



Food and Agriculture Organization
of the United Nations

Status of the World's Soil Resources

Main Report

Chapter 2
The role of soils
in ecosystem
processes

© FAO / Giuseppe Bazzani



GLOBAL SOIL
PARTNERSHIP



INTERGOVERNMENTAL
TECHNICAL PANEL ON SOILS



2015
International
Year of Soils

Disclaimer and copyright

Recommended citation:

FAO and ITPS. 2015.
Status of the World's Soil Resources (SWSR) – Main Report.
Food and Agriculture Organization of the United Nations
and Intergovernmental Technical Panel on Soils, Rome, Italy

The designations employed and the presentation of material in this information product do not imply the expression of any opinion whatsoever on the part of the Food and Agriculture Organization of the United Nations (FAO) concerning the legal or development status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries.

The mention of specific companies or products of manufacturers, whether or not these have been patented, does not imply that these have been endorsed or recommended by FAO in preference to others of a similar nature that are not mentioned.

The views expressed in this information product are those of the author(s) and do not necessarily reflect the views or policies of FAO.

ISBN 978-92-5-109004-6

© FAO, 2015

FAO encourages the use, reproduction and dissemination of material in this information product. Except where otherwise indicated, material may be copied, downloaded and printed for private study, research and teaching purposes, or for use in non-commercial products or services, provided that appropriate acknowledgement of FAO as the source and copyright holder is given and that FAO's endorsement of users' views, products or services is not implied in any way.

All requests for translation and adaptation rights, and for resale and other commercial use rights should be made via www.fao.org/contact-us/licence-request or addressed to copyright@fao.org.

FAO information products are available on the FAO website www.fao.org/publications and can be purchased through publications-sales@fao.org.

2 | The role of soils in ecosystem processes

Soils play a critical role in delivering ecosystem services. Management to change an ecosystem process in support of one regulating ecosystem service can either provide co-benefits to other services or require trade-offs (Robinson *et al.*, 2013; Dominati, Patterson, and Mackay, 2010). Recent reviews have provided examples of some of these synergies and trade-offs (Smith *et al.*, 2013) and illustrated the role of soils in supporting ecosystem services and underpinning natural capital (Robinson, Lebron and Vereecken, 2009, Robinson *et al.*, 2014, Dominati, Patterson and Mackay, 2010). In this chapter, we present current knowledge – and knowledge gaps – on the role of soils in the carbon, nitrogen and water cycles, and on their role as a habitat for organisms and as a genetic pool.

2.1 | Soils and the carbon cycle

Carbon (C) storage is an important ecosystem function of soils that has gained increasing attention in recent years due to its interactions with the earth's climate system. Soil is a major C reservoir that holds more carbon than is contained in the atmosphere and terrestrial vegetation combined. All three of these reservoirs are in constant exchange. In many soils, soil organic matter (SOM), which contains roughly 55–60 percent C by mass, comprises most or all of the C stock – referred to as soil organic carbon (SOC). In arid and semi-arid soils, significant inorganic C (IC) can be present as pedogenic carbonate minerals or 'caliche' (typically Ca/MgCO₃), formed from the reaction of biocarbonate (derived from CO₂ in the soil) with free base cations, which can then be precipitated in subsoil layers (Nordt, Wilding and Drees, 2000). Also soils derived from carbonate-containing parent material (e.g. limestone) can have significant amounts of inorganic carbon. However, in most cases changes in inorganic C stocks are slow and not amenable to traditional soil management practices. Hence inorganic carbon does not play a significant role in terms of management of ecosystem services. For this reason, the further discussion of soil C in this chapter will focus on soil organic carbon.

A general overview of the ecosystem C cycle as it interacts with soils is given in Figure 2.1. The major input of organic C to soils is provided by the uptake and fixation of CO₂ by plants (the net result of photosynthesis and above- and below-ground plant respiration), and by the subsequent incorporation of plant residue C (both above- and below-ground) into soil. Some of the fixed plant C may be removed by harvest before entering the soil. Conversely, C additions from offsite sources (e.g. compost, manure) may occur. Organic matter on and in the soil is subject to comminution and mixing by soil fauna and to enzymatic breakdown and metabolism by microorganisms, resulting in release of CO₂ via microbial respiration (also referred to as organic carbon mineralization). Microbial transformations as well as interactions of organic matter with soil minerals greatly influence the stabilization of organic C and its rate of mineralization. In flooded soils, emissions of methane (CH₄) from microbial metabolism can represent a significant gaseous C efflux. Erosion can also directly

affect the soil C balance through the removal and/or deposition of the C contained in the transported soil. Leaching of dissolved organic (DOC) and dissolved inorganic carbon (DIC) through the soil profile and out into groundwater and surface water represents an additional loss pathway that can be significant in some soils.

Maintaining and increasing SOC stocks through improved land use and management practices can help to counteract increasing atmospheric CO₂ concentrations (Paustian *et al.*, 1998, Smith *et al.*, 2007; Whitmore, Kirk and Rawlins, 2014). Increasing soil C content also improves other chemical and physical soil properties, such as nutrient storage, water holding capacity, aggregation and sorption of organic and/or inorganic pollutants (Kibblewhite, Ritz and Swift, 2008). Carbon sequestration in soils may therefore be a cost-effective and environmentally friendly way to store C. It can also enhance other ecosystem services derived from soil, such as agricultural production, clean water supply, and biodiversity by increasing SOM content and thereby improving soil quality (Lal, 2004).

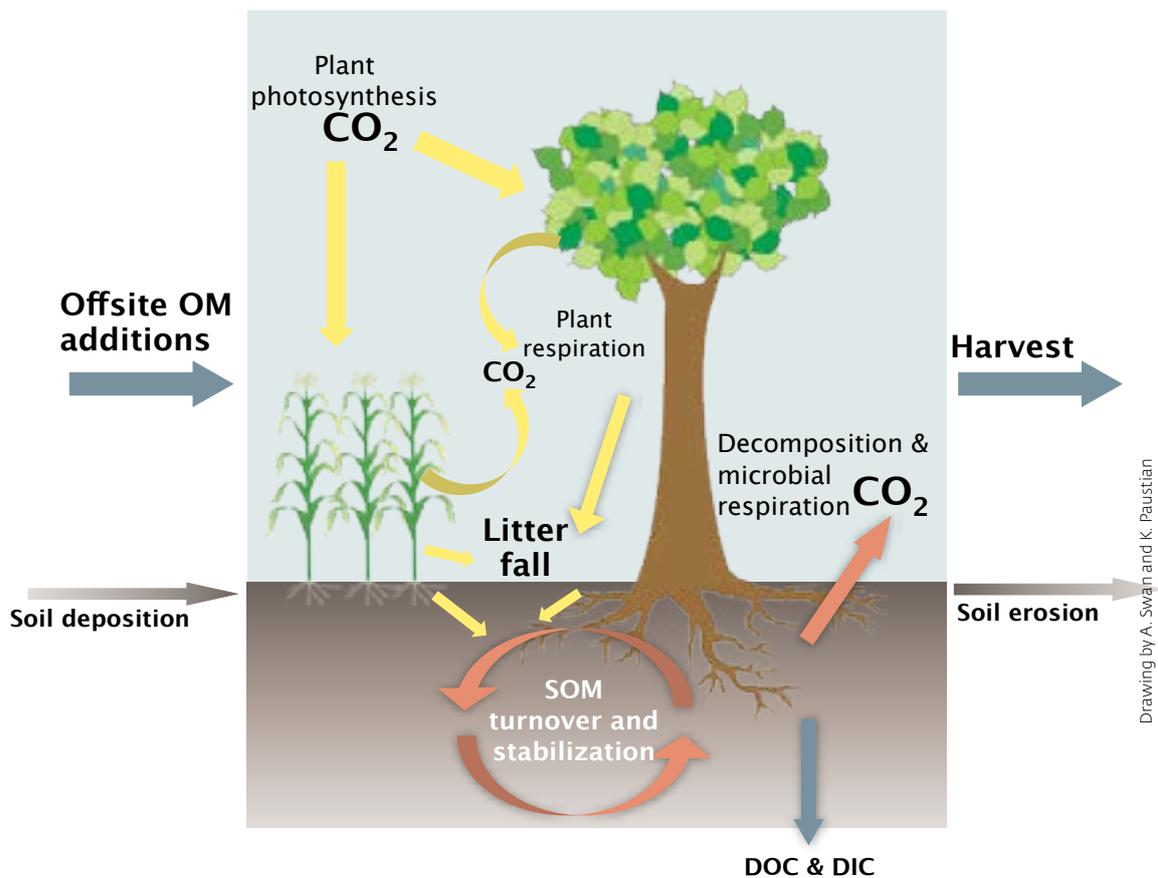


Figure 2.1 | Overview of ecosystem processes involved in determining the soil C balance.

2.1.1 | Quantitative amounts of organic C stored in soil

Organic C stocks in the world's soils have been estimated to comprise 1 500 Pg of C down to 1 m depth and 2 500 Pg down to 2 m (Batjes, 1996). Recent studies, based on newer estimates for the C stored in boreal soils under permafrost conditions, suggest that soil C storage may be even greater, accounting for as much as 2000 Pg to 1 m depth (Tarnocai *et al.*, 2009). Although the highest C concentrations are found in the top 30 cm of soil, the major proportion of total C stock in many soils is present below 30 cm depth (Batjes, 1996). In the northern circumpolar permafrost region, at least 61 percent of the total soil C is stored below 30 cm depth (Tarnocai *et al.*, 2009).

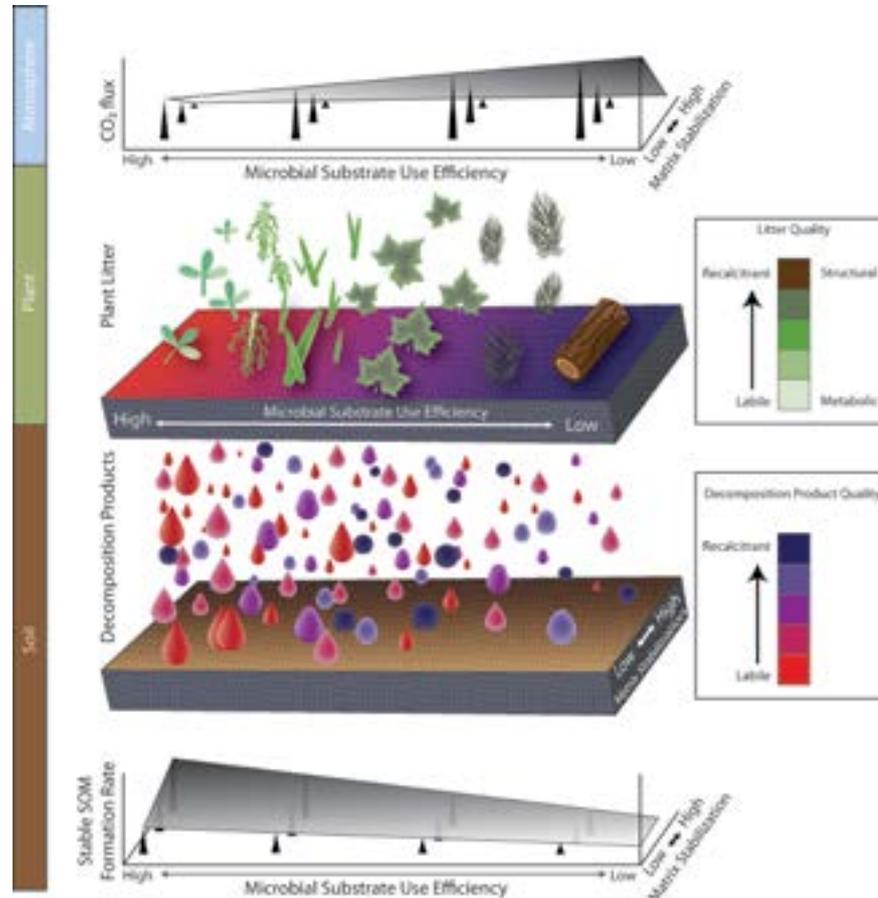
2.1.2 | Nature and formation of soil organic C

Soil organic matter (SOM) is composed of plant litter compounds as well as of microbial decomposition products. SOM is thus a complex biogeochemical mixture derived from organic material in all stages of decomposition (von Lützow *et al.*, 2006; Paul, 2014). Due to microbial degradation and mineralization to CO₂ (and CH₄ in anaerobic environments), the majority of plant litter compounds added to soil remain for a relatively short time (from a few days to a few years). This is particularly the case if the organic compounds are added on the soil surface. However, some organic matter compounds may persist in the soil for decades or centuries or even for millennia (Paul *et al.*, 1997; von Lützow *et al.*, 2006). It is increasingly accepted that, despite their recalcitrant nature, plant litter compounds (e.g. lignin) themselves do not substantially contribute to SOM persistence in soil (Thévenot, Dignac and Rumpel, 2010). Longer term stabilization is generally conferred through interactions with soil minerals (e.g. through surface binding or occlusion within microaggregates), which reduce SOM exposure to enzymatic degradation (Sollins, Homann and Caldwell, 1996; Six, Elliott and Paustian, 2000; Schmidt *et al.*, 2011). Thus, the location of SOM within the soil matrix has a much stronger influence on its turnover than its chemical composition (Chabbi, Kögel-Knabner and Rumpel, 2009; Dungait *et al.*, 2012)

One consequence of the role of reactive mineral surfaces in SOC stabilization is that the surface area of the soil mineral fraction, which is finite and a function of soil texture (e.g. clay, silt or sand content) and of mineralogy, may set an upper limit for the amount of SOM that a particular soil can hold (Six, Elliott and Paustian, 2002). A recent conceptual model (Figure 2.2) by Cotrufo *et al.* (2013), based on studies showing that microbially-derived decomposition products make up most of the mineral-stabilized organic matter, postulates that relatively labile litter compounds with higher microbial growth yield efficiency contribute proportionally more to the stable mineral-associated SOM pool than do more recalcitrant plant compounds with low microbial growth yield efficiency. This concept is in agreement with the current understanding that microbial material is building up much of the stabilised SOM pool (Miltner *et al.*, 2012).

Figure 2.2 | Conceptual model of interactions between litter quality, microbial products and soil mineral interactions affecting the formation and stabilization of organic matter.

Source: Cotrufo *et al.*, 2013.



With this emphasis on the importance of SOM location within the soil, microbial accessibility of organic material at very small scales has become a focus of research in recent years (Lehmann, Kinyangi and Solomon, 2007). The development of powerful new tools like X-ray spectroscopy and secondary ion mass spectrometry now allows the visualisation of organo-mineral interactions at nanoscale. As a result, the location and distribution of organic matter within the soil mineral matrix may now be assessed in more detail (Lehmann *et al.*, 2008). However, before the results obtained with these tools yield information concerning soil C formation at macroscale, upscaling and integration of spatial heterogeneity is necessary (Mueller *et al.*, 2013).

As well as exhibiting tremendous heterogeneity in terms of its composition, the distribution of SOC within the soil is also very heterogeneous, particularly with respect to depth within the soil profile. Whereas upper soil layers receive greater amounts of aboveground litter ('shoot C' from leaves and stems), subsoil C originates primarily from root-derived C as well as from plant- and microbial-derived dissolved organic carbon (DOC) transported down the soil profile. Root C has a greater likelihood of being preserved in soil compared to shoot C (Balesdant and Balabane, 1996) and studies suggest that root C therefore accounts for a larger proportion of SOM (Rasse, Rumpel and Dignac, 2005). In general, C cycling and C formation is most active in topsoil horizons, whereas stabilised C with longer turnover times makes up a greater proportion of the total SOC found in deep soil horizons (Scharpenseel and Becker-Heidmann, 1989; Trumbore, 2009). The accumulation of stabilised C with long residence times in deep soil horizons may be due to continuous transport, temporary immobilisation and microbial processing of DOC within the soil profile (Kalbitz and Kaiser, 2012) and/or efficient stabilisation of root-derived organic matter within the soil matrix (Rasse, Rumpel and Dignac, 2005).

An additional long-term C pool in many soils is pyrogenic carbonaceous matter, formed from partially carbonised (e.g., pyrolysed) biomass during wildfires (Schmidt and Noack, 2000). A portion of this material has a highly condensed aromatic chemical structure (often referred to as pyrogenic carbon or black carbon) that resists microbial degradation and can persist in soils for long periods (Lehmann *et al.*, 2015).

2.1.3 | Soil C pools

For modelling purposes, soil C is usually divided into a number of pools (typically from two to five) in order to represent the heterogeneity in residence time of the vast mixture of different organic compounds in soil (Smith *et al.*, 1997). A useful three pool split of soil C (excluding litter) – into a labile pool, an intermediate pool, and a refractory (stable) pool – is employed in several soil C models, including the Century model (Parton *et al.*, 1987). The *labile pool* represents easily degradable plant material, microbial biomass and labile metabolites, and may turn over within a few months or years. Conceptually, the *intermediate pool* comprises microbially-processed organic matter that is partially stabilized on mineral surfaces and/or protected within aggregates, with turnover times in the range of decades. The *refractory pool*, including highly stabilized organic matter-mineral complexes and pyrogenic C, may remain in soils for centuries or millennia.

Individual model pools (as opposed to the total C stock) are typically not defined as measurable pools *per se*. The kinetics of the model conceptual pools are instead inferred from C dating and tracer studies, laboratory incubations and total SOC dynamics in long-term field experiments (McGill, 1996; Paustian, 1994). Many carbon cycle, ecosystem and crop growth models successfully employ this type of functional representation of SOM (Krull, Baldock and Skjemstad, 2003; Stockman *et al.*, 2013). Nonetheless, ways to reconcile 'measurable' and 'modelable' pools have been under discussion for a number of years (Elliott, Paustian and Frey, 1996; Smith *et al.*, 2002; Dungait *et al.*, 2012). This reconciliation remains a desirable goal for improving understanding of SOC dynamics (Schmidt *et al.*, 2011).

2.1.4 | Factors influencing soil C storage

Fundamentally, the amount of SOC stored in a given soil is determined by the balance of C entering the soil, mainly via plant residues and exudates, and C leaving the soil through mineralization (as CO₂), driven by microbial processes, and to a lesser extent leaching out of the soil as DOC. Locally C can also be lost or gained through soil erosion or deposition (Figure 2.1), leading to a redistribution of soil C at local, landscape and regional scales.

Consequently, a main control on SOC storage is the amount and type of residues that are produced by plants as the primary producers in the ecosystem. Plant productivity and subsequent senescence and death lead, through plant necromass breakdown, to the input of organic C to the soil system. Thus, broadly speaking for a given pedoclimatic condition, higher levels of plant residue inputs will tend to support higher SOC stocks, and vice versa. C levels of many soils are also influenced by fertiliser additions, which are indispensable for sustaining plant productivity in agricultural systems.

In addition to productivity and plant C inputs, climatic factors, such as soil temperature and water content greatly influence soil C storage through their effect on microbial activity. In general, higher soil temperatures increase microbial decomposition of organic matter. Temperature is, therefore, taken as major control of SOM storage in soil C cycle models, although the temperature sensitivity of decomposition for different SOM fractions remains an area of uncertainty (Conant *et al.*, 2011).

Water also influences soil C storage through several processes. Moist but well-aerated soils are optimal for microbial activity. Decomposition rates consequently decrease as soils become drier. However, flooded soils have lower rates of organic matter decay due to restricted aeration (e.g. O₂ depletion due to limited O₂ diffusion in water) and thus may often yield soils with very high amounts of soil C (e.g. peat and muck soils). High precipitation may also lead to C transport down the soil profile as dissolved and/or particulate organic matter. During extreme events, such as drought, SOM decomposition may initially decrease but may subsequently increase after rewetting (Borken and Matzner, 2008). Fire may decrease soil C storage at first, but over the longer term may increase C storage through positive effects on plant growth and through input of very stable pyrogenic C (Knicker, 2007).

The quantity and composition of SOC in mineral soils is also strongly dependent on soil type, with clay content influencing not only the amount but also the composition of soil C. In clay rich soils, higher organic matter content and a higher concentration of O-alkyl C derived from polysaccharides may be expected, compared to sandy soils which are characterised by lower C contents and high concentrations of alkyl C (Rumpel and Kögel-Knabner, 2011). Aliphatic material may contribute to the hydrophobicity of soils, which could lead to reduced microbial accessibility and therefore increased C storage.

Bioturbation (the reworking of soils by animals or plants) may further influence the amount as well as the chemical nature of soil C. It may greatly influence the heterogeneity of soils by creating hotspots. On biologically active sites, incorporation and transformation of organic compounds into soil is usually enhanced by bioturbation, leading to organo-mineral interactions and increase of C storage (Wilkinson, Richards and Humphreys, 2009).

Microbial decomposition of SOM may be stimulated (or reduced) by labile organic matter input through the 'priming effect' (Jenkinson, 1971; Kuzyakov, 2002). Positive priming refers to mineralisation of otherwise stable C through shifts in microbial community composition (Fontaine, Mariotti and Abbadie *et al.*, 2003). However, in some cases, the addition of organic matter to soil may also cause changes in the soil microbial communities with regard to the preferentially degraded substrate and therefore impede mineralisation of native SOM (Sparling, Cheschire and Mundie, 1982; Kuzyakov, 2002).

Plant communities are main controlling factors of these processes because they influence organic matter input and microbial activity by their effects on soil water, labile C input, pH and nutrient cycling.

2.1.5 | Carbon cycle: knowledge gaps and research needs

Substantial progress has been made in recent years towards a deeper understanding of the processes controlling soil C storage. There has been progress also in improving and deploying predictive models of soil C dynamics that can guide decision makers and inform policy. However, it is equally true that many new (and some old) gaps in our knowledge have been identified and the need for further research has been assessed. Recent research on soil C dynamics has been driven in part by increasing awareness of: (1) the importance of small scale variability for microbial C turnover (Vogel *et al.*, 2014); (2) interactions between the C cycle and other biogeochemical cycles (Gårdenäs *et al.*, 2011); and (3) the importance of soil C not only at the field scale but at regional to global scales (Todd-Brown *et al.*, 2013).

The most cited knowledge gaps and research needs include:

Basic understanding

- Controls on microbial efficiency of organic matter processing, including biodiversity
- The degree of association or separation of organic matter and microbial decomposer communities in the mineral soil matrix
- Role of soil fauna in controlling carbon storage and cycling
- Dynamics of dissolved organic carbon and its role in determining C storage and decomposition
- Pyrogenic C stabilization and interactions of pyrogenic C with native soil C and mineral nutrients
- Role of soil erosion in the global C cycle

Predictive modelling and assessment

- Reconciliation of measured and modelled SOM fractions
- More explicit representation of microbial controls
- Improved modelling of C in subsurface soil layers
- Distributed soil C observational and monitoring networks for model validation
- More realistic and spatially-resolved representation of soil C in global-scale models

2.1.6 | Concluding remarks

Both biotic and abiotic factors control soil C content and dynamics through their effect on plant litter inputs and microbial decomposer communities. The understanding of the C cycle and the role of soils as a sink or source of CO₂ depends on our ability to integrate knowledge of physical, chemical and biological processes operating at small scales (nm, µm, soil profile) and of the spatial heterogeneity of SOM distribution and decomposition processes at increasing scales (field, region, globe). At the global scale, soils are a major component of the planet's C cycle and can have a strong influence on the concentration of CO₂ in the atmosphere. Thus, land management needs to be based on an understanding of the controls on SOM distribution, stabilisation and turnover in order to safeguard and increase the organic matter content of our soils. This will be an important contribution to both food security and the mitigation of greenhouse gases.

2.2 | Soils and the nutrient cycle

Soils support plant growth and so are vital to humanity. They provide nutrients such as nitrogen (N), phosphorous (P), potassium (K), Calcium (Ca), Magnesium (Mg), Sulphur (S) and many trace elements that support biomass production. Biomass is important for food supply, for energy and fibre production and as a (future) source for the chemical industry. Since the 1950s, higher biomass production and yield increases have been supported through mineral/synthetic fertilization (Figure 2.3). However, intensification of agricultural practices and of land use has in many regions resulted in a decline in the content of organic matter content in agricultural soils. In some areas, extensive use of mineral fertilizers has resulted in atmospheric pollution, greenhouse gas emissions (e.g. CO₂ and N₂O), water eutrophication and human health risks (Galloway *et al.*, 2008).

In coming years, human population and demand for food, feed and energy will continue to rise. In order to sustain biomass production in the future and to mitigate negative environmental impacts, fertile soils need to be preserved. Where soil fertility has declined, it needs to be restored by maintaining sufficient amounts of organic matter in soils (Janzen, 2006). This can be achieved by measures of sustainable management (see Chapter 8 of this volume), including by targeted additions of mineral and organic amendments to soils.

The soil function 'fertility' refers to the ability of soil to support and sustain plant growth, including through making N, P and other nutrients available for plant uptake. This process is facilitated by: (I) nutrient storage in soil organic matter; (II) nutrient recycling from organic to plant-available mineral forms; and (III) physical and chemical processes that control nutrient sorption, availability, displacement and eventual losses to the atmosphere and water.

Managed soils represent a highly dynamic system and it is this very dynamism that makes soils function and supply ecosystem services. Overall, the fertility and functioning of soils depend on interactions between the soil mineral matrix, plants and microbes. These are responsible for both building and decomposing SOM and therefore for the preservation and availability of nutrients in soils. To sustain soil functions, the balanced cycling of nutrients in soils must be maintained.

After carbon (Section 2.1), N is the most abundant nutrient in all forms of life, since it is contained in proteins, nucleic acids and other compounds. Humans and animals ultimately acquire their N from plants, which in terrestrial ecosystems occurs mostly in mineral form (e.g. NH_4^+ and NO_3^-) in soils. The parent material of soils does not contain significant amounts of N (as opposed to P and other nutrients). New N enters the soil through the fixation of atmospheric N_2 by a specialized group of soil biota. However, the largest flux of N

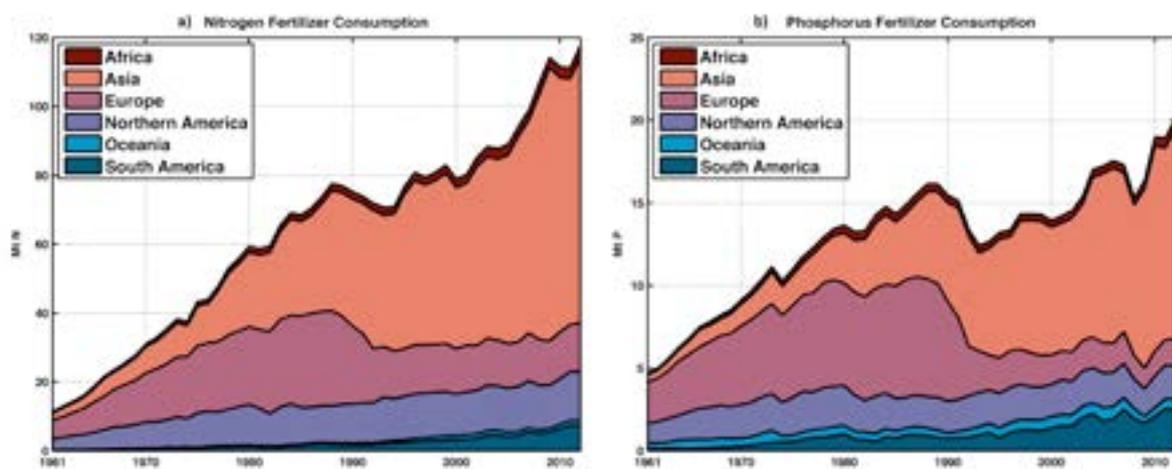


Figure 2.3 | Global (a) nitrogen (N) and (b) phosphorus (P) fertilizer use between 1961 and 2012 split for the different continents in Mt P per year. Source: FAO, 2015.

in soils is generated through the continuous recycling of N internal to the plant-soil system: soil mineral N is taken up by the plant, it is fixed into biomass, and eventually N returns in the form of plant debris to the soil. Here soil biota decompose it, mineralizing part of the N and making it newly available for plant growth, while transforming the other part into SOM, which ultimately is the largest stock of stable N in soil. Nitrogen is lost from the soil to the water system by leaching and to the atmosphere by gas efflux (NH_3 , N_2O and N_2).

In most natural ecosystems, N availability is a limiting factor to productivity and N cycles tightly in the system with minimal losses. Through the cultivation of N_2 fixing crops, the production and application of synthetic N fertilizer, and the deposition of atmospheric N, humans have applied twice as much reactive N to soils as the N introduced by natural processes, thereby significantly increasing biomass production on land (Vitousek and Matson, 1993). However, since mineral fertilizer use efficiency is generally low and far more

fertilizer is often used than plants actually need, a high percentage of N fertilizer is lost from the soil. This is generating a myriad of deleterious cascade effects on the environment and on human health (Galloway *et al.*, 2008). This phenomenon is spread over most of the globe. However, in some regions of the world, in particular Sub-Saharan Africa, which are characterized by eroded soils and where economic constraints limit the use of fertilizers, productivity is still strongly constrained by low levels of soil-available N and other nutrients, notably P (Figure 2.2).

Phosphorus is an essential element for all living organisms. It cycles internally in the plant-soil system, moving from the parent material through weathering to biochemical molecules (e.g. nucleic acid, phospholipids) and back to mineral forms after decomposition (e.g. H_3PO_4). In natural soils P is among the most limiting nutrients, since it is present in small amounts and only available in its soluble forms, which promptly react with calcium, iron and aluminum cations to precipitate as highly insoluble compounds. Adsorbed on those compounds, P can be lost from soils, entering the aquatic system through erosion and surface runoff. To correct this lack of available P, 'primary' P is mined and added to soils in the form of mineral fertilizer. This external input has led to positive agronomic P balances (McDonald *et al.*, 2011). There are, however, large variations in the world, with large surpluses in the United States, Europe and Asia, and deficits in Russia, Africa and South-America (Figure 2.4). Additionally, since plant P uptake is a relatively inefficient process with roughly 60 percent of the total P input to soils not taken up, it has been estimated that the amount of P exported from terrestrial to aquatic systems has tripled, with significant impacts on the environment (Bennett, Carpenter and Caraco, 2001).

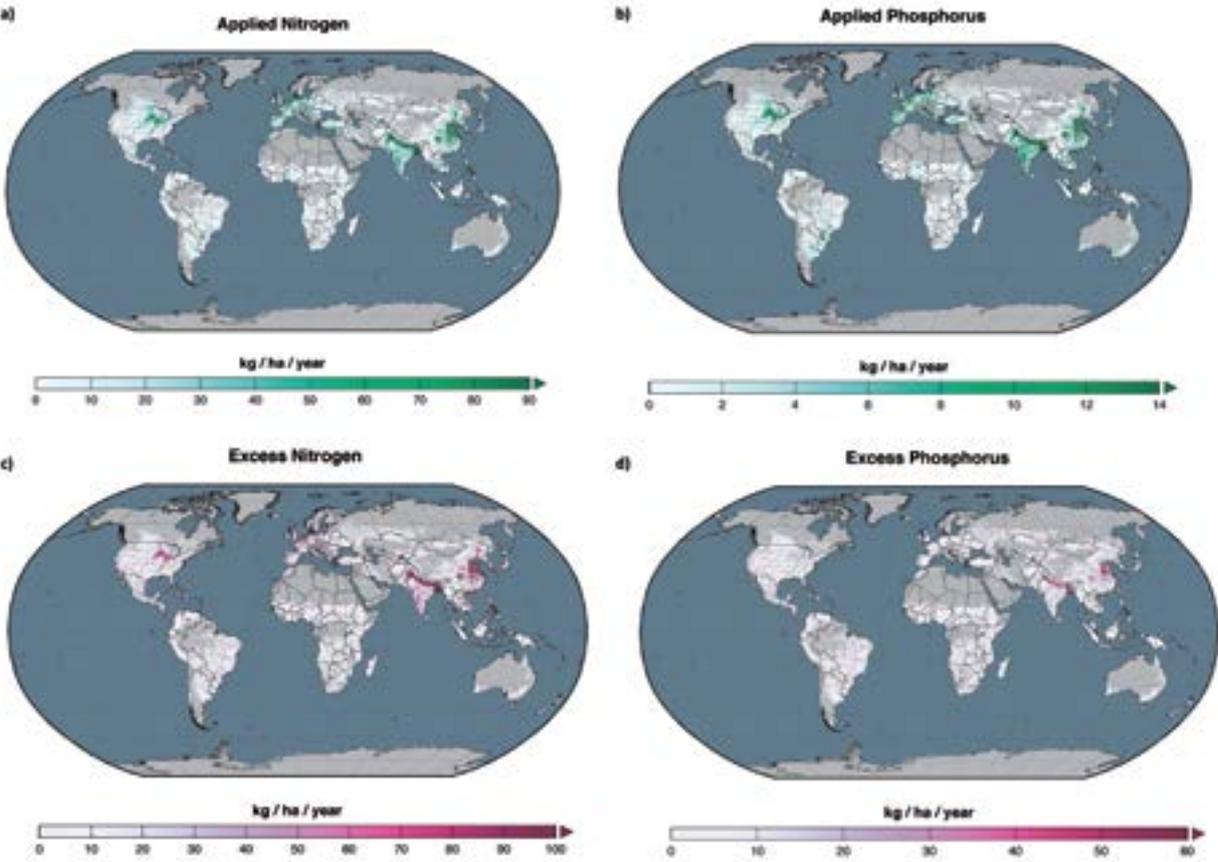


Figure 2.4 | Applied and excess nitrogen and phosphorus in croplands. Nitrogen and phosphorus inputs and excess were calculated using a simple mass balance model, extended to include 175 crops. To account for both the rate and spatial extent of croplands, the data are presented as kg per ha of the landscape: (a) applied nitrogen, including N deposition; (b) applied phosphorus; (c) excess nitrogen; and (d) excess phosphorus. Source: West *et al.*, 2014.



Management practices need to be implemented that sustain, restore or increase soil fertility and biomass production while limiting associated negative impacts. This can be achieved by promoting the accrual of soil organic matter and nutrient recycling, applying balanced C amendments and fertilization of N, P and other nutrients to meet plant and soil requirements, while limiting overuse of fertilizer. Carbon, N and P cycling in soils is coupled by tight stoichiometric relationships (e.g. relatively fixed C:N:P in plants and microorganisms). This means that an enduring increase or decrease of carbon in soils cannot be achieved without a proportional change in nitrogen and phosphorous (and several other nutrients). This is a fundamental consideration in any programs for carbon sequestration and land restoration because of the significant costs. Therefore, their management needs to be planned in concert.

Nutrient management has been extensively studied, with the aim of identifying and proposing management practices (e.g. precision agriculture) that improve nutrient use efficiency and productivity while reducing potentially harmful losses to the environment (van Groenigen *et al.*, 2010; Venterea, Maharjan and Dolan, 2011). However, our ability to predict the ecosystem response to balanced fertilization is still limited and the relationship requires continued monitoring. Further benefits are anticipated from improved plant varieties with root morphologies that have better capacity to extract P from soils or use it more efficiently.

More generally, further research is needed into organic matter responses to agricultural C inputs and into the potential for restoring and increasing soil organic matter to promote long term soil fertility (e.g. Lugato, Berti and Giardini, 2006). Hence, we stress the importance of an integrated approach to nutrient management which supports plant productivity while preserving or enhancing soil organic matter stocks and reducing nutrient losses to the atmosphere or aquatic systems. Prediction and optimization of performance would benefit from continued data acquisition across the whole range of climate and environmental and agro-ecological conditions.

2.2.1 | The nutrient cycle: knowledge gaps and research needs

In the second half of the 20th century, higher biomass yields were supported by higher use of fertilizer (N, P) inputs. This is now considered unsustainable in many situations. Alternatives are required that make better use of inherent soil fertility, improve resource use efficiency, and prevent losses of N and P. Examples in agriculture include sustainable intensification and new crop varieties that have root systems with improved extraction capability or which have higher internal P use efficiency. At the food system level, more effective nutrient management would benefit from a focus on a '5R strategy': (1) *realign* P and N inputs; (2) *reduce* P and N losses to water, thereby minimizing eutrophication impacts; (3) *recycle* the P and N in bio-resources; (4) *recover* P and N from wastes to use as fertilizer; and (5) *redefine* use and use-efficiency of P in the food chain (Withers *et al.*, 2015).

In addition, a better understanding of biogeochemical processes at the molecular level is needed. This should include: (I) research into the role of plant symbionts on the weathering of minerals and support of nutrient uptake, and (II) development of target-specific 'smart' agrochemical agents that enhance nutrient uptake.

2.3 | Soils and the water cycle

Soils provide important ecosystem services through their function within the water cycle. These services include provisioning services of food and water security, regulating services associated with moderation and purification of water flows, and cultural services such as landscapes and water bodies that meet recreation and aesthetic values (Dymond, 2014). Water stored in soil is used for the evapotranspiration and plant growth that supply food and fibre. Soil water also stabilizes the land surface to prevent erosion and regulates nutrient and contaminant flow. At a catchment and basin scale, the capacity of the soil to infiltrate water attenuates stream and river flows and can prevent flooding, while water that percolates through soil can replenish groundwater and related streamflow and surface water ecosystems.

The soil functions of accepting, storing, transmitting and cleaning of water shown in Table 2.1 are inter-related. Soil water storage depends on the rate of infiltration into the soil and on soil hydraulic conductivity that redistributes water within and through the soil profile. Similarly, infiltration and hydraulic conductivity are dependent on the water stored in the soil. The initially high rate of infiltration into dry soil declines as the soil water content increases and water replaces air in the pore space. Conversely, hydraulic conductivity increases with soil moisture content as a greater proportion of the pores are transmitting water. Water content and transmission times are also important to the filtering function of soil because contact with soil surfaces and residence time in soil are controls on contaminant supply and removal.

Optimum growth of most plants occurs when roots can access both oxygen and water in the soil. The soil must therefore infiltrate water, drain quickly when saturated to allow air to reach plant roots, and retain and redistribute water for plant use. The ideal soil for plant production depends on climatic conditions and on the soil requirements of the crop. For instance, in dry regions it can be an advantage to have soils with a high clay content to retain water, while sandier soils that drain quickly are better suited to wetter regions.

Soil structural stability and porosity are also important for the infiltration of water into soil. Organic matter improves soil aggregate stability. While plant growth and surface mulches can help protect the soil surface, a stable, well-aggregated soil structure that resists surface sealing and continues to infiltrate water during intense rainfall events will decrease the potential for downstream flooding. Porosity determines the capacity of the soil to retain water and controls transmission of water through the soil. In addition to total porosity, the continuity and structure of the pore network are important to these functions and also to the further function of filtering out contaminants in flow.

Table 2 | Soil functions related to the water cycle and ecosystem services

| Soil Function | Mechanism | Consequence | Ecosystem service |
|------------------------------------|--|--|---|
| Stores (Storage) | Water held in soil pores supports plant and microbial communities | Biomass production Surface protection | Food Aesthetics Erosion control |
| Accepts (Sorptivity) | Incident water infiltrates into soil with excess lost as runoff | Storm runoff reduction | Erosion control Flood protection |
| Transmits (Hydraulic conductivity) | Water entering the soil is redistributed and excess is transmitted as deep percolation | Percolation to groundwater | Groundwater recharge Stream flow maintenance |
| Cleans (Filtering) | Water passing through the soil matrix interacts with soil particles and biota | Contaminants removed by biological degradation/retention on sorption sites | Water quality |



Another important role of soil water is its support of biota that can degrade compounds into beneficial forms that may also retain nutrients. Soil organic matter is important to this role - together with mineral soil (especially the clay fraction), SOM provides sorption sites, but sorption capacity is finite. Flow through macropores that bypass the soil matrix where biota and sorption sites are generally located can quickly transmit water and contaminants through the soil to groundwater or artificial drains. However, for filtering purposes a longer, slower route through the soil matrix is more effective.

Soil management alters the ecosystem services provided by water (Table 2.2). Soil conservation practices and sustainable management help to retain regulating ecosystem services such as soil organic matter and structural stability. Similarly, the promotion of soil as a C-sink to offset greenhouse gas emissions helps to maintain or improve soil functions. On the other hand, deforestation, overgrazing and excessive tillage of fragile lands lead to deterioration of the soil structure and to loss of soil function and surface water quality (Steinfeld *et al.*, 2006).

Anthropogenic modifications to the water cycle can aid soil function. In dry regimes, inadequate soil moisture can be mitigated through supplementary irrigation, and where excessive precipitation causes problems, waterlogging can be relieved by land drainage. However, irrigation and drainage can have consequences for water regulation services. Irrigation that enables a shift to intensive land use can increase the contaminant load of runoff and drainage water (McDowell *et al.*, 2014). Furthermore, drainage of wetland soils has been shown to reduce water and contaminant storage capacity in the landscape and can increase the potential for downstream flooding. The abstraction of surface or groundwater for irrigation disrupts the natural water cycle and may stress downstream ecosystems and communities. Irrigation of agricultural lands accounts for about 70 percent of ground and surface water withdrawals, and in some regions competition for water resources is forcing irrigators to tap unsustainable sources. Irrigation with wastewater may conserve fresh water resources but brings the risk of water-borne contaminants in soil and crops (Sato *et al.*, 2013) and the accumulation of salts in some environments.

Table 2.2 | Examples of global trends in soil management (Steinfeld et al., 2006; Setälä et al., 2014) and their effects on the ecosystem services mediated by water.

| Management (global trend) | Provisioning | Regulating | Cultural |
|---|---|--|--|
| Land use change (agricultural to urban) | Decreased biomass, decreased availability of water for agricultural use | Increased impervious surface, decreased infiltration, storage, soil-mediated water regulation | Decreased natural environment |
| Land use change (increase in change of arable to intensive grassland) | Land use change (increase in change of arable to intensive grassland) | Increased C sequestration, greater requirement for water, stress on ecosystem health of downstream waterways | |
| Irrigation (increase) | Increased biomass over dryland agriculture, decreased availability of water for urban use | Increased C sequestration, but decreased filtration potential | Infrastructure alters landscape |
| Drainage (increase in marginal land) | Decreased soil saturation, increased biomass, reduction in wetlands | Decreased C sequestration, denitrification and flood attenuation | Decreased recreational potential (e.g. ecotourism) |

The soil management practices to maintain the ecosystem services of food and water security and flow regulation within the soil and water cycle are reasonably well established. However, their application is not universal and poor management leads to a loss of function. Under climate change scenarios of increased climatic variability with more extremes of precipitation, soil functions will be stressed and better soil management will be required (Walthall *et al.*, 2012).

2.4 | Soil as a habitat for organisms and a genetic pool

Soils represent a physically and chemically complex and heterogeneous habitat supporting a high diversity of microbial and faunal taxa. For example, 10 g of soil contains about 10¹⁰ bacterial cells of more than 10⁶ species (Gans *et al.*, 2005), and an estimated 360 000 species of animals are dwellers in soil (Decaëns *et al.*, 2006). These complex communities of organisms play critical roles in sustaining soil and wider ecosystem functioning, thus conferring a multitude of benefits to global cycles and human sustainability. Specifically, soil biodiversity is critical to food and fibre production. It is also an important regulator of other vital soil services including nutrient cycling, moderation of greenhouse gas emissions, and water purification (Wall *et al.*, 2012). It is also recognized that the stocks of soil biodiversity represent an important biological and genetic resource for biotechnological exploitation (Brevik and Sauer, 2015). Previous methodological challenges in characterizing soil biodiversity are now being overcome through the use of molecular technologies. As a result significant progress is currently being made in opening the 'black box' of soil biodiversity (Allison and Martiny, 2008), particularly in assessing the normal operating ranges of soil biodiversity under different soil, climatic and land use scenarios. Addressing these knowledge gaps is of fundamental importance, both as an entry point to understanding wider soil processes and as a way to gauge the likely consequences of land use or climatic change on both biodiversity and soil ecosystem services.



The development of molecular technologies has aided morphological characterisations and allowed quantification of stocks and changes in soil biodiversity. This has led to a surge in studies characterizing soil biodiversity at different scales – from large landscape-scale surveys to locally focused studies. The large-scale surveys yield the broader picture, and conclusions are emerging identifying the importance of soil parameters in shaping the biodiversity of soil communities (Fierer and Jackson, 2006). In essence, the same geological, climatic and biotic parameters that ultimately dictate pedogenesis are also involved in shaping the communities of soil biota and thus in regulating the spatial structure of soil communities observed over large areas (Griffiths *et al.*, 2011). Locally focused experimentation then typically reveals more specific changes in broad taxonomic features with respect, for example, to local changes in land use or climate. Many studies have focused on assessing one component of soil diversity, but even greater advances utilizing next-generation high throughput sequencing now allow the analysis of ‘whole soil foodwebs’. This permits a thorough interrogation of trophic and co-occurrence interaction networks. The challenge is to consolidate both approaches at different scales to understand the differing susceptibility of global soil biomes to change.

Alongside these new developments in assessing biodiversity, it is essential to link the biodiversity characteristics measured to specific soil functions. This helps understanding the pivotal roles of soil organisms in mediating soil services. The development of stable isotope tracer methodologies (e.g. Radajewski *et al.*, 2000) to link substrate utilization to the identified active members *in situ* serves to clarify the physiological activity of these soil organisms. Additionally, improved sequencing techniques are now becoming an increasingly cost-effective for assessing the biodiversity of functional genes in soils for both eukaryotes and prokaryotes (Fierer *et al.*, 2013). This potentially allows a more trait-based approach to understanding soil biodiversity, akin to recent approaches applied to larger and more readily functionally understood organisms above-ground. It is becoming increasingly apparent that often, as is typical in natural ecosystems, functionality and biodiversity co-vary with other environmental parameters. Further manipulative experimentation is required to determine the fundamental roles of soil biodiversity versus other co-varying factors in driving soil functionality.

Clearly, we are learning more and more about how global change affects soil biodiversity and functioning. Global-scale syntheses on soil biodiversity are still lacking, but projects such as the Global Soil Biodiversity Atlas (European Commission, 2015) are combining information from across the globe and making it publicly available. However, much remains to be done. More than 20 years ago, many of these issues were raised (for example, in Furusaka, 1993), and to date many of the factors involved have yet to be unravelled. A key barrier to achieving syntheses is the lack of concerted soil surveys that address multiple functions using standardized methodologies.

New technologies for soil biodiversity assessment generate large sequence datasets that are typically archived in publicly accessible databases. However, morphological datasets remain largely unpublished. The best approach to addressing the gaps would be to adopt agreed standard operating procedures for soil function measurements (e.g. as developed in the recent EU-funded EcoFINDERS project) and to ensure that results are widely accessible.

Ultimately the new methods are revealing the high sensitivity of changes in soil biological and genetic resources to threats such as poor management. We now need to recognize the distinct types of organisms found in different soils globally, and to understand their functional roles in order to predict vulnerability of these resources to future change.

References

- Allison, S.D. & Martiny, J.B.H.** 2008. Resistance, resilience, and redundancy in microbial communities. *Proc. Natl. Acad. Sci. US*, 105: 11512-11519.
- Balesdant, J. & Balabane M.** 1996. Major contribution of roots to soil carbon storage inferred from maize cultivated soils. *Soil Biol. Biochem.*, 28:1261-1263.
- Batjes, N.H.** 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, 47: 151-163.
- Bennett, E., Carpenter, S.R. & Caraco, N.F.** 2001. Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective. *BioScience*, 51: 227-234.
- Borken, W. & Matzner, E.** 2008. Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. *Global Change Biology*, 15: 808-824.
- Brevik, E.C. & Sauer, T.J.** 2015. The Past, Present, and Future of Soils and Human Health Studies. *Soil*, 1: 35-46.
- Chabbi, A., Kögel-Knabner, I. & Rumpel, C.** 2009. Stabilised carbon in subsoil horizons is located in spatially distinct parts of the soil profile. *Soil Biology and Biochemistry*, 41: 256-271.
- Conant, R.T., Ryan, M.G., Agren, G.I., Birge, J.H.E., Davidson, E.A., Eliasson, P.E., Evans, S.E., Frey, S.D., Giardina, C.P., Hopkins, F.M., Hyvonen, R., Kirschbaum, M., Lavalley, J.M., Leifeld, J., Parton, W.J., Steinweg, J.M., Wallenstein, M.D., Wetterstedt, J.A.M. & Bradford, M.A.** 2011. Temperature and soil organic matter decomposition rates – synthesis of current knowledge and a way forward. *Global Change Biology*, 17: 3392-3404.
- Cotrufo, M.F., Wallenstein, M.D., Boot, C., Deneff, K. & Paul, E.** 2013. The microbial efficiency-matrix stabilisation (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable organic matter? *Global Change Biology*, 19: 988-995.
- Decaëns, T., Jiménez, J.J., Gioia, C., Measey, G.J. & Lavelle, P.** 2006. The values of soil animals for conservation biology. *Eur. J. Soil. Biol.*, 60: 807–819.
- Dominati, E., Patterson, M. & Mackay, A.** 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecological Economics*, 69: 1858-1868.
- Dungait, J.A.J., Hopkins, D.W., Gregory, A.S. & Whitmore, A.P.** 2012. Soil organic matter turnover is governed by accessibility not recalcitrance. *Global Change Biology*, 18: 1781-1796.
- Dymond, J.** 2014. Ecosystem services in New Zealand. New Zealand, Lincoln, Manaaki Whenua Press. 540 pp.
- Elliott, E.T., Paustian, K. & Frey, S.D.** 1996. Modeling the measurable or measuring the modelable: A hierarchical approach to isolating meaningful soil organic matter fractionations. In D.S. Powlson, P. Smith & J.U. Smith, eds. *Evaluation of soil organic matter models using existing, long-term datasets*. pp.161-179. NATO ASI Series, Global Environmental Change. Vol. 38. Berlin, Springer Verlag.
- European Commission.** 2015. *Global Soil Biodiversity Atlas*. Luxembourg, Publications Office of the European Union. 176 pp.
- FAO.** 2015. Food and Agriculture Organization of the United Nations - Statistic Division (FAOSTAT) (also available at <http://faostat3.fao.org/home/E>)
- Fierer, N. & Jackson, R.B.** 2006. The diversity and biogeography of soil bacterial communities. *Proc. Natl. Acad. Sci. US*, 103: 626-631.



- Fierer, N., Ladau, J., Clemente, J.C., Leff, J., Owens, S.M., Pollard, K.S., Knight, R., Gilbert, J.A. & McCulley, R.L. 2013. Reconstructing the microbial diversity and function of pre-agricultural tallgrass prairie soils in the United States. *Science*, 342: 621-624.
- Fontaine, S., Mariotti, A. & Abbadie, L. 2003. The priming effect of organic matter: a question of microbial competition. *Soil Biology & Biochemistry*, 35: 837-843.
- Furusaka, C. 1993. Global environment and microorganisms. *Bull. Jpn. Soc. Microb. Ecol.*, 8(2): 127-131.
- Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P. & Sutton, M.A. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. *Science*, 320: 889-892.
- Gans, J., Wolinsky, M. & Dunbar, J. 2005. Computational improvements reveal great bacterial diversity and high metal toxicity in soil. *Science*, 309: 1387-1390.
- Gårdenäs, A.I., Ågren, G.I., Bird, J.A., Clarholm, M., Hallin, S., Ineson, P., Kätterer, T., Knicker, H., Nilsson, S.I., Näsholm, T., Ogle, S., Paustian, K., Persson, T. & Stendahl, J. 2011. Knowledge gaps in soil carbon and nitrogen interactions - From molecular to global scale. *Soil Biology and Biogeochemistry*, 43: 702-717.
- Global Change Biology. 2013. The microbial efficiency-matrix stabilisation (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable organic matter? *Volume 19*: 988-995.
- Griffiths, R.I., Thomson, B.C., James, P., Bell, T., Bailey, M. & Whiteley, A.S. 2011. The bacterial biogeography of British soils. *Environmental Microbiology*, 13(6): 1642-1654.
- Janzen, H.H. 2006. The soil carbon dilemma: Shall we hoard it or use it? *Soil Biology and Biochemistry*, 38(3), 419-424.
- Jenkinson, D.S., 1971. Studies on the decomposition of ¹⁴C-labelled organic matter in soil. *Soil science*, 111: 64-70.
- Kalbitz, K. & Kaiser, K. 2012. Cycling downwards – dissolved organic matter in soils. *Soil Biology and Biochemistry*, 52: 29-32.
- Kibblewhite, M.G., Ritz, K. & Swift M.J. 2008. Soil health in agricultural systems. *Phil. Trans. Royal Soc.*, 363: 685-701.
- Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. *Biogeochemistry*, 85: 91-118.
- Krull, E.S., Baldock J.A. & Skjemstad J.O. 2003. Importance of mechanisms and processes of the stabilisation of soil organic matter for modelling carbon turnover. *Funct. Plant Biol.*, 30: 207-222.
- Kuzyakov, Y. 2002. Review: factors affecting rhizosphere priming effects. *Journal of Plant Nutrition & Soil Science*, 165: 382-396.
- Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304: 1623-1627.
- Lehmann, J., Kinyangi, J. & Solomon, D. 2007. Organic matter stabilization in soil microaggregates: implications from spatial heterogeneity of organic carbon contents and carbon forms. *Biogeochemistry*, 85: 45-57.
- Lehmann, J., Solomon, D., Kinyangi, J., Dathe, L., Wirrick, S. & Jacobsen, C. 2008 Spatial complexity of soil organic matter forms at nanometre scales. *Nature Geoscience*, 1: 238-242.
- Lehmann, L., Abiven, S., Kleber, M. Pan, G., Singh, B.P., Sohi, S. & Zimmerman, A. 2015. Persistence of biochar in soil. In J. Lehmann & S. Joseph, eds. *Biochar for Environmental Management – Science, technology and implementation.*, 2nd edition. pp. 235-283. Routledge. 944 pp.



- Lugato, E., Berti, A. & Giardini, L.** 2006. Soil organic carbon (SOC) dynamics with and without residue incorporation in relation to different nitrogen fertilisation rates. *Geoderma*, 135: 315-321.
- MacDonald, G. K., Bennett, E. M., Potter, P. A. & Ramankutty, N.** 2011. Agronomic phosphorus imbalances across the world's croplands. *Proceedings of the National Academy of Sciences of the United States of America*, 108(7): 3086-3091.
- McDowell, R.W., Cox, N., Daughney, C.J., Wheeler, D. & Moreau, M.** 2014. A national assessment of the potential linkage between soil, and surface and groundwater concentrations of phosphorus. *Journal of the American Water Resources Association*. In press.
- McGill, W.B.** 1996. Review and classification of ten soil organic matter (SOM) models In D.S. Powlson, P. Smith and J.U. Smith, eds. *Evaluation of soil organic matter models using existing, long-term datasets*. pp. 111-132. NATO ASI Series, Global Environmental Change, Vol. 38. Berlin Springer Verlag.
- Miltner, A., Bombach, P., Schmidt-Brücken, B. & Kästner, M.** 2012. Som genesis: Microbial biomass as a significant source. *Biogeochemistry*, 111: 41-55.
- Mueller, C.W., Weber, P.K., Kilburn, M.R., Hoeschen, C., Kleber, M. & Pett-Ridge, J.** 2013. Advances in the analysis of biogeochemical interfaces: NanoSIMS to investigate soil microenvironments. *Advances in Agronomy*, 121: 2-39.
- Nordt, L.C., Wilding, L.P. & Drees, L.R.** 2000. Pedogenic carbonate transformations in leaching soil systems: Implications for the global C cycle. In R. Lal, J.M. Kimble, H. Eswaran, & B.A. Stewart, eds. *Global Climate Change and Pedogenic Carbonates*. pp. 43-64. USA, FL, Boca Raton, Lewis Publishers.
- Parton, W.J., Schimel, D.S., Cole, C.V. & Ojima, D.S.** 1987. Analyses of factors controlling soil organic matter levels in great plain soils. *Soil Science Society of America Journal*, 51: 1173-1179.
- Paul, E.A.** 2014. *Soil microbiology, ecology and biochemistry*. Academic press. 598 p.
- Paul, E.A., Follett, R.F., Leavitt, S.W., Halvorson, A., Peterson, G.A. & Lyon, D.J.** 1997. Radiocarbon dating for determination of soil organic matter pool sizes and fluxes. *Soil Sci. Soc. Amer. J.* 61: 1058-1067.
- Paustian, K.** 1994. Modelling soil biology and biogeochemical processes for sustainable agriculture research. In C. Pankhurst, B.M. Doube, V.V.S.R. Gupta & P.R. Grace, eds. *Management of Soil Biota in Sustainable Farming Systems*. pp. 182-196. Melbourne, CSIRO Publ. 262 pp.
- Paustian, K., Cole, C.V., Sauerbeck, D. & Sampson, N.** 1998. CO₂ mitigation by agriculture: An overview. *Climatic Change*, 40: 135-162.
- Radajewski, S., Ineson, P., Parekh N.R. & Murrell J.C.** 2000. Stable-isotope probing as a tool in microbial ecology. *Nature*, 403: 646-649.
- Rasse, D.P., Rumpel, C. & Dignac, M.F.** 2005 : Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant and Soil*, 269: 341-356.
- Robinson, D.A., Fraser, I., Dominati, E.J., Davíðsdóttir, B., Jónsson, J.O.G., Jones, L. Jones, S.B., Tuller, M., Lebron, I., Bristow, K.L., Souza, D.M., Banwart, S. & Clothier, B.E.** 2014. On the value of soil resources in the context of natural capital and ecosystem service delivery. *Soil Sci. Soc. Am. J.* In press.
- Robinson, D.A., Hockley, N., Cooper, D.M., Emmett, B.A., Keith, A.M., Lebron, I., Reynolds, B., Tipping, E., Tye, A.M., Watts, C.W., Whalley, W.R., Black, H.I.J., Warren, G.P. & Robinson, J.S.** 2013. Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry*, 57: 1023-1033.
- Robinson, D.A., Lebron, I. & Vereecken, H.** 2009. On the Definition of the Natural Capital of Soils: A Framework for Description, Evaluation, and Monitoring. *Soil Sci. Soc. Am. J.* 73: 1904-1911.
- Rumpel, C. & Kögel-Knabner, I.** 2011. Deep soil organic matter – a key but poorly understood component of terrestrial C cycle. *Plant and Soil*, 338: 143-158.



- Sato, T., Qadir, M., Yamamoto, S., Endo, T. & Zahoor, A. 2013. Global, regional, and country level need for data on wastewater generation, treatment, and use. *Agric. Water Manage.* 130: 1-13.
- Scharpenseel, H.W., Becker-Heidmann, P., Neue, H.U. & Tsutsuki, K. 1989. Bomb-carbon, ¹⁴C dating and ¹³C measurements as tracers of organic matter dynamics as well as of morphogenetic and turbation processes. *The Science of the Total Environment*, 81/82: 99-110.
- Schmidt, M.W.I. & Noack, A.G. 2000. Black carbon in soils and sediments: Analysis, distribution, implications, and current challenges. *Global Biogeochemical Cycles*, 14: 777-793.
- Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kogel-Knaber, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S. & Trumbore, S.E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature*, 478: 49-56.
- Setälä, H., Bardgett, R. D., Birkhofer, K., Brady, M., Byrne, L., de Ruiter, P. C., de Vries, F. T., Gardi, C., Hedlund, K., Hemerik, L., Hotes, S., Liiri, M., Mortimer, S. R., Pavao-Zuckerman, M., Pouyat, R., Tsiafouli, M. & van der Putten, W. H. 2014. Urban and agricultural soils: conflicts and trade-offs in the optimization of ecosystem services. *Urban Ecosyst*, 17: 239-253.
- Six, J., Conant, R.T., Paul, E.A., & Paustian, K. 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil*, 241: 155-176.
- Six, J., Elliott, E.T. & Paustian, K. 2000. Soil microaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology and Biochemistry*, 32 (14): 2099-2103.
- Smith, J.U., Smith, P., Monaghan, R. & MacDonald, J. 2002. When is a measured soil organic matter fraction equivalent to a model pool? *European Journal of Soil Science*, 53: 405-416.
- Smith, P., Ashmore, M., Black, H., Burgess, P.J., Evans, C., Quine, T., Thomson, A.M., Hicks, K. & Orr, H. 2013. The role of ecosystems and their management in regulating climate, and soil, water and air quality. *Journal of Applied Ecology*, 50: 812-829.
- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. & Smith, J.U. 2007. Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society, B*, 363: 789-813.
- Smith, P., Smith, J.U., Powlson, D.S., McGill, W.B., Arah, J.R.M., Chertov, O.G., Coleman, K., Franko, U., Frolking, S., Jenkinson, D.S., Jensen, L.S., Kelly, R.H., Klein-Gunnewiek, H., Komarov, A.S., Li, C., Molina, J.A.E., Mueller, T., Parton, W.J., Thornley, J.H.M. & Whitmore, A.P., 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. *Geoderma* 81: 153-225.
- Sollins, P., Homann, P. & Caldwell, B.A. 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74: 65-105.
- Sparling, G.S., Cheshire, M.V. & Mundie, C.M. 1982. Effect of barley plants on the decomposition of ¹⁴C-labelled soil organic matter. *Journal of Soil Science*, 33: 89-100.
- Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. & Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23, GB2023.
- Thévenot, M., Dignac, M.-F. & Rumpel, C. 2010. Fate of lignins in soils: a review. *Soil Biology and Biochemistry*, 42: 1200-1211.
- Todd-Brown, K.E.O, Randerson, J.T., Post, W.M., Hoffman, F.M., Tarnocai, C., Schuur, E.A.G. & Allison, S.D. 2013. Causes of variation in soil carbon simulations from CMIP 5 Earth system models and comparison with observations. *Biogeosciences*, 10: 1717-1736.



- Trumbore, S.** 2009. Radiocarbon and soil carbon dynamics. *Annual Review of Earth Planetary Sciences*, 37: 47-66.
- Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J. & Van Kessel, C.** 2010. Towards an agronomic assessment of N₂O emissions: a case study for arable crops. *Eur J Soil Sci*, 61: 903-913.
- Venterea, R.T., Maharjan, B. & Dolan, M.S.** 2011. Fertilizer Source and Tillage Effects on Yield-Scaled Nitrous Oxide Emissions in a Corn Cropping System. *J. Environ. Qual.*, 40: 1521-1531.
- Vitousek, P. M. & Matson, P. A.** 1993. In R. S. Oremland, ed. *The Biogeochemistry of Global Change: Radiative Trace Gases*. pp. 193-208. New York, Chapman and Hall.
- Vogel, C., Mueller, C.W., Hoschen, C., Buegger, F., Heister, K., Schulz, S., Schloter, M. & Kogel-Knabner, I.** 2014. Submicron structures provide preferential spots for carbon and nitrogen sequestration in soils. *Nature Communications*, 5: 2947. 7 pp.
- von Lützow, M., Kögel-Knabner I., Ekschmitt K., Matzner E., Guggenberger G., Marschner B. & Flessa H.** 2006. Stabilization of organic matter in temperate soils: Mechanisms and their relevance under different soil conditions – a review. *European Journal of Soil Science*, 57: 426-445.
- Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, T.H., Ritz, K., Six, J., Strong, D.R. & van der Putten, W.H.** (eds.). 2012. *Soil Ecology and Ecosystem services*. Oxford University Press, UK. 424 pp.
- Walthall, C.L., Hatfield, J., Backlund, P., Lengnick, L., Marshall, E., Walsh, M., Adkins, S., Aillery, M., Ainsworth, E.A., Ammann, C., Anderson, C.J., Bartomeus, I., Baumgard, L.H., Booker, F., Bradley, B., Blumenthal, D.M., Bunce, J., Burkey, K., Dabney, S.M., Delgado, J.A., Dukes, J., Funk, A., Garrett, K., Glenn, M., Grantz, D.A., Goodrich, D., Hu, S., Izaurrealde, R.C., Jones, R.A.C., Kim, S-H., Leaky, A.D.B., Lewers, K., Mader, T.L., McClung, A., Morgan, J., Muth, D.J., Nearing, M., Oosterhuis, D.M., Ort, D., Parmesan, C., Pettigrew, W.T., Polley, W., Rader, R., Rice, C., Rivington, M., Roskopf, E., Salas, W.A., Sollenberger, L.E., Srygley, R., Stöckle, C., Takle, E.S., Timlin, D., White, J.W., Winfree, R., Wright-Morton, L. & Ziska, L.H.** 2012. *Climate Change and Agriculture in the United States: Effects and Adaptation*. Washington, DC, USDA Technical Bulletin 1935. 186 pp.
- West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K. & Siebert, S.** 2014. Leverage points for improving global food security and the environment. *Science* 345: 325-328
- Whitmore, A.P., Kirk, G.J.D. & Rawlins, B.G.** 2014. Technologies for increasing carbon storage in soil to mitigate climate change. *Soil use and Management*, 31(S 1): 62-71.
- Wilkinson, M.T., Richards, P.J. & Humphreys, G.S.** 2009. Breaking ground: Pedological, geological and ecological implications of soil bioturbation. *Earth Science Reviews*, 97: 257-272.
- Withers, P.J., van Dijk, K.C., Neset, T.S., Sesme, T., Oenema, O., Rubæk, G.H., Schoumans, O.F., Smit, B., Pellerin, S.** 2015. Stewardship to tackle global phosphorus inefficiency: the case of Europe. *Ambio*, 44: 193-206.



