at which different carbon substrates in soils can be processed and protected either physically, biologically or chemically (Jastrow *et al.*, 2007), soil fractionation has long been recommended to separate particulate organic matter (POM) from mineral-associated organic matter (MAOM) when evaluating soil carbon sequestration (Cambardella and Elliott, 1992). However, a growing body of empirical research has led to a shift in understanding of the mechanisms for SOM formation and persistence (Cotrufo *et al.*, 2013). This new understanding highlights two discrete biologically mediated pathways to each of the POM and MAOM fractions (Cotrufo *et al.*, 2015). A physical pathway to POM formation is dominated by litter fragmentation and bioturbation as a result of larger soil fauna and microarthropods (ecosystem engineers), whereas the microbially-mediated pathway to MAOM formation is dominated by bacteria, archaea and microbial communities that biochemically alter carbon substrates through decomposition and condensation/polymerization.

This new understanding has brought several additional insights about microbial community dynamics that can increase microbial carbon utilization efficiency (CUE), increase soil carbon sequestration in the relatively stable MAOM fraction, and even help to identify the microbial taxa associated with healthy soils that are more resilient to the impacts of climate change (Dubey *et al.*, 2019; Sokol *et al.*, 2019). While this level of mechanistic understanding is not typical in biogeochemical soil models, recent attempts to include representations of microbial function and POM/MAOM pools in mathematical models show promise (Wieder *et al.*, 2014; Robertson *et al.*, 2019). Being able to use ecosystem models to represent soil biodiversity and leverage the wealth of data that metagenomic techniques provide is a crucial step in being able to predict any potential feedbacks with climate change (Dubey *et al.*, 2019).

Box 3.5.3.1 | Keeping the carbon in the soil

Keeping the carbon already contained in the soil is more effective than any other practice aiming at capturing atmospheric carbon. Peatlands where peat-forming vegetation is intact or has been restored are likely to be more resilient to climate change impacts than degraded ones (Parish *et al.*, 2008). Therefore, the conservation of C-rich soils like peatlands, forest soils and permanent grasslands should become a priority.

Recent estimates indicate that agricultural land uses have resulted in the loss of 133 Pg C from the soil (Sanderman *et al.*, 2019). On 1 December 2015 at the COP21, the international initiative “4 per 1000” was launched, with the aim of demonstrating that agriculture, and in particular, agricultural soils can play a crucial role where food security and climate change are concerned. Accordingly, good agricultural practices that revolve around increasing C retention and decrease nutrient losses should be adopted (such as no-till and conservation agriculture). However, many limitations to achieving SOC increase at a rate of 4‰ yr⁻¹ over large areas (de Vries *et al.*, 2018; Poulton *et al.*, 2018) and likely to require large investments (Paustian *et al.*, 2016). Therefore, soil management practices should aim to ensure all soil functions rather than specifically maximise carbon sequestration. General guidelines to make soils more resilient to climate change should include the following:

- (1) Conservation of C-rich soils like peatlands, forest soils and permanent grasslands should become a priority.
- (2) Improve SOC pool by adopting less intensive agricultural practices that limit/exclude agrochemical use and maintain a live plant cover (such as conservation agriculture) or use organic amendments such as mulch, compost.
- (3) Adopting ‘paludiculture’ practices to cultivate organic soils without drainage, whereas land use practices that require drainage should, if possible, be relocated to areas with mineral soils.
- (4) Increase soil biodiversity by protecting natural areas, keeping habitat heterogeneity and diversifying land use options.

The great recent advances in collating and mapping soil biodiversity data at global scales (see for example Tedersoo *et al.*, 2014; van den Hoogen *et al.*, 2019; Phillips *et al.*, 2019) means that it will now be possible to link specific soil biota groups to terrestrial carbon stocks. From this, new management strategies that enhance C retention across different ecosystems can be developed and finally incorporated in environmental policies.

Box 3.5.3.2 | Biochar as C-sequestration

Biochar is carbon-rich solids produced during pyrolysis of biomass at oxygen-limited conditions and may be beneficial soil amendments and increase soil quality as biochar sequester carbon, increase water infiltration, mediate soil pH, elevate CEC, increase nutrient uptake, and modify GHG emissions (Li *et al.*, 2019; Zhang *et al.*, 2018). The feedstock is important for the chemical and physical properties of the produced biochar. Examples of feedstock are agricultural waste as straw and coconut shells, wood, animal manure and wastewater sludge. The pyrolysis conditions such as temperature in combination with oxygen availability are significant for the biochar properties and have to be carefully controlled. When controlling the pyrolysis conditions, problematic compounds such as heavy metals can be removed while production of problematic polycyclic aromatic compounds can be kept to a minimum. Biochar is reported to reduce availability of polluting compounds such as heavy metals and organic contaminating compounds by absorption, contributing to soil remediation (Kavitha *et al.*, 2018).

Biochar has a priming effect on soil organic matter and microbial activity, which can be both negative and positive, probably depending on the physical structure of the soil. This is suggested to be due to a higher stimulation of microbial activity in sandy soil (Wang *et al.*, 2016).

After introduction into the soil, about 3 percent of the biochar is relatively easily degraded, with a lifetime of 108 days, while the remaining 97 percent has an estimated lifetime in soil of >500 years (Wang *et al.*, 2016). Hence, it will remove carbon from the current carbon cycle, and in this way reduce atmospheric CO₂ and mitigate climate change.

3.5.4 | SOIL FORMATION AND EROSION PREVENTION

While parent rock determines soil type as well as soil texture through the size of primary particles, biological activity directly affects the aggregation of mineral grains into secondary units. Farmers have observed the role of soil organisms in soil structure modifications for a long time. The crucial role of organic binding agents often produced by bacteria and fungi are affected by size, quality and stability of soil aggregates (Tisdall and Oades, 1982). The activity and diversity of soil biota are essential to soil structuring, thus contributing to soil formation and regeneration. When grassland is converted into cropland or when peatland is converted into agriculture, loss of soil structure can be observed due to mechanical disturbances and increased oxidation. Soil stability is strongly depending on soil texture, as finest soil texture (clay, loam) leads to highest stability, and soil organic matter content (Le Bissonais, 1995). Nevertheless, soil organisms, including microorganisms and fauna, have a large share of responsibility for soil stabilization (Pérés *et al.*, 2013). A good and stable soil structure is one of the main objectives of farmers and is also the goal of SDG 12 (responsible consumption and production), SDG 13 (climate action) and SDG 15 (life on land).

Stabilization processes are required in organic carbon degradation, along with complex chemical, physical, and biological interactions within the soil matrix (see for example Kögel-Knabner *et al.*, 2008). Soil macrofauna play an important role in soil structure modification through bioturbation and the production of biogenic structures (Brussaard *et al.*, 1997; Lavelle and Spain, 2001). Earthworms, ants and termites, but also small mammals, modify their surrounding environment and thus affect soil water and nutrient dynamics through their interactions with other soil organisms. In terms of biomass and function, earthworms play a key role in agricultural soils with strong effects on soil structure and processes through borrowing, feeding and casting (Blouin *et al.*, 2013, Pulleman *et al.*, 2012). The burrow network allows increased water infiltration, especially when they are provided by anecic earthworm species (see the following section on water flux) and by consequence could take part of decreasing soil erosion by 50 percent (Shuster *et al.*, 2002). Deposited cast of anecic species, by increasing the soil roughness, decrease the speed of runoff and therefore lead to a fourfold decrease in soil erosion (Le Bayon and Binet, 2001). However, other authors suggest that over longer time-scales (thousands of years or more), the erosion of cast material could lead to vast amounts of sediment accumulation in alluvial soil or floodplains (Feller *et al.*, 2003). No matter what, the effect of earthworm biostructure on soil erosion has to be reconsidered with respect to the rain event intensity, the slope, the soil surface cover, and at larger scale, the landscape organization (presence of hedges). Enchytraids also form aggregates, but at a smaller spatial scale (Marinissen and Didden, 1997).

Soil crusting is considered to be an important form of soil degradation that limits land productivity because it considerably reduces soil water infiltration and root growth. Soil-crusting risk depends on the texture, as loamy soils present the highest risk (Le

Bissonnais, 1996). The application of organic mulch has allowed the recuperation of surface crusted soils in Burkina Faso (Mando *et al.*, 1996). Many other examples have been described, highlighting the importance of appropriate residue management on farmland to foster biological activity benefiting soil structure.

Despite the important role of soil organisms in soil formation and erosion, little attention has been given to the essential organisms in intensively used agricultural soils, where natural and biologically mediated processes have been largely replaced by human activity. In particular intensive tillage practices affecting soil structure with mechanical soil loosening and winter furrow leaving a coarse surface to be broken up by the frost have significant negative impacts. Stabilization of aggregate depends on many parameters such as biotic (microorganism, fauna) and abiotic (texture, organic matter) and formation and stabilization of macroaggregates result from combinations of functional groups (Frazão *et al.*, 2019). Macroaggregate breakdown increases with age because the action of organic binding agents is progressively disrupted. Microaggregates are far more stable, and as long as soil organisms are active, in particular bacteria producing organic binding agents in combination with fungal hyphae, they can become building blocks during the formation of new soil macroaggregates (Barrios, 2007). If this is no longer the case, erosion processes will accelerate with soil particles from disrupted soil aggregates or even soil horizons being lost or transported vertically (for example by gravity or water) and horizontally (by water, ice or wind). Aggregate stability is linked to plant diversity. Plant mixtures with high frequency of grasses increase soil aggregate stability by increasing root biomass, soil organic carbon concentrations, and soil microbes, while legumes may be less favourable for soil aggregate stability (Pérès *et al.*, 2013). A healthy soil ecosystem with high biodiversity will help reduce erosion and secure soil formation. If the soil quality is deteriorating and erosion processes are increasing, it might take a very long time for the regulating service to be restored again. Consequently, in order to maintain long-term sustainability of soil fertility and soil structure, soil management plans need to more carefully address soil as a habitat and not only as a substrate for cropping.

3.5.5 | REGULATION OF WATER FLOW

The ability of soils to store and release water is a widely acknowledged regulating ecosystem service and is part of SDG 6 (clean water and sanitation) and SDG 15 (life on land). Furthermore, soil moisture is the driver of many chemical and biological processes and is therefore essential for soil development and functioning (Dominati, 2013). As we are experiencing dramatic climatic changes, we must enhance water storage to increase resilience to weather extremes. Only soils with diverse and abundant soil life and high biodiversity can act as efficient filters to produce clean drinking water. Soil organisms, both fauna and flora, facilitate water infiltration and drainage thanks to the creation of macro- and micropores. In addition, a greater accumulation of organic matter in the soil

by biota increases the water holding capacity of the soil. Soil organisms thereby improve soil aeration and water infiltration (Brussaard *et al.*, 2007, Pulleman *et al.*, 2012, El Mujtar *et al.*, 2019). By regulating the water flow, they contribute to flood mitigation and climate control (Vogel *et al.*, 2019). These multiple water services are closely connected to continuously undergoing soil formation processes. A favourable soil structure facilitates the germination and the establishment of crops, helps to prevent water logging, reduces the risks of water shortage and maximises resistance against physical degradation (El Mujtar *et al.*, 2019).

Regulation of water flow is threatened by compaction of agricultural soil, which can be caused by traffic and loss of soil carbon due to agricultural practices. Also xenobiotic compounds as pesticides and organic contaminants can reduce the soil structure and reduce the regulation of the water flow.

The water flow regulation is increased by increasing the soil carbon content by return of organic matter including plant residues to soil, by reduced and low tillage, which both will increase biodiversity. Interestingly, sequestering carbon by incorporation of biochar in soil has a very positive effect on the water filtration and water holding capacity of the soil (Mao *et al.*, 2019).

3.5.6 | WASTEWATER TREATMENT

Ecosystems such as wetlands filter effluents, decompose waste through the biological activity of microorganisms, and eliminate harmful pathogens. Despite this direct regulating service of soil biodiversity, the most efficient treatment of wastewater is in man-made installations. Wastewater treatment processes are engineered installations that receive wastewater (agricultural, domestic, and industrial) and use biological and chemical methods to produce aqueous effluents that can be discharged safely to the environment; largely oceans, rivers and lakes (Olsson and Newell, 1999). A “solids” component comprised of biological cells/cellular debris and/or chemical precipitates is also generated and handled by different methods depending upon composition and volume. The activated sludge process (Nielsen and McMahon, 2014) and wetlands (Wu, *et al.*, 2015) are common treatment options, with the choice of process dependent upon many variables including wastewater volume and pollutant composition (concentration and types) (Prasse, *et al.*, 2015).

Wastewater from agricultural and domestic sources will likely contain animal and/or human pathogens and the treatment process chosen will need to mitigate pathogens. Biological processes dominate wastewater treatment methods, particularly of agricultural and domestic wastewaters, which have substantial organic components that can be bio-decomposed largely by microorganisms but aquatic plants contribute in wetland

processes. The source of microbes in biological treatment processes come partially from the wastewater influent but are also opportunistically sourced from soils in the vicinity of the installations. Thus, soil biodiversity substantially contributes to this major biotechnological industry. Nutrients (such as nitrogen and phosphorus) are common pollutants in wastewater. Nitrifier/denitrifier and phosphorus accumulating microbes largely sourced from soils ensure that most nitrogen in wastewater is converted to N_2 gas and most orthophosphate is bio-accumulated inside specific bacteria as polyphosphate. For example, polyphosphate accumulators were identified as Betaproteobacteria, *Candidatus Accumulibacter phosphatis* (Crocetti *et al.*, 2000; Hesselmann *et al.*, 1999), and found responsible for mitigating discharge of phosphorus, which is a substantial environmental eutrophying agent (Oehmen *et al.*, 2007). These bacteria were likely sourced from soils rather than the wastewater itself. Likewise, nitrogen removal is mediated by a community of soil-sourced bacteria and archaea (Limpiyakorn *et al.*, 2013), with wide-ranging nitrogen metabolic capacities (Schmidt *et al.*, 2003, van Loosdrecht and Jetten, 1998). Soil microbes dominate wetland ecosystems, underscoring their high value (Rajan *et al.*, 2019).

The discharge of wastewater effluents to lake and river environments impact soil microbial communities. Particularly of concern is the rise in antimicrobial resistant genes from wastewater treatment systems, which could be a source of these genes into other ecosystems (Subirats *et al.*, 2019, Yin *et al.*, 2019), especially if the treated effluent is used for agricultural irrigation.

3.5.7 | INVASIVE SPECIES

The majority of our knowledge of invasive soil species concerns agricultural pests, of which many contribute to huge economic losses globally (Coyle *et al.*, 2017; SDG 1 (no poverty), SDG 2 (zero hunger)). In addition, when certain species are introduced to pristine areas as biocontrol agents, this can lead to massive biodiversity loss, affecting terrestrial ecosystem functioning (SDG15). Invasive soil species are also often found in urban or disturbed environments (for example, after fire), which have habitats that encourage the spread and existence of invasive species. However, due to undersampling and a general lack of taxonomic knowledge, the general distribution of invasive species in the soil are largely unknown, while the invasion status of many species is uncertain, being listed as having a ‘cosmopolitan’ distribution.

Soil taxa listed under the 100 of the World’s worst invasive alien species (Lowe *et al.*, 2004) include five species of ants, including the Argentine ant (*Linepithema humile*), crazy ant (*Anoplolepis gracilipes*), big-headed ant (*Pheidole megacephala*), little fire ant (*Wasmannia auropunctata*) and red imported fire ant (*Solenopsis invicta*). Others on the list include the Rosy wolfsnail, (*Euglandina rosea*), soilborne fungal pathogens such as

Phytophthora cinnamomimi, a termite (*Formosan subterranean termite*, *Coptotermes formosanus shiraki*) and a soil-dwelling flatworm species (New Guinea flatworm, *Platydemus manokwari*).

Impacts

Although gaps in our understanding of invasive species in soil exist, there are numerous examples of soil microorganisms and invertebrates that have substantial ecological and socioeconomic impacts when they invade ecosystems (reviewed in Cameron *et al.*, 2016; Thakur *et al.*, 2019). The effects of invasive species in soil vary depending on the invader's trophic position and functional role, including whether they are pathogens, herbivores, detritivores, omnivores, and predators. As a result, soil invaders are likely to affect our ability to meet a number of the SDGs. First, through their impacts on species, communities, and ecosystems, invasive organisms in the soil are affecting both life on land (SDG 15) and life below water (SDG 14). For example, the Argentine ant (*Linepithema humile*) is listed as one of the world's worst invaders and can cause substantial declines in diversity of native ant species (Suarez *et al.*, 1998). The effects of organisms invading soil can further extend to impact aquatic systems. For instance, *Phytophthora lateralis* is a soilborne plant pathogen that causes Port Orford cedar root disease. It has a range of impacts including reducing native diversity and reducing shading along streams, which increases water temperature and leads to invertebrate and salmon killing, as well as increasing soil erosion (Robin *et al.*, 2011). Secondly, effects of invasive soil organisms can directly impact food resources and thus their control is important for progress on addressing SDG 2 (zero hunger) and SDG 3 (good health and well-being). For example, *Phytophthora cinnamomi* disperses through soil and plants and causes root rot. It has more than 500 host species and consequently can cause changes in vegetation that may cascade to affect entire food webs (Hardham and Blackman, 2018). In particular, it affects food crops and cash plants such as avocado, pineapple, and eucalyptus. Often, invasions of soil organisms may have impacts that affect multiple SDGs. For example, non-native earthworms are ecosystem engineers and their invasions can cause cascading effects that impact plant communities (SDG 15), forest productivity (SDG 15), carbon sequestration (SDG 13), wildlife and human disease (SDG 3), and soil and water quality (SDG 2, SDG tab6) (Frelich *et al.*, 2019).

Pathways of introduction

With the broadening of international trade since the early twentieth century, an increased trade and importation of plants globally caused a progressive increase in the number of alien terrestrial invertebrates introduced (Faulkner *et al.*, 2016). The majority of soil invertebrates' introductions appear to have been accidental, as contaminants or stowaways (Faulkner *et al.*, 2016). A wide range of human activities can act as vectors for transport of soil, and consequently of organisms living within the soil. For instance, soil can be transported as a contaminant of shipping containers (Godfrey and Marshall 2002), shoes (McNeill *et al.*, 2011), and plants (McNeill *et al.*, 2006). Soil animals easily can be transported through the horticultural trade surviving as eggs, while many

small soil arthropods are desiccation resistant, surviving months in an anhydrobiotic state. The majority of non-indigenous taxa found in soil associated with commodities, cargo, vehicles, or footwear, included fungi, bacteria, moss, invertebrates, nematodes, and rotifers (reviewed in McNeill *et al.*, 2011). There are few border and quarantine interception records for soil organisms but, in general, the majority of soil invertebrates' introductions appear to have been accidental, as contaminants or stowaways (McNeill *et al.*, 2011). This is partly due to their small size, which has resulted in many species having been imported undetected, but it is also because taxonomic identification of some introduced taxa remains problematic and it is often difficult to determine whether a soil organism is non-indigenous in a particular region (McNeill *et al.*, 2011).

Control and biosecurity

Soil is recognized as a biosecurity risk; for example, the International Plant Protection Convention states that soil is a high-risk pathway for the introduction of invasive species/pests and consequently that guidelines are needed to minimize risk of introduction of pests with soil movement (IPPC, 2007). Given the impacts of invasive soil organisms on other organisms and ecosystem services on land and in aquatic systems, appropriate biosecurity methods are critical for maintaining life on land (SDG 15), food production (SDG 2), and health and well-being (SDG 3), as well as clean water and life below water (SDGs 6 and 14). However, the relative risk levels of different pathways and vectors are poorly understood for invasive soil organisms, which limits the ability of officials and regulators to target the pathways that are most likely to lead to introductions of invaders (McNeill *et al.*, 2017). A further issue is the lack of taxonomists trained in identification of soil organisms, and consequently training of taxonomists is a key priority to facilitate the detection of newly introduced species and to aid in their eradication and control (Convention on Biological Diversity, 2014). Taxonomic skills can also be supplemented, although not replaced, by identification of species through molecular approaches, such as DNA barcoding (www.boldsystems.org), which has been successfully used globally as an early detection and management tool for invasive species (Armstrong and Ball 2005; Bergstrom *et al.*, 2018). Ongoing survey work is needed to increase detection of new invasive species and to understand risk levels of different pathways, in order to improve management and control of the impacts of soil invaders on terrestrial and aquatic biodiversity, and ecosystem services.

Climate change and invasive species

It has long been suggested that climate change (SDG 13) will exacerbate the impact of invasive species (Cannon, 1998), facilitating the spread and establishment of invasive species. Understanding the physiological traits of invasive species may shed some light on how to better manage or prevent the introduction of invasive species (Karsten *et al.*, 2016), especially pest species which are predicted to change in distribution with climate change (Bebber *et al.*, 2013; Pecl *et al.*, 2017). Several physiological studies using Collembola (springtails) as model organisms have indicated that invasive species are generally more tolerant of warmer, drier conditions than are indigenous species (Chown

et al., 2007; Slabber *et al.*, 2007; Janion *et al.*, 2010; Janion-Scheepers *et al.*, 2018). A predicted increase in frequency of extreme temperature events will likely have an impact on soil community composition and functioning (IPCC, 2007). Indeed, some recent experimental evidence suggests that variation in thermal tolerance traits could lead to trophic mismatches, although some interspecific variation could lessen the severity of this (Franken *et al.*, 2017).

3.5.8 | BIODIVERSITY REGULATION AND BIOCONTROL

The interactions of soil organisms, directly and indirectly through competitive, facilitative, mutualistic, pathogenic or predatory effects, affect the overall structure of the soil food web; these interactions are self-reinforcing and self-regulating processes that lead to emergent community properties such as community stability and biological control or biocontrol. Biocontrol of plant disease is the reduction in the numbers and activity of a plant pathogen or pest, using one or more organism. This process often occurs below ground within plant roots, at the rhizosphere, or more generally within bulk soil.

The interactions that lead to biodiversity regulation (that is, the reinforcement of stable populations within the soil system, including pest control) can be positive or negative. For example, between soil microorganisms, facultative (positive or beneficial) associations between different bacterial strains (including *Bacillus*, *Paenibacillus*, *Pseudomonas* and *Rhizobia*), and different arbuscular mycorrhizal (AM) fungal species stimulate growth of AM fungi and the germination of their spores, leading to increased root colonization of the host plant by AM fungi, increased solubilization of phosphate, and the suppression of pathogens around the rooting zone. Biological regulation does not require direct contact between organisms, and rather often arises through indirect antagonistic interactions.

Microbial biological control agents (MBCAs) employed for commercial plant production act via multiple modes of pest-host interference mechanisms (see Kohl *et al.*, 2019 for current review). Some MBCAs act via nutrient or space competition (that is, exploitative competition) that can modulate the growth conditions for the pathogen or pest (such as insects or weeds). For instance, strains of *Fusarium oxysporum* that are non-pathogenic can be superior competitors for carbon and root colonization sites. Another mechanism of biocontrol is growth interference of a pathogen through antibiosis, where volatile or non-volatile substances produced by one organism, such as enzymes or other metabolites, kill or inhibit growth of another organism.

Such interactions are highly regulated by compounds such as signalling compounds, enzymes and other interfering metabolites. Antibiotics are both volatile and non-volatile

substances produced by one species of organism that in low concentrations kill or inhibit growth of another organism.

Volatile organic compounds (VOCs), are typically small, odorous compounds (<C15) with low molecular mass (<300 Da). Microbial volatile organic compounds belong to different chemical classes including alkenes, alcohols, ketones, benzenoids, pyrazines, sulfides and terpenes. Because many of these compounds are secondary metabolites or simple products of metabolism such as alcohols and aldehydes, they can diffuse easily through gas-filled pore space, and can operate at larger spatial scales as they do not need direct contact to play an important role in long-distance microbial interactions. *Trichoderma* (Fungi) species are particularly active producers of several antifungal volatiles, and are an effective biocontrol agent against several phytopathic fungi (Saravanakumar *et al.*, 2017).

Soluble antibiotics are secondary metabolites with high polarity, which makes them soluble in water, a characteristic that can lead to strong biocontrol efficacy but at smaller spatial scales than VOCs. Examples of organisms that produce soluble antibiotics and have been used as MBCAs are several soil fungi (such as *Aspergillus* and *Penicillium*, *Trichoderma*), which produce soluble antibiotics such as citrinin, patulin, gliotoxin and penicillic acid, peptide antibiotics and trichodermin. Fluorescent pseudomonads also have been shown to suppress a variety of plant pathogens by secretions of soluble antibiotics.

Siderophores – specialized iron-binding molecules – can also play a role in biocontrol. Siderophores are low molecular weight secondary metabolites produced by microbes under iron deficiency, to help bind and supply iron to the organism. Siderophores produced by plant growth promoting bacteria (PGPB) can play a role in the prevention or reduction of the effects of pathogenic microbes in plants by depriving such pathogens of iron if they produce higher rates of siderophores than the pathogen. (Figure 3.5.8.1). Both *Pseudomonas* and *Bacillus* spp. are important biocontrol agents because they produce siderophores that are very competitive for Fe binding.

There are different classes of siderophores such as hydroxamate, catecholate and mixed ligand siderophores. Siderophores produced by PGPB can play a role in the prevention or reduction of the effects of pathogenic microbes in plants by depriving such pathogens of Fe. In this case, PGPBs must produce higher rates of siderophores, and they must be specific to the producing organism and very competitive for Fe binding, relative to the pathogen. This is especially true for *Pseudomonas* and *Bacillus* spp, which are important biocontrol agents.

Direct antagonisms

Several microbial groups are effective biocontrol agents because they exhibit direct contact phenomena that lead to the destruction of organism and the elimination of competitors. For example, aggressive litter-decomposing and wood decay basidiomycetes (fungi) can destroy the mycelia of other fungi by growing over them, thus capturing their

resources by replacement. This can be physical or combined with chemical antibiosis. For instance, *Collybia peronata* will overgrow and out-compete *Cladodporium cladosporioides* in laboratory cultures *in vitro*, by first producing hyphal runners which traverse the mycelia of *Cladodporium cladosporioides* for the purpose of capturing resources and spreading.

In addition to strong competitive interactions, antagonistic microbial interactions also include acting through parasites, which invade and kill microfauna, mycelium, spores and resting structures of fungal pathogens and cells of bacterial pathogens, or direct predation. The most common example of predation in soil is protozoa and nematodes preying on bacteria and fungi, particularly in the rhizosphere, where numbers of bacteria and fungi are high. Additionally, many soilborne fungi have been demonstrated to be antagonists of nematodes, including predacious fungi, endoparasitic fungi, parasites of nematode eggs and cysts, and fungi that produce metabolites toxic to nematodes. More than 150 fungal species have been isolated from the females or cysts of *Heterodera glycines* (the soybean cyst nematode), including *Exophiala tusarium* and species of *Glicocladium*, *Neocosmospora*, *Paraphoma*, *Phoma*, *Stagonospora*, *Verticillium*, *Dictyochaeta* and *Pyrenochaeta*. *Fusarium oxysporum* was able to colonize sclerotia of *Sclerotinia sclerotiorum* surfaces under soil substrate conditions, and antagonist re-isolation from sclerotia and viability reduction were low. Sclerotia were planted in the *Fusarium oxysporum* colony, and a significant reduction in sclerotia viability was detected over time, resulting in a reduction of the inoculum source.

Biological control is a way to relieve the pressure on soil biodiversity and restore the ecological balance (Ruiu, 2018). The United States Environmental Protection Agency (US EPA) defines biological pesticides as naturally occurring substances that control pests (biochemical pesticides), microorganisms as bacteria, fungi, viruses, protozoa and nematodes that control pests (microbial pesticides), and pesticidal substances produced by plants containing added genetic material (plant-incorporated protectants) (US EPA, 2019). The basic concept of biological control is to facilitate the natural ecosystem to counteract the potential of pests (Ruiu, 2018) and generally increase biodiversity and ecosystem functioning. Biological control agents can be very target-specific and can limit non-target effects while being benign to the ecosystem. Ruiu (2018) reviewed entomopathogenic microorganisms used for biological control; these organisms include bacteria, fungi, baculoviruses and nematodes.

Worldwide, the largest commercial success of a biological control agent is without doubt *Bacillus thuringiensis* (Bt), a common bacteria isolated from soil (Jouzani *et al.*, 2017). Bt is a biological control agent with insecticidal activity against a range of different insects, and different strains and marketed products have different specificity towards different pest insects which increases the specificity against the target organisms (reviewed by Bravo *et al.*, 2011). Bt produces an intracellular toxin that upon ingestion by an insect, is released in the insect gut where the deadly activity happens. (Figure 3.5.8.2). This specificity means that the Bt toxin does not affect warm-blooded animals such as birds and humans, and Bt products have even been sprayed from aircrafts over cities and other

populated areas as late as May 2019 (Williams 2007, CityNews 2019). The Bt toxin producing genes have been inserted into agricultural crops, especially maize, and the GMO Bt-maize is presently grown in large areas of the world (Romeis *et al.*, 2019). Lately, the soil bacterium Bt has been found to have other plant growth-promoting properties and may in the near future be involved in other environmentally friendly plant growth-stimulating products and agricultural practices (Azizoglu 2019; Jouzani *et al.*, 2019). This is not surprising, as many other *Bacillus* species are known to have plant-beneficial traits (Saxena *et al.*, 2019).

◇ **Box 3.5.8.1 | Anaerobic soil disinfection (ASD) - an alternative fumigation technique** ◇

An alternative method to fumigants for managing soil pathogens is a technique that manipulates the soil microbiome. Anaerobic soil disinfestation (ASD) is a biologically based chemical fumigant alternative. ASD consists of incorporating organic soil amendments into the soil, covering the soil with a plastic mulch, and saturating the soil to field capacity with water (Roskopf *et al.*, 2015). The soil undergoes a shift from aerobic to an anaerobic environment, during which facultative anaerobes and anaerobes increase in population. These microbes produce anti-bacterial and fungal metabolites such as short chain organic acids, methyl sulphide compounds, hydrocarbons, and dimethyl disulphide (Hewavitharana *et al.*, 2019). Plants grown in ASD treated soil had yields similar to or in excess of those for plants planted in chemically fumigated soil, and ASD yields were 30 times greater than the untreated controls (Shrestha *et al.*, 2016). Multiple studies have demonstrated that the microbial community changes with the use of ASD. The technique has been proven to work in multiple crops worldwide, yet the mechanics behind ASD are not fully understood and there remains a significant knowledge gap in pest control research. ASD offers broad-spectrum and non-specific pest control, like methyl bromide, and has been shown to manage soilborne bacterial and fungal plant pathogens and root-knot nematodes, and to provide weed control. However, just like chemical fumigants ASD impacts most soil life and biodiversity, so with the ambition of preserving biodiversity ASD should have limited use

◇ **Box 3.5.8.2 | Biological control and crops** ◇

Conventional agriculture with low crop diversity and extensive use of chemical pesticides tends to reduce soil biodiversity, unbalance the ecosystem with an oversimplification of the species present, and pave the way for pathogenic organisms to prevail.

Soilborne pathogens are one of the major biological causes of yield loss, decreases in fruit quality, and plant mortality. Plant pathogens are challenging to manage, as the same microbe can infect multiple hosts and it can remain in the soil for years. Growers will lose entire crops and grow less profitable crops or even abandon their fields because of the severity of soilborne diseases.

The abundance of pathogen suppressive microorganisms is significantly greater in a diverse crop rotation compared to monocultures (Peralta *et al.*, 2018). Biological control of pathogens cannot be attributed to just one aspect but rather by multiple factors of both biochemical, physical and biological origin. Hence, many strategies to fight the plant pathogens should be considered.

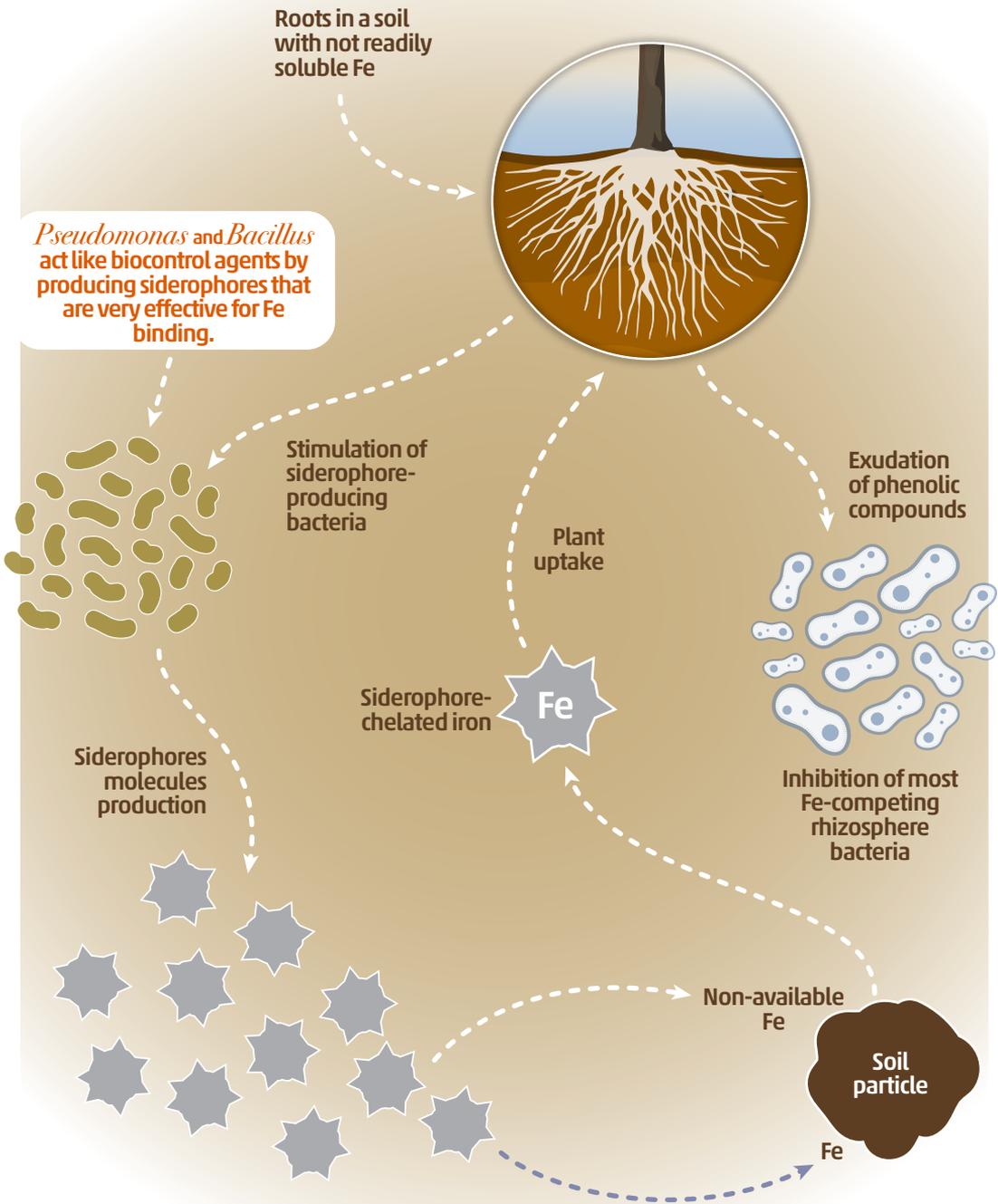


Figure 3.5.8.1 | Synergistic and antagonistic effects of soil microbes and plants

Chelating agents play important roles in soil ecology. Some plants, when growing in soils with non-readily soluble iron (Fe) (as calcareous soils), can either produce it by themselves or stimulate a specific group of soil bacteria to synthesize Fe-chelating agents named siderophores. They are organic molecules able to capture and carry iron into the plant roots. (Adapted from Wail and Brady, 2017).

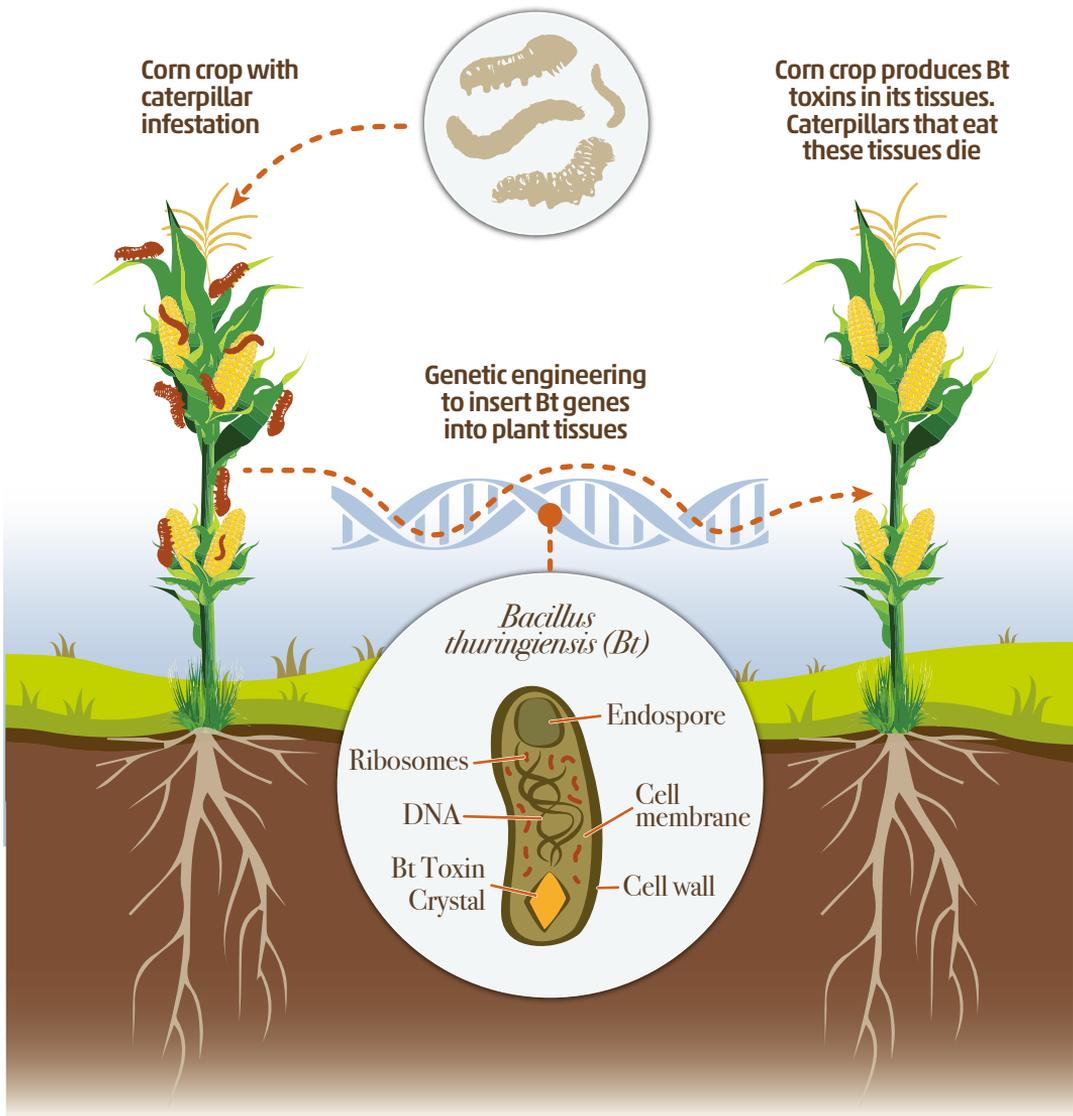


Figure 3.5.8.2 | Biological control

Bacillus thuringiensis (Bt), a bacterium species isolated from the soil, has been successfully used as a biological control agent against insects. Bt produces an intracellular toxin that, when ingested by an insect, is released in the insect's gut, killing it. The genes that produce the Bt toxin are inserted into agricultural crops, particularly maize, giving the plant the ability to avoid attack by certain pathogenic organisms.

3.5.9 | GENETIC DIVERSITY

Provisioning and regulating ecosystem services arise from a biodiverse ecosystem. Maintenance of genetic diversity, a supporting ecosystem service, underpins provisioning and regulating services, providing the raw material that enables humans to modify our environment in beneficial ways in response to change. Soil biodiversity is essential to the maintenance of genetic diversity as a supporting ecosystem service. Soil microbial and faunal biomass is equivalent to or exceeds above-ground biomass in most ecosystems (van der Heijden *et al.*, 2008; Fierer *et al.*, 2009). These microbes and animals have been referred to as the “unseen majority” (Whitman *et al.*, 1998; van der Heijden *et al.*, 2008) and likely encompass at least as much genetic diversity as has been documented in above-ground ecosystems. Wagg *et al.* (2014) demonstrated that as soil biodiversity is lost, ecosystem functions are reduced. With at least half of terrestrial genetic diversity housed below ground, conservation of soil biodiversity is a key component of maintaining genetic diversity.

Soil biodiversity can play a protective role in the emergence of disease. Biodiversity has repeatedly been shown to be negatively correlated with disease prevalence through what has been called the ‘dilution effect’ (Civitello *et al.*, 2015). Greater soil biodiversity is associated with enhanced control of pest and pathogen populations and reductions in disease incidence (Ferris and Tuomisto 2015; Wall *et al.*, 2015). There is some evidence that soil biodiversity may even directly mitigate human illness. In a study of teenagers in a small town in Finland, Hanski *et al.* (2012) found that exposure to greater soil biodiversity was associated with a reduced risk of allergies. Fungal plant pathogen diversity increases with plant diversity, yet fungal disease incidence and severity decline (Rottstock *et al.*, 2014). One possible mechanism for this observed decline is interference between pathogens (Ostfeld and Keesing, 2017), which is more likely in more diverse systems. The same relationship is expected for soil biodiversity and disease incidence and severity, although data are currently lacking.

Because most species are rare, taxonomic diversity tends to be negatively correlated with species abundances. Consequently, if higher soil biodiversity reduces pathogen abundance, then disease risk should be lowest in the most diverse communities (Ostfeld and Keesing, 2017). Extending to soil ecosystems, the finding that more diverse microbiomes within hosts suppress strains resistant to antimicrobial compounds (Keesing *et al.*, 2010) suggests that conservation of soil biodiversity is of critical importance to slowing the emergence of new diseases and antimicrobial resistance of disease agents for crops, domesticated animals and humans. Recent work by Kinkel and others (Schlatter *et al.*, 2017) advances the hypothesis that soils that are generally suppressive of plant disease result from competitive interactions among pathogen populations and diverse networks of saprophytic populations. Such general suppression would depend on the diversity of potential competitors for pathogens. If soil organisms at higher trophic levels are more susceptible to extinction than those at lower trophic levels (such as plant or animal pathogens), as has been observed in animal systems (Duffy, 2003), then conservation of soil biodiversity is also essential for the maintenance of organisms capable of natural regulation of populations of disease agents.

3.6 | CULTURAL SERVICES

Decades ago, the influential pedologist, Hans Jenny, noted the growing cultural gap between soils and humans, further heightened by the decline in the ability of soil scientists to feel connected to and communicate about soils: “we don’t mention our emotional involvements. In fact, our soil language is lifeless, and the soil descriptions in our publications are utterly boring.” He goes on to say that our “intellectual isolation and ... invisibility have to do with the lack of formulating exciting ideas about soils themselves and their relations to people” (Jenny, 1999).

Cultural ecosystem services in the Millenium Ecosystem Assessment refer to the “the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences” and include not only services, but benefits and values (Chan *et al.*, 2012, Milcu *et al.*, 2013). The FAO Ecosystem Services and Biodiversity (ESB) framework includes the following among cultural services: aesthetic inspiration, cultural identity, sense of home, and spiritual experience related to the natural environment. Cultural services are deeply interrelated and also connected to provisioning and regulating services (Chan *et al.*, 2012). Good health and well-being (SDG 3) is also strongly connected to many cultural services. The knowledge that soil is alive expands the possibilities for human-soil relationships. Discovery of the commonalities shared by the diverse microbiomes that thrive in the bodies of both humans and soils makes these connections even more pertinent and compelling. The concept of “soil health” reflects society’s increasing recognition of soil’s aliveness which stems, of course, from soil biodiversity (Kirschenmann, 2005). With that has come concern for and increasing desire to take better care of soils (Krzywoszynska, 2016, 2019).

3.6.1 | SPIRITUAL EXPERIENCE AND SENSE OF PLACE

A sense of place is tied, in part, to its landscapes and soils. The sensory information that defines a place - its sights, sounds, smells - stems in part from biodiversity of its soils. This includes the local insects that spend part of their life cycle below ground and the scents of post-rain geosmin from actinomycetes or hydrogen sulfide from sulfate reducers in swampy soils. Among rural communities connected to land, consideration of soil is less as an asset and more as a partner in a relationship (Puig de la Bellacasa, 2019). Values and associated principles emerging from their specific landscapes and ways of life have guided and sustained the traditions and practices of farmers (Fitter *et al.*, 2010; McNeill and Winiwarter, 2004). Globally, farmers value the importance of soil biodiversity, either for its own sake or as translated through its impact on soil properties and functions. In

many cases, the traditional knowledge of farmers has been disconnected, or lost, with the introduction of “improved” seeds, agrichemicals, equipment and cultural practices brought about by development and engagement in global markets (Barrera-Bassols and Zinck, 2003). In reaction to widespread degradation and loss of soils via erosion and nutrient depletion over the past decades, however, the concept of soil health has emerged, with many farmers recultivating human-soil relations (Krzywoszynska, 2019).

Soil has long been part of philosophical systems and religions throughout the world (Minami, 2009). The origin or creation stories of many cultures include involvement of clay, soils or earth. Throughout the world, soil is the home or origin of numerous spirits who protect the land, crops and harvest. The “energy” or “vitality” that some cultures believe give rise to soil’s generative power (Minami, 2009) would appear to be associated with soil’s life; with its biodiversity.

Specific soil properties and functions combine to define a sense of place in crop-growing regions via the foods and wines grown there. *Terroir*, or the interaction of climate, soil, and the vine in a given place (van Leeuwen *et al.*, 2004, 2018) determines the character and quality of grapes grown in a particular region. For wine, we know that the soil microbiome plays a central role in *terroir*, both in viticulture (for example, in soil fertility and crop health) and enology (for example, in fermentation and flavour; Belda *et al.*, 2017). The concept of *terroir* has also been described for other foods including vegetables, fruits and cheeses (Trubek, 2008).

Historically and continuing today, many human conflicts and wars are fought over land and often arise in regions with seriously degraded soils (due to such factors as erosion or drought; Lal, 2015). Though not called out explicitly, declines in the below-ground biodiversity of these regions undoubtedly contributed to soil degradation and provided additional reasons for conflict. Collapses of civilizations in some regions of the world have followed periods of long-term conflicts over land and soil (Hillel, 1992; Minami, 2009).

3.6.2 | AESTHETIC APPRECIATION AND INSPIRATION FOR CULTURE, ART AND DESIGN

For centuries, a diverse variety of art forms and traditions have engaged with soils, using the soil as a medium itself (for example in pottery and as pigment) or as the subject of artwork (such as in landscape paintings). A number of publications have compiled and catalogued many of these creations and activities (Landa and Descola, 2010; Toland *et al.*, 2018; Feller *et al.*, 2015). Soil is featured or embodied in paintings, ceramics, sculpture, literature, philosophy, cinema, architecture, performance art and multi-media productions, among other forms of art (Minami, 2009). Direct engagement with soil is part of many recent art exhibitions and public engagements (Puig de la Bellacasa, 2019). Soil biodiversity contributes to many art works, either responsible for some of the shapes, reliefs, and colours depicted in landscapes, or as the origin of earth-derived colours that wind up on palettes or cave walls. (Figure 3.6.2.1). Examples are pigments originating from iron and manganese-oxidizing or reducing bacteria (Tuli *et al.*, 2015).



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Figure 3.6.2.1 (next pages) | Soil biodiversity and inspiration for art

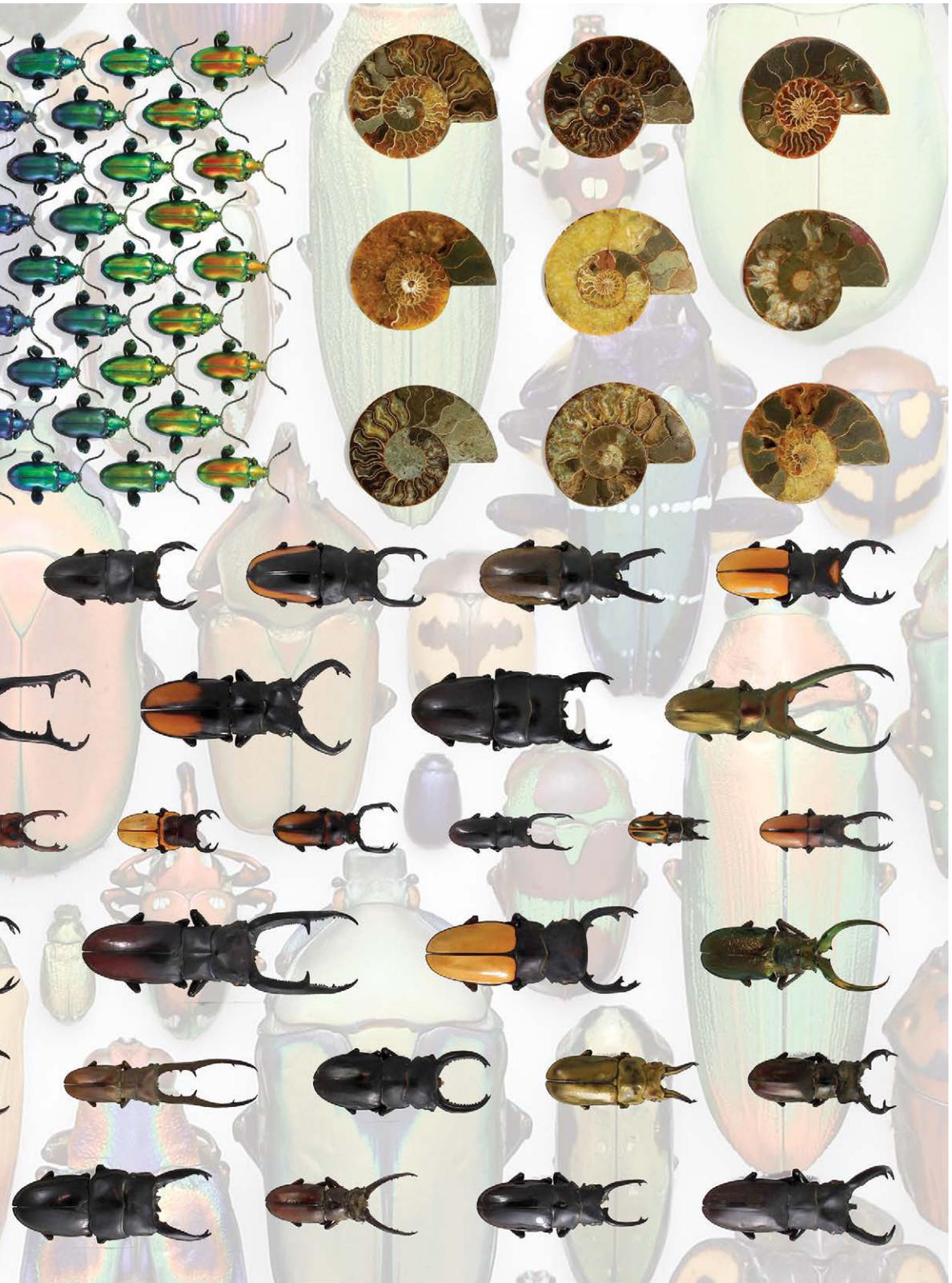
The beauty of organisms, their wide variety of shapes, colors and patterns has inspired different forms of art and soil biodiversity is not the exception. Author: Christopher Marley.



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3.6.3 | CULTURAL HERITAGE, KNOWLEDGE AND EDUCATION

Soils are major repositories where physical artefacts of human cultural heritage (such as structures, objects and organisms) become archived. These sites are important for understanding culture and history, and are the subject of archaeology and paleoecology (Adhikari and Hartemink, 2016). Soil biota play important roles in both preservation and deterioration of these artefacts. The microbial processes involved are both direct – via decomposing organic material or corrosion – and indirect – through governing soil environmental conditions such as redox, pH, and nutrient availability. Understanding these soil biological processes could lead to tools and practices to better preserve these historical sites and archives.

Ethnopedology is the study of local knowledge of soils, and recognizes the value of the cultural context in local sustainable land management. This includes “beliefs, myths, rituals and other symbolic meanings, values and practices related to land management, and soil quality evaluation” (Barrera-Bassols and Zinck, 2003). Ethnopedology underscores how local technical knowledge cannot be abstracted from its cultural context, as well as how essential both the natural and social sciences are in understanding how humans “engage symbolically, cognitively and practically with soil and land resources” (Barrera-Bassols and Zinck, 2003).

3.6.4 | RECREATION AND MENTAL AND PHYSICAL HEALTH

Natural spaces provide social, recreational, spiritual, and sometimes therapeutic benefits for society. There has long been considerable public support for green spaces, national parks, and wilderness areas. More recently, the many human health benefits associated with contact to Nature - including soil - have been documented, and these encounters are being prescribed, for example with Nature Rx and creation of gardens at clinics, prisons, schools and places of employment (Frumkin *et al.*, 2017). Many positive attributes of these spaces (such as tranquility, wildness and visual and olfactory stimulation) depend on soil biodiversity, particularly in less managed areas. Soil can be an essential part of what makes a tourist destination desirable, for example in agrotourism and wildlands. Impoverishing biodiversity not only jeopardizes these spaces but negatively impacts human enjoyment and the intrinsic value of such experiences (Fitter *et al.*, 2010). Soil is vulnerable and can be severely degraded through compaction by feet and vehicles, use of chemicals, careless waste disposal, and destruction of vegetation (McCool and Moisey, 2001). Sustaining heavily impacted locations year after year depends on the activities of

soil organisms to regenerate them through re-building organic matter and soil structure, cleaning up pollution, and supporting reestablishment of plant communities. Though not the primary goal, the economic value of these ecosystems for tourism and recreation is often greater than their ecological value for provisioning services (Fitter *et al.*, 2010).

Human-soil relationships

Kirschenmann (2005) quotes the philosopher Aldo Leopold, who reminds us that “we are not “conquerors” of the biotic community, we are simply “plain members and citizens” of it. Leopold notes that we can learn from soil: “it is the soil that helps us to understand the self-limitations of life, its cycles of death and rebirth, the interdependence of all species. And our task as humans is not to ‘save’ the environment, nor preserve things as they are, but to engage the environment in ways that revitalize the biotic community” (Kirschenmann, 1997).

The scholars Anna Krzywoszynska and Maria Puig de la Bellacasa, bringing new perspectives from anthropology and geography, describe the valuing of soils and soil biodiversity through the idea of “care networks” that connect humans and non-humans, such as soil biota (Krzywoszynska, 2016). Soils are viewed not simply as providers of services for human exploitation and consumption but as “living worlds with an intrinsic value for themselves beyond human use” (Puig de la Bellacasa, 2019). Krzywoszynska (2019) writes about the need for more ethical relations between humans and non-humans, especially with soil organisms who “pervade and create liveable environments” and where “care for non-human lives (e.g. soil biota) becomes part and parcel of caring for human well-being.”

Krzywoszynska and Puig de la Bellacasa’s ideas have been inspired, in part, by scientific advances in our understanding of soil food webs, and through application of non-invasive technologies for visualizing soil (Puig de la Bellacasa, 2015). Building on the knowledge that soil biota is organized into food webs provides a platform to integrate thinking about the role of all organisms and flows of material into ethical systems that are not solely human-centred and materialist.

Importance for future

Inclusion of social-cultural perspectives in considerations of soil biodiversity broadens public engagement by increasing social acceptance and legitimacy of management decisions, finds common ground to engage actors with different values and goals, and bridges gaps among different disciplines and cultures. Public support and policy change for soil and its biodiversity will not be driven by science alone. For society at large, the sense of discovery, connection and inspiration rooted in soil biodiversity - knowing that soil is alive – is the best means to expand enthusiasm and advocacy for soils.

3.7 | ECONOMIC VALUE OF SOIL BIODIVERSITY

Soil biodiversity provide a plethora of values to human society by contributing significantly to ecosystem services from soil biota. Values in this context are purely anthropocentric, as opposed to ethical or bio-centric reasoning on biodiversity. It follows the framework of the Millennium Ecosystem Assessment and TEEB.

Soil biodiversity can be considered a natural capital asset from which a flow of soil ecosystem services is produced (Turner and Daily, 2008). When the soil biodiversity asset is reduced, for example due to degradation of soils, this will lead to costs to society (such as requiring mitigation of environmental impacts such as GHG emissions) or to landowners (through increases of costly inputs to alleviate the decline in soil ecosystem services).

If society omitted to account for the value of soil biodiversity and the costs of running down the natural capital from soil biota, policies would be misguided and society would misallocate scarce resources (Pascual *et al.*, 2015). Continuing along the economic concept of soil biodiversity as a natural capital asset, soil biodiversity represents a portfolio of resources that build up soil natural capital; the flow of ecosystem services provides a return or interest received from the asset (Perrings *et al.*, 2006) cited by (Pascual *et al.*, 2015).

Society and private landowners directly or implicitly make trade-offs between enhancing soil biodiversity or investing in other economic activities on the land, depending on how they perceive the expected net returns. In other words, landowners and society contribute directly and indirectly through regulation and consumption choices to maintaining or reducing the natural capital asset of (high) soil biodiversity. Shedding light on the values at stake across space and time is crucial for making optimal decisions. These optimal decisions may differ depending on whether we consider public or private goods flowing from soil biodiversity to be assets.

An example of a private good from soil biodiversity is crop yield, which is dependent on the support of nutrient delivery and water regulation provided by soil biodiversity.

Examples of a public good from well-functioning soil biodiversity are carbon sequestration and nutrient retention, which contribute directly to two essential planetary boundaries: global nitrogen cycle and climate (Rockström *et al.*, 2009; Steffen *et al.*, 2011).

Different approaches exist to attribute monetary values to changes in soil biodiversity, depending on the type of value. Pascual *et al.* (2015) offer a framework based on the Total Economic Value of soil biodiversity, which is the cumulative value of total output value. This consists of use and non-use values and the natural insurance value, which relates to the capacity of soil biodiversity to maintain the production of ecosystem services over time under risk and uncertainty. It is important to recognize that the economic values of ecosystem services do not exist without the human values, and that human inputs into land

utilizations also contribute to the generation of value of the ecosystem service provision. Ecosystem services are not generated purely from natural processes (Figure 3.7.1).

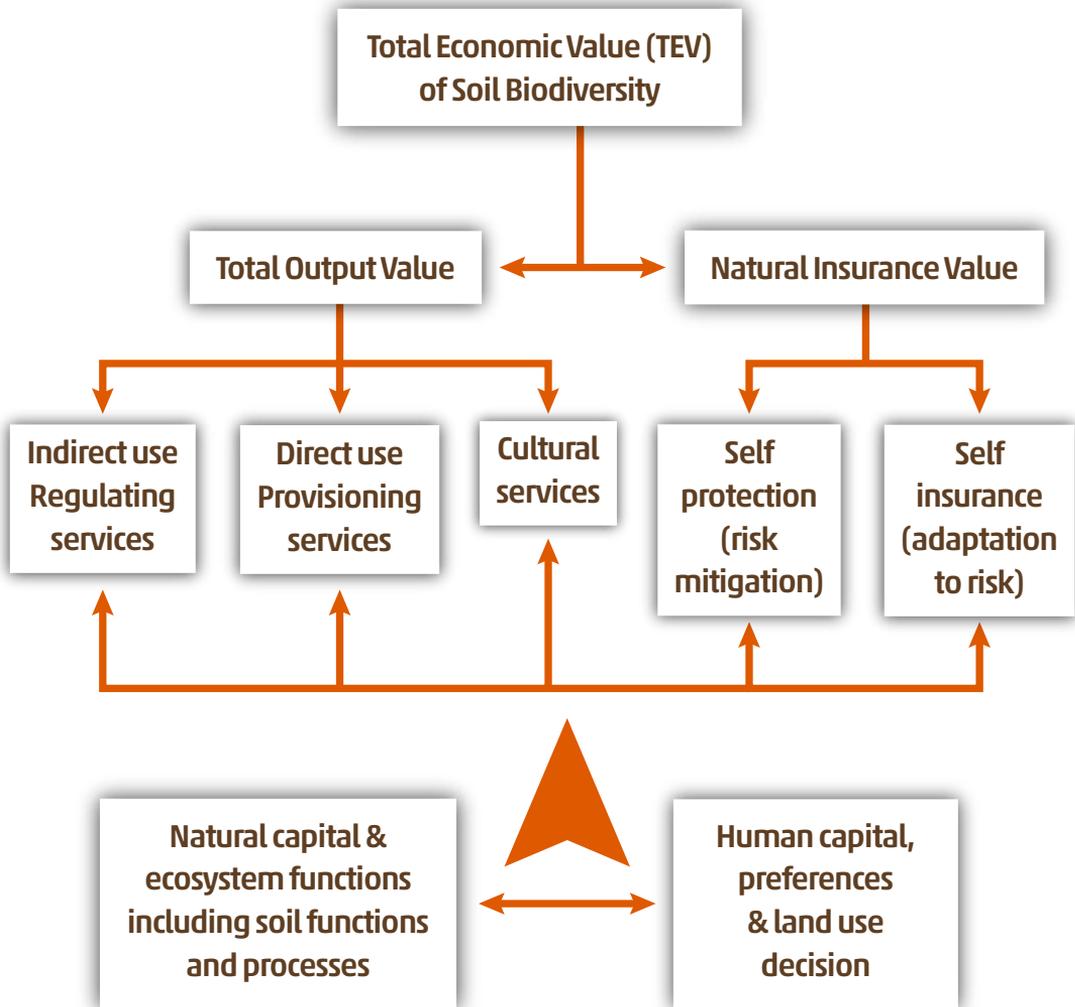


Figure 3.7.1 | Economic value of ecosystem services

Components of Total Economic Value in relation to ecosystem services, ecosystem functions and land use decisions making. Adapted from Pascual *et al.* (2015).

While the Total Economic Value framework is accepted and widely used in environmental assessments (such as UKNEA), the emphasis has been on quantifying the Total Output Value (Figure 3.7.1) and less attention has been given to evaluation of the value of accounting for risks and uncertainty in the provision of ecosystem services, that is, the natural insurance value.

Furthermore, the literature on economic values of ecosystem services has focused on measuring different *components* of value rather than on quantifying the role of the supporting functions underpinning those values. One of the main reasons for this has been the concern that quantifying supporting services, intermediate services and final services independently would lead to double (or triple) counting and result in estimates that would not be usable for economic prioritizations in decision-making processes (Fisher and Turner, 2008). This remains a valid concern (Yang *et al.* 2018) as the value of the final services such as food production already include the value of the biological processes underpinning the production. However, this does not imply that the interdependences among the multiple services supported by soil biodiversity can be ignored or that they are not economically relevant. It is essential that valuations of alternative land utilization plans take account of how investments in the provision of some services may have synergistic effects on other services, whereas investment in other services may result in trade-offs.

A fairly recent development in natural resource economics has been the concept of natural insurance value, originally proposed in the context of biodiversity economics by Baumgärtner (2007). Baumgärtner shows that biodiversity can be interpreted as a natural insurance value to the risk-averse natural resource manager and that this value component is an additional component to use and non-use values in the total economic value framework. This implies that including natural insurance values from biodiversity would not lead to double counting and that accounting for natural insurance would lead to higher investments in biodiversity.

In more recent work, the concepts have been developed further and have distinguished insurance value into two types – a component relating to mitigation of risk (reducing the probability of adverse outcomes) and adaptation to risks (lowering the magnitude of impacts in the case of an adverse event; Baumgärtner, 2007). The empirical quantifications of natural insurance values remain scarce, but research is emerging using integrated ecological-economic models on the value of investment in regulating soil functions (Sidibé *et al.*, 2017). A real policy concern in this field is the argument that provision of market insurance to adapt to increases in risk due for example to climate change may potentially reduce farmers' incentives to invest in sustainable soil practices. There is some empirical evidence that this should be a concern (Wu, 1999). In response to this it has been suggested that development of market insurance products should consider implementation of sustainability conditions to ensure that long-term provision of private and public goods from soils is protected (Jørgensen *et al.*, 2020).

Public and private goods from soil biodiversity

The optimal decisions about preserving or enhancing soil biodiversity may differ depending whether we consider public (collective) or private goods flowing from soil biodiversity assets through bundles of soil ecosystem services.

Private goods are characterised by being exclusive and rivalrous, such as crop production that is appropriated by private landowners, while collective goods are the opposite (non-exclusive and non-rivalrous), such as soil biodiversity's contribution to reducing soil erosion or eutrophication, benefitting people outside the farm gate. Optimization of private goods (for example, maximizing yields through intensive farming practices) typically leads to an undersupply of public goods, causing negative externalities such as nutrient pollution and GHG emissions. The notion of public and private goods is therefore central to the management of soil biodiversity. The present soil biodiversity crisis is aggravated by the property rights regime (Bartkowski *et al.*, 2018), although there is a growing recognition of the social obligations of property owners to manage land more oriented towards the common good.

The natural insurance value of soil biodiversity exhibits both private and public goods characteristics. A private landowner can benefit from the natural insurance value of mitigating and/or adapting to risks on his or her own land. Also, future owners benefit from current sustainable management practices, giving incentives to adopt a long-term perspective. This is however complicated by the extended practice in developing countries of rented agricultural land, where the tenant, compared to the landlord, is found to be disincentivised to invest in soil biodiversity assets generating long-term private benefits and instead overexploit soil biota services (Foudi, 2012). In the face of risk and uncertainty, the natural insurance value of soil biodiversity for collective goods include for instance the capacity over time to maintain carbon sequestration services or reduce eutrophication and soil erosion, which is appropriable by society at large, but controlled by private landowners (Pascual *et al.*, 2015).

3.7.1 | THE CHALLENGE OF VALUING SOIL ORGANISMS

Many would say that it is not possible to place a value on ecosystems, nor on their natural capital and on the services they provide to people, or that it is meaningless to estimate the value of natural capital stocks at large scales because this value is essentially infinite (Costanza *et al.*, 1997; Robinson *et al.*, 2013). Valuing total ecosystem services at a national level invites a similar criticism (Robinson *et al.*, 2013). It is the effect of change in stocks or flows, or margins, rather than estimating a total value, that is important. Nevertheless, Costanza *et al.* (1997) did estimate the annual value of services provided by ecosystems of the world, and this was reckoned to be somewhere between USD 16 and USD 54 trillion, in comparison to an annual global gross national product of around USD 18 trillion. These ecosystem services included all the benefits that humans derive from processes acting upon natural capital (that is, ecosystem assets and natural resources). Since this landmark paper, there has been a steady rise in the number of

publications on ecosystem services, but only a small proportion include studies that directly link soil properties to the services (Adhikari and Hartemink, 2016).

Making a robust assessment of the economic value of nature and biodiversity is a huge challenge because there are many different ways to derive value and because value is not the same as price (*Nature* Editorial, 2019). The concept of economic valuation of ecosystem services leads to a hierarchy of categories of value with their sum equalling what is termed the total output value (see Baveye *et al.*, 2016; Jónsson and Davíðsdóttir, 2016). Total output value can be divided into instrumental value (that value directly benefiting humans) and intrinsic value (the non-anthropocentric value, for example conferred by genetic diversity and ecological processes). Instrumental value itself can be split into use value and non-use value. Use values are those related to direct consumption or use of services, or regulation services. Non-use values are those detached from the actual use of the service (for example option, bequest and existence values).

The soil ecosystem service valuation framework in Pascual *et al.* (2015) included a further category of “natural insurance value.” This is value that one places on the ability to reduce risk of reductions in ecosystem service flows in uncertain conditions. Hence, in addition to using stocks to value ecosystem services, the management of soil natural capital and link to services may also be used as an approach to increasing system resilience and mitigating production risks in agriculture (Cong *et al.*, 2014). We can also therefore think about the massive diversity and likely functional redundancy found in soil communities from the perspective of greater stocks of soil biodiversity delivering resilient ecosystem service flows under disturbance or environmental stress.

The variety of values is accompanied by a variety of valuation methods. Market-based valuation can be made using market prices, production functions or replacement costs; valuations based on market prices require a well-functioning market (POST, 2011) and, for example, regulating services are not typically marketed and their valuation is more challenging. Non-market valuation can be made using a revealed preference approach (such as hedonic pricing or travel costs) or a stated preference approach (such as contingent valuation or price modelling; Baveye *et al.*, 2016). Ecosystem service valuation is also dependent on market values; these differ geographically and culturally, with services having different values (Robinson, 2013). It is also argued that assets managed for monetary values only, as opposed to more intangible values, may risk being mismanaged, with increased chances of degradation (POST, 2011).

The actions of soil organisms are strongly linked to important tangible and intangible economic values. Focusing on the stocks of soil natural capital is valuable because change in soil natural capital stocks can be used to estimate the value of potential gains or losses of ecosystem service flows. We cannot, however, estimate natural capital stocks by examining changes in ecosystem service flows. Baseline data on the stock of organisms provides a powerful gauge against which future changes in soil properties and threats to their function, including climate change, may be quantified. Regular measurements over time can provide evidence as to whether the soil organism stock is in decline, and can provide inference on changes in the value of ecosystem services.

Examples of valuations

At the local scale, Dominati *et al.* (2014) used a range of valuation methods (market price, replacement costs, provision cost, defensive expenditure) to value soil services in a pasture system in New Zealand. For example, flood mitigation was valued using the costs associated with building an on-farm water-retention dam, to substitute for the water retention capacity of the soil. They presented six guiding principles for such valuation: differentiating soil services from supporting processes; identifying key soil properties and processes behind each service; distinguishing natural capital from added/built capital; identifying how external drivers affect natural capital stocks; analyzing the impact of degradation processes on soil properties; and basing the economic valuation on measured proxies (Dominati *et al.*, 2014). The recent paper by Pascual *et al.* (2015) demonstrated two examples of valuation of soil biodiversity and ecosystem services, with the specific cases of the value of earthworms to water infiltration, and the value of earthworms to crop productivity and greenhouse gas regulation. This study was carried out in the context of soil biodiversity as an insurance against climatic variability and the implications for agricultural outputs.

Agricultural activities have been depleting or degrading soil natural capital for centuries. Improving or increasing soil natural capital is therefore a major step towards more sustainable agricultural systems (Pretty, 2008). Our land management practices determine whether the stocks of natural capital are being degraded (in the case of non-sustainable practices), maintained or improved. Degradative changes in soil ecosystems impinge on their condition or quality and, therefore, on their functional capacity. In Europe, there has been much focus on the impacts of soil erosion and organic matter depletion.

Eight major threats that were identified and based on the Impact Assessment (European Commission, 2006) were estimated, on an annual basis, to cost the following to the European Union: erosion: EUR 0.7 to 14.0 billion; organic matter decline: EUR 3.4 – 5.6 billion; salinization: EUR 158 – 321 million; landslides: up to EUR 1.2 billion per event; contamination: EUR 2.4 – 17.3 billion. No estimate was possible for compaction, sealing and biodiversity decline. Degradation of soils in England and Wales was estimated to cost between GBP 0.9 billion and GBP 1.4 billion per year (Graves *et al.*, 2015). Having a thorough knowledge of the stock of soil organisms means that the impacts of degradation threats on ecosystem service provision and valuation can be better understood. Indeed, Adhikari and Hartemink (2016) suggest that future research on ecosystem services should focus on exploring functional diversity of soil biota.

Opportunities and knowledge gaps

Understanding the value of ecosystem services linked to soil organisms is vital for decision-makers when considering soil use and land management changes. We need to sift through the multitude and complexity of approaches when thinking about economic valuation of services mediated by soil organisms. It is vital that clear and influential messages be presented to different groups of stakeholders. The solution is not to generate another framework to value the benefits derived from soil organisms, but to make better use of the data and examples we have and to communicate these in a coherent and effective manner.



Two general (an not mutually exclusive) routes towards this goal are suggested, with contrasting coverage and depth. For the first option, a key group of soil organisms should be selected and their diversity and abundance considered as holistic indicators of a capacity for provision of sustainable soil services. The economic value of these indicator organisms should then be derived at a global scale. A good example would be to use earthworms. Economic values for the ecosystem services that earthworms provide were shown to be substantial. Bullock *et al.* (2008) suggest that earthworms add EUR 723 million per year to livestock production in Ireland; adding the equivalent value for food crops could raise the total value of earthworms to over EUR 1 billion. Bailey *et al.* (1999) estimated the value of earthworms for soil structuring service at GBP 0.48 per kilogram of earthworms under reduced tillage. If such valuations of earthworm-mediated ecosystem services could be generalized across the globe, we could begin to develop an appreciation for their value. For the second option, efforts must be made to generate a global map of case studies based on a pluralistic valuation approach (Pascual *et al.*, 2015b), where location-based valuations for soil biodiversity are made for areas representing different biomes and different land management practices. This approach should take into account the fact that soil ecosystem service assessment is dependent on market values, which can change over time, as well as between countries or regions where services have different values (Robinson, 2013). We must invest in soil biodiversity.



