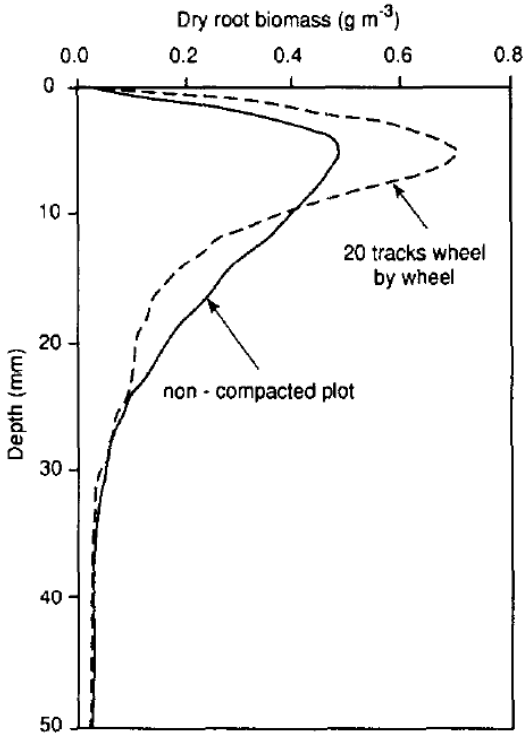

Figure 4-7: Distribution of maize dry root biomass in the soil profile in spring compaction experiments (Whalley, Dumitru et al. 1995)

Table 4-1: Effects of laboratory compaction of silt loam grassland soil on Acari (mean of 20 samples) at a soil water content of 22.4 g per 100 g (Whalley, Dumitru et al. 1995)

Family	Control				Pressure 2.80 kPa				Pressure 5.60 kPa			
	Total alive	Total dead	% dead	SE	Total alive	Total dead	% dead	SE	Total alive	Total dead	% dead	SE
Parasitidae	48	3	5.9	3.3	28	8	22.2 NS	6.9	15	16	51.6**	9.0
Rhodacaridae	88	42	32.3	4.1	140	82	36.9 NS	3.2	96	54	36.0 NS	3.9
Eviphididae	172	3	1.7	1.0	78	5	6.0 NS	2.6	105	13	11.0	2.9
Uropodidae	1	8	88.9	10.5	5	14	73.7 NS	10.1	3	13	81.3 NS	9.8
Scheloribatidae	10	24	70.6	7.8	13	2	40.0 NS	12.7	10	78	88.6*	3.4
Pelopsidae	103	144	58.3	3.1	65	188	74.3**	2.8	70	234	77.0**	2.4

NS, = not significant; *P = 0.05; **P = 0.01.

An important portion of European soils have high (28%) to very high (9%) risks of compaction (Jones, Spoor et al. 2003). Central and Eastern European soils are particularly affected by compaction. Soil compaction areas are continuously increasing in Europe due to the use of heavier wheel pressure in agriculture. To date, pressures of up to 13 t (which is more than what is allowed on some roads) are currently used, and their impact on wet soils in particular is tremendous, causing compaction of up to almost one metre deep. European soils and soil organisms are thus increasingly threatened by soil compaction.

4. 2. 5. SEALING

Soil sealing is the process through which soils are covered by an impermeable layer, which impedes exchanges between aboveground and belowground worlds. Natural soil surface sealing occurs when fine particles form a surface crust that may impede water

infiltration in the deeper soil layers. This kind of surface crusts is, for example, an important structure in semi desert environments that generally host active although fragile microbial communities. However, today, most sealing is of anthropogenic origin, and linked to urbanisation. In this case, soils are covered by impermeable layers of asphalt, concrete or other sealing materials. Such artificially sealed soils are not functional anymore. An additional negative consequence of sealing is that natural processes, such as water infiltration, become concentrated on a much smaller soil surface and that water will need to run-off directly to canals and rivers. As a result, local soil sealing also has negative effects on other places, where disproportional water discharges have to be processed.

Sealing can lead to a slow death of most soil organisms. Soil biota can initially survive on the moisture and organic matter that was present in the soil before sealing, until these resources are exhausted. Then, soil bacteria enter an inactive state and larger soil fauna may either disperse or, when sealing covers vast areas, die off.

Human-driven sealing often concerns soils that are rich in nutrients and valuable for agricultural production, because rich soils are close to urban areas and, therefore, more readily subject to the pressure of expanding urban areas. Moreover, sealing contributes to the elimination of 'buffer zones'. These are the semi-natural zones between urban and agricultural areas which connect natural ecosystems. Thus, soil sealing and degradation by urbanisation may also affect surrounding natural areas through habitat fragmentation.

The degree of soil sealing is variable throughout Europe and it is, of course, particularly high in extremely urban areas such as the Ruhrgebiet (Germany), or in the Mediterranean region where the pressure of tourism has led to a vast expansion of urbanised areas along the coast. In the future, soil sealing is expected to continue at an increasing rate all over Europe. Indeed, even in already highly urbanised areas, sealing is still progressing: between 1990 and 2000 in France, natural soils sealing due to urbanisation increased by 4.8% (IFEN 2005) and the sealing rate is still increasing. However, the most noticeable increases in sealing may occur in Central and Eastern Europe, due to the economic development, and in areas which up to now had low sealing levels, such as Finland or Ireland.

New initiatives in spatial and urban development planning could already limit the levels of soil sealing through keeping space for semi-natural areas within urban zones and considering the possibility to have green roofs to favour the reconstitution of natural environments.

4.3. LAND-USE MANAGEMENT

The densely populated European landscapes are dynamic structures that have experienced continuous redistributions and modifications for centuries. Land use is changing according to the ever-evolving needs of human populations for critical ecosystem services such as food, fresh water, and housing. Thus forests are grown and cut for construction materials, paper or fuel; crop fields are cultivated with a variable intensity, depending on population growth and needs, and may be fallowed, or completely abandoned, for economic or other reasons. For example, in the nineteen-nineties, when the world market prices dropped, much arable land was fallowed. Another example is the abandonment of agricultural land currently taking place for biodiversity conservation and restoration practices e.g. in case of natural disasters or with the farmer changing for a more productive parcel or leaving for a better paid

activity. Grasslands are also being turned into crop fields, and several of these fields are gradually consumed by growing urbanisation. In addition to the continuous changes in the shape and composition of the landscape mosaic (the amount of specific parcels building the landscape), changes in intensification of land use are also common. Also, the demand for producing biofuel crops involves the risk that intensification of land use will increase, leading to a decrease in soil biodiversity and corresponding ecosystem functioning.

In the scientific literature, land-use changes are the first most commonly cited cause of general biodiversity extinction, as they are immediate and often take place at large scales, thereby not allowing species to adapt, or to move away to other areas. As land use is highly susceptible to changing policies and as the effects of land use on biodiversity are so strong, land use change is an important policy tool for managing and conserving (soil) biodiversity and the corresponding services.

The changes in land use may affect soil communities mainly by changing the quality and quantity of inputs available to them and by modifying soil micro-habitats (Bardgett and Cook 1998). For example, forest clearing eliminates the leaf and woody surface litter that is home to a wide fungal and invertebrate diversity. In the same manner, conversion of grasslands to agricultural fields or tree nurseries involves tillage that destroys the habitats of large invertebrates that for example act as soil engineers (producing burrows or galleries). Soil tillage also destroys mycorrihzal networks and other fungal hyphae and it brings the soil community in a disturbed state, thereby eliminating many soil organisms that have a relatively long life span (Helgason, Daniell et al. 1998).

As human population keeps growing, the demand for soil services and the ensuing need for changes in the type and intensity of land use are expected to continue at an ever increasing rate. Eventually, and if no action is taken, this could alter the abundance and diversity of soil fauna and also of soil microbes, especially soil fungi, which reduces the capacity of soils to provide the expected goods and services.

4. 3. 1. SOIL BIODIVERSITY FOR DIFFERENT LAND USES

Despite the fact that a majority of Europe's population lives in urban areas, 91% of the EU territory is composed of rural areas, forests and (semi-)natural areas. The rural areas consist mainly of a mosaic of croplands, grasslands, and orchards. In the coming 10 years, between 2000 and 2020, there is an expected increase in the total forest area by 5%, and a matching decrease in cropland (EEA 2007). Studies have shown that the responses of soil communities to land-use change are taking much more time than the initial changes in vegetation (Korthals, Smilauer et al. 2001; Hedlund, Regina et al. 2003). It can take years, if not decades, before the soil community has become adapted to the changed environmental conditions. This is partly because the growth and development of populations is so slow and, in part, because it takes much time for soil organic matter to build up and, in part, because some soil organisms need to disperse to the land that is changed in use. Thus, land use can be changed from one day to another, but it may take years to decades for soil biodiversity to follow and to establish new equilibria (van der Wal, van Veen et al. 2006). It will also take years, or more likely decades, for the ecosystem services to be changed accordingly.

→ **FORESTS**

Natural forests are the most common type of natural area in Europe, covering over 35% of the EU territory. Forest soils are characterised by extensive root systems and leaf litter layers that provide both habitat and food to soil fauna. Forests also offer a protective microclimate, mostly characterised by reduced temperature extremes, decreased light availability, reduced wind speed and increased moisture. Two main types of forest soils can be distinguished: coniferous forest soils, which are more acidic (mor), and deciduous forest soils which are non acidic (mull).

Forest soils are usually quite buffered and sometimes very old environments, which tend to host highly diverse soil communities (Hagvar 1998). Deciduous forest soils have a good aeration and allow ion exchanges, favouring a high soil biodiversity. They exhibit high C:N ratios (10:15) (see Box 18) and are characterised by fungal-dominated food-webs (the ratio of fungal to bacterial biomass ranges from 5:1 to 10:1 in deciduous forests), fungi-eating protists and nematodes, and high densities of microarthropods and anecic earthworms (Bardgett 2005). However, earthworm forest communities are not very diverse: only three out of 27 species commonly found in Europe are clearly associated with the forest environment, and other species may only be found at low densities (Watt 2004).

Coniferous forest soils in contrast have lower biological activity, since the acidic conditions restrict microbial activity, and hence the functional groups above. Compared to deciduous forest soils, these soils are more heavily fungal-dominated, with fungal to bacteria biomass ratios reaching 100:1 or 1000:1 (Ingham, Coleman et al. 1989). Ecosystem-engineer communities are dominated by epigeic earthworms and enchytraeids (Lavelle et al. 1997).

Box 18: The C:N ratio and the fungal: bacterial ratio

The C:N ratio is the amount of carbon relative to the amount of nitrogen present in SOM. There is always more carbon than nitrogen in organic matter, and a low ratio (close to one) means that the amount of carbon is close to that of nitrogen, whereas a high ratio means that there is a considerably higher mass of carbon for each gramme of nitrogen in organic matter. The C:N ratio of leaves is typically much lower than that of wood, by at least an order of magnitude (Snowdon 2005).

The C:N ratio determines what happens when organic matter is incorporated into soils. Indeed, the C:N ratio is a measure of the quality of SOM, which influences its rate of decomposition. Decomposition may occur either through fungal- or through bacterial-based pathways. Fungi have a higher C:N ratio than most bacteria (de Vries et al. 2006). Accordingly, fungi tend to prefer food rich in carbon, such as cellulose, whereas bacteria tend to prefer food rich in nitrogen, such as plant leaves. Moreover, fungi usually have slower turnover rates than bacteria. **As a result, high C:N ratios may lead to fungal-dominated decomposition and lower nitrogen mineralisation than bacterial decomposition.** At any stage of decomposition, nutrient deficiencies may limit microbial activity and thereby block the release of nutrients and other elements to plants and other soil organisms. This occurs when the C:N ratio of the decomposing resource is high compared to that of the chemical engineers. Indeed, in this case nitrogen is limiting, and is used by the chemical engineers for their own growth, and not available to plants (Lavelle and Spain 2001). This explains why agricultural systems can require nitrogen fertiliser, as well as enrichment of organic matter with a moderate C:N ratio.

→ **GRASSLANDS**

Grasslands are ground covered by grass-dominated vegetation, and little or no tree cover. Various types of grasslands exist in Europe, ranging from almost desert-like in south-east Spain, through steppe and mesic types to humid grasslands and meadows which dominate in the North and North-West. Most European grasslands can be defined as 'semi-natural grasslands', since they are covered with sown and grass leys aimed at producing forage for livestock. They are modified and maintained through grazing and/or farmers' cutting regimes. In 2005, grasslands represented 13% of the EU territory, most of which were permanent grasslands (87% of European grasslands). However, these grasslands are not evenly distributed across Europe, with 60% of permanent grasslands found in only four countries (U.K., France, Spain and Germany). The intensity and type of agricultural management practices vary according to the land use. Most European grassland systems, particularly in Western Europe, are moderately to heavily managed (Bardgett and Cook 1998), while most cropped systems are intensively managed. Grassland management varies in particular, with respect to the nature and quantity of fertiliser inputs (Bardgett and Chan 1999).

Grassland soils present the richest soil biodiversity and it is worthwhile to consider including longer-lasting grasslands in an arable crop rotation in order to restore carbon levels and soil biodiversity, as well as disease-suppressing services (Garbeva 2004). Grasslands are characterised by extensive root systems and generally limited amounts of leaf litter which favour a high diversity and biomass of nematodes, microarthropods and earthworms in particular. Given the low level of leaf litter, grasslands are characterised by fungal-dominated food webs with microbial biomass similar to that of forests, but missing wood-decaying fungi (Tugel 2000). The communities of biological regulators are very active, and dominated by fungal-feeding microarthropods, protists and nematodes. Grasslands generally host the most diverse and abundant earthworm communities in Europe (Watt 2004), with some communities dominated by endogeic species and others by anecic species (Lavelle and Spain 2001). In temperate grasslands, most of the biomass can be explained by one family of earthworms, *Lumbricidae*, which can represent 70-80% of the total soil biomass in low tillage systems (Bardgett and Cook 1998; Ruiz Camacho 2004). When used as pastures, grasslands face soil compaction and pressure on crop growth.

→ **CROPLANDS**

Croplands are managed at moderate to strong intensity, with irrigation, deep tillage and ploughing, and systematic use of chemical inputs, such as fertilisers and pesticides aimed at enriching the soil, controlling parasites and diseases, and eliminating crop competitors. Croplands represent 22% of the EU territory[33], over 95% being conventional agricultural land. The key feature of this type of agriculture is the specialisation of the production process, often resulting in monocultures and choice of fast-growth and high-yield plants that allocate most of their biomass to the harvested parts. In other words, conventional agriculture may push ecosystems in the direction of performing one single service, food provisioning, at the expense of the other, related services, such as the maintenance of soil structure, water quality and climate control. Such intensive agricultural practices contribute to the homogenisation of the landscape and are unfavourable to most soil organisms, leading to large scale soil biodiversity changes (Freckman, Duncan et al. 1979; Ingham and Detling 1984; Bardgett, Frankland et al. 1993; Bernier and Ponge 1994). It is not necessarily so that soil biodiversity of

[33] Eurostat, 2008, Agricultural statistics - Main results – 2006-2007.Website: epp.eurostat.ec.europa.eu/cache/ITY_OFFPUB/KS-ED-08-001/EN/KS-ED-08-001-EN.PDF Downloaded the 10th September 2009

croplands is so much less than of for example grasslands, but some essential species groups with special functions can drop out. For example, cropland soil contains relatively few arbuscular mycorrhizal fungi and also few earthworms. The soil community is adapted to regular disturbance and the food chains are mainly based on bacteria-based pathways (De Ruiter et al. 1995).

Especially conventionally cropped soils result in stressed and depleted soil food webs. When intensively cropped, arable soils are characterised by low organic matter inputs (leaf litter and stubbles are largely removed), and thus low soil fungal/bacterial ratios, and depleted bacteria-dominated chemical engineers communities. Consequently, biological regulator communities are themselves reduced and dominated by opportunistic bacterial-feeding fauna. Finally, strong mechanical and chemical disturbance cause reduction of earthworm and mycorrhizal fungi communities. Earthworms are only present at moderate densities (10 à 20 individuals per m²) and mostly composed of endogeic species (Patrick Lavelle, personal communication), as epigeics are missing due to a lack of litter layer. Together, these conditions are indicative of low resilience and low sustainability (de Vries, Hoffland et al. 2006).

Crop systems generally vary with latitude and growth seasons are relatively short in the Mediterranean (where summer drought prevents crop growth, unless irrigation is used) and northern Europe, where the summers are relatively short. In the temperate zone, crop systems are often based on rotations, for example of cereals, sugar beet, cereals and potato. Normally, cereals (like wheat) enable the soil to recover from high-intensity crops (like sugar beet and potato). When prices on the world market drop and particular crops become economically unprofitable, such as happened with the prices for cereals in the past 20 years and no alternative rotational crop is available, the rotations will be narrowed, with fewer crops that enable soils to recover. A major disadvantage is then that levels of specific soil-borne diseases do not drop anymore between the successive cultures of the same crop species. For example, beet cyst nematode, or potato cyst nematode do need three years of non-crop in order to have their populations declined. If the rotations become shorter, the nematode populations do not decline and biocides need to be used for nematode control, which is also very negative for other soil invertebrates and for ground and surface water quality (Scholte 1985).

Continuous cropping of soils is often applied in cereal fields. During the first decade, continuous cropping results in a decline of harvest, because of emerging soil-borne diseases. However, after a decade of continuous cropping, disease suppressiveness may develop, such as against root pathogens of wheat (Weller 1995). Some crops, like maize, can withstand enormous amounts of nutrients, which makes that these crops have been used in the past for rather excessive fertilization by manure, leading to phosphate-saturated soils in north-western Europe (Koopmans 2004).

Although each type of land use is characterised by its specific soil biodiversity, the intensity of management practices may also vary within a certain land use and severely impact soil biota. Typically, soil biodiversity peaks at intermediate management intensities (see Box 3). Thus, species diversity and abundance increase from low to intermediate disturbance (e.g. extensive grasslands to organic agriculture), peak at moderate agricultural disturbance (e.g. organic agriculture) and then decrease with strong agricultural disturbances (e.g. conventional agriculture)(Freckman, Duncan et al. 1979; Ingham and Detling 1984; Bardgett, Frankland et al. 1993; Bernier and Ponge 1994). Therefore, reducing management intensity of an intensive cropping practice

with some degree of organic inputs, continuous plant cover and limited tillage, typically leads to an environment in which soil biodiversity is enhanced.

A number of major long-term studies have investigated the impacts of tillage on soil biodiversity under intensive, reduced and no-tillage conditions. Tillage consists of preparing the soil for cultivation by ploughing, ripping, or turning it with a chisel plough or heavier duty field cultivators. Tillage can vary in intensity, with deep tillage leaving less than 15% of the crop residue cover on the soil, whereas softer tillage typically leaves between 15% and 30% of crop residues behind. The most obvious effect of soil tillage is the negative correlation between the size of organisms and their biomass, since tillage mechanically disrupts the soil structure, particularly at the scales of action of biological regulators and ecosystem engineers). Earthworms and other large surface soil-dwelling organisms are often damaged or killed by intensive soil tillage, which promotes soil compaction and reduced water infiltration (Citeau 2008). This effect depends of course on the intensity and depth of tillage, with light tillage leaving a higher diversity of large earthworms than conventional tillage for instance (Ernst and Emmerling 2009). Fungi can also be physically disrupted in tilled soil, as their mycelia are broken up.

Soil tillage is clearly devastating for some key groups of soil organisms, especially for (arbuscular mycorrhizal) fungi and earthworms. The solution would be no tillage, however, that also involves a number of complications, such as reduced potential for weed control, which is especially a bottle neck in organic farming (Berner 2008). Moreover, the effects of soil tillage on earthworms differ between earthworm species (Ernst and Emmerling 2009), which is probably due to altered distribution of soil organic matter across the soil profile (Ernst and Emmerling 2009) and altered soil habitat quality (Metzke, Potthoff et al. 2007). Soil tillage definitely has considerable potential, but it clearly needs more long-term and comparative experimentation before it can be practically recommended as a means of conserving soil biodiversity and enhancing ecosystem services (Peigne 2009).

However, one of the problems of non-tillage is that crop left-overs may promote disease transmission from one to another year. Another potential disadvantage is that weeds are less controllable, and in the case of root- or tuber crops, non-tillage is much more difficult. Still, non-tillage would promote that population levels of beneficial soil organisms remain high and that soil water holding capacity may improve. Therefore, this avenue, or other novel techniques reducing the negative impacts of soil tillage on soil biodiversity, might need to be further explored (Waid 1999; Brussaard 2007).

Mechanical tillage also disrupts soil structure, rendering previously protected organic matter available to microbial decomposition, and previously inaccessible prey available to predation (Van Veen and Kuikman 1990). As a consequence, soil tillage results in increased mineralisation and cycling of organic matter (Ogle, Breidt et al. 2002) and intensive soil tillage enhances erosion on slopes, especially when soils are not ploughed parallel to elevation along the elevation lines in order to prevent runoff.

Organic farming practices can be considered to provide a lower level of stress and higher organic inputs for food webs than conventional agriculture. This increases the potential niches for soil fauna, however, it may not necessarily lead to improved soil biodiversity. Organic farming may change the relative abundance among groups of soil organisms and promote only some specific taxa, such as earthworms, more specifically (Birkhofer, Bezemer et al. 2008).

Indeed, in a meta-analysis, no impact of organic management practices was found on soil organisms, although for most functional groups organic management resulted in increased abundances (Bengtsson, Ahnstrom et al. 2005). Overall though, there is a tendency for long-term soil management in organic agriculture to promote better soil structure (aggregates stability and organic matter supply), nutrition (organic matter supply) and foster pest control compared to conventional agriculture (Birkhofer, Wise et al. 2008).

Another major issue in cropland management concerns the use of mineral versus organic manure as a main source for nutrient supply. Organic, sustainable farming practices are increasingly favoured in Europe, although cereal crop yields under organic management are typically 60 to 80% of those under conventional management (Mader, Fliebbach et al. 2002; David 2004). The switch from low to intensive management typically reduces the diversity, although not necessarily the density of soil fauna, as some faunal species, for example bacterial feeding and root-feeding nematodes often become increasingly abundant (Bardgett and Cook 1998; Bloem, Schouten et al. 2003).

Extensive trials at the Frick site in Switzerland showed that organically manured arable land produced slightly less yield, but that other ecosystem characteristics were much more favourable (Mader, Fliebbach et al. 2002). Organic manure promotes soil microbial biomass, but slows down microbial activity. Farmyard manure also promotes the abundance of biological regulators (for example bacterivorous nematodes) and ecosystem engineers (earthworms) in the soil and generalist predators above ground (Birkhofer et al. 2008). In a further examination of the Frick experiment, conventional crop production using mineral fertilizer gave 23 % more straw and wheat production (Birkhofer et al. 2008), but the environmental costs of using biocides and herbicides may reduce this profit when analyzing all economic costs and benefits. Clearly, what is currently lacking are studies that consider the various aspects of organic and conventional agriculture in comparison in a much more integrated way (Bengtsson 2005).

Concerns are being raised about the long-term environmental consequences and sustainability of intensively managed systems. It is now clear that the intensification of agriculture can have negative consequences at local scales (e.g. increased erosion, lower soil fertility), regional scales (e.g. pollution of groundwater) and global scales (e.g. reduced climate regulation). Moreover, cultivation systems have long-term effects on microbial community structures (Buckley and Schmidt 2001), and soil communities in general, such that several decades after abandonment, agricultural fields still show modified or depleted soil activity (van der Wal, van Veen et al. 2006).

→ **URBAN LANDS**

Europe is highly urbanised, with over 75% of its population living in cities and a projected 80% by 2020 (EEA 2006; EEA 2007). Over a quarter of Europe's territory is already urban, but cities are expanding faster than populations, in what is known as 'urban sprawl'. Urban areas consist of highly modified habitats, with over 80% of most central urban areas covered by pavement and buildings. As a result, urban soils are subject to sealing and compaction, and face environmental stresses such as air pollution, heavy metals pollution, and increased temperatures in the urban cores (2–3°C warmer)(McDonnell, Pickett et al. 1993) (Hansen, Knight et al. 2002). The buildings and other paved areas modify the chemical and physical equilibrium of soils, as well as their connectivity to other types of ecosystems (see section 4. 2. 5). However, in the

remaining space, urban systems can involve nature, in the form of tree lanes, lawns, cultivated areas and parks, or even natural remnants such as urban forests, wetlands, lakes and streams. But even these soil patches face intensive management and disturbance, involving intensive use of chemicals and little to no litter.

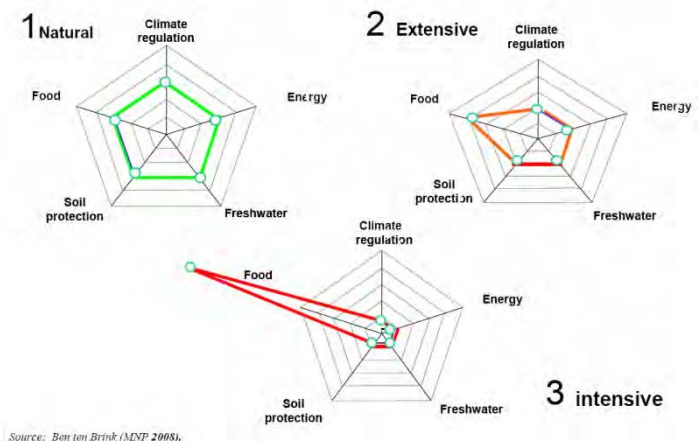

Figure 4-8: Trade-offs between agriculture and other ecosystem services under different management intensities[34]

Sealed urban areas have a severely depleted soil biodiversity, given that soil sealing stops exchanges between soil fauna and all external inputs. The chemical engineers might go in dormancy under sealed soils, when they do not simply die off. Non-sealed urban soils are bacteria-dominated given the high chemical inputs used for pest control. Biological regulators are dominated by microarthropods. Earthworms are mostly absent, and present only in urban parks or forests.

The table below summarise the distribution of functional groups in different land-use types.

[34] Figure from: Ben ten Brink, The Cost of Policy Inaction: The case of not meeting the 2010 biodiversity target, European Commission, DG Environment, 2007

Table 4-2: Distribution of functional groups by land-use types

Soil biodiversity (Dominance/diversity)	Forest	Grassland	Cropland	Urban land
Total	++	++	+	-
Chemical engineers	Fungi dominated	Fungi dominated Fungi: 10-100 m Bacteria: 10^8-10^9 g/soil	Bacteria dominated Bacteria: 10^8-10^9 g/soil	Bacteria dominated
Biological regulators	Fungal-feeding protists and nematodes (100-1000 g/soil) Micro-arthropods (10^6/m²)	Protists and nematodes dominated Protists: 1000/g Nematodes: 10-100/g Microarthropods: 5000-20000/m²	Opportunistic bacterial- feeding fauna Protists: 1000/g Nematodes: 10-20/g Microarthropods: < 100/m²	Negligible
Ecosystem engineers	Earthworm and ant-dominated Anecic earthworms (100/m²)	Earthworm dominated Endogeic/Anecic earthworms	Epigeic and endogeic earthworms (50-300/m²)	Negligible

4. 3. 2. IMPACT OF LAND-USE CHANGE ON SOIL BIODIVERSITY

Each square metre of used land may face some, to many changes. These changes usually involve an initial disturbance phase, before a new ecosystem equilibrium is established. In the course of this process, the quantity and quality of available organic and mineral inputs are modified, leading to a redistribution of soil communities. Consequently, large biodiversity changes may occur locally, as species escape to neighbouring resource-rich plots or go extinct if they cannot escape. Accordingly, native soil communities may experience reductions in their diversity and abundance, while in contrast non-native, sometimes invasive, species may find suitable conditions and resources, and progressively replace native species. In the new ecosystem, biodiversity may be more, equally or less abundant than in the previous one, depending on numerous and complex parameters, e.g. ecosystem characteristics, the intensity of the disturbance and the pool of species present in the landscape.

Some land-use changes may reflect natural disturbances, that can induce shifts from old mature systems to newly created systems, where species often assemble at random and productivity is relatively limited (Bardgett, Bowman et al. 2005). Others may be human induced instead, and involve modifications in plant cover and/or management regime, such as switches from forests to agriculture, with a sudden disappearance of leaf-litter and of the microclimate created by trees, or conversion of grasslands to arable land, with major losses in carbon and soil biodiversity.

→ CHARACTERISTICS OF LAND-USE CHANGES IN EUROPE

Europe has experienced relatively fast and drastic changes in landscapes throughout its history. For instance, in the 20th century alone, European landscapes have changed drastically, as rapid industrialisation has led to escalating urbanisation, while growing populations have been pushing for increased agricultural productivity. These changes

can occur over very short timescales, for instance, the Corine Land-cover database[35] shows significant changes in land use in Europe between 1990 and 2000, where at least 2.8% of the European land was subjected to a change in use.

As a result of fast land-use changes, current communities are often composed of generalist species that have been able to adapt to changes well, whereas more specialised species that were unable to adapt to changes may have become extinct or have been maintained only in few, isolated habitats. For instance, European earthworm communities are surprisingly homogeneous from Finland to Portugal, with 25 species making up a large part of all communities across the different types of land uses, although the structure of the communities varies locally (Watt 2004). Whereas homogenisation processes may not always be so drastic, they are likely to be common for small organisms with low dispersal ability, such as soil organisms. The homogenisation of biological communities at landscape scales reduces the insurance potential and thus the resilience of communities to future changes.

Moreover, the original pool of species from which soil communities originate is not very old. Indeed, historically, the most drastic land-use changes that affected Europe were the ice ages. In those times, the glaciers destroyed the soil, killing most of the earthworms in Northern Europe for instance, such that re-colonisation could only start after the retreat of the glaciers, ca -10000 BC.

Nevertheless, some of the largest changes in (semi-)natural systems are also due to natural succession processes. Ecological successions have been particularly well studied in forests, where natural or human induced disturbances (such as storm, fire, or logging) recurrently cause forest clearings. These clearings are then re-colonised by a suite of species in a series of succession steps. First, pioneering fast-growing species with good colonisation capacity occupy the site. Slowly however, these species are replaced by better competitors, and a reorganisation phase starts during which the composition of communities is highly variable, and depends essentially on the resources available (such as light, nutrients), on the species present inside the patch, as well as on the inputs from outside the patch. In their mature phase, communities are usually dominated by a few species of the locally superior competitors. This phase is usually followed by a senescence phase, where organisation is progressively lost, probably as a result of reduced nutrient availability. In the next cycle, the same community, or a very different one, may establish, depending on equilibrium attained after the initial reorganisation phase (Bernier and Ponge 1994). These four phases have been generalised to all natural as well as economic or institutional processes (Gunderson, Holling et al. 1997). As communities change naturally, so do, depending on abiotic factor, soil nutrients, pH, and organic matter accumulation in litter, and therefore soil biota. For instance, during the early stages of succession, litter is usually essentially composed of leaf tissue, whereas in the later stages it is mostly composed of wood, which takes longer to decompose (by a factor of 10 to 100).

→ **IMPACT OF THE MAIN LAND-USE CHANGES ON SOIL BIODIVERSITY AND RELATED SERVICES**

When forests are converted to grasslands, grasslands to crop lands, or agricultural lands turned into urban areas, a sharp switch from one type of soil community to another occurs. The switch is even greater when forests are converted directly to agricultural lands or urban areas. During the transition phase, a general decline in soil biodiversity is observed (Decaens, Jimenez et al. 2004)(see also table at the end of section 4. 3. 1, Table 4-3 and Table 4-3).

[35] www.terrestrial.eionet.europa.eu/CLC2006/ last retrieval 16/09/2009

Forest → Grassland

This is a switch within a fungi-dominated system, but increased disturbance in grasslands results in lower soil microbial activity in grasslands than in forests. The reduction in the activity of chemical engineers is also reinforced since the reduced activity of anecic earthworms in grasslands hinders their movements (Edwards 2002; Decaens, Jimenez et al. 2006). However, communities of biological regulators and ecosystem engineers tend to be more diverse in grasslands than in forests.

The switch from forests to grasslands thus results in a reduction of nutrient cycling and of the regulation of carbon flux and climate control services. However, while reductions in carbon storage have been frequently observed in microcosm studies, this process is highly variable in field conditions and also may depend on land use and the level of nitrogen deposition (Setälä H, Haimi J et al. 1988; Setala, Haimi et al. 1988; Setälä 1990; Setala, Martikainen et al. 1990; Setala and Huhta 1991; Lavelle and Spain 2001). Severe reductions in organic matter decomposition have only been reported so far in tropical soils where processes are much faster and biological impacts much more important than in temperate soils (Rose and Woods 1980; Chauvel, Grimaldi et al. 1999). Although not explicitly tested, there are some suggestions that fungal-dominated grassland soils retain more nutrients in the microbial biomass under stress (e.g. dry-wet) and that more nutrients are held in the microbial biomass in general, which might be important for nutrient retention (Gordon 2008).

Grassland → Cropland

This is essentially a switch from low to moderate or high management intensity (intensification, see introduction). As such, microbial biomass can be approximately 40% greater in native grasslands and pastures than in cropped fields (Dominy and Haynes 2002), at all depths from 0 to 40 cm, with a higher diversity and abundance of bacteria (Garbeva et al. 2002). The more grasslands are managed, the more their soil communities bear resemblance to those of cropped sites (Steenworth 2002). However, the conversion of grassland into agricultural land can suddenly render available the previously protected organic matter and provoke local bursts of microbial activity and significant losses in soil C (Van Veen and Kuikman 1990). Impoverished microbial communities result in reduced communities of biological regulators in cropped fields, in particular with fewer nematode species (Hodda and Wanless 1994). Cropped fields are usually dominated by bacteria-feeding nematodes, at the expense of plant-feeding ones (Yeates 1999). Frequent disturbance due to machinery use and low food availability are known to cause reductions in the abundance of springtails (Heisler and Kaiser 1995) and earthworms (Didden 2001), to the point that some cropped fields may have no earthworm populations at all. Earthworm communities in agricultural fields are probably a subset of grassland communities, as similar species are found, just at lower abundances (Boag, Palmer et al. 1997; Didden 2001).

Loss of organic matter and biologically simplified food webs in agricultural areas compared to grasslands can result in the reduction of the provision of services. The disruption of chemical engineers and earthworm communities hinders nutrient cycling, carbon regulation and thus climate control. Impoverished communities of mutualist and non-pest microbes and biological regulators also reduce plant protection and growth, impacting soil fertility. In agricultural cultures, this is often remediated through artificial means (e.g. fertilisers, pesticides). The change in chemical engineers communities may also alter the self-regulation of ecosystems so that toxicities may develop in soils or in water tables and effluents (Altieri 1999). As a matter of fact, given

the relatively low efficiency of crops to absorb nutrient inputs (50 to 80% in the best of cases for mineral N, for example), large amounts of nutrients brought as fertilisers in cropped fields leak to water tables, surface effluents and to the seas, resulting in eutrophication. Finally, mechanical practices and machinery increase soil compaction in agricultural fields, thus impairing soil mixing and aggregation and impairing water transfer, as explained in the section on soil tillage.

Grassland/Cropland → Urban land

In this switch, the abundance and diversity of all native soil species reduces dramatically, and more so with increasing urbanisation, mostly as a result of the prevalence of soil sealing. In contrast, some exotic soil species or some urban exploiters (species able to adapt to the human environment) may flourish (Germaine, Rosenstock et al. 1998; Hansen, Knight et al. 2002). Sewage sludge can also severely deplete soil invertebrate communities and soil trophic structure (Andres and Domene 2005), mainly reducing parasitic and predatory mites and predatory and omnivorous nematodes (Pavao-Zuckerman and Coleman 2007). Whether this is due to altered nutrient availability, or to adverse environmental conditions is not well known. However, urban forests can represent havens of soil biotic activity inside urban areas. For instance, although they have little organic residues and litter, and are heavily impacted by humans (resulting e.g. in soil sealing and compaction), urban forests favour earthworms which are able to dig down to deep soil organic matter stores (Kostel-Hughes 1995) (Pouyat, McDonnell et al. 1995).

The overall decline in all components of soil biodiversity, from already impoverished soil communities, results in the loss of almost all services provided by soil biodiversity. Litter decomposition is made almost redundant, given the reduced litter quantity and man-made management practices involving e.g. soil sealing or litter cleaning. As a result, carbon storage and climate control services are impaired. Moreover, the reduction in natural water regulation is often not fully compensated for by urban management towards surface water regulation and streaming.

Table 4-3: Impact of land-use change on the diversity of the three functional groups

Functional group	Forest → Grassland	Grassland → Cropland	Cropland → Urban land
Chemical engineers	↘ ↘fungi, ↗ bacteria	↘ (but some local ↗)	↘
Biological regulators	= / ↗ ↗ nematodes ↘ microarthropods	↘ Plant-feeding -> bacteria-feeding nematodes	↘
Ecosystem engineers	↗ anecic -> endogeic earthworms	↘ / 0 ↘ anecic earthworms	↘

Table 4-4: Impact of land-use change on the services provided by soil biodiversity

Ecosystem service	Forest → Grassland	Grassland → Cropland	Cropland → Urban land	Affected soil functions
Soil fertility and nutrient cycling	=/↘	↘	↘	Reduced decomposition of soil organic matter Reduced biological control
Regulation of carbon flux and climate control	↘	↘	↘	Reduced decomposition and mixing of soil organic matter
Regulation of the water cycle	-	↘	↘	Reduced burrowing activity
Decontamination and bioremediation	-	↘	↘	Impaired self-regulation of ecosystems
Pest control	-	↘	↘	Reduced biological control
Human health effects	-	-	-	

Box 19: Switching from forest to plantations

Natural forests may be converted into tree plantations, for production of trees. In this process, forests are partly or completely cut and tree seeds are planted and cultivated in order to give young trees, later planted in cities and gardens. Therefore, the ecosystem changes from a natural and biodiverse ecosystem to a managed, often mono-culture ecosystem. Many forest species have specific habitat requirements, and may have their populations drastically reduced in case of simplification of their habitat. The impacts are typically related to a reduction in litter or to the loss of old and dead trees.Forest management practices typically involve the reduction and change of the diversity and quantity of litter inputs. For instance, rotten logs are usually removed. As a result, all functional groups are affected by a reduction of their food and habitat resources in the switch to monoculture. A switch from forest to plantation in Malaysia resulted in the simplification of community structure in several taxonomic groups, and to an increased biomass of endogeic earthworms and of two other soil engineers. In contrast, communities of other soil engineers (termites, ants, beetle larvae) were depressed by the vegetation change (Tsukamoto 2005). However, many species can use food sources in plantations if colonisation is made possible from nearby native vegetation (Lindenmayer 2004). Tree plantations are also characterised by a suite of management cycles, involving the removal of aged and dead trees. As a result, plantations follow succession-like dynamics, with successional phases of instability and equilibrium, and soil communities present temporal patterns similar to those observed in natural successional patterns. The impacts depend on the species. For instance, observations in German and French forests showed that the diversity of microflora and Collembola dropped steadily after tree harvesting, and then increased continuously during the growing phase. In contrast, the richness of larger chemical engineers remained stable in the first phases, and decreased afterwards (Chauvat, Zaitsev et al. 2003; Hedde, Aubert et al. 2007). However, some successional stages are often inhibited in tree plantations (e.g. senescence and/or decay of dead trunks). This may preclude the occurrence of some essential soil functions, such as bioturbation by earthworms (Bernier and Ponge 1994). This can actually impact the fertility service, since earthworms typically use part of the litter accumulated during the mature phase, resulting in a massive release of nutrients and in the macro-aggregation of soil from the upper 10 cm, thereby creating suitable conditions for rapid growth of seedlings.

4. 3. 3. SCALE OF IMPACT

→ **SPATIAL SCALE: THE IMPORTANCE OF LANDSCAPE SCALE**

The impacts of land-use change on soil biodiversity may not be the same at the landscape scale as at the plot scale. At the plot scale, land-use changes impact soil food webs and biotic interactions, and thus the dynamic equilibrium within a soil community. At the landscape scale, dispersal is the key factor determining the amount and type of transfers of soil biotas among different communities. As part of this process, non-native species may colonise new ecosystems and potentially become invasive. But as a result of this re-arrangement of communities at the landscape scale, it is possible that soil services can still be provided over the whole landscape, although they are impaired in certain plots. However, this requires that the landscape composition and structure offers conditions for the resilience of soil communities.

To date, most scientific literature has focused on the impacts of land-use changes at the plot scale and on local soil food webs, often not taking into account the surrounding landscape. Solutions as regards the maintenance of biodiversity and ecosystem services are probably to be found at landscape level, by creating refuges for living organisms or focusing on stocks of seeds in soil. For example, grassy margins around cultivated fields, and riverine forests can help soil biota recover and recondition adjacent soils (Hansen, Knight et al. 2002; Bloem, Schouten et al. 2003), mainly larger invertebrates. Also, biodiversity at the landscape level provides sources of living organisms, which are crucial to increase the resilience of ecosystems, by allowing re-colonisation of degraded ecosystems, e.g. after a fire.

The landscape dimension is starting to be increasingly studied, in particular through the development of monitoring systems. For example, the BIOASSESS (BIOdiversity ASSESSment tools) project of the European Union (Watt 2004) focused on the impacts of land-use intensification for soil biodiversity at the landscape scale.

→ **TEMPORAL SCALE: SOIL BIODIVERSITY RESTORATION**

The effects of land-use changes are cumulative, as each land-use change comes with further modifications and potential damages to soil diversity, e.g. for grasslands changed into cultivated fields and then into an urban area, the native soil fauna of the grasslands will be subject to two consecutive and cumulative impacts.

The effects of land-use change may also only occur after a certain time lag. As an example, some consequences of urbanisation processes may present thresholds in their biological response and only be noticeable several decades following the start of urban development while others (e.g. sealing) will be immediately noticed (Hansen, Knight et al. 2002), depending also on the soil organisms considered. Soil organisms that depend on living plant roots will be gone within days to months following sealing, whereas mineralizing microbes may survive for years until all suitable organic matter has become mineralized. However, very little is known about what actually happens under such sealed conditions.

The time a disturbance takes to damage a soil is often much less than the time it takes to restore that ecosystem. Lowering the intensity of land use practices enhances soil biodiversity, but this process takes several years to take effect (Korthals et al. 2001). For instance, impacts of agricultural practices are often still felt on grasslands 25 years after their restoration. Moreover, not all groups reach their equilibrium number at the same pace. For instance, following a switch from agricultural to grassland, it take years,

sometimes more than 25 years for fungal to bacterial biomass ratios to recover to pre-disturbance levels (van der Wal, van Veen et al. 2006; Smith 2008). Some fungal-feeders did not come back at time scales for which study sites are available (Siepel 1996), and others did not reach their natural level again, like diverse predatory mites and nematodes (Holtkamp, Kardol et al. 2008). Similarly, for earthworms, although their total biomass significantly increased in the first couple of years, after seven years, epigeic earthworm communities were still depleted in favour of dominant endogeic earthworms (Citeau 2008). Therefore, restoration of soil biodiversity is not an easy task and when the soil chemistry and structure has been altered too far, the soil community rather develops towards a new state than to the original conditions. Very few studies (if any) have systematically analyzed consequences for ecosystem services and for economy of such barriers in soil transition.

4. 3. 4. FUTURE TRENDS

By 2035, it is predicted that rural areas will not be the dominant land-use type in Europe anymore, and grasslands are expected to decrease to around 10%, being replaced by surplus land. During the period 2000-2020, arable land is expected to decrease by 5% and grasslands and permanent crops by 1% each (EEA 2007), although these figures can become outdate very easily, due to sudden changes in land demand for, for example, biofuel production. Thus land use in rural areas may be changing faster than soil biodiversity can follow.

But while projections suggest that the agricultural surface will decrease slightly by 2035, organic agriculture surfaces are expected to continue increasing, as the rate of conversion from conventional to organic agriculture varies from 0.1% in Malta to 11.9% in Austria[36]. Indeed, the trend towards organic agriculture is growing, with organic agriculture surfaces moving from 1.8% to 4.1% of cultivated surfaces between 1998 and 2005 in the EU-25, and reaching 8% in some European countries[37]. Thus organic agriculture in Europe is already twice as much extended than it is in the rest of the world and effects on soil biodiversity, ecosystem processes and services have been outlined in the section on arable land above.

Total forest area has consistently increased over the recent decades and is expected to continue increasing, to grow by around 5% between 2000 and 2020. This will be due to a mixture of afforestation and natural processes, and likely to occur mostly on former agricultural land (EEA 2007), as well as along the tree margins in mountain and boreal areas. Moreover, an increase of approximately 1% for urban areas is expected until 2020 (EEA 2007), although large differences exist between Member States and regions within Europe. Given the poor soil biodiversity of urban soils, this would result in drastic reductions of the whole soil biomass and soil biodiversity. However, large differences exist across Member States and regions, with the proportion of the surface sealed ranging from 0.3% to 10%.

4.4. CLIMATE CHANGE

Global climate change can have important impacts on all the soil biodiversity and related services. These impacts can be direct or indirect effects linked to the alteration

[36] Eurostat, 2007
[37] www.organic.aber.ac.uk/statistics/index.shtml; last retrieval 15/09/09

of the climatic parameters (e.g. temperature, humidity). Here some examples are provided for each of the services discussed earlier.

4. 4. 1. IMPACTS ON CARBON STORAGE AND CLIMATE CONTROL

As previously discussed in section 3.3, an important fraction of carbon is stocked in soil which has important implications for climate regulation. This relationship is actually bidirectional equilibrium, such that climate change can also affect the soil carbon storage capacity.

Since soil is the largest store of carbon, one of the major issues related to climate change is that it will alter the activity of soil organisms leading to increased breakdown and loss of C to atmosphere, with positive feedback to climate change (Jenkinson 1991). In particular, climate change-driven modifications on:

- Temperature – freeze/thaw cycles
- Precipitation rate – wet/dry cycles
- CO_2 concentration

Climate change alters the soil carbon storage and climate control service directly, through a modification of:

- soil organic matter (SOM) decomposition

And indirectly through an alteration of:

- litter quality and quantity
- erosion
- photosynthesis

For example, a long term increase in temperature, such as observed under climate change has been shown to influence microbial respiration in laboratory experiments. The respiration of soil microbes is an important factor modulating the overall organic matter decomposition and thus the carbon storage service. The more respiration is efficient, the more organic matter is decomposed with in parallel, a release of CO_2. However, this direct relationship among soil organic matter decomposition and atmospheric temperature is still a subject of debate, and contradictory results are produced in laboratory and open field experiments. In contrast to laboratory studies, long-term field experiments on forest soils have shown that the organic matter decomposition is constant at different latitudes having different temperatures. Similarly, in grass prairies field experiments, an artificial warming of 2°C has been observed to provoke a microbial acclimatisation, thus basically an adaptation to the new conditions rather than an altered respiration rate. These two studies have weakened the idea that a positive feedback between increasing temperature and CO_2 release could exist due to microbial activity (Giardina and Ryan 2000; Luo 2001). Thus, depending on the model, the quantity of released carbon under modified climatic conditions can be differently evaluated (Schils 2008). In any case, the optimal climatic conditions for enzymatic activity of chemical engineers always vary locally, depending on the specific species assemblage in the considered geographical area (Desanto, Berg et al. 1993). Thus, global optimal conditions for the delivery of this service cannot be defined.

In addition to temperature, the soil moisture and the frequency of wet/dry and freeze/thaw cycles can modify the soil aggregation and have potential important

impacts on the availability of organic matter and, as a consequence, on the microbial community structure and activity (Figure 4-9).

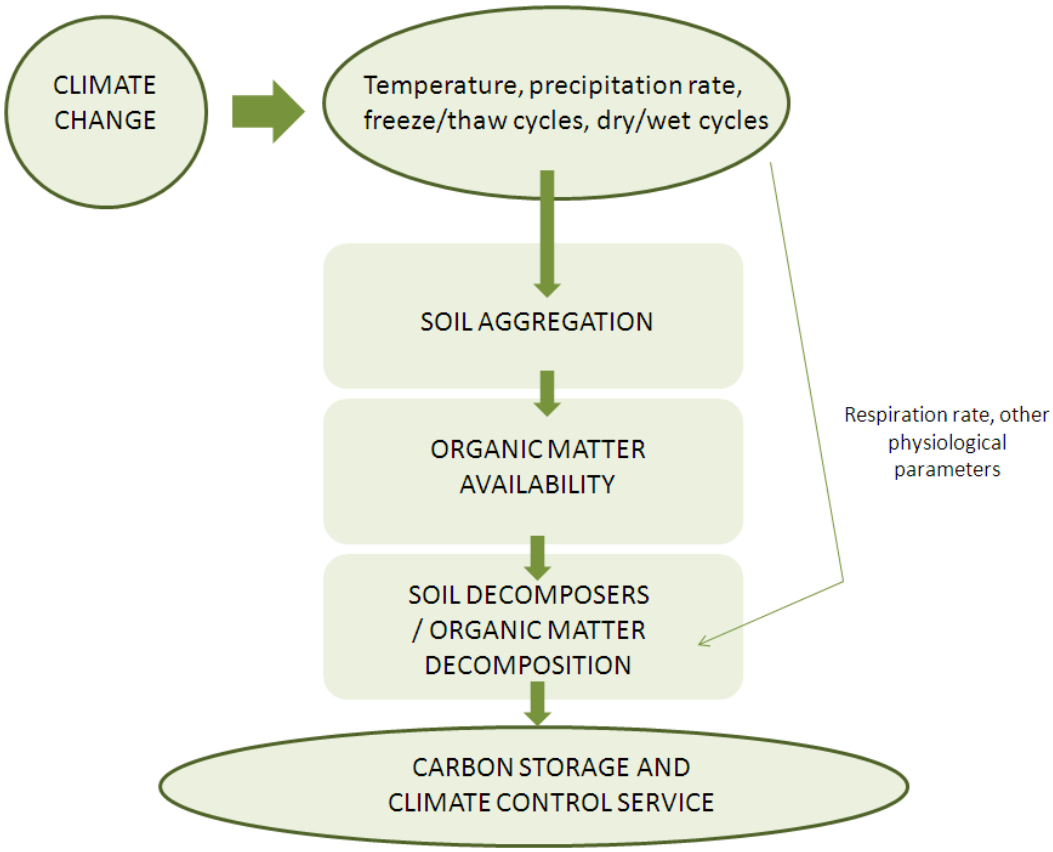

Figure 4-9: Simplified representation of the potential influence of climate change on climate control/carbon storage service

However, current understanding of the sensitivity of the decomposition rate to humidity is limited. A number of studies have shown a range of possible effects of the precipitation rate on the carbon cycle, with a special focus on wet/dry cycles. In general, the effects of the alternation of dry and wet conditions on this service depend on the local hydrological conditions. Thus, for example, depending on the water status of soil, the soil respiration rate can either be enhanced or repressed in European shrub lands during drought: in water limited ecosystems microbial respiration is repressed, while it is increased in ecosystems having a high relative humidity (Sowerby, Emmett et al. 2008). Heavy precipitation and drought events can also mobilise organic matter that was previously unavailable and stabilised through freezing, thawing or water logging events. This, in general, decreases soil aggregation and thus modifies the activity of microorganisms in the soil.

Future climate change may also affect land and ocean efficiency to absorb atmospheric CO_2, thus leading to a final positive feedback effect (Figure 4-10). As a consequence, an increased concentration of CO_2 can be considered both as a cause and an effect of climate change.

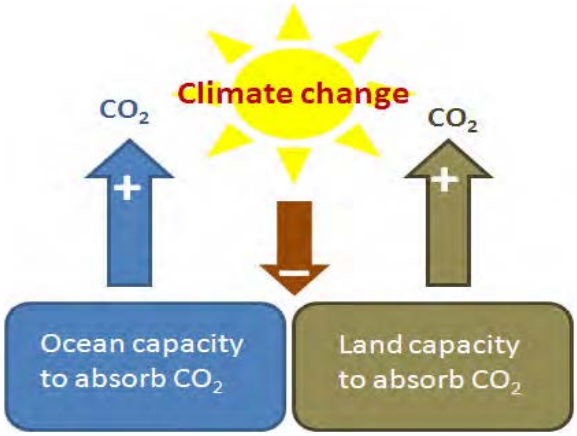

Figure 4-10: Positive feedback of climate change on CO_2 stored in land and ocean reservoirs

A number of experiments have demonstrated that an increase in atmospheric CO_2, which may be one of the effects of climate change, can significantly change soil environment mainly by modifying the distribution of above and belowground nutrients. For example, an increase of atmospheric CO_2 could lead to an increased plant growth, since CO_2 is the molecular building block for photosynthesis. This may lead to an increase in litter production rate and a modification in litter chemical composition, which may in turn lead to a change in its digestibility. Such modifications will then influence the nature of organic matter available for soil microorganisms (Figure 4-11)(Zak, Pregitzer et al. 2000). As a consequence, a modified litter production may modify the overall carbon supply and the nitrogen flow between plants and microorganisms (Berntson and Bazzaz 1997). In addition, elevated CO_2 may lead to an increased root growth which will have a significant impact on soil structure and major consequences for soil biota.

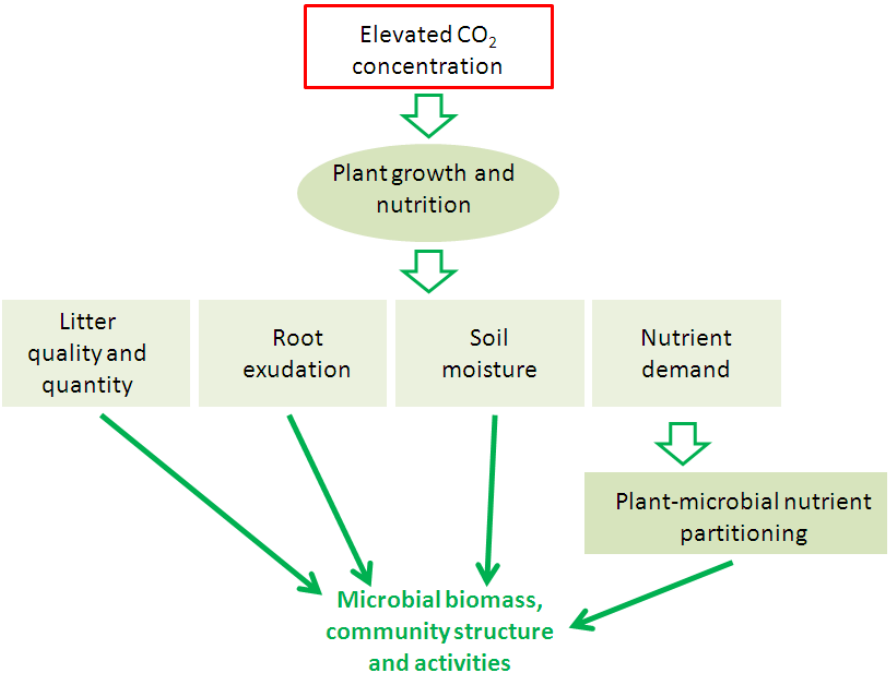

Figure 4-11: CO_2-induced alteration of resource availability for soil microbes. In this conceptual model, atmospheric CO_2 enrichment indirectly affects soil microbial biomass, community structure and activities by altering carbon, nutrient and water availability (Hu, Firestone et al. 1999)

4. 4. 2. IMPACTS ON NUTRIENT CYCLING AND FERTILITY

Climate change may not only affect the carbon cycle, but also the nitrogen cycle. It has been shown in a natural forest soil, that soil warming increases the nitrogen availability for plants through an increase in net nitrogen mineralisation (Melillo, Steudler et al. 2002)(Figure 4-12).

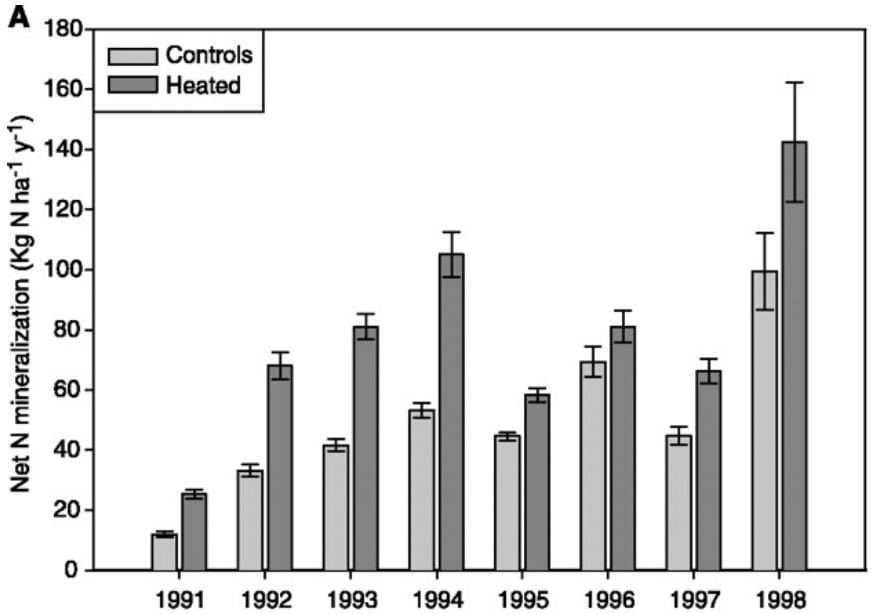

Figure 4-12: Average yearly net nitrogen mineralisation rates measured in the heated and disturbance control plots at the Harvard Forest soil warming experiment (Melillo, Steudler et al. 2002)

The observed effect on nitrogen mineralisation is probably due to an effect of soil warming on microbial activity. The impacts of temperature on microbes regulating the nitrogen cycle within soil depend on the considered ecosystem and the analysed species. For example, a study (Horz, Barbrook et al. 2004) shows that a reduced microbial activity in grasslands may be expected under climate change. Indeed, the authors show that an increase of atmospheric CO_2 (to 700 pm) and a high precipitation rate (50% of relative humidity) affects the Ammonia Oxidising Bacteria (AOB) community and structure negatively (Figure 4-13). The association of an elevated ambient temperature and high precipitation rate have been observed to have similar effects. In conclusion, these observations suggest that climatic factors susceptible to be altered by climate change, such as CO_2 concentration, temperature and precipitation rates can significantly alter soil chemical engineers growth and activity and that such modifications can have implications for nutrient cycling and fertility services. However, specific studies on individual soils hosting different soil bacterial communities should be performed, in order to anticipate the nature of impacts on this service at the local level.

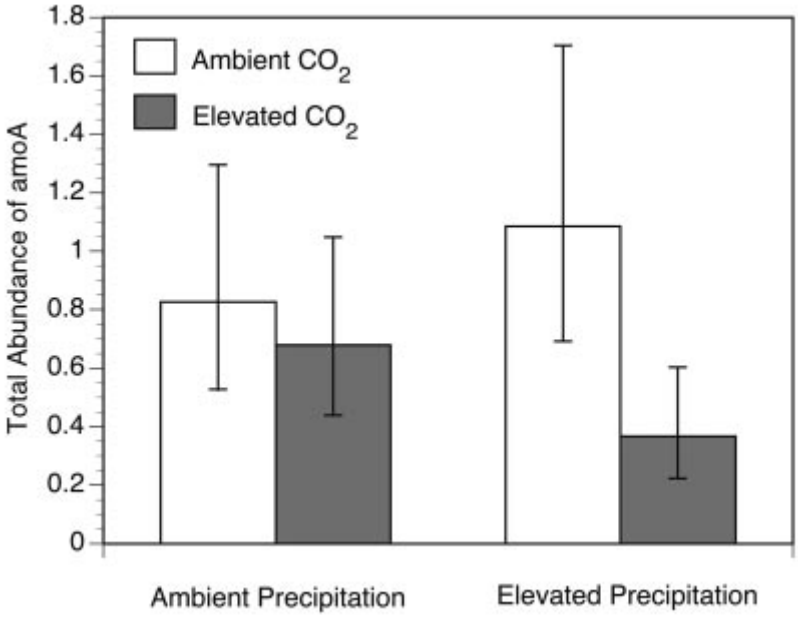

Figure 4-13: The effect of CO_2 and precipitation levels on AOB population (Horz et al. 2004)

4. 4. 3. IMPACTS ON WATER CONTROL

Climate change can have profound impacts on:

- **Soil properties and structure**: As shown in (a) in Figure 4-13, changes in temperature and precipitation rates may have important implications for soil properties (e.g. pH, porosity)
- **Soil organisms**: as discussed in section 2. 2. 1 and as shown in (b) in Figure 4-13, the ecology of all three functional groups of soil organisms is sensitive to climatic parameters

In turn, as shown in (c) in Figure 4-13 ,a bidirectional relationship exists between soil properties, soil structure and soil organisms (Young, Blanchart et al. 1998).

In particular, regarding the impacts of climate change on the biodiversity of soil organisms, any modification of the ecology of ecosystem engineers, which through their biogenic structure and their 'engineering' action are crucial in ensuring the infiltration of water underground, could alter the water control service. But the quantity and the quality of water stored in soil is not only a function of infiltration, but also of several other processes including drainage, capillary rise, evapo-transpiration, etc. All of these processes are at least partly dependent on plants diversity. The aboveground/belowground relationships are thus also crucial in the provision of this service and likely to be altered by climate change, given that both plants and soil organisms are sensitive to climatic parameters (d in Figure 4-13). Thus, even if they have not been quantified yet, some important impacts of climate change on this service could be expected.

4. 4. 4. IMPACTS ON PEST CONTROL

As argued in section 3. , more diverse soil communities ensure better pest control. Climatic factors susceptible to be altered by global climate change, can affect soil organisms differentially, favouring some groups while being deleterious to others.

Favoured species can sometimes become pests. The impacts of climate change on pests can be:

- **Direct:** climate change can provoke massive migrations, range expansion, and geographical or seasonal shifts in species ranges that alter the distribution of several deleterious pest species.
- **Indirect**: climate change can alter the biotic interactions of pest species within the ecosystem

In the case of insects, for example, most studies have concluded that insect pests generally become more abundant as temperatures increase, through a number of inter-related processes, including range expansions and phenological changes, as well as increased rates of population development, growth, migration and overwintering (Cannon 1998).

Biological interactions may also be disrupted in the soil as a result of climate change, releasing some pest species from their biotic control and enabling them to spread. This can lead to pest outbreaks of e.g. bacteria, fungi, nematodes or insects species. However, sometimes the synchrony between crops, pest and their biotic control will be kept in climate change scenario, if the three are sensitive to climate in similar ways, or able to adapt accordingly. An increase of the average temperature of 2°C in UK, for example, will result in an advance of the growing season of 2-3 weeks (Rowntree 1991). As a consequence, some pest species, such as the spittlebug, will respond by anticipating their life cycles of 2–3 weeks, leading to no main changes to the spittlebug invasion frequency (Whittaker 1996).

Thus, climate change can have important impacts on pest control mediated by soil biodiversity (Garrett 2006), but these effects are very context dependent. Therefore, in order to foresee possible impacts and take precautionary measures, a case by case approach (associating geographical condition, pest species and plant species) should be considered.

4. 4. 5. CURRENT AND FUTURE TRENDS

Today, global warming is a well known fact, with an overall increase of both air and ocean temperatures, and evidence of a significant melting of snow and a rise in average sea level. Several scientific studies show that almost all natural systems are being affected by this process, including soils.

On the basis of several scenarios exploring alternative development pathways, and covering a wide range of demographic, economic and technological driving forces, future GHG emissions trends can be estimated. A range of scenarios all concur to predict a warming of 0.2 °C per year for the next two decades, along with a modification in the rate and intensity of precipitations.

Such modifications of climatic factors could strongly impact soil functional groups of organisms either directly, through an effect on their ecology, or indirectly, through increased floods, droughts, wildfires, insects distributions and land-use changes, and fragmentation of natural systems. An increase in soil erosion rate is also expected.

In conclusion, climate change is likely to have significant impacts on soils that may affect all of the services provided by soil biodiversity, indeed the quantification of these impacts is not possible at the moment (Schils 2008). In any case, all mitigation and attenuation measures taken to limit global climate change are expected to have a

beneficial impact on soil biodiversity preservation, soil functioning and associated services.

4.5. CHEMICAL POLLUTION AND GMOS

A large range of chemical pollutants can reach the soil of both natural and modified ecosystems through various routes (direct application, atmospheric fall out, waste disposal, etc.) and influence the functioning of soils on a wide spatio-temporal scale, from individual organisms to landscapes.

4. 5. 1. TYPES OF CHEMICAL POLLUTANTS

→ **PESTICIDES**

The deleterious impacts of pesticides on soils depend on their chemical characteristics. The persistency of pesticides in soil can be highly variable, ranging from hours (e.g. fumigant nematicides) to decades (e.g. organochlorine insecticides). Similarly, their toxicity can be restricted to a class or affect a broad spectrum of organisms, either directly or indirectly. In addition, some pesticides can bio-accumulate, which means that they can be concentrated in the bodies of soil organisms and taken up into higher levels of the food chain.

→ **INDUSTRIAL CHEMICALS**

A number of industrial chemicals can pollute the land near their manufacturing sites or be transported as gaseous emissions or through water to other soils (TGD 2003). The industrial chemicals which can constitute a threat for soil biodiversity include, for example, heavy metals, inorganic gaseous emissions (e.g. NO_2), persisting oil and fats (e.g. petroleum) and the polychlorinated biphenyls which, similarly to some class of pesticides, can be bio-accumulated by some species of soil organisms.

4. 5. 2. IMPACTS OF CHEMICAL POLLUTION ON SOIL BIODIVERSITY AND RELATED SERVICES

The impacts of chemical pollution on soils can be extremely heterogeneous, and either direct or indirect.

Effects on survival or reproduction are measured in acute or chronic tests. The direct effects include an impaired survival or reproduction of soil organisms due to acute toxicity or bioaccumulation. Toxicity and bioaccumulation affect the metabolism, growth, development or longevity of soil organisms, and even possibly cause some genetic effects (e.g. leading to genetic modifications of the target organism). These direct effects affect individual species of soil organisms, like microorganisms, invertebrates or plants depending on the nature of pollutants and on its distribution into the soil matrix. Thus, the direct impacts of chemical toxicity on soil organisms can have important detrimental effects on their population dynamics, by influencing basic reproduction and survival parameters, and thereby modifying the size, sex ratio, and stability of soil organism populations.

Alternatively, chemical pollution can have indirect effects. In general, the indirect effects are more difficult to evaluate and are less well studied than the direct effects on specific organisms. The indirect effects can be due to a contamination of soil organisms' food supply and in general, involve a modification of the functions of soil organisms. For example, pesticides can alter or disrupt dynamic soil processes which are crucial for the delivery of soil services (e.g. the decomposition of the organic

matter)(Hendrix and Parmelee 1985) leading to an impairment of the nutrient cycling and fertility service. Sometimes, chemical pollutants can also have strong indirect impacts on predator/prey relationships, thus altering the food web (Edwards 1999). Acid emissions, such as NO_2 or SO_2 emissions can alter the availability of soil organic matter, and therefore its decomposition and the soil pH, which in turn modify the community structure and composition of soil organisms. In addition, pollutants, through indirectly influencing the relationships between belowground organisms and plants, can affect the structure and composition of plants communities (Edwards 1996).

Thus, in fact chemical pollutants can influence soil functioning at all trophic levels, altering individual organisms, populations or communities, and at different spatio-temporal scales. Here we present the impacts of chemical pollution on each of the three main functional groups.

→ **IMPACTS ON CHEMICAL ENGINEERS**

Chemical pollutants can strongly alter the ecology and the physiology of chemical engineers such as bacteria and fungi. Several studies have demonstrated the effects of pollutants (e.g. pesticides such as fungicide tebuconazole, pyrethroid insecticide lambda-cyhalothrin) on (Sturz and Kimpinski 1999; Cycon, Piotrowska-Seget et al. 2006):

- microbial survival and growth: the pesticide sulphonylureas, for instance, targets the enzymes involved in the synthesis of the amino acids valine, leucine and isoleucine; non-target organisms such as bacteria and fungi can be harmed by the compounds in high concentrations. Moreover, during degradation of the pesticide fenpropimorph, active saprotrophic fungi are substantially affected,
- microbial respiration
- enzymatic activity (i.e. alteration in the efficiency in nutrient transformation): for instance, pesticides such as trichloronate, linuron, thrimethacarb have been observed to have some effects (Bollag 1993).

These studies have sometimes reported conflicting results and the mechanisms underlying the observed effects are not always understood. In fact, the characteristics of a chemical pollutant in soils can be altered by the action of soil organisms and by the presence of other pollutants. For example, Cadmium can be present as an impurity in certain phosphate fertilisers, and can be captured by hyperaccumulator bacteria species like Thlaspi caerulescens or by fast-growing plants, such as Salix and Populus spp. that accumulate above-average concentrations of only a smaller number of the more mobile trace elements, including Cadmium. The longer-term effectiveness of phytoextraction and associated environmental issues are still studied and not forseen with certainty (Dickinson 2009). But in specific cases, mycorrhizal fungi can absorb Cadmium and modify mine residues, and hence contribute to soil formation (Gonzalez-Chavez 2009). However, to date, interacting effects between pesticides and biotic factors received little attention. In addition, in some cases a pesticide can surprisingly favour microbial growth. This occurs for example in the case of Fosthiazate, which, being an organophosphate, may serve as an energy source for microorganisms (Eisenhauer, Klier et al. 2009).

When considering the impacts of chemical pollution on chemical engineers, following aspects should be taken into account:

- A single chemical can have different effects on different soil microbial species and communities, which can disturb the interactions within and among functional groups
- Microbial organisms have a very short reproduction time (e.g. an average of 20 minutes for bacteria in optimal conditions), thus an exposition to some toxic chemical could rapidly lead to a resistant microbial population. Chemical resistance evolves via natural selection acting upon random mutation. Thanks to this process, heritable traits (genes) codifying for such resistance and making it more likely for a microbial organisms to survive and successfully reproduce, become more common in a population over successive generations. In the case of bacteria, an additional mechanism can facilitate the development of a chemical resistant population. Once a gene carrying the information for the resistance is generated, bacteria can transfer the genetic information in a horizontal fashion (between individuals) by DNA exchange.
- On the other hand, the positive aspect of chemicals-microorganisms interaction is that some chemicals can be transformed by soil microorganisms into non- or less toxic compounds; in this case we speak of bioremediation (see also section 3.5).

In conclusion, it is clear that the microbial community structure in soil may be markedly changed by chemical pollution. Some microorganisms may be suppressed and others may proliferate in the vacant ecological niches. This may in turn lead to successions in the microbial community and thus to altered activities at a later point in time.

→ **IMPACTS ON BIOLOGICAL REGULATORS**

Industrial chemicals, such as heavy metals and petroleum, have been reported to have various deleterious impacts on biological regulators. Some studies have been performed on nematodes. Industrial chemicals can affect the lifespan of individuals (Figure 4-14), and as a result the abundance and the structure of soil nematode communities (Ettema and Bongers 1993; Chen 2009). Such changes can seriously impair the functioning of nematode communities and affect the provisioning of related services.

The responses of a species to individual pollutants can vary depending on the dose and the exposure time (e.g. the sensitivity of nematodes to pentachlorophenol after 72 hours of exposure can be 20 to 50 times higher than their sensitivity to cadmium). Therefore, for each considered chemical pollutant and species, a specific dose-response curve should be determined.

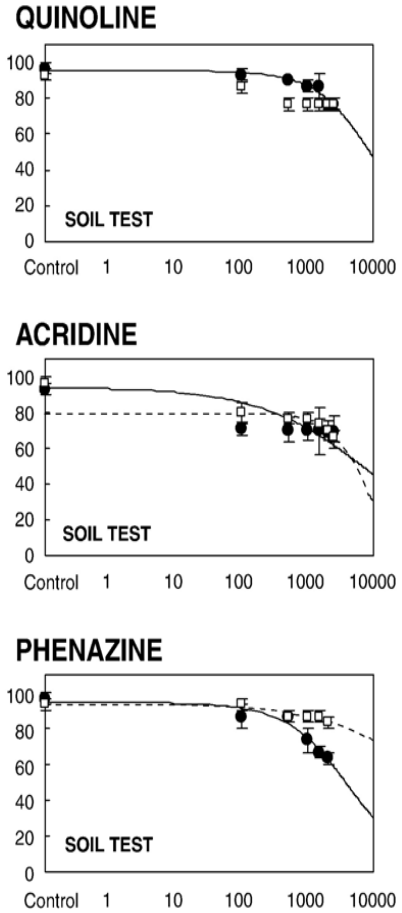

Figure 4-14: Effects of three pesticides on nematodes (C. Elegans) survival after 24 h (black) and 48 h (white) at different concentrations (Sochová 2007)

→ **IMPACTS ON SOIL ECOSYSTEM ENGINEERS**

Earthworms, in contrast to ants and termites which tend to be resistant to several pollutants (Eeva, Sorvari et al. 2004), are often highly sensitive to soil pollution. Their sensitivity is due both to:

- Their close contact with the pore water and their high water permeable epidermis: water soluble pollutants can easily be internalised
- The fact that they swallow high quantities of soil

The influence of heavy metals and their bioaccumulation by earthworms has been, for example, the subject of many studies in the past (Bouche 1984; Morgan and Morgan 1999; Kennette, Hendershot et al. 2002). Metals have been shown to cause mortality and reduce fertility, cocoon production, cocoon viability and growth of earthworms. Rather than the total metal content of soils, it is worth considering the metal fraction that is mobile and thus available for earthworms. For instance, Cadmium from the industries and from the production and the application of artificial phosphate fertilizers mainly ends up in soils, giving rise to cadmium-rich sludge. Cadmium strongly absorbs the organic matter in soils and can be taken up by plants. This is a potential danger to the animals that are dependent upon the plants for survival. Earthworms and other essential soil organisms are also extremely sensitive to cadmium poisoning. They can die at very low concentrations and this has consequences on the soil structure. When cadmium concentrations in soils are high they can influence the soil processes of

microorganisms and threaten the whole soil ecosystem. However, it is still not possible to predict with a high degree of confidence the body burden of an earthworm at a metal contaminated site, and more research is needed.

Pesticides can also affect several physiological parameters of earthworms, including neuronal activity, immune response capacity (Sorvari, Rantala et al. 2007), and reproduction rate (Bustos-Obregon and Goicochea 2002). In addition, pesticides can be at the origin of deleterious effects on DNA causing genetic mutations and increased productions of free radicals resulting in cellular degeneration (Song 2009).

A number of factors should be considered when evaluating the impacts of chemical pollution on earthworms:

- Earthworms are selective consumers (Edwards, Bohlen et al. 1996), and food selectivity depends on the considered species. Thus, rates of heavy metal bioaccumulation for instance will differ according to the species (Morgan and Morgan 1999).
- The distribution of a pollutant may vary among soil phases: it can be absorbed to the solid phase or dissolved in the liquid (e.g. water pore) phase, depending on its chemical form. Different forms of the same pollutant can then be more or less available for uptake by earthworms, thus having different levels of 'chemical availability'. Since the environmental conditions (e.g. soil pH) can influence the chemical availability of a pollutant, any modification of soil properties may alter pollutants distribution.
- Earthworms are able to eliminate excess metals in their bodies, thanks to a physiological control mechanism. Depending on the pollutant this elimination pathway can be more or less efficient. For example, in the case of metals, copper and zinc are easily eliminated by physiological pathways based on carrier systems, which naturally exist for the physiological control of these elements. However, the mechanism of excretion is much slower for cadmium and lead. Thus, for these metals the main detoxification pathways are much more complex and involved intracellular granules which act as metal storage compartments (Spurgeon and Hopkin 1999).

In conclusion, in order to evaluate the sensitivity of earthworms to a chemical pollutant, information on the local earthworm species, their feeding and habitat preferences are needed, as well knowledge of their ability to expulse specific pollutants.

→ IMPACTS ON SOIL BIODIVERSITY RELATED SERVICES

As discussed earlier, many pollutants can have seriously adverse effects on soil systems. Indeed, there are very few terrestrial ecosystems worldwide that are not exposed to chemical pollutants. In general, research into the effects of pollutants on soils is relatively limited, most studies simply showing the susceptibility of particular organisms to certain pollutants. As a matter of fact, to predict effects of pollutants on the communities of soil organisms, information must be extrapolated from a small subset of the species, because it is impractical to conduct a large number of tests on a large number of species. Thus, holistic integrated studies that evaluate the impacts of chemical pollutants on soil functioning as a whole and the related services are still at their beginning and the issue of threats to soil biodiversity has only recently been covered by ecological risk assessments (Box 20).

4. 5. 3. THE IMPACTS OF GENETICALLY MODIFIED ORGANISMS (GMO) ON SOIL BIODIVERSITY

In the case of soils, when we mention GMO this refers to plants in the majority of cases. Genetic modifications are used to improve crop quality (e.g. pest resistance, timing of ripping process) and productivity (e.g. growth capacity). The existing molecular techniques involve the insertion and integration of a short segment of DNA from another organism (e.g. plant, microbe or animal) into the genome of the plant, to add single characteristics to the plant breeding line and variety.

However, GM plants can also be considered as a source of pollution for soil organisms, because they can have an indirect impact on soil biodiversity, and favour the development of genetic resistance in target pest organisms (Eastham 2002).

So far, the majority of studies on the effects of GM crops on soil biodiversity, have focused on Bt-modifications (Bacillus thurigiensis) (Icoz and Stotzky 2008). However, these modifications mainly target insect resistance, and their impact on the bulk of soil biodiversity is likely to be indirect and negligible (Kowalchuk, Bruinsma et al. 2003).

In contrast, the main question related to soil biodiversity is whether the effects of the GM-crops fall outside the normal operating range (NOR) of soil organisms, which defines their normal metabolic and physiological fluctuations within agricultural systems.

Large studies in this area include farm-scale analyses in the UK38, the EU-project Ecogen (Krogh and Griffiths 2007) and the Dutch ERGO-programme39. But since soil biodiversity is so variable and diverse, the identification of such NORs is highly complicated. Moreover, soil tillage, fertiliser application and pesticide use already exert large effects on soil communities. If the effects of such practices are also considered to fall within the NOR, then it is expected that the effects of most currently known GM-crops will also easily fall within this range (Kowalchuk, Bruinsma et al. 2003; Weinert 2009).

Out-crossing of GM traits to wild plant species has been studied intensively, but the question how these introduced genes may influence soil biodiversity in nature is still an open question. Indeed, few studies have identified some impacts on soil organisms. The main identified impacts of GM crops on soil communities can be divided depending on the considered functional groups.

→ IMPACTS ON CHEMICAL ENGINEERS

GM crops can influence microbial communities by several ways: altering the quality and the quantity of growth substances (Oger, Petit et al. 1997), the structure of the bacterial community (Di Giovanni, Watrud et al. 1999), the efficiency of microbial mediated processes (Hopkins, Webster et al. 2001), or the genetic transfer between GM crops and bacteria.

However, this last impact has recently been debated by the scientific community and the transformation frequencies under field conditions are supposed to be very low (Demaneche, Sanguin et al. 2008). It is important to underline that the public debate about antibiotic resistant genes in transgenic plants should not divert the attention from the real causes of bacterial resistance to antibiotics, such as the continued abuse and overuse of antibiotics prescribed by physicians and in animal husbandry (Lynch,

[38] www.defra.gov.uk/Environment/gm/fse/ ; last retrieval 10/08/09
[39] www.defra.gov.uk/Environment/gm/fse/ ; last retrieval 10/08/09
[39] www.nwo.nl/NWOHome.nsf/pages/NWOA_6N4LKX_Eng ; last retrieval 10/08/09

Benedetti et al. 2004). In any case, most of the studies that have been conducted have detected some effects (e.g. transfer of transgene to soil bacteria) (Bruinsma, Kowalchuk et al. 2003). In addition, GM crops have been reported to alter the mycorrhizal colonisation of roots (Turrini 2008).

→ **IMPACTS ON BIOLOGICAL REGULATORS**

Small impacts are in general highlighted. For example, the analysis of the soil fauna in agricultural fields cultivated with genetically modified tobacco plants has shown an increased number of nematodes and a decreased number of collembola (Donegan, Seidler et al. 1997). The majority of the studies are focused on nematodes and long-term studies on microarthropods included in this functional group are rarer (Heckmann, Griffiths et al. 2006).

→ **IMPACTS ON ECOSYSTEM ENGINEERS**

The influence of GM crops on earthworms varies depending on the considered genetic modification and earthworm species, ranging from no effects to slightly significant effects. For example a type of GM plants has been shown to influence the cocoon hatchability of an earthworm species (Vercesi, Krogh et al. 2006), while another type has no significant effect on all the analysed earthworm species.

For the future, studies may need to focus on specific functions, rather than on biodiversity as a whole. Decomposer functions and enzymatic functions are important candidates for such studies, as they are crucial for the cycling of elements through ecosystems. Also, we need to move more towards a predictive system that can help to estimate how specific modifications can influence soil biodiversity and functions inside and outside cropping systems. Until a system is devised to establish the full consequences of GM crops on soil functioning, for example on litter decomposition and carbon and nutrient cycling (Powell 2009), the consequences of specific modifications will need to be evaluated on a case by case basis (Kowalchuk, Bruinsma et al. 2003).

This is likely to become an issue of increasing importance in the future. Indeed, for obvious reasons of increased crop productivity, the area employed for culturing GM crops has been constantly increasing in the last years (Figure 4-15).

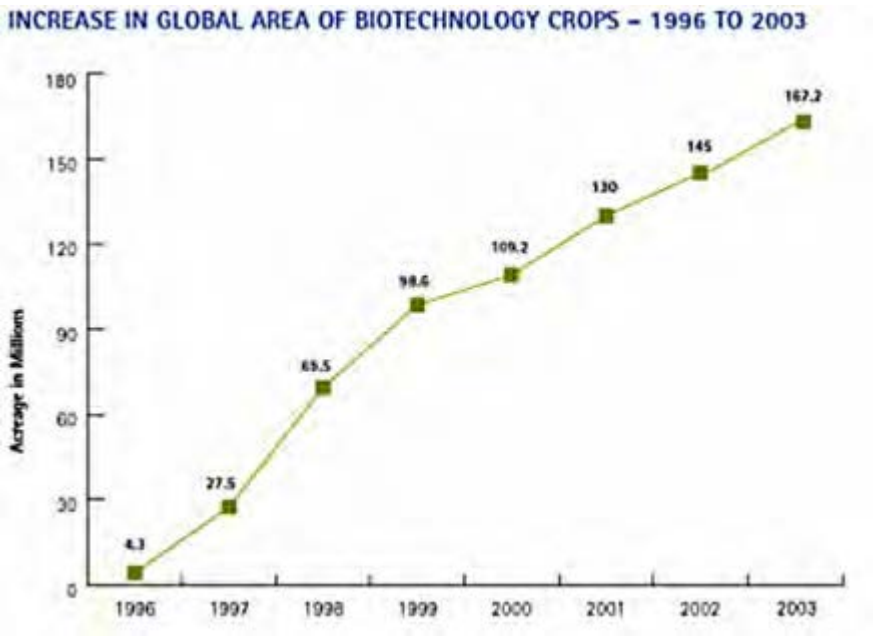

Figure 4-15: Area employed for culturing GM crops from 1996 to 2003 (James 2003)

Box 20: Taking into account biodiversity in ecosystem risk assessments

The main objective of the Ecosystem Risk Assessment (ERA) methodology is to identify the origins and quantify the impacts of human activities on natural ecosystems. To this end, ERAs must assess the ecological integrity of ecosystems. Although ecological integrity is tightly linked to the state of biodiversity, since it can be defined as the condition of relatively unaltered ecosystems, which contain a full suite of native species, **biodiversity has so far been neglected in environmental assessments**. This is because biodiversity is often considered too broad and vague a concept to be applied to real-world regulatory and management problems.

Measurable indicators can be selected to assess the status of biodiversity over time. For instance, useful, measurable indicators can be chosen that correspond to the different functions (e.g. soil organic matter decomposition) or different levels of biological organisation (e.g. landscape, community, species). Particular attention should be paid to validate the relationships between indicators and the components of biodiversity they represent (Noss 2000), and to ensure that they allow to answer the specific question that the assessment is intending to answer. A comprehensive indicator considering all the aspects of biodiversity does not exist.

Some attempts have been made. Biodiversity assessment is not so uncommon in site specific ecological risk assessment (e.g. using the TRIAD approach). Recently, a methodology was proposed for performing a **qualitative** assessment of soil quality, based on the analysis of the possible effects of soil contamination on ecosystem biodiversity. Such methods could be first good tools for policy makers (Semenzin, Critto et al. 2009). However, methods capable of assessing the impairment of soil biodiversity **quantitatively** are still lacking.

However, given the different impacts of chemical pollution on soil organisms, significant impacts of chemical pollution on soil biodiversity functioning and related services can be expected (Table 4-5).

Table 4-5: Possible impacts of chemical pollution on soil biodiversity related services, on the basis of its impacts on soil organisms

Chemical pollutant	Affected soil organisms	Affected soil function	Affected soil service
Pesticides	Biological regulators, ecosystem engineers	Organic matter decomposition, residue fragmentation	Nutrient cycling, soil fertility
Pesticides	Chemical engineers (microorganisms), biological regulators (micro-fauna)	Mineralisation, immobilisation	Nutrient cycling, soil fertility
Pesticides	Ecosystem engineers	Bioturbation, Soil structure regulation Soil organic matter production	Nutrient cycling, soil fertility, Water regulation
Pesticides	Biological regulators	Population control	Pest control
GM plants	Chemical engineers	Mineralisation, organic matter decomposition	Nutrient cycling, soil fertility
Industrial chemicals (heavy metals)	Chemical engineers		Nutrient cycling, soil fertility
Industrial chemicals (heavy metals)	Biological regulators (Nematodes)	Soil structure regulation Soil organic matter production and transformation, regulation predation	Nutrient cycling, soil fertility, pest control, water control, climate control
Industrial chemicals (heavy metals)	Ecosystem engineers (Earthworms)	Soil structure regulation Soil organic matter production and transformation	Nutrient cycling, soil fertility; water control
GM plants	Ecosystem engineers (Earthworms)	Soil structure regulation Soil organic matter production and transformation	Nutrient cycling, soil fertility; water control

In conclusion, when analysing what are the most affected soil functions by chemical pollution, we can see that organic matter degradation and soil structure regulation are highly altered functions. As a consequence, the soil fertility and nutrient cycling service together with the water control service are the most affected services by this category of threat.

4. 5. 4. CURRENT AND FUTURE TRENDS

Humans currently use more than a third of the production of terrestrial ecosystems and about half of usable freshwaters. To obtain such productivity, terrestrial nitrogen supply and phosphorus liberation have been doubled through the addition of fertilisers and deleterious species have been controlled through pesticides. Likewise, we have significant quantities of pesticides have been created and released globally, causing a mounting pressure on soil organisms. Pesticide production in the world has increased by 400% between 1960 and 1990. At this rate, pesticide production will be almost twice that of today by 2020, and three times the present amount by 2050. Indeed, this probably gives a much higher figure than in a realistic scenario and even if the EU consumption is high, reaching a total of almost 300 000 tonnes in 2001 in EU-15 (see

following table) recent EU legislation to limit the use of pesticides will probably help in controlling EU pesticide consumption in the future.

Projections for 2050 show an increase from 1.9- to 4.8-fold. Should trends continue, by 2050, humans and other organisms in natural and managed ecosystems would be exposed to markedly elevated levels of pesticides (Tilman, Fargione et al. 2001).

Regarding GMO, since the first large-scale cultivation of transgenic insect-resistant crops in 1996, the global area of transgenic crops has increased 47-fold, from 1.7 million hectares to 81 million hectares in 2004 (James 2003). The dominant trait introduced in cultivated transgenic crop plants is herbicide tolerance, followed by *Bacillus thuriengiensis* (Bt) based insect resistance. Today, the four major genetically modified crops are herbicide tolerant soybean and canola, and Bt maize and cotton (James 2003). For vegetable crops, tomatoes with delayed fruit ripening and potatoes with insect and virus resistance are the two commercialised transgenic crops. However, several novel traits have been already introduced in vegetable crops, but are not yet commercialised. For example, in recent years considerable success has been achieved in introducing abiotic stress tolerance, quality traits and expression of various proteins and enzymes of pharmaceutical and industrial importance. Indeed, in the future the commercial trends of GMO may be less impressive than thought at the beginning, due to active opposition of public and environmental NGOs which plead in favour of the precautionary principle and ask for a moratorium on GMO products.

Table 4-6: Trends in EU pesticide consumption rates in 2001 (source: INRA)

	Fungicide		Herbicides		Insecticides		Others		Total	% (a)	SAU/ SAU UE 15 (b)	Ratio (a)/(b)
	Tonnes	%	Tonnes	%	Tonnes	%	Tonnes	%				
France	54130	54,3%	32122	32,2%	2487	2,5%	10896	10,9%	99635	34,3%	21,0%	**1.6**
Italy	23288	51,8%	8191	18,2%	9747	21,7%	3741	8,3%	44967	15,5%	11,0%	**1.4**
Spain	13790	33,7%	10374	25,4%	11631	28,4%	5099	12,5%	40894	14,1%	21,1%	0.7
Germany	8418	32,1%	13337	50,9%	868	3,3%	3601	13,7%	26224	9,0%	12,1%	0.7
Portugal	13915	56,0%	6399	25,7%	2616	10,5%	1926	7,7%	24856	8,5%	2,9%	**2.9**
UK	3628	18,0%	11817	58,6%	857	4,2%	3874	19,2%	20176	6,9%	12,0%	0.6
Greece	4860	43,7%	2650	23,9%	2638	23,7%	963	8,7%	11111	3,8%	6,0%	0.6
Netherlands	3628	46,1%	2172	27,6%	227	2,9%	1840	23,4%	7867	2,7%	1,4%	**1.9**
Belgium	1595	31,5%	2345	46,3%	560	11,1%	566	11,2%	5066	1,7%	1,1%	**1.5**
Austria	1088	38,6%	1317	46,7%	94	3,3%	322	11,4%	2821	1,0%	2,4%	0.4
Denmark	511	19,5%	1925	73,5%	66	2,5%	116	4,4%	2618	0,9%	1,9%	0.5
Sweden	339	18,2%	1462	78,4%	24	1,3%	40	2,1%	1865	0,6%	2,2%	0.3
Finland	192	13,4%	1120	78,2%	42	2,9%	78	5,4%	1432	0,5%	1,6%	0.3
Ireland	410	30,7%	795	59,6%	84	6,3%	45	3,4%	1334	0,5%	3,1%	0.2
EU 15	129792	44,6%	96026	33,0%	31941	11,0%	33107	11,4%	290866	100%	100%	

4.6. INVASIVE SPECIES

4. 6. 1. IMPACTS OF INVASIVE SPECIES ON SOIL BIODIVERSITY

Biological invasions are one of the fifth key worldwide threats for biodiversity and ecosystem functioning. Exotic species are called invasive when they become disproportionally abundant in their new environment. Traditionally, biological invasions are species that cross artificial barriers, for example helped by transport or tourism. Currently, global warming enhances the spreading of species from lower to higher latitudes and altitudes.. As invasive species may disturb ecological relationships or nutrient and energy flows through ecosystems, they can have major direct and indirect impacts on ecosystem goods and services and on native biodiversity.

Invasive species are known for any life form: vertebrate animals, invertebrates (e.g. insects), plants, and microbes. Invasive vertebrate animals occur on all continents, but their largest effects are in areas where species with such traits did not occur before. For example, deer and rats in New Zealand have major impacts on vegetation structure and nutrient cycling (Wardle, Yeates et al. 2001; Fukami, Wardle et al. 2006). Invasive plants are widespread both in Europe and in all other continents. Their impact seems larger in for example North America, Australia, New Zealand and South Africa, but the number of invasive plant species in Europe and their impact on ecosystems are also considerable. Invasive insects are numerous in Europe, as well as in other continents. A famous example is the Colorado beetle, which threatened potato crops in the 1950s and 1960s, whereas currently corn in south-eastern Europe is attacked by western corn root worm that originates from North America (Ciosi, Miller et al. 2008). This root-feeding insect is still expanding north-west wards. Invasive earthworms are mainly known in North America (most of them coming from Europe) and in the tropics (Bohlen, Groffman et al. 2004; Gonzalez 2006). Soil biodiversity can be influenced by all these types of invasive species. Indeed, the application of a realistic strategy based on biodiversity to effectively fight this threat would need further studies.

→ IMPACTS ON CHEMICAL ENGINEERS

The strongest impact on chemical engineers comes from invasive plants that have traits that differ from the resident vegetation (van der Putten, Klironomos et al. 2007). In most cases, such plants will be nitrogen fixing (Liao, Peng et al. 2008). For example, nitrogen fixing Acacia trees in Portuguese sand dunes can make use of local symbiotic nitrogen fixing microbes (Rodriguez-Echeverria, Crisostomo et al. 2009), which change the soil conditions by enhancing carbon and nitrogen stocks, as well as changing catabolic soil properties (Marchante, Kjoller et al. 2008). These changes may result in altered soil conditions that can promote the invasion (Marchante, Kjoller et al. 2008). However, in the case of invaders that do not have such particular traits, the effects observed on nutrient dynamics can also be neutral, or even negative (Ehrenfeld, Ravit et al. 2005; Liao, Peng et al. 2008). As plants have species-specific effects on the microbial rhizosphere community composition (Grayston, Wang et al. 1998; Kowalchuk, Buma et al. 2002), invasive plants will alter the relative abundance of microbial species in the soil. However, as all plants can do this, including native plants, those effects will have little impact on soil biodiversity.

Invasive plants can influence the community structure and the abundance of mycorrhizal fungi. For example, European crucifer plants reduce mycorrhizal inoculum, which can have negative effects on natural tree recruitment in North American forests (Stinson, Campbell et al. 2006), and impair the re-establishment of native grassland plants in Californian grasslands (Vogelsang and Bever 2009). Introduced, non native plants, such as the European forest understory forbs (*Alliaria petiolata*) in Canada, even if they are not invasive, are also known to suppress local arbuscular mycorrhizal fungi, thereby limiting natural forest regeneration (Stinson, Campbell et al. 2006). Suppression of native mycorrhizal fungi by invasive plants may be due to their selectivity; some invasive plants only become infected with a small portion of all native mycorrhizal species present (van der Putten, Kowalchuk et al. 2007). For example, St. John's Wort in North America has lower mycorrhizal dependency than populations of the same species in Europe (Seifert, Bever et al. 2009). Interestingly, such effects on arbuscular mycorrhizal fungi have been reported mostly from outside Europe; whether they also occur in Europe is not well known.

Introduced invertebrates and vertebrates can alter the soil microbial community and their functioning. For example, in a large number of long-term exclosure experiments in New Zealand, it has been shown that the introduction of large grazers in nature reserves influences vegetation development and soil nutrient dynamics (Wardle, Yeates et al. 2001). In another case, introduced rats killed shore birds that nest on the ground. In rat-free islands, these birds transfer nutrients from sea to land, which stimulates soil microbial activity. On islands with rats, the stimulation of microbial activity was strongly reduced (Fukami, Wardle et al. 2006). In Europe, soil microbial activity can also be reduced by invasive New Zealand flatworms. As these flatworms are predators of earthworms, they indirectly reduce microbial activity. Whether or not they reduce microbial diversity is not known (Boag, Yeates et al. 1998).

→ IMPACTS ON BIOLOGICAL REGULATORS

Plant invasions may be favoured by the release of pathogen control in the introduced range. Indeed, in natural conditions, root herbivores and soil pathogens are important regulators of spatial and temporal changes in the composition of natural vegetation (Yeates, Wardle et al. 1999; Wardle 2002; Bever 2003; Van der Putten 2003). Evidence shows that soil pathogens and root herbivores play important roles in controlling plant abundance (Klironomos 2002), plant species diversity (Packer and Clay 2000; De Deyn, Raaijmakers et al. 2003), and vegetation succession (Van der Putten, Van Dijk et al. 1993; De Deyn, Raaijmakers et al. 2003). Root-feeding insects may cause dramatic decline of plant populations (Blossey and Hunt-Joshi 2003). In contrast, effects of root-feeding nematodes vary from marked, generalised reduction in plant production (Stanton 1988), to localised damaged patches, and their effects may depend on interactions with, for example, pathogenic soil fungi (De Rooij-van der Goes 1995; Van der Putten and Van der Stoel 1998).

In their introduced range, invasive plants have fewer pathogens, parasites or viruses than in their natural range or than other native plant species around them. Evidence is rising that this may be due to escape from soil pathogens. For instance, (Levine, Vila et al. 2003), overall invasive exotic plants tend to have less nematodes than related natives (van der Putten, Yeates et al. 2005). For example, Marram grass, which has been intensively planted for sand stabilisation in European coastal dunes since the 19th century, has eight species of plant parasitic nematodes in the central part of its range (coastal fore dunes in north-western Europe). However, this number drops towards the extreme parts of the range, in the Mediterranean and in southern Sweden and northern Germany. Interestingly, marram grass is more abundant and persistent in non-native dunes than in stabilised dunes of its natural range. This seems to be because in its new range, marram grass still has a number of generalist root-feeding nematodes, whereas the specialists, such as cyst nematodes and root knot nematodes, are absent (van der Putten, Yeates et al. 2005). However, other studies have shown contrasting patterns, with lower densities of root-feeding nematodes in the invaded range compared to the native range (Virginia, Jarrell et al. 1992), higher richness of nematodes on the invasive weed *Tradescantia fluminensis* compared to areas without this invasive weed (Yeates and Williams 2001), or similar densities of root-feeding nematodes, shared among the invasive and the native plant species (Knevel, Lans et al. 2004).

There have been very few reports on the effects of invasive species on microarthropods. Microarthropods seem to be influenced most by invasive earthworms (Migge-Kleian, McLean et al. 2006), but such examples are quite rare in Europe. Invasive exotic plants can also influence microarthropod communities when

they produce different litter than the native plant species. For example, the Japanese stilt grass that invades south-eastern United States has more phosphate-rich litter than native plants and this enhances mite abundance, which reduces other microarthropods (McGrath and Binkley 2009). Also in other examples invasive plants reduced microarthropod communities (Pritekel, Whittemore-Olson et al. 2006). Microarthropods from warmer climate regions invade the Antarctic, partly due to human traffic, e.g. intensified tourism (Hugo, Chown et al. 2006; Sinclair, Scott et al. 2006).

Overall therefore, it appears that soil biological regulators populations tend to be reduced by invasive species. This may actually favour the invasiveness of some introduced plants, which find themselves released by their plant pathogens and root–herbivores. Although these effects may be substantial, some native plants may in fact have comparable effects (De Deyn, Raaijmakers et al. 2004).

→ IMPACTS ON SOIL ECOSYSTEM ENGINEERS

Probably the strongest effect on ecosystem engineers reported thus far is that of invasive New Zealand flatworms in the UK, which are a predator of indigenous earthworms (see also the impact on chemical engineers). Through their effects on native earthworms, invasive flatworms may have large effects on soil microbial communities and their activities, as well as the effects on soil moisture dynamics, soil properties and plant community composition (Boag, Yeates et al. 1998). Moreover, as earthworms can control plant enemies, such as nematodes (Blouin, Zuily-Fodil et al. 2005), invasive flatworms could also have an impact on agricultural crop protection. This may also stretch to plant-aboveground insect relationships, which can be controlled by earthworms (Wurst, Dugassa-Gobena et al. 2004; Wurst, Langel et al. 2005).

4. 6. 2. CURRENT AND FUTURE TRENDS

→ SOIL ORGANISMS CONTROL PLANT INVASIONS

Soil organisms are a key factor in controlling plant abundance. When plants run out of this control, they can become invasive (Van der Putten 2003). This applies to both classic invasions (Klironomos 2002) and to global warming influenced invasions (van Grunsven, van der Putten et al. 2007; Engelkes, Morrien et al. 2008). Plants that are spreading faster, or further, than their natural soil pathogens can become invasive in their new range. An example is black cherry, which was planted in the north-western European forest. In North America, this tree is controlled by native soil pathogens (Packer and Clay 2000), but this control is absent in Europe, as the trees have been introduced but not their pathogens (Reinhart and Dollahon 2003). However, very few of the introduced plant species really become invasive. The estimate is that one to ten of every thousand introduced plant species is becoming invasive (Williamson 1996). Probably, in many of the exotic plants that do not become invasive, soil biodiversity contributes to plant control by root herbivores or soil pathogens that switch from native plants to the invaders. In that respect, soil biodiversity serves as a reservoir of potential enemies against invasive plants. This is crucial, as the control of invasive plants costs the European Community billions of Euros on an annual basis.

There have been biological control programmes involving release of root-feeding insects, for example to control Knapweed (Centaurea maculosa) in the USA (Clark, Brown et al. 2001; Clark, Brown et al. 2001), although such introduced enemies also

can exert negative effects on other plants (Callaway, DeLuca et al. 1999). However, no programmes involving the release of soil pathogens or root-feeding nematodes are known in order to control plant invasions; the few plant-feeding nematodes tested parasitize aboveground plant structures (Robinson, Orr et al. 1979).

→ GLOBAL CHANGES AND INVASIONS

The incidence of biological invasions is increased by other large-scale changes in the environment. Urban areas are a major source of invasive species and they also can act as stepping stones for plant range expansion under climate warming; because of the relatively high temperatures in cities, many plants from warm regions can survive there and they may break out when temperatures in the surroundings of the cities increase. Other disturbances, for example due to land-use change, open up possibilities for invasive plants (Ward and Masters 2007).

It will be impossible to control all exotic species. Most of them, such as the large number of exotic garden plants, have not yet made it to invasive exotic weeds. High soil biodiversity will enhance the chance that potential control organisms are present in the soil from the invaded range and that this contributes to biotic resistance against the invasive plants. What would be far more risky is to consider the introduction of pathogens from the native range, as they may switch to other hosts and reduce the performance of native plant species. This may become an important subject when plant range migrations towards previously colder regions will take place.

The consequence of successful range-expansions of plants as a consequence of the current global warming for plant invasiveness can be massive. Thus far, there have been some first studies showing that these range expansions indeed can make plants escape from their native soil pathogens (Van Grunsven, Van der Putten et al. 2009). These developments need to be carefully studied, in order to assess their possible consequences for nature management and biodiversity conservation.

→ PREVENTION

There are many strategies for controlling invasive species. However, very few, if any, make use of soil biodiversity, although it could provide an interesting source of local control mechanisms. Recent studies have shown that many native plants are controlled by native soil pathogens and release from these pathogens can enhance plant invasiveness in a new range. The possible strategy of using pathogens from the invaded range to control exotic plants clearly requires further studies. Nevertheless, this strategy, instead of introducing pathogens from the native range, would mean reducing the risks due to one problem (that of the invasive plant) by potentially creating another one (that of introducing an alien pathogen).

One way to prevent that potential invaders are introduced is through the limitation of imports on plant materials. A complication is that so relatively few species become invasive and that there is so little general predictability about which species are going to become invaders. There are some rules of thumb, for example that exotic plants forming many seeds, which grow on wasteland and that have become invasive in other areas with the same climate conditions, can become invasive more easily than slow growing, fast reproducing species from undisturbed habitats. In the near future, more and more States will come up with import limitations, but such permits only operate well when executed at the scale of the whole Europe.

4.7. MANAGEMENT PRACTICES

Soil management practices have important and sometimes immediate effects on soil biodiversity and the resulting ecosystem services. Often, the impacts of land management are organised into syndromes. For example, intensively managed land is often fertilized by mineral fertilizers, intensively tilled, heavily compacted during harvesting, and cropped with a limited variety of crop species. This results in high pest and pathogen pressure, which requires regular spraying with biocides, as well as weed eradication by herbicides. These conditions are unfavourable for carbon storage in the soil, they enhance the risk of leaching of nutrients and reduce the water holding capacity, which pollutes ground- and surface water and which enhances the risk of flooding downstream during and after heavy rainfall. Therefore, management practices should aim at diminishing these accumulations of disturbances (e.g. in the case of intensive agriculture), as well as provide opportunities for enhancing the resilience of soil ecosystem services by conserving soil biodiversity. These activities may lead to what can be called sustainable management practices. Given that management practices are typically applied by farmers, who have long-term and influential contacts with the land, their interest and motivation in addressing the threats to soil biodiversity will have a strong influence on the maintenance of this diversity.

The main mechanism explaining the changes in soil biodiversity with increased intensification of management practices is linked to organic matter input. Organic matter drives the soil foodweb, and depending on the type, it will drive bacteria- (low C:N) or fungi- (high C:N) dominated food webs (Box 18). Greater litter inputs in grasslands encourage fungal-dominated microbial communities (Yeates, Bardgett et al. 1997), and a greater diversity of nematodes (Wasilewska 1994; Yeates, Bardgett et al. 1997) and microarthropods (Siepel 1996). The enhanced microbial activity may also enhance the biological regulators, and thus reduce nematode and soil pathogen incidence (Freckman 1988; Griffiths, Ritz et al. 1994). In contrast, in intensively managed (fertilised) grasslands or croplands, microbial communities are depressed (Lovell, Jarvis et al. 1995) and shift to opportunistic bacteria-dominated communities (Bardgett, Frankland et al. 1993; Bardgett and Leemans 1996). In turn, this tends to favour opportunistic bacteria-feeding fauna. Soil tillage practices disturb fungal hyphae and the larger earthworm species that visit the soil surface to obtain plant material for food, such as anecic earthworms (Emmerling, Schloter et al. 2002). Biomass and abundance of anecic earthworms are reduced by a factor of 1.3-3 in conventionally managed soils when compared to organic management types (Pfiffner and Mader 1997; Siegrist, Schaub et al. 1998; Mader, Fliebbach et al. 2002). Conventional management also results in poorer soil aeration and soil drainage.

These trends are illustrated in the following figure, where extensive management represents an intermediate state between organic (biological) and intensive management.

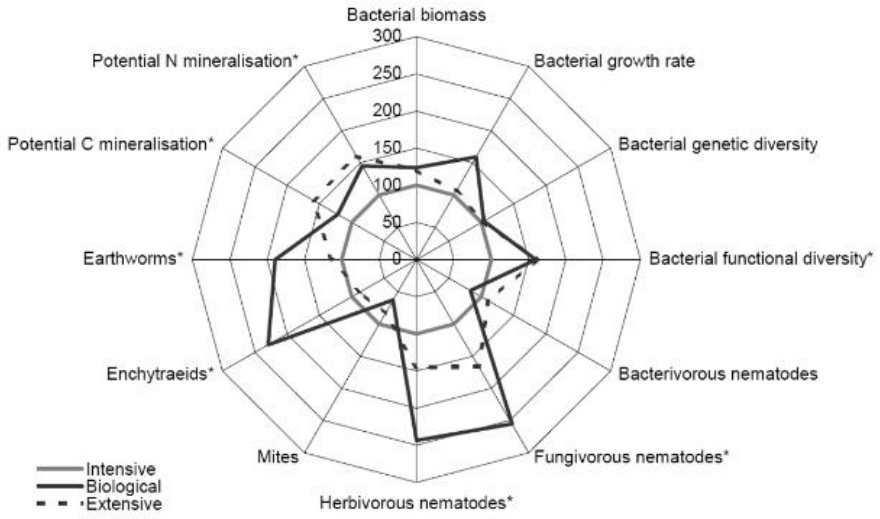

Figure 4-16: Soil biotic variables in biological, extensive and intensive grassland farms on sand. Intensive is set to 100%; * indicates statistically significant difference (p=<0.05) between categories (Bloem, Schouten et al. 2003)

4. 7. 1. SOIL MECHANICAL FARMING PRACTICES

→ MULCHING / LIGHT SOIL SEALING

Mulching consists of covering the soil surface to protect against erosion and to enhance its fertility. Mulch is usually applied towards the beginning of the crop growing season, and may be reapplied as necessary. It serves initially to warm the soil by helping it retain heat and moisture. A variety of materials can be used as mulch, including organic residues (e.g. crop residue, hay, bark), but also manure, sewage sludge, compost, rubber or plastic films. Mulching limits weed growth while conserving soil moisture and moderating soil temperature, virtually eliminating erosion.

Organic mulches are also a source of nutrients for the soil when they break down, thereby stimulating chemical engineers activity (Bush-Brown 1996). In an experience of wide extension trials using manure compost in Australia, the first year of mulching application led to 70% mineral fertilisers savings (Moral, Paredesa et al. 2009).

In addition, mulches can improve soil architecture by providing pore spaces which can support fungi and root growth, serve as habitat and refuge from predators for biological regulators and soil engineers. In the long-term, mulching systems favour anecic and epigeic earthworms (they can have a biomass up to 12 times higher in mulch cropping systems compared with conventional systems), whereas organic and conventional farming systems are favourable to endogeic earthworms (Pelosi, Bertrand et al. 2009).

It is critical that mulches are used that do not contain contaminants. Other options, for example *in situ* mulching where plant residues are left in the field instead of burned, should be done such that it does not promote re-infestation sources of pathogens.

→ **APPLICATION OF ORGANIC RESIDUES (COMPOST/ MANURE/ SLUDGE)**

Application of animal manure, sludge or other carbon-rich wastes, such as coffee-berry pulp or compost, improves the organic matter content of the soil. For agricultural purposes, it is usually better to allow for a period of decomposition of the organic residues before applying them to the field. This is because addition of carbon-rich compounds immobilises available N in the soil temporarily, as micro-organisms need both C and N for their growth and development. Composting is precisely the recycling and transformation of organic material (usually from plant residues), into humus form to improve agricultural production.

Organic waste on crop fields increases the food available to chemical engineers and soil engineers. Thus soil structure is stabilised and soil organic matter decomposition improved (Six, Bossuyt et al. 2004). In India, introduction of organic compounds derived from composted urban organic wastes increased the biomass of earthworms by a factor of four (from 4 g/m² to 18 g/m²) and other macroinvertebrates by a factor of five (from 3 g/m² to 16 g/m²) (Senapati 2000).

The different types of organic residues do not all have the same impact on soil fauna. The density of soil macrofauna was higher in farmyard manure (FYM) and in municipal solid waste compost (MSW) than in control soil with no organic inputs (C), biowaste (BW) and green waste composted with sewage sludge. Both manure and solid waste compost include much air space, suggesting they help create good habitat conditions for macrofauna.

Application of organic residues is a cheap, efficient and sustainable treatment.

4. 7. 2. CHEMICAL TREATMENTS

→ **PESTICIDES**

Pesticides are used as a prevention or remediation measure against crop pests and diseases. Most of the chemical treatments are preventive, with seeds treatments (fungicides, insecticides, birds' and wild boars' repellents) and periodic preventive treatments (herbicides, fungicides, insecticides). One-off curative treatments allow avoiding sudden pest invasions, with a locally high chemical spread locally destroying soil biodiversity. Curative treatments are applied among developed and fragile crops, and are more constraining and expensive to farmers, who tend to limit their usage and opt for a more preventive management of pests.

All pesticides, whether applied directly, or targeted at the aboveground parts of the plant or the pests, are likely to end up in the soil and in contact with soil organisms. Their effects are highly variable, depending on the type and amount of pesticide, soil environment, and biotic group considered. Generally however, the impacts are not restricted to the target but can have disruptive effects on the entire biological regulatory capacity of the soil community.

→ **FERTILISERS**

Mineral fertilisers are chemical compounds usually applied through the soil to promote plant growth by providing plants with the necessary nutrients for their growth. They can be organic (manure, compost) or inorganic and produced industrially from chemicals. Nitrogen fertilisers are the most common form of fertilisers used in Europe.

High levels of some inorganic nitrogenous fertilisers provide chemical engineers with easy to use nitrogen, thereby boosting their activity. This increases the rate of decomposition of low quality organic inputs and soil organic matter, resulting in the continuing decline of soil organic matter content which, ultimately, results in loss of soil structure and waterholding capacity. In addition, inorganic nitrogen fertilisers also result in the presence of high ammonium concentrations, that stimulate nitrification (Box 7), resulting in excess nitrate leaching from the soil and contaminating water tables.

4. 7. 3. CROP MANAGEMENT

→ CHOICE OF THE CROPS SPECIES

The choice of the cultivated crop is important as it defines the kind of habitat available to soil fauna. For example legumes can act as natural fertilisers, improving the nitrogen concentration in soil, thanks to the symbiotic relationship they establish with Rhizobia (Box 1).

The growth rate and the yield of the crop also determine how much soil and soil fauna are impacted by agricultural practices. Rapid growth crops and high-yielding crop varieties like maize or fast growing energy crops demand an amount of energy and resource from soil, which then need to have time to recover. Such soil recovery cannot occur if the following crop is another rapid growth or high-yielding crop, such as maize, or elephant grass, which is used for biofuel production. In that way, successive cycles of fast-growing/high yield crops will result in a depletion of soil organic matter and thus of the soil fauna which feeds on it with all negative consequences for soil structure and the related ecosystem services.

→ CROP ROTATIONS

Crop rotations are used to counter the negative effects of monocultures, which end up draining nutrients from the soil, as the same crops are grown year after year by varying the crops in a given field. Crop rotations can also help avoid the build-up of pathogens and pests, as the alternation of crops modifies the associated communities of biological regulators. For instance, crop rotation often involves the replenishment of nitrogen through the use of leguminous crops in sequence with cereals. A common form of rotation is a three year cycle, where wheat is grown the first year, leguminous plants in the second year, effectively turning the field into a pasture, and finally the land is left to rest (fallow) in the third year. Long-term studies have shown that such management practice generates great variations of the soil carbon level and total soil nitrogen, depending on the period of the rotation. Soils have higher carbon levels in pasture lands and pasture lands which were previously cereal fields than in permanent cereal fields (Boellstorff 2008). Continuous leguminous cropping can increase soil carbon storage and total soil nitrogen by up to 20% in the 0-15 cm soil depth compared with rotation including cereals (Bhattacharyya, Prakash et al. 2009).

4. 7. 4. LANDSCAPE MANAGEMENT

→ HEDGEROWS AND GRASSY FIELD MARGINS

Establishing hedgerows or grassy strips at the edge of arable fields are commonly used methods in Europe. They offer a stable habitat, food, and a protective environment for soil fauna next to the intensively managed fields. For example, six metre wide strips

increase the number and variety of species such as earthworms, beetles, and various biological regulators, resulting in improved soil fertility, and possibly improved pest control. These strips can be placed around fields, so-called field margin strips, but they can also be installed across fields, so-called beetle banks. This far, such ecological islands have been considered mainly for aboveground purposes, such as the promotion of natural enemies of aboveground plague insects. Hedgerows are even more favourable to soil organisms, in particular biological regulators, than grassy field margins, however, due to their low mobility; the soil organisms will have only limited dispersal into the fields. That also counts for field margins, in which 10% of the soil-dwelling species present in farmland were found to occur exclusively. This makes these habitats important sources of biodiversity (Smith, Potts et al. 2008), albeit that their effects on soil biodiversity in the adjoining fields will decline sharply with distance, due to poor mobility of the soil biota.

4. 7. 5. TOOLBOX OF MANAGEMENT PRACTICES

Some soil management practices can be used to restore soil functions, through the action of soil fauna. Some examples are presented in Table 4-7 for a series of characteristic soil problems. A (+) indicates a practice which can have a positive impact on remediating to the problem and a (-) a practice with no impact on the problem. For instance crop-rotation helps pathogen build up, which reduces the need for pesticides; it may also improve carbon storage and thus soil structure.

Table 4-7: Soil biological problem and remediation role of different management practices. Legend: + = Positive effect; - = No effect; +/- = Intermediate effect

	Poor structure	Low SOM	High pesticide levels	High salinity	High pollutant levels
Bioremediation	-	-	+	+	+
Compost	+	+	+/-	+	+/-
Manure	+	+	-	+/-	-
Crop rotation	+/-	+/-	+	+	-

4. 7. 6. CONCLUSIONS

In this section, major threats and the soil degradation processes have been presented. However, it is important to highlight that, in reality several threats can interact with each other. For instance, the effects of a chemical pollutant (or a mix of chemical pollutants) will have even stronger impacts on a soil subject to land use change that triggers one or more soil degradation processes (e.g. organic matter depletion). It is known that a high soil biodiversity *per se* may help deal with interactive threats and increase the resilience of soil. But to date there is a clear need for further studies on those potential interactions (e.g. how climate change can influence the impacts of chemical pollution).

Moreover, large amount of work exist on the impact of a specific threat on an individual species or group of organisms, but studies to better understand the impacts of threats to specific functions and related services are still lacking. This is especially true for certain threats, such as GMOs. Additional research on this issue will help in answering the question if it would be better to preserve biodiversity *per se* or if it is possible to preserve specific biodiversity in priority.

Finally, it would be useful to determine the optimal scale of action (e.g. species, patch or landscape scale) to protect soil from deleterious impacts. This of course will depend on the considered threat. Thus, for climate change the scale of action is global, while for chemical pollution it can be local. When we protect soil biodiversity at a spatio-temporal scale targeting a specific functional group, we should also consider how the other functional groups, at lower or higher spatio-temporal scales, and their related services, are affected.

→ **MAIN RESEARCH GAPS**

- **Impacts of climate change on soil ecosystems, biodiversity and related functions, including impacts of altered precipitation rate, not limited to temperature**
- **Interactions between threats, and within the same class of threats, between similar deleterious factors (e.g. pollutants)**
- **Interactions between pesticides and biotic factors**
- **Research into the effects of pollutants on soils**
- **More research on the impacts of threats on soil functions and the services delivery**
- **Determination of the optimal scale of action, depending on the threat**
- **Research on the impacts of land-use changes taking into account the surrounding landscape**
- **Research on the sensitivity of decomposition rate to humidity**

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5. INDICATORS AND MONITORING SCHEMES FOR SOIL BIODIVERSITY

Establishing the state of soil biodiversity requires the use of reliable indicators and the setting up of monitoring schemes -ongoing research is contributing to these two elements. Examples of research networks involved in soil biodiversity are presented in Section 7.

Much work has already been done in Europe on indicators of soil biodiversity and quality. A rather large number of papers and books show the usefulness of most soil organisms as indicators of soil quality and/or describe their responses to different effects (Paoletti 1999). The BIOASSESS project (1999-2002) produced a significant amount of work on the response of soil invertebrate communities (Collembola, earthworms and macrofauna) and other elements of biodiversity (plants, butterflies, birds and Carabidae) to landuse intensification. The ENVASSO (2006-2007) project addressed threats on soil biodiversity and proposed a set of minimal indicators based on their significance (based on sound science), acceptance of the methodology (existence of standardised methods), measurability and costs (Huber 2008). This section provides a synthesis of this work and describes the monitoring schemes that different European countries have developed.

5.1. INDICATORS

5. 1. 1. USEFULNESS AND SELECTION OF INDICATORS

→ USEFULNESS OF INDICATORS

Why indicators

Assessing the actual and predicting the future performance of ecosystems that are or may be influenced by human activities is essential to ensure a sustainable development. Changes in the environment (either natural or human induced) need to be highlighted as soon as possible, and their impact on the short and long term assessed, in order to predict the consequences on soil functioning and ecosystem services delivery. Thus managers and policy makers need tools to estimate and class the risk linked to ecosystem degradation and management practices in a holistic way.

In this context, there is a need for assessment tools that can capture the trends in biodiversity and ecosystem services. Unfortunately, direct measurements are often impossible to perform, due to methodological problems or practical reasons of cost and time. Simulation models which are developed as an alternative to direct measurements are also often highly impractical (Bockstaller and Girardin 2003). Therefore, there is a need for indicators to assist us in establishing baseline conditions and trends. Indicators also allow to establish threshold effects and to know the acceptable level of pressure exerted on soil.

Indicators are a way of presenting and managing complex information in a simple and clear manner. Essentially, ecological indicators have two main functions: an informative function, i.e. to decrease the number of measures and parameters that would normally

be required to represent a complex situation (e.g. an agro system), and a decision-aid function, to simplify the communication process through which information is conveyed to final users and to help achieve the initial objectives, (e.g. the sustainability of a farming system).

The development of indicators at the national, regional or local field level has become a common approach to meet the crucial need for assessment tools.

Value of indicators

Two levels of information may be distinguished when using indicators: **headline indicators** that provide an overview of the situation at a high level of aggregation and **detailed indicators** that are needed to better understand underlying trends or existing links between policy measures and their effects. The challenge is to find an appropriate balance between simplification and completeness.

Indicators can serve different purposes. Many indicators are not aimed at being used to predict an actual impact, but to supply information about a risk or a potential effect (Halberg 1999). Indicators can also inform policy makers about the progress that is being made towards achieving a policy objective (Vos 2000), inform about specific questions, focus research, provide a basis for discussion and act as a communication tool. The role of the indicator may be to signal positive movements as well as negative ones. Others are also aimed at 'raising the alarm', meaning they should give information on negative impacts before they actually occur (Reus, Leenderste et al. 1999). They can be also a useful tool for prioritising which environmental information is most useful as part of an environmental information system.

Each purpose may be associated to a different target group. Different stakeholders have different information needs, and different indicators have to be developed to answer their specific needs. It does not mean that new monitoring needs to be developed, only that the data may be interpreted differently for different purposes.

Scale issues

Many indicators relating to some aspect of biodiversity exist but none of them capture biodiversity in its entirety. Despite the need to agree and implement a method for measuring biodiversity status, no scientific consensus measure exists. The main difficulties in establishing operational indicators are due to the multidimensional nature of biodiversity which can be defined in terms of composition, structure and function at multiple scales (Noss 1990). For instance, while indication methods have been proposed that combine a number of factors related to biodiversity status (Jenkins 2003; Scholes and Biggs 2005), these methods allow for comparisons on large changes on the global biodiversity between different environments but may be insensitive to diffuse impacts like for instance the long term effects of habitat fragmentation, climate change or pollution.

The main challenge of indicators is to capture the variety of spatio-temporal scales over which environmental changes occur. The extrapolation of process measurements from one scale to another is extremely difficult and can lead to substantial errors. These extrapolations are generally based on the development of suitable models: sufficient information is usually available for one of the scales but extrapolating to the global level requires the use of modelling. Many of the problems with extrapolation stem from the presence of nonlinear relationships between processes and scale (King 1991; White and Running 1994). Some processes may operate only at certain scales, becoming redundant at other scales. The recognition and verification of the domains

within which each process operates represent an important research area attached to environmental monitoring. However, for management purposes and political decision making, it is important to consider a number of spatial scales. Indicators and most of the management activities should be planned in the context of the landscape level at least.

→ SELECTION CRITERIA

To measure soil biodiversity, many different aspects need to be assessed, which requires the use of a set of various indicators. As a result, investigators have tried to design comprehensive indicators that combine a number of indicator parameters such as individual densities of indicator species or physico/chemical soil parameters. But for reasons of efficiency, data quality and repeatability, the number of indicators should be limited. Thus, the aim is to select the minimum set of indicators that adequately characterise soil biotic properties.

The choice of these indicators varies across a range of temporal and spatial scales and can be based on the following criteria:

- **Meaningful**: indicators must relate to important ecological functions and use good surrogates (e.g. recognised high value organisms as indicator groups). This ensures the indicators will serve their purpose accurately, i.e. monitor trends in soil biodiversity.
- **Standardised**: the selected parameters should be readily available and (almost) standardised. This ensures the comparability of data among sites.
- **Measurable and cost-efficient**: the selected parameters must be easy to investigate in the field and to sample, affordable, and must not be restricted only to experts or scientists, but should also be assessable by interested public (e.g. citizens). This ensures the indicators will be used in practice, and can be routinely collected.

Other relevant criteria for the selection of core set of indicators that accommodate environmental agencies and management practices needs as well as environmental experts, have been put forward (EEA 2005) Convention on Biological Diversity, Montreal, 2003)[40]:

- **Policy-relevance**: the selected parameters should be sensitive to changes at policy-relevant spatio-temporal scales, enable to capture progress towards policy targets, and allow for comparisons between a baseline situation and a policy target.
- **Spatio-temporal coverage**: the selected parameters should occur in the different soil types and land uses, e.g. at natural and managed sites. They should also be amenable to aggregation or disaggregation at different spatial scales, from ecosystem to national and international levels.
- **Understandability**: the indicators should be simple and easily understood (avoiding contradictory messages)
- **Accuracy**: the value of the indicators should reflect precisely and robustly the changes they monitor.

Indicators must be selected depending on the question to answer. Thus, despite the fact that the majority of criteria usually cited for the selection of appropriate indicators

[40] www.biodiversity-chm.eea.europa.eu/stories/STORY1068016983; last retrieval 16/09/2009

are suitable for every situation, the priority given to one or another aspect largely depends on the assessment purposes and the endpoint users.

5. 1. 2. MEASURING SOIL BIODIVERSITY

A huge number of methods exist to measure the activity, biomass and biodiversity of soil organisms. Some methods directly count the number of species and individuals present in a sample to calculate diversity, while others are based on a community approach, and rather estimate the activity of soil organisms, or of specific functional groups. In the past few years, considerable efforts have been made towards the standardisation of some methods. A working group of the ISO Technical Committee 190 Soil Quality reviewed appropriate candidates and proposed five methods for inclusion within the working programme, covering the main classes of soil invertebrates.

→ **ABUNDANCE AND DIVERSITY**

The methodology used to estimate species diversity varies depending on the soil organism considered. The largest organisms are directly observable with the naked eye or with a microscope, while the presence of the smallest can only be estimated by complex molecular techniques.

Microorganisms

The diversity of microorganisms (including bacteria, fungi, but also protists) can be estimated through either of two approaches: cellular cultures and molecular biology methods.

- **Cellular cultures:**

Cellular cultures are used to encourage the controlled growth of microorganisms under laboratory conditions (e.g. in incubators or flasks containing appropriate growth medium). The main drawback of this method is that it is a selective protocol favouring the growth of some species compared to others. The proportion of cells that can currently be cultured is estimated to be only between 0.1% and 10% of the total population in a given soil sample. As a consequence, cellular cultures only reveal a subset of the original soil microbial community. Depending on the considered microorganism, the cost of these methods can vary, but it is in general lower than for molecular biology methods. An incubator capable to keep a constant temperature and CO_2 pressure cost 1000-2000 Euros and an appropriate growth medium cost 10-100 Euros per litre.

- **Molecular biology methods:**

More recently, several methods based on molecular biology have been developed to characterise the genetic information contained in the DNA and RNA of microbes or other soil organisms. The main advantages of these methods are:

- They can be applied directly on the nucleic acids extracted from the soil samples
- They give very precise information regarding the genetic structure and the diversity of soil communities
- They provide the information even if the genetic material is present in small quantities because the genetic material can be amplified (e.g. by PCR - Polymerase Chain Reaction).

- It is possible to amplify specific portion of the genetic information (e.g. genes) thus obtaining selective information
- It is possible to follow the evolution of soil communities under different environmental conditions

The main disadvantages are:

- There are no standardised DNA extraction procedures and the efficiency may vary depending on the nature of soil sample. Indeed, the DNA extraction from soil samples is currently under standardisation at ISO level (ISO/CD 11063 'Soil quality – Method to directly extract DNA from soil samples').
- The efficiency of the amplification of genetic material depends on the genetic sequence and on the experimental conditions

Other molecular methods are based on fatty-acids analysis (this process is also in the process to be standardised at ISO level: ISO/NP TS 29843-1&2). Phospholipids are signatures of specific groups of microorganisms, such as the different types of bacteria involved in nitrogen cycling. Although this type of analysis does not allow for identification at the species level, it provides an overview of the community as a whole, and of specific functional groups of microorganisms. Some of these methods can be quite labour-intensive and time consuming.

Thus, methods based on molecular biology can be viewed as a useful complement to culturing methods to measure the microbiological diversity of soil (relative and absolute diversity). The cost of molecular biology methods can vary depending on the chosen method, ranging from a few hundred Euros (in the case of simple DNA extraction) to tens of thousands of Euros (in the case of quantitative DNA amplification with a real time PCR cycler).

Soil invertebrates

The diversity of soil invertebrates can be estimated directly, through direct sampling. The number of species, their abundance, as well as community composition can be assessed in that way. These estimates can be combined to calculate diversity in various ways, depending on the importance given to changes in community structure. Different methods can be used to measure the abundance and diversity of soil invertebrates according to taxonomic and functional groups (biological regulators and ecosystem engineers). The measure of invertebrate abundance and diversity is often used for monitoring purposes. For example, the Maturity Index (Bongers 1990), based on the composition of nematode communities, measures the abundance and diversity of soil nematode communities in relation to soil health. However, the results are often not comparable because sampling methods and study designs can considerably differ. For this reason several methods have been standardised under the ISO classification (e.g. ISO 23611-2: Hand sorting and formalin extraction of earthworms).

→ MEASURES OF BIOMASS AND ACTIVITY

Biomass

The idea behind this indicator is that soil microbial biota can be treated as a single entity. Estimation of total soil microbial biomass or biomass of some specific groups can be measured with many methods, such as fumigation. The biomass of soil invertebrates can be estimated based on their abundance and species numbers, and can be pooled over species.

Activity

The importance of the measurements of microbial activity in soil is rather neglected. One reason for this is that it is often difficult to interpret changes in microbial activity. For example, the presence of organic pollutants may have many direct and indirect effects on it. Moreover, resistant microorganisms could grow using debris of sensitive microorganisms and enhance their activity (Nannipieri 2002). Due to the high number of soil biodiversity functions, various methods have been developed to cover soil functional diversity. Most of these methods determine microbial activity. The soil biological activity is determined by measuring the amount of chemical products (e.g. CO_2) generated or disappeared by the overall soil community or by specific populations. In particular it is possible to measure:

- Soil decomposition rates through measuring the rate of organic residue consumption
- Soil respiration rate through measuring the CO_2 production
- Soil nitrification rate performed by specialised bacteria
- Soil enzymatic activity[41]

In addition, some integrative methods are used to measure functions performed by the soil biota as a whole. For instance, the litter-bag method measures the mass loss of organic matter, while the bait-lamina[42] test measures the feeding rate of soil invertebrates. However, their use is limited by the lack of appropriate negative control (Gardi, 2009).

5. 1. 3. INDICATOR POTENTIAL OF THE FUNCTIONAL GROUPS

→ CHEMICAL ENGINEERS

The highest amount of soil biotic capital is in the form of microbial biomass (bacteria and fungi). The absolute and relative amounts of these organisms give a good indication of the activity and stability of the ecosystem. Moreover, some of their measurements are automatised and standardised, which means they can easily be performed on large scales.

→ BIOLOGICAL REGULATORS

Biological regulators are very abundant, and the area covered during their life cycle is typically representative of the site under examination. Thus, their life histories permit insights into soil ecological conditions and, several species have already been recognised as useful biological indicators of soil quality.

Nematodes are present in high numbers, have high species diversity, and their relative or absolute amount give good information on the diversity and stability of the ecosystem. Indeed, since they have a range of feeding habits, they react differently to a range of soil changes. For instance, productivity of the belowground system may be indicated by the number of bacteria-feeding nematodes. However, identification of nematodes at the species level is still time-consuming, technical and relatively

[41] An ISO standard is currently under development : ISO/DTS 22939 'Soil quality – Measurement of enzyme activity patterns in soil samples using fluorogenic substrates in micro-well plates')

[42] The bait-lamina test is a practical mean to assess soil faunal feeding activity. The test consists of vertically inserting 16-hole-bearing plastic strips filled with a plant material preparation into the soil.

expensive (e.g. depending on the distance to the sampling one should count at least 20-40 Euro per sample).

Springtails can also be considered good candidates, since they have high species diversity and respond to a variety of ecological and environmental factors, like changes in soil chemistry, micro-habitat configuration (e.g. soil moisture, litter quantity and quality), land-use practices (e.g. forestry and agricultural practices) and landscape composition (Hopkin 1997). Although species identification can be time-consuming and requires expert knowledge, the number of Collembola experts in Europe is rather large. Moreover, while taxonomic identification is needed for exhaustive biodiversity monitoring, surrogate taxonomic levels (e.g., genus level) can be used for rapid biodiversity assessments. For the evaluation of soil quality, a trait base classification can be adopted.

→ **SOIL ECOSYSTEM ENGINEERS**

These organisms are broadly accepted predictors of biodiversity in a large proportion of European soils, with the notable exception of large extensions of the Mediterranean areas and reflect their outstanding influence as soil engineers. They indicate the diversity and intensity of physical and chemical ecosystem engineering operated by soil invertebrates, and subsequent microbial activities (Lee and Foster 1991; Lavelle 1997; Pulleman, Six et al. 2005). Their activity conditions the habitat and activity of several chemical engineers and biological regulators, which also typically form their diet, so their abundance reflects an integration of a range of biological processes occurring in soils. Among ecosystem engineers, earthworms are the most frequently used indicator species.

Earthworms can be very abundant, but are not very diverse and easy to characterise and count. However, they can be appealing to a range of stakeholders:

- Farmers, who know that earthworms are important for soil structure and who know that more is better
- Conservationists, since earthworms are the main food source for many aboveground conspicuous species
- Ecotoxicologists, since earthworms are sensitive to pollutants

Moreover, earthworms are sensitive to soil type and secondarily to land use.

5. 1. 4. INVENTORY OF INDICATORS AND SUITABILITY

So far, no comprehensive index has been proposed that would combine all the aspects of soil complexity into a single formula and allow accurate comparison among sites and plots. Existing indicators comprise rather long lists of potentially relevant variables to be measured, although no general agreement has been reached on their interpretation (Doran and Zeiss 2000). Some attempts to combine groups of variables into indicators of soil biotic activity have recently been proposed. The different concepts that have been developed for soil protection may be basically classed into three main approaches:

- Shopping list approach, where a set of different soil parameters are assessed
- Benchmark approach, where the degree of deviation between reference situations and the actual measurements are evaluated
- Numerical approach, where synthetic indices are developed for the assessment of soil status

→ **SIMPLE INDICATORS**

Several indicators, based on individual organisms groups or taking into account the whole soil community have been used to characterise soil biodiversity (Table 5-1). These indicators are directly based on the different measures available (see section 5. 1. 2).

Concerning chemical engineers, for example, the characterisation of microbial communities has been mainly based on the determination of fungal or bacterial biomass (Beare, Neely et al. 1990; ISO 1997) or on functional variables (Table 5-1). Sometimes, indices are calculated based on microbial activity, to assess the values determined with respect to soil quality. Some examples are the quotient of microbial carbon in the biomass to organic carbon content (Cmic / Corg) as an indicator for carbon dynamics in soil (Kaiser, Müller et al. 1992); the metabolic quotient as an indicator of energetic efficiency (Insam and Haselwandter 1989); or the respiratory activation quotient as an indicator of the presence of contaminants (ISO 2001). The pattern of degradable carbon sources (BIOLOG Identification System[43]) is applied for the comparison of sites with respect to their microbial communities. Recently, efforts have been spent on using structural aspects for the characterisation of the microbial community diversity. Different molecular methods (Lukow, Dunfield et al. 2000) as well as the determination of single microorganisms or microbial groups using cell components have been successfully applied (Frostegard, Baath et al. 1993; Waite, O'Donnell et al. 2003). These methods usually have good measurability (Table 5-1), and some have been proposed for use in assessment systems (Mulder, Cohen et al. 2005).

The use of biological regulators or ecosystem engineers as indicators of soil quality has a long tradition (Volz 1962). Many invertebrate taxa have been proposed as indicators, including protists (Louisier and Parkinson 1981; Aeschht and Foissner 1991; Bobrov, Charman et al. 1999), nematodes (Bongers 1990; Mulder, De Zwart et al. 2003; Yeates 2003; Mulder, Dijkstra et al. 2005), Enchytraeidae (Beylich and Graefe, 2002)(see also Box 9), earthworms (Philipson, Abel et al. 1976; Beylich and Graefe 2002), and mites (Beck, WOAS et al. 1997; Behan-Pelletier 1999), springtails (Van Straalen and Verhoeff 1997; Fromm 1998; Filser, Wittmann et al. 2000; Filser, Mebes et al. 2002). Their abundance and diversity is typically measured, and these groups usually provide good indicators of changes in soil types and land uses (Table 5-1).

However, these approaches are all insufficient by themselves, as no single group can cover the huge variety of environments and soils. Accordingly, none of these group-specific indicators are routinely used. The information provided by the whole zoological community offers a better resolution of information (van Straalen 1998).

→ **COMPOUND INDICATORS**

In the last decades, a considerable number of compound indicators related to soil biodiversity or using concepts based on soil communities have been developed. However, these indicators have usually been developed with the intention to assess soil health status and to establish ecological soil classifications for the purpose of soil quality assessment, rather than with soil biodiversity assessment as an aim per se. As a result, these soil biodiversity indicators typically encompass multi-factorial aspects of

[43] The Biolog Identification System is a bacterial identification method that establishes an identification based on the exchange of electrons generated during respiration, leading to a subsequent tetrazolium-based color change. This system tests the ability of a microorganism to oxidize a panel of different carbon sources.

soil, including biotic to abiotic conditions, which makes them more meaningful indicators of soil biodiversity (Table 5-1). Each proposal has its own advantages and disadvantages. Most indicators are based on benchmarks, where soil biodiversity in the sampled site is compared to that in a reference, baseline site. The reference sites are typically defined based on expert assessments, and only the most recent integrated indicators propose more robust, objective assessments. Moreover, few indicators actually propose an integrated measure, that is easy to use and report, most are based on complex multi-factorial representations. The main compound indicators for soil biodiversity are detailed below.

Compound indicators: benchmark approach

In the early 1960s, a first proposition to characterise soils from a pedozoological point of view arose in Germany (Volz 1962). Biomass of some macro-invertebrate groups (earthworms, beetle and dipteral larvae, snails and slugs, isopods, myriapods, etc.) was used to characterise forest soils and to class them into different groups. The main problem of this proposal deals with the fact that the measurement of the biomass requires great effort, the lack of important groups of soil macro-invertebrates and that meso-fauna groups are underestimated. The proposal was widely ignored by scientist and authorities because at that time, nobody saw the need for a biological classification of the soil system (Breure, Mulder et al. 2005).

Another Dutch proposal was made in 1992 by Sinnige et al., 1992. It was based on the definition of ecotopes and the identification of their characteristic "soil fauna communities". In fact, only myriapods, ants, collembolans, enchytreids and earthworms were considered. Each factor describing a site or a soil was classified in three classes. This proposal do not consider quantitative data like abundance but the species spectrum and the priorities in soil protection at the moment of its publication were not in line with this kind of approach.

Graefe and Schmetz (Graefe and Schmelz 1999) proposed a classification soil system using enchytraeids and earthworms as indicators for a "typical community of saprophagous microorganisms and animals" by using numbers in analogy to plants. Species are classified according to moisture, pH, salinity and life strategy (depth in the humus profile, gradient occurrence and reproduction type). The measurement endpoints are species composition, abundance, frequency and characteristic species. Problems associated with this approach concern:

- the impossibility to associate most animal species with the categories for plants merely because their occurrence is determined by different factors,
- the difficulty to identify enchytraeids
- the fact that it focuses only on Oligochaeta, leading to a low differentiation between sites.

Table 5-1: Simple indicators of soil biodiversity. Meas.= measurability

Functional group	Organisms	Indicator	Method	Standard	Sensitivity to soil type	Sensitivity to land use	Meas..
Microbial Decomposers	Microorganisms	Biomass / activity	SIR, fumigation-extraction ATP concentration, initial rate of mineralisation of glucose	Yes Yes	Good	Good	Good
		Activity	Respiration rate/quotient/ratio, Nitrification, N mineralisation, C mineralisation Denitrification N-fixation Mycorrhizae (% of root colonised)	Yes Yes No No No	Good Medium Medium Good Good	Medium Medium Medium Medium Good	Good
		Enzymatic activity	Dehydgenase activity Other enzymatic activity tests: phosphatase, sulphatase, etc. Enzyme index	Yes No No	Good Good Very good	Good Good Very good	Medium Good
		Diversity	Culture-dependent methods: direct count, community-level physiological profiles Culture independent methods: fatty acids analysis, nucleic acid analysis	No No	Poor Poor	Poor Very good	Good Good (technical)
Biological regulators	Protists, nematodes	Abundance and Diversity	Culture-dependent methods: direct count (diversity index, functional or trophic diversity) Culture independent methods: fatty acids analysis, nucleic acid analysis	Yes	Good	Very good	Low (time, expertise)
	Microarthropods (springtails, mites)	Counting	Litter-bag technique (colonisation capacity) Soil coring	No	Good	Good	Low (time, expertise)
		Abundance and Diversity	Community composition, ecological groupings	Yes	Very good	Very good	Low (time, expertise)
Soil ecosystem engineers	Earthworms, isopods	Abundance Diversity	Species richness, diversity, evenness	Yes (ongoing)	Very good	Good	Good (low expertise, simple)

Soil Invertebrate Prediction and Classification Scheme (SOILPACS)

The Soil Invertebrate Prediction and Classification Scheme (SOILPACS, Weeks et al., 1998) was developed in the U.K. to assess ecological soil quality. The scheme is based on the approach developed for the assessment of fresh water quality (RIVPACS, River Invertebrate Prediction and Classification System), and is aimed at assessing the biotic differences among communities in stressed sites (e.g. polluted soils) vs. undisturbed sites. Thus the observed soil biotic communities of selected invertebrate groups (e.g., earthworms, spiders, isopods, collembolans) in a polluted site are compared against a benchmark, expected community at a reference site, not submitted to environmental stress. The main advantage of this approach is that it uses standardised sampling and statistical evaluation (TWINSPAN) methods.

However, it also presents several problems:

- There are a small number and low quality of reference data for soil organisms in the United Kingdom (UK). A great effort is necessary to create a reliable set of reference data
- Even when a site has never been polluted or degraded by human activities, 'natural' stresses may have a temporary impact on the observed fauna and thus influence the result obtained (lowering of the observed/expected ratio)
- The environmental stress responsible for the value of the observed/expected ratio of soil fauna for a given site cannot be clearly identified
- So far, the system has only been used for the assessment of heavy-metal contaminated sites in Wales.

Biological Indicator System for Soil Quality (BISQ)

In the Netherlands, a Biological Indicator System for Soil Quality (BISQ) was developed to assess soil quality based on the ecological status and ecosystem services provided by soil (Schouten, Brussaard et al. 1997). The system develops a distance-to-target indicator for soil biotic communities, based on the idea that, in a given area, the threat to ecosystem processes can be estimated by comparing the number of species in a functional group with its reference (undisturbed) area.

The indicator is based on ecological processes and biotic interactions. Five ecosystem services are considered:

- Decomposition of organic material
- Nutrient cycling
- Soil fertility
- Soil structure
- Stability of the biotic communities

A range of soil biotic variables (indicator values) are measured to reflect the functions responsible for those services. These include the abundance and diversity of earthworms, nematodes, micro-arthropods, as well as measures of microbial activity and biomass. These biotic characteristics are correlated to the abiotic conditions in the site of measure. The resulting data can be presented in graphical form, as the deviations of each indicator value from the benchmark. Alternatively, the indicator values can be aggregated into a single indicator, using the average factorial deviation of the biological reference value (Breure, Mulder et al. 2005).

One advantage of the IBQS or BISQ is that the soil quality concept is related to the concept of 'ecological status' and ecosystem services, and explicitly includes a broad

range of soil taxa and functions. Moreover, the benchmark approach describes real soil systems and the methods used to assess soil biological parameters constitute an interesting starting point to build a harmonised framework for sustainable land use.

However the IBQS or BISQ presents some weaknesses. Macro-invertebrate communities are under-represented. They are only represented by earthworms, which cannot be considered as representative of the whole soil macro-invertebrate groups and are too sensitive to changes in some soil parameters. In addition, the use of benchmarks to determine soil quality references may be useful but presents some severe limitations:

- Benchmarks are established on the basis of a set of existing locations. The presumed good ecological status is derived from these locations and is thus describing only a relative state (Rutgers, Schouten et al. 2009). Since benchmarks are used to provide guidance to improve soil quality, the land management goals are constrained to this relative ecological status and not to an absolute good status.
- Since the benchmarks are established using a limited set of sites that cannot be representative of the whole heterogeneity of soils and site deviations, it will be difficult to adequately define land-use management goals.

Biological Soil Classification Scheme (BBSK)

The Biological Soil Classification Scheme (BBSK) proposes a biological method to assess the function of soil as a habitat for soil organisms (Ruf, Beck et al. 2003). It also develops a distance-to-target indicator for soil biotic communities. This indicator is based on the principle that similar soils should have similar soil fauna. This means that it is possible to define a reference biotic community for each site. The reference community can then be used as a baseline against which the actual quality of a sampled site is weighed.

Thus, to calculate the BBSK indicator, reference sites are first selected based on qualitative expert assessments to define groupings of sites sharing similar land use, soil and climatic properties. The reference soil biotic communities are then defined for each reference site. Finally, the indicator measures the deviation between the reference and the actual sampled community, in a given site. The main advantage of this system is that it accounts for multiple biotic factors, by including diversity and abundance of meso- and macro-fauna. However, these data are not integrated, which makes them difficult to interpret by non experts. For instance, the user has to decide what it means if the activity in the sample site is e.g. half that of the reference site for one type of organism but not for another.

Moreover, the sensitivity to detect soil quality differences between sites with this indicator is limited. This is because, despite the fact that this approach includes a variety of soil taxa that belong to different size, life-form and trophic groups, the organisms taken into account do not completely represent the different soil habitats and ecological interactions in soil. Thus the number of soil parameters used to classify sites into similar groups may be too reduced (pH, organic matter content, C/N, texture and soil moisture) to adequately characterise the entire soil status.

Moreover, this indicator has received limited validation. So far, it has only been tested in a limited number of study sites, with a biased diversity of environments (e.g. for 15 study sites, there were 10 forest sites, 4 grassland sites and only a single arable field). This number might however increase rapidly depending on support given to this

approach. With a small number of reference sites, the whole system does not work because of its low ability to detect any deviations between observed and expected communities. Dividing soils in only ten types means that any stress would have to be dramatic to be detected because each ecotype would be very broad (Breure, Mulder et al. 2005).

Finally, reference values are defined by expert's knowledge and are not based on real observations from "reference sites".

Biological Soil Quality (BSQ)

The Biological Soil Quality (BSQ) index (Parisi 2001) is an indicator of the activity of biological regulators. It is based on the idea that high soil quality is associated to the number of microarthropod groups well-adapted to the soil habitat. Thus the BSQ is applied to soil microarthropods, and based on the life-form approach. The life forms include groups of microarthropods characterised by the same convergent morphological features, which enable to assess the degree of specialisation of microarthropods without requiring complex taxonomic identification (Parisi, Menta et al. 2005). Indeed, microarthropods show morphological characters that reveal their adaptation to the soil environment, such as reduction or loss of sight, pigmentation, jumping or running adaptations. Thus, the presence of these characters enables the characterisation of organisms at the morphological level, rather than at the species level. This means non-specialists can also use the BSQ.

The main steps for obtaining the BSQ are:

- Sampling and extraction: soil cores and funnel extraction
- Determination of the biological forms: classification of the different types according to homogeneous morphological characters.
- Calculation of the BSQ index (Parisi 2001): Each morpho-type receives an eco-morphological index (EMI) proportionate to its soil adaptation level. The scoring ranges from 1 (surface-living forms) to 20 (deep soil living forms). The most highly adapted microarthropods belonging to a group determine the overall EMI score for that group. The BSQ of a sample is the sum of the EMIs of that sample.

Two different BSQ are proposed, one based on microarthropods, and the other based only on springtail species.

Some of the strengths of this indicator are that it is sensitive to land-use change (Parisi, Menta et al. 2005) and to short-term variations in management practices (Gardi, Tomaselli et al. 2002), but it is less sensitive to large variations in some soil parameters, such as SOM (Gardi, Tomaselli et al. 2002). Its versatility and relative ease of use (no need for determining species or estimating their abundance) mean that it could be used in large scale sampling and monitoring schemes. Qualitative ranking of BSQ could also be used for soil quality cartography.

Compound indicators: numerical approach

Another approach consist to develop indices intended to synthesise the information collected from a range of soil physical, chemical and biological parameters into a "quality score". The main advantage of this kind of approach is to allow comparisons between different soils using a numerical approach which simplifies interpretations.

General Indicator of Soil Quality (GISQ)

The General Indicator of Soil Quality (Velasquez, Lavelle et al. 2007) is an indicator of the different ecosystem services provided by soils, one of them being the conservation of soil biodiversity. It is a synthetic indicator combining five sub-indicators representing five ecosystem services, each quantified by a range of variables:

- Physical quality or the ability of soils to provide infiltration and storage of water. It includes measures of porosity and moisture
- Chemical fertility or the ability of soils to provide the nutrients necessary for plant production. It can be estimated via nutrient concentrations and pH.
- Morphology that measures soil macro aggregation, soil ecosystem engineers, and describes characteristics of the litter system.
- Organic matter and the ability of soils to participate in climate regulation. It can be estimated through C and N concentrations.
- Macro-invertebrate community composition, as an indicator of soil biological activities. It can be estimated by the structure and abundance of macroinvertebrate communities. Soil macro-invertebrates include all the invertebrates that belong to a group where more than 90% of the individuals can be seen with a naked eye.

The GISQ is calculated statistically, using multi-variate analysis. First, for each of the five sub-indicators, all the different measures taken are simplified into a smaller number of variables that best summarise their variability. Second, the sub-indicators are calculated based on these summary variables, and can range from 0.1 to 1. The GISQ is a weighed sum of the five sub-indicators.

The soil macro-fauna sub-indicator is the only one to be correlated to the other four sub-indicators, suggesting that the abundance and diversity of soil macro-fauna might be a valuable indicator of soil quality (Velasquez, Lavelle et al. 2007).

Some of the strengths of this indicator are that it integrates a range of soil physical, chemical and biological measurements to characterise soil properties. The GISQ also offers an interesting possibility in soil assessment since it allows knowing the status of each compartment separately (by the score obtained with each sub-indicator) and thus gives the possibility to modify management practices in order to improve the compartment getting a low score. Moreover, the approach followed for developing this index could be extended to other compartments of the ecosystem depending on the question to answer. Thus, pollution or socio-economic sub indicators for example could be added to the general formula. However, a limitation is that this indicator is regionally specific (Velasquez, Lavelle et al. 2007) and the evaluation of soils different from those used to its development implies the collection of new data sets. The validation of this kind of indicator is necessary.

Biotic Indicator of Soil Quality (IBQS)

The Biotic Indicator of Soil Quality (Ruiz Camacho 2004) uses macro-invertebrate indicator taxa to assess soil status. A set of soil physico-chemical measurements is first used to identify groups of soils sharing similar properties. In each type of soil, the indicator taxa are then selected from the whole community of soil macro-invertebrates based on two criteria of ecological interest: the specificity and the fidelity of each organism for that environment (Dufrene and Legendre 1997). The procedure used for the identification of indicator taxa allows excluding rare species that could not be

found with a reasonable sampling effort. The abundance and indicator value of each indicator taxa re further combined to obtain a 'quality score'.

Thus, the IBQS allows comparing different soils from a biological point of view and uses a realistic soil system classification (based on a set of physico-chemical parameters routinely used to describe soil status). Moreover, it considers the whole communities of macro-invertebrates which offer a better resolution of information than using single groups. Its development is based in robust statistical evaluations.

However, the robustness of this index still needs to be improved by increasing the number of observations so as to get a wider representation of different soil types and land management practices. It has not yet been validated, and this step will be necessary to establish the extent of its application and to identify threshold levels for different parameters related to management practices and ecosystem functioning.

Table 5-2: Main compound indicators of soil biodiversity

Indicator	Functional groups	Soil biotic indicators	Soil abiotic indicators	Integrated	Robustness	Measurability	Sensitivity
Benchmark indicators							
BISQ (Biological Indicator System for Soil Quality)	-Chemical engineers -Biological regulators - Soil ecosystem engineers	- Microbial activity and biomass - Diversity and abundance of nematodes, mites, earthworms		No, but can be	Poor	Good	Limited to reference sites
BBSK (Biological Soil Classification Scheme)	-Biological regulators -Soil ecosystem engineers	- Abundance and diversity of meso-fauna and macro-fauna	pH, C/N ratio, soil moisture, soil texture	No	Poor	Good	Limited to reference sites
BSQ (Biological Soil Quality)	-Biological regulators	- Diversity of micro-arthropods morphotypes	No	Yes	Good	Very good	Large scale only
SOILPACS (Soil Invertebrate Prediction and Classification Scheme)	-Invertebrates	-Stress of soil communities	No	Yes	Good		Good – but difficult to separate natural from human induced stresses
Numerical indicators							
IBQS (Biotic Indicator of Soil Quality)	- Soil ecosystem engineers	- Structure and abundance of macro-fauna	Physical classification of soil, based on routinely measured parameters (e.g. pH, cation concentration)	Yes	Yes	Good	Good – to be validated
GISQ (General Indicator of Soil Quality)	- Soil ecosystem engineers	- Diversity of macro-fauna	- Physical (porosity, moisture) - Chemical (nutrient concentrations) - Morphological (aggregation) - Organic matter (C and N concentrations)	Yes	Good	Good	Good – to be validated

5. 1. 5. RECOMMENDATIONS

Sustainable use of soil should be indicated by an ecological indicator, based on a holistic approach that integrates data on physical, chemical and biological characteristics of the soil. Such approaches recognise the complexity of ecological interactions and the importance of ecosystemic processes as a reflection of underlying functions, including soil characteristics. The combination of biotic and abiotic measurements leads to the possibility of deducing response models for individual indicators. With such models, predictions can be made concerning the effects of environmental and human impact scenarios. The relation between abiotic conditions, management practices and the composition and functioning of soil organisms offers opportunities to adapt political and management practices towards an optimal (sustainable) use of the soil biodiversity and the ecological processes that are governed by soil organisms. To establish the scale in which indicators fluctuate, it is necessary to make reference to descriptions and determine the effects of severe disturbance.

In general a useful indicator should be:

- Meaningful
- Standardised
- Measurable and cost-efficient
- Relevant for policy makers
- Cover a wide spatio-temporal scale
- Understandable
- Accurate

Detailed and headline indicators should be used in combination, depending on the targeted public (e.g. scientists, policy makers, etc.). For policy makers, the use of risk indicators could be important to orient decisions concerning the application of the precautionary principle.

5.2. MONITORING SCHEMES

At an international scale, the need for soil biodiversity monitoring schemes is identified in the Soil Biodiversity Initiative, set up by the Food and Agriculture Organisation (FAO). It was established following a decision at the 6th meeting of the Conference of Parties to the Convention on Biological Diversity in order to promote the conservation and sustainable use of soil biodiversity. In this context, monitoring of soil biodiversity is encouraged as a method of assessing soil health, in order to better inform management and policies related to the use of soil[44].

Currently, there is no consensus on biological soil monitoring initiatives worldwide. The reason is that soil monitoring activities vary widely in their scope, goal, duration, efforts and in the parts of the soil system that they represent.

However, any efficient monitoring should follow some basic principles. Most importantly, monitoring ought to result in robust parameters. Thus, it must be very clear how and when monitoring should take place, how the sampling should be standardised, and which indicators should be used. For instance, soil biological parameters change over time, and standardised guidelines must clarify at which time

[44] www.fao.org/ag/agl/agll/soilbiod/initiative.stm; last retrieval 2/9/2009

of year samples are to be taken. However chemical and physical parameters of soil are more constant in time than biological measures, and thus may not need to be sampled as frequently.

5. 2. 1. SOIL BIODIVERSITY MONITORING IN EUROPE

At an EU level, the soils section of the European Commission's Joint Research Centre set up a Biodiversity Working Group which has been charged with evaluating existing monitoring schemes and standardising methods between them[45].

The current situation in Europe is that despite there being a well-established system of soil monitoring networks, very few of these networks consider soil biodiversity as a parameter that should be measured (Saby, Bellamy et al. 2008). For instance, only 5 of 29 European countries have monitoring sites for earthworms (Saby, Bellamy et al. 2008). The networks that have been specifically set up to measure soil biodiversity or which include the monitoring of biological parameters are outlined in Table 5-3.

→ **SOIL BIODIVERSITY MONITORING INITIATIVES**

Some of the most characteristic examples are presented in more detail below.

France

In order to integrate biological parameters in soil quality monitoring, a French programme RMQS-biodiv (Soil Quality Measurement Network) has been recently (2009) developed at the regional scale (Brittany)(Cluzeau 2009). This programme, which assessed biological parameters, was connected to a larger soil monitoring network developed at the national scale (Soil Quality Measurement Network- RMQS) which assessed agro-pedological parameters[46]. The connexion of both programmes allowed the monitoring of soil biodiversity (species and function) in relation to land use (mainly agricultural practices) and pedoclimatic parameters.

The final objectives of this RMQS-Biodiv Programme was more particularly to contribute to a better definition of soil biota sampling procedures for their necessary standardisation at national or European level and to define the relevance of some criteria in term of performances/cost in order to propose them to field actor.

Thus, this programme measured a large range of biological parameters, requiring an important research network: macro-fauna (earthworms, total macro-fauna), meso-fauna (nematodes, acarina and springtails), microorganisms (microbial biomass, bacterial and fungal diversity), and also functional biological parameters (soil respiration, humus index). The pilot area covered more than 27 000 km² and the sampling was realised by a systematic approach based on a grid 16 X 16 km. 115 sites were sampled in 2006 and 2007. The sampling methods, adapted to the study context, were more or less closed to ISO standards.

Based on the same French monitoring network a project called ECOMIC-RMQS aims to characterise telluric bacterial communities on about 2 200 soil samples based on molecular tools such as quantitative PCR, DNA microarray and DNA fingerprint directly on DNA extracted from soil. This project wants to build up and maintain a national soil DNA library (in the platform GenoSol)[47] that could be available to the whole scientific

[45] www.eusoils.jrc.ec.europa.eu/library/themes/biodiversity/wg.html; last retrieval 2/9/2009
[46] www.gissol.fr/programme/rmqs/rmqs.php; last retrieval 16/09/2009
[47] www.dijon.inra.fr/plateforme_genosol; last retrieval 12/11/2009

community in order to assess microbial diversity in the future with more powerful tools and/or other molecular analysis (Gardi 2009).

Another French monitoring network called RENECOFOR (National network for the long term tracking of forest ecosystems) was created by the ONF (National Forest Office) in 1992 in order to complete the French network for health forest monitoring. This network represents the French part for the monitoring of forest ecosystems of a European network composed by 34 countries[48]. The main objective of RENECOFOR network is to detect any change on the long term on the wide range of ecosystems monitored and to identify the reasons of these changes. The network is composed by 102 permanent observation sites representative of the region where they are found and that will be studied during 30 years at least. While the measurements are mostly of abiotic parameters (e.g. pedological descriptions, meteorological, ozone, and ammonia measurements), numerous research projects have been and are currently being developed in order to complete the extent of the monitoring and to increase the variety of biological measurements. Thus for example, soil macro-invertebrates have been studied to be monitored in a routine way. Since 2008, the issue of biodiversity has been included in the programme.

Germany

Soil monitoring activities are not centrally coordinated in the country, but in total 800 BDFs (permanent soil monitoring sites), are run by each of the 16 Länders, at which a wide range of abiotic (plus some biotic) parameters are measured. Each Länder may thus use a different approach. Soil biodiversity monitoring in particular is only performed on a case by case basis. This results in gaps and taxonomic sampling biases (Gardi 2009). However, according to new agreements, the information on soil biodiversity available on the level of the individual Länder, is currently compiled on the federal level. Based on this information, further activities are planned in order to improve soil protection. The legal basis is § 1 of the German Soil Protection Act (1998) which requires to protect the function of soil as a habitat for organisms. The 2nd paragraph presents a definition on soil functions.

Since 1998, after the Federal Soil Protection Act became effective, the German Federal Environmental Agency (UBA) supported the development of soil biological site classification concepts (BBSK), as a promising tool for the assessment of the habitat function of soils.

The BBSK relies on a classification whereby each region can be characterised by a limited number of sites with characteristic soil communities, which can be characterised by its abiotic parameters (e.g. soil properties, climatic factors). The assessment relies on the differentiation between the sampled soil community, and the one expected under the reference state for that site. The sampling of sites should be performed with standardised measures, use easy measurements and a reference (undisturbed) site should exist. So far, about 50 sites have been sampled (mainly forests) for a wide range of soil organisms. Two case studies have been performed (one in 11 forest sites, the other in 15 sites – 10 forest, 4 grasslands, 1 arable field) The sampling included ecosystem engineers (e.g. earthworms, isopods) and biological regulators (e.g. mites). These studies showed that a habitat function of soil by the BBSK concept was possible, although better definition of the assessment criteria (reference

[48] www.onf.fr/renecofor; last retrieval 16/09/2009

state, inclusion of chemical engineers) would still need to be improved (Römbke 1997; Rombke, Breure et al. 2005).

Netherlands

The Dutch Soil Quality Network (DSQN), a national monitoring network, offers a usable infrastructure and the advantage of available, comprehensive abiotic measurements. The objective of the network is to increase the knowledge of the effects of soil type and management on diversity and functioning of soil organisms and mineralisation processes. Moreover, the development and application of new biological indicators for soil quality is expected.

The monitoring network is composed of 200 sites in a stratified grid design that represents 70% of the total surface area of the Netherlands, with respect to soil type and land use. The programme started in 1993 with an inventory of the free living nematodes on the 200 locations, within a period of 5 years. The 200 locations represent 10 categories of land-use / soil type combinations. The major part of these locations was cattle- or arable farms.

In 1999 the 5 year sampling programme was repeated and extended to a foodweb approach, in which nematodes still play a central role. Moreover, organic farming and new soil categories were added, enlarging the monitoring network to circa 300 locations.

In addition to the original framework, 50 to 100 sites from outside this network are regularly sampled, for instance biological farms or polluted areas which are supposed to be good and bad references, respectively. Each year two categories are sampled (40 sites plus reference sites). That is the reason why it takes five years to complete one round of monitoring. The obtained results on micro-organisms are combined with data on soil fauna and soil chemistry from related projects.

Since 1997, the Biological indicator-system for Soil Quality (BiSQ) is designed to make the link between soil biological diversity and ecosystem function in DSQN. The dominant soil organisms groups and ecological process parameters are therefore brought together in a practical indicator set to be used in a nation-wide monitoring programme for soil quality. Each location is sampled and analysed every 6 years.

The outcome from this monitoring scheme should help identify:

- Key environmental processes on soil biodiversity and ecosystem functioning
- Taxa and habitats that are most vulnerable to the loss of soil biodiversity
- Soil habitats that are most amenable to soil restoration

With the dataset available, the main question remains whether the health of the soil biodiversity can be assessed. Strictly speaking, no guidance comes from just determining values for parameters on a given location. These values should be benchmarked against a certain reference value, in order to assess soil biological health (e.g. judge it to be bad, normal or healthy) and to guide policy measures.

Portugal

The National Forestry Service (Ministry of Agriculture) has established 16 monitoring plots (using ICP Forest Level II plots) for forest biodiversity representing the major forest types in the country. This activity is funded under the Forest Focus programme, that deals with the implementation of an operational methodology for biodiversity monitoring (soil indicators include soil macrofauna, collembola and carabids).

EU

No EU-wide monitoring system of soil biodiversity currently exists, but the setting-up of one is in the pipelines. As a preliminary step, the EU FP6 (6th Framework programme) project ENVASSO was launched with the ambitious aim to provide the basis for a comprehensive, harmonised soil information system in Europe, by designing and testing an integrated and operational set of indicators. Regarding soil biodiversity, a set of EU-wide indicators were selected based on their significance, the existence of standard measurement methodology for them, and their measurability and costs[49] (Bispo 2007; Bispo 2009; Gardi 2009) .

The minimum set of surrogate measures selected to assess the overall changes in soil biodiversity cover the three functional groups:

- Soil ecosystem engineers: earthworm biomass and diversity
- Biological regulators: springtails biomass and diversity
- Chemical engineers: microbial activity (respiration)

This minimum set of indicators could be extended in some regions, according to the availability of resources, to include e.g. all macro-fauna or nematodes.

To ensure the monitoring results in robust parameters, the procedures and protocols used for the different indicators are all based on ISO standards, which have been adapted for assessment at the EU scale. Pilot tests in sites distributed in four countries (France, Ireland, Poland, Italy) have been conducted and proved the effectiveness of each indicator, and its sensitivity to detect change across a range of land-use categories at EU level (Gardi 2009).

It is recognised that soil biodiversity monitoring should be accompanied by measurements of soil abiotic characteristics, so as to be interpretable. These include:

- Habitat characteristics: detailed geographical classification, land-use type, climate data, groundwater level
- Soil properties: pH, SOC content, Nitrogen content, C:N ratio, texture, Cation exchange capacity, usable capacity of the root layer
- Contamination and human-induced stress: concentration of heavy metals, other soil degradation processes

One issue for this monitoring scheme remains the lack of an established methodology to derive baseline indicator values for given soil types (depending e.g. on land use, texture, climate) that are not based on subjective expert opinion.

[49] See Envasso report : www.eusoils.jrc.ec.europa.eu/projects/envasso/

Table 5-3: Monitoring schemes in the EU that measure biological parameters of soil (Bloem, Schouten et al. 2003; Breure 2004; Jones 2005; Parisi, Menta et al. 2005; Rombke, Breure et al. 2005)[50]

MS	Name of monitoring scheme	Aim of scheme	Initiating organisation	Date of initiation	Scale	Indicator used	Sampling scheme	Frequency of sampling	Organisms monitored
Austria	Environmental soil survey		Provincial governments		Regional		Initial Environmental Soil Survey (6000 sites across the country) – regularly monitored	Regular intervals	Microbes (biomass), earthworms, pot worms and springtails.
Czech Republic	Basal Soil Monitoring Scheme		Ministries of Agriculture and the Environment	1992	National		217 plots across the country, divided by land use and soil type Four samples taken from each monitored plot.	Annual for microbiological parameters	Microbes (biomass, C, N biomass; basal respiration; anaerobic ammonification; nitrification)
France	RMQS (Soil Quality Measurement Network)-biodiv	Biological monitoring of soil quality – improve soil biota sampling procedures	Environmental ministry, French environmental agency (ADEME), and French agronomic research institute (INRA GIS SOL)	2006	Regional (27000 km²)		115 sites of 16km x 16 km,	Annual	Microbes (biomass, bacterial and fungal diversity, soil respiration), biological regulators, macro-fauna (earthworms, total macro-fauna), humus index
France	ECOMIC-RMQS	Biological monitoring of soil quality – improve sampling procedures	INRA, ANR,,ADEME, GIS Sol	2006	National		2,200 sites of 16km x 16 km,	Not yet decided	Microbes (bacterial and fungal diversity,)
Germany	Soil Biological Site Classification	Soil biological classification to assess the habitat function of soil	Umweltbundesamt (Federal Environmental Agency)	2000	Regional	Soil Biological Site Classification	Approx. 50 sites (mainly forests, but also grasslands and crops)		
Italy		To assess soil quality	ISPRA		Regional	Qualità Biologica Suolo (QBS)			Microarthropods
Latvia	Agricultural Land	To allow the		1992	National		12 research plots,	Annually	Meso-fauna and

[50] Document available at : www.rubicode.net/rubicode/RUBICODE_Report_on_Indicators_of_Disturbance.pdf ; last retrieval 16/09/2009

MS	Name of monitoring scheme	Aim of scheme	Initiating organisation	Date of initiation	Scale	Indicator used	Sampling scheme	Frequency of sampling	Organisms monitored
	Monitoring Programme	assessment of the anthropogenic impact on agricultural land					which represent the 20 soil variations, the types of farming and the climatic conditions		epigeic fauna
Netherlands	Biological indicator system for Soil Quality	Role of biodiversity in the maintenance of ecological functions in soil	Dutch Ministry of Environment (National Soil Quality Network)	1997 (nematodes, 1993)	National	Biological Indicator for Soil Quality	200 locations sampled, divided into ten categories of soil type and land-use type	Two soil/land-use types sampled annually, so one cycle to cover whole network takes 6 years	Microbes (nitrification, microbial biomass, activity, functional and genetic diversity, C:N mineralisation), biological regulators (protists, nematodes, springtails, mites), ecosystem engineers (earthworms, fungal hyphae), community structure
Romania		To monitor soil quality		1992	National		16 km by 16 km grid set up to cover whole country. At each intersection samples are taken from a 400 m by 400 m square (942 sites)	Every four years, unless site is particularly degraded in which case monitoring is annual	Bacteria and fungi (number of)
UK	Countryside Survey, Work Package Soils	The Soil section of the survey has several aims. Aim in relation to soil biodiversity is to determine if there is evidence of a decline in soil biodiversity	Centre for Ecology and Hydrology (research institution), funded by Department for Environment, Food and Rural Affairs	Countryside Survey started in 1978, biological measurements on soil started 1998	National	Invertebrate diversity	629 1km squares to represent all major habitat types 4 samples taken from each square	Since biological measurements were added, surveys have been done in 1998 and 2007	Invertebrates

→ **BARRIERS AND LIMITATIONS TO THE DEVELOPMENT OF SOIL BIODIVERSITY MONITORING NETWORKS**

Absence of awareness of soil biodiversity conservation

In the majority of cases (with the possible exception of Germany, Italy and the UK), the measuring of soil biodiversity did not involve the establishment of a separate monitoring scheme. Instead, soil biodiversity measurements, such as microbial biomass or the diversity of other soil fauna, are typically included within the set of parameters which are measured in existing soil monitoring schemes. For instance, in 52 soil monitoring programmes across the world, 18 of them include soil biological attributes (review by the Alberta Ministry of Agriculture and Rural Development, 2003). The advantage of inserting soil biodiversity monitoring within existing soil monitoring networks is that a combined picture of soil abiotic and biotic conditions can be gained, which is necessary to adequately interpret soil biological data.

However, soil biodiversity monitoring per se is not usually an explicit aim, even when it is monitored. Instead, where the purpose of the scheme can be determined (Table 5-3), all but one scheme used soil biota as a means to assess the quality of the soil, usually in combination with other chemical and physical parameters. Only the UK's Countryside Survey appeared to include the monitoring of soil biodiversity as an explicit aim. However, the Dutch monitoring scheme was initiated as part of the country's efforts to meet the requirements of the Convention on Biological Diversity (Rombke, Breure et al. 2005). Therefore, the assessment of biodiversity is likely to be a final aim of this scheme as well.

Other limitations to widespread biodiversity monitoring include a lack of awareness of the importance of biological parameters of soil quality, which is most commonly thought of as a physical or chemical resource.

Absence of reference system and indicators

In many cases, the monitoring of soil biodiversity is still in its early stages, and therefore practices have not yet become standardised. For example, the Austrian scheme is implemented at a regional level, and while some regions have included biological parameters (e.g. microbial biomass in Upper Austria), others are yet to do so. However, it is expected that as new methods are developed, such parameters will be a requirement of soil monitoring programmes (Jones 2005).

Several sampling systems have been utilised, which generally follow those used in the existing soil monitoring networks. The lack of long-term data makes the definition of optimum values of soil biodiversity difficult. Thus, the assessment of changes in biodiversity over time is likely to be more useful than assessing the current state of biodiversity (Bloem, Schouten et al. 2003).

Some of the existing monitoring schemes, including those of Germany, Italy and the Netherlands, involved the use of specially developed indicators of soil biodiversity. However, in other cases (e.g. Latvia and Romania), the biomass or number of microbes was the only biological parameter measured. The complexities and difficulties in developing a set of indicators that are not too sensitive to site differences or contamination is likely to be hindering the more thorough monitoring of biodiversity than just of biological parameters (Breure 2004). Difficulties in the identification of soil organisms can also limit the extent to which they can be fully assessed.

Costs

The complexity of biodiversity implies that time and money are major impediments for thorough monitoring. Intensive monitoring schemes are relatively expensive and time-consuming. The main cost in soil monitoring is field sampling. For example, taking and analysing samples for one site in the Dutch scheme cost 5500 Euros in 2002, with the entire programme costing 330 000 Euros each year for only 60 sites (Bloem, Schouten et al. 2003). Moreover this programme does not include some of the most relevant soil organisms such as protozoa. Another example is the RMQS programme for Brittany (France) which cost Euros 580 000 per year for 120 sites. Even if these costs may appear extremely high, in reality, when they are considered per hectare, they are relatively low. For instance, a good coverage of the French territory could be carried out with around 2000 sites. With an average cost of 5000 Euros per site this would amount to a cost of 11 000 000 Euros per year, which could be considered expensive. Indeed, if the cost per hectare is considered, France covering a surface of 29 000 000 of hectares, this would amount to a cost of 0.37 Euros per hectare, which is not a high cost for the preservation of soil ecosystem functioning and related services[51].

The main factors influencing the costs of monitoring programmes are the salary of the personnel performing the sampling and the analyses, which may vary significantly, if academic or private personnel are employed. Added to this, in order to have a complete picture of soil biodiversity, at least one technician is needed for each group of soil organisms within the same research team. Thus, currently, the training of such personnel is one of the blocking points.

Another barrier is the lack of knowledge regarding the DNA sequences useful for species monitoring of micro and macro-fauna. This knowledge is much stronger for soil microorganisms and allows, for this class of organisms, an automatic screening (e.g. DNA microarrays). Even if the investment cost for such molecular screening can be elevated, it is generally paid off in the long term. An identification of the informative DNA sequences and the development of DNA extraction protocols adapted for these groups of organisms will thus help in the development of molecular techniques which could contribute to the cost reduction of monitoring schemes.

5. 2. 2. SOIL BIODIVERSITY MONITORING OUTSIDE EU

Following are examples of biodiversity monitoring schemes located outside the EU, which employ different approaches to those identified in Europe.

Monitoring in Australia occurs as part of national or regional biodiversity monitoring programmes, as opposed to soil monitoring programmes. For example, in Australia terrestrial biodiversity has been assessed as part of the National Land and Water Resources Audit in 2002 and again in 2008. The soil biodiversity assessment which is included in this programme focuses on earthworms, termites and ants. These groups are included due to their functional roles in ecosystem process and their influence on soil properties, and the fact that they are well known and constitute a large proportion of the biomass of soil invertebrate (Woodman 2008).

In Canada, the diversity of earthworms is monitored as an indicator of soil biodiversity and health by the Worm Watch programme, which is coordinated by the Ecological Monitoring and Assessment Network (EMAN) of Environment Canada, the Canadian Nature Federation and the University of Guelph. The programme is an example of a community-based monitoring scheme, which still ensures that the data collected is

[51] Antonio Bispo, personal communication.

scientifically sound. To achieve this, participants, which include researchers, education institutes and the general public, are required to follow standard sampling methods and submit their data using a standardised form. The data collected is used by EMAN (Ecological Monitoring and Assessment Network) to contribute to their national monitoring programmes which aim to assess long-term changes over time and compare them spatially. Using a community-based approach allows a greater spatial coverage of data across the country to be obtained, while at the same time collecting local-scale information which is useful for local decision makers. The scheme also has an educational function and can raise awareness of both soil biodiversity and ecological monitoring[52].

5. 2. 3. CONCLUSIONS, KNOWLEDGE GAPS AND RECOMMENDATIONS

Given the complexity of soil biota, indicators are useful to translate trends in soil biodiversity and related services in a simple and clear manner. This is a key factor to **communicate the value of soil ecological capital** to decision-makers. Suitable indicators must be **meaningful** or clearly relate to an important ecological function, **standardised**, so as to allow comparisons among different sites, and **easy to use**, so as to ensure they can be routinely used.

To date however no reference set of indicators or synthetic indicators are available, despite the fact that a multitude of indicators estimating some specific aspects of soil activity or diversity, many of which ISO-certified, exist. But recently, much progress has been made in the development of compound indicators that account for both factors affecting soil biodiversity and soil biodiversity per se. The most promising avenue may lie in the development of **numerical indicators** which are objectively defined, such as GISQ and IBQS, since these do not rely on expert opinion or the definition of reference sites.

A Europe-wide monitoring system must be developed to answer the questions of decision-makers on status and trends in soil biodiversity and their implications. But widely accepted reference sets of indicators, reference ecosystems and standardised sampling protocols are still missing. Thus, while several regional monitoring programmes have been developed, they remain of limited use, since no consensus exists on their scope, duration, or on the parts of the soil system that they represent. The **ENVASSO programme** is the first attempt to a comprehensive, harmonised soil information system in Europe. It offers a minimum set of reference indicators of soil biodiversity that can constitute a standard against which future monitoring schemes should be developed. The future EU-wide monitoring programmes could be integrated into existing biodiversity or soil monitoring networks, and participative schemes could be encouraged.

There is still a need for good quality data on soil organism abundance and distribution over a wide range of situations, in order to establish baselines and thresholds allowing the definition of 'good' or 'bad' soil status. This kind of database is also essential to develop models and predict the impact of management practices and politics on soil biodiversity and protection accurately. Moreover there is a need for methods permitting an easier extrapolation of data from one spatio-temporal scale to another, and permitting long term forecasting of impacts.

[52] www.eman-rese.ca/eman/program/about.html ; last retrieval 3/9/2009

The choice of appropriate bio-indicators is dependent on the assessment goals and on the availability of standard methods, laboratories that can perform the analysis, and an appropriate budget. Any system for the assessment and monitoring of soil status based on soil organisms needs simple practical keys for taxa identification, regardless of the taxonomic resolution considered.

In Europe, the monitoring of soil-dwelling species can be integrated into existing national biodiversity monitoring networks. If methods are found to ensure the scientific reliability of data, community monitoring schemes could be encouraged, particularly in those countries where active volunteer programmes already exist. A regulatory obligation to monitor soil biodiversity could be, of course, a strong incentive for such training, as has occurred in the past for water quality monitoring.

An effective monitoring strategy could be to perform a comprehensive monitoring in each EU country covering the whole territory, following by a partial monitoring each year covering only a portion of the sites. The ideal frequency of monitoring for soil biodiversity could be 3-5 years for each site.

This would have the added benefit of raising awareness of the existence and importance of soil biodiversity. The complexity of biodiversity implies that time and money are major impediments for thorough monitoring. Intensive monitoring schemes are expensive and time-consuming. The main cost in soil monitoring is field sampling. Indeed, since the value of the provided services is not valuated in a homogenous way, it is not possible to say which is a relative high cost for the preservation of soil ecosystem functioning.

→ **MAIN RESEARCH GAPS:**

- **Development of accepted synthetic indicators**
- **Indicators for long term impacts**
- **More precise models for extrapolation of process measurement from one scale to another**
- **An identification of the informative DNA sequences and the development of DNA extraction protocols adapted for soil organisms would help in the development of molecular techniques which could contribute to the cost reduction of monitoring schemes. This already exists for microorganisms, but is not developed enough for other soil organisms such as earthworms or insects.**

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6. EXISTING POLICIES RELATED TO SOIL BIODIVERSITY

6.1. EU AND INTERNATIONAL POLICIES

6. 1. 1. POLICIES HAVING A DIRECT LINK WITH SOIL BIODIVERSITY

→ **PROTECTING SOIL BIODIVERSITY**

The FAO Soil Biodiversity Initiative

The Conference of Parties (COP) to the Convention on Biological Diversity (CBD)[53] decided, at its 6th meeting in Nairobi April 2002, "to establish an International Initiative for the Conservation and Sustainable Use of Soil Biodiversity as a cross-cutting initiative within the programme of work on agricultural biodiversity, and invites the Food and Agriculture Organisation of the United Nations, and other relevant organisations, to facilitate and coordinate this initiative".

The International Technical Workshop on Biological Management of Soil Ecosystems for Sustainable Agriculture, which was organised by EMBRAPA-SOYBEAN and FAO, and held in Londrina, Brazil during 24-27 June 2002, was organised in this context as a contribution to the joint programme of the CBD (Convention on Biological Diversity) and FAO in accordance with FAO's mandate on sustainable agriculture and food security and with Decision V/5 of the COP to the CBD.

There are two main objectives for the Soil Biodiversity Initiative[54]. The first one is the promotion of **awareness raising, knowledge and understanding** of key roles, functional groups and impacts of diverse management practices in different farming systems and agro-ecological and socio-economic context. The second, even more important, is the promotion of **ownership and adaptation** by farmers of integrated soil biological management practices as an integral part of their agricultural and sustainable livelihood strategies. Finally, the initiative also aims at strengthening collaboration among actors and institutions and mainstreaming soil health and biological management into agricultural, land management and rehabilitation programmes.

This **recommendation** originated from the **Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA)** of the CBD, which resulted in the document 7/INF/11 on **Soil Biodiversity and Sustainable Agriculture**[55] which was submitted by FAO.The initiative for conservation and sustainable use of soil biodiversity was formally established by decision VIII/23, section B of the Conference of the Parties, in March 2006. The mandate for establishing the initiative was provided by decision VI/5.

[53] At the 1992 Earth Summit in Rio de Janeiro, world leaders agreed on a comprehensive strategy for 'sustainable development'. One of the key agreements adopted at Rio was the Convention on Biological Diversity. The Convention establishes three main goals: the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits from the use of genetic resources.

[54] Available at: www.fao.org/ag/agl/agll/soilbiod/fao.htm ; last retrieval 02/09/2009

[55] Available at: www.cbd.int/doc/meetings/sbstta/sbstta-07/information/sbstta-07-inf-11-en.pdf ; last retrieval 02/09/2009

To date, work on soil biodiversity has been conducted in the areas of assessment, monitoring and mainstreaming under programmes and projects conducted by institutions, e.g. Tropical Soil Biology and Fertility of the International Centre of Tropical agriculture (TSBF-CIAT) [56], the Institut pour la Recherche et le Développement (IRD)[57], CAB International[58], etc. In general, some groups of soil biota have been studied more than others, especially earthworms, termites, ants and nitrogen-fixing bacteria.

Regarding the first objective of the initiative, *sharing of knowledge and information and awareness-raising,* while some case studies exist[59], new case studies would allow for the needed updated information. To date, limited work has been undertaken to compile, synthesise, and evaluate case studies for practical advice and active dissemination. There is still limited coordinated effort to gather data and information specific to soil biodiversity. Databases and information systems exist that contain relevant information, but these are intrinsic to project work being carried out. In general there is considerable potential for the development of information systems and augmentation of networking regarding sharing of knowledge and information. Moreover, much more work is required in this area and efforts are needed to enhance public awareness and make relevant information widely available. Regarding the second objective, *capacity-building for the development and transfer of knowledge of soil biodiversity and ecosystem management into land use and soil management practices,* the promotion of adaptive management approaches, as well as capacity building efforts and some targeted participatory research is ongoing, e.g. in Brazil, the AMAZ-BD (IRD) is conducting participatory farmer-oriented learning-by-doing processes on soil life and function. Some very relevant work has been undertaken on indicators, which has the potential to contribute to broader efforts to develop tools, build information and identify and develop datasets on soil biodiversity at national level that are important for agriculture. FAO has provided support for the development of extension field guides on soil macro-fauna and soil health in collaboration with IRD. Taxonomic expertise lacks in many countries for most of the soil biota groups; therefore collaboration with the Global Taxonomy Initiative could be strengthened, to fill specific gaps. Technical expertise and capacity building is provided at the technical level and only for some groups of soil organisms. Nonetheless, there is a need for training on soil biodiversity and functions at the farmer level with advocacy material and training manuals. Regarding the third objective, *strengthening collaboration among actors and institutions and mainstreaming soil biodiversity and biological management into agricultural and land management and rehabilitation programmes,* activities have so far been limited. There is also a need to strengthen collaborative mechanisms between sectors in order to ensure mainstreaming of soil biodiversity and biological management. In conclusion, the work carried out to date has highlighted the very real need and considerable potential for work under this Initiative to develop further partnerships and in particular to make available relevant research findings for

[56] Website of the Centro Internacional de Agricultura Tropical, available at www.ciat.cgiar.org/tsbf_institute/index.htm ; laste retrieval 15/12/2009

[57] IRD is a French public research institute working for the development of Southern countries. The website is available at: www.en.ird.fr/the-ird ; laste retrieval 15/12/2009

[58] CABI is a not-for-profit international organization. The website is available at www.cabi.org/; laste retrieval 15/12/2009

[59] Some case studies are available on the FAO soil biodiversity portal: www.fao.org/landandwater/agll/soilbiod/default.stm and through the CBD Secretariat website www.biodiv.org/programmes/areas/agro/case-studies.asp ; laste retrieval 15/12/2009

application for promoting sustainable and efficient agricultural development (UNEP/CBD 2007).

UN Convention to Combat Desertification (UNCCD)

Soil protection falls directly within the aims of the **UN Convention to Combat Desertification in Countries Experiencing Serious Drought and/or Desertification (UNCCD)**, adopted in 1994 and entered into force in December 1996. The UNCCD aims to prevent and reduce land degradation, rehabilitate partly degraded land, and reclaim partly desertified land. In 2002, a review of the UNCCD[60] made a series of recommendations regarding the need for more coherent policy and legislative instruments and strategies to deal with sustainable land management.

Affected Member States are required to assess and evaluate the main drivers of desertification in their country in a report. This will provide a basis to combat desertification, through activities aimed at the prevention and/or reduction of soil degradation, rehabilitation of degraded soil, and reclamation of desert soils. All developed countries from the European region, regardless of whether they are affected by desertification or not, are requested by the UN to support the activities of UNCCD worldwide and to report on their financial assistance.

Although desertification is widely recognised to lead to a decline in soil biodiversity, only three of the 12 EU-27 Member States affected by desertification reported any information on soil biodiversity (Greece, Hungary and Latvia). Each of these three Member States detected a decline in soil biodiversity, which was clearly identified as a soil degradation process in Hungary, while Greece and Latvia did not provide any information on the spatial extent or intensity of biodiversity decline (Hannam and Boer 2004).

EU Soil Thematic Strategy

Until recently, soil had not been subject to a specific protection policy at the Community level, although several Community policies contribute to soil protection. For these reasons, in the context of the Sixth Environment Action Programme (EAP[61]), the Commission adopted a Thematic Strategy on Soil Protection, with the aim to halt and reverse soil degradation. This comprehensive strategy aims to account for all the different functions that soils can perform, their variability and complexity, and the range of different degradation processes to which soils can be subject (European Commission 2006). The strategy is based around four pillars: (i) a framework legislation, (ii) the integration of soil protection in other national and Community policies, (iii) increased research on soils as a foundation for policies, and (iv) raising public awareness of the need to protect soils.

In September 2006, the Community made a proposal for a framework Directive to protect EU soils. The Directive, which is still being evaluated at the EU level[62], establishes common objectives and principles, but leaves it to each MS to decide on its

[60] See UNCCD Secretariat, Report of the Committee for the Review of the Implementation of the Convention (2002)

[61] Decision No 1600/2002/EC of the European Parliament and of the Council of 22 July laying down the Sixth Environmental Action Programme (OJ L 242, 10.9.2002, p. 1–15).

[62] For more information see www.ec.europa.eu/environment/soil/process_en.htm; last retrieval 02/09/2009

level of intervention, allowing for an efficient use of the national and administrative capabilities (European Commission 2006). The overall objective of the Directive is the protection and sustainable use of soils based on (1) the prevention of further soil degradation and the preservation of its functions, (2) the restoration of soils to a level of functionality consistent with current and intended use. In addition, the Directive calls for an evaluation of the impacts of other sectoral policies on soil functions, and also requires the identification of areas where soils are at risk of degradation and the establishment of national programmes of measures, as well as measures to identify and avoid contamination of soils.

Nevertheless, such proposals within the Directive do not include provisions specially aiming at the protection of soil biodiversity (to fight the decline of soil biodiversity),but rather address it indirectly by limiting soil degradation processes such as soil sealing, contamination, compaction, organic matter decline, salinisation and landslides..

EU Biodiversity policies

As a party to the 1992 CBD, the European Community accepted an international obligation to achieve a significant reduction of the loss in biodiversity rates by 2010, and went further by deciding to halt biodiversity loss by the same year (European Commission 2001). This is a recognition at the EU level of the 'no net loss principle', requiring quantitative and qualitative aspects of biodiversity to be maintained at a status quo. This aim is also enshrined in the legally binding Decision on the 6th EAP[63]. Yet, managing soil biodiversity is largely neglected, and is a completely different issue to that of managing aboveground biodiversity.

MS need to adopt measures to conserve biological diversity, especially through *in situ* conservation. Given the importance of soil and soil activity for the maintenance of all biodiversity, the protection of nature should not neglect soil biodiversity. In order to attain the 2010 target to 'halt biodiversity loss', the core of European nature conservation and biodiversity policy lies in the Natura 2000 EU wide network of protected areas (de Sadeleer 2006). Natura 2000 now forms the largest coherent network of protected areas in the world, covering more than 20% of the EU territory, with over 26 000 protected areas[64]. Innovatively, Natura 2000 combines conservation and development aims, such that most of the land in the network continues to be privately owned. The network of protected areas was established under the 1992 Habitats Directive[65] and aims to ensure the long term survival of Europe's most valuable threatened species and habitats, thereby fulfilling a Community obligation under the UN Convention on Biological Diversity for restoring endangered habitats and species of Community interest.

Natura 2000 relies mainly on a designated areas approach within each Member State, which remains the core legal technique of nature conservation in Europe. Member States design Special Areas of Conservation (SACs) established under the Habitats Directive, to maintain habitats of community interest that are in danger of disappearing within their natural range, that occur mainly in the EU, or represent an

[63] Decision No 1600/2002/ EC of the European Parliament and of the Council of 22 July laying down the Sixth Environmental Action Programme, (OJ L 242, 10.9.2002, p. 1–15)..

[64] Natura 2000: www.ec.europa.eu/environment/nature/natura2000/index_en.htm; last retrieval 09/09/2009

[65] Council Directive 92/43/EEC on the conservation of natural habitats of wild fauna and flora (the Habitats Directive)

outstanding example of one or more of the nine European eco-regions. The network also encompasses Special Protection Areas (SPAs) established under the older Birds Directive[66]. SPAs are the most suitable sites to conserve particularly vulnerable bird species (as listed in Annex 1 of the Directive) and regularly occurring migratory birds, but also contribute to maintain healthy populations of all bird species.

In general terms, the Habitats and the Birds Directives address to a very limited extent soil threats or soil biodiversity. Indeed, Annex 2 of the Habitats Directive includes a few soil families, mainly peaty soils and raised bogs and mires, and only a very few exceptions of the soil species included in the Annex 1 of the same Directive are considered in the present study (e.g. the beetle O. Eremite). The creation of SACs or SPAs does, however, have indirect beneficial impacts on soil biodiversity. In particular, some of the concerned SAC habitats are based on specific soil types, thus protecting these habitats will indirectly protect soil biodiversity. Furthermore, protecting certain areas from agricultural intensification or deforestation through the Habitats Directive can have an important effect on soil biodiversity. Indeed, soil biodiversity tends to be greater in undisturbed natural lands compared to cultivated fields (SoCo 2009).

In this context, LIFE is the main funding mechanism at an EU level from which environmental and nature conservation projects can benefit. Projects funded under the LIFE+ Nature and Biodiversity category are required to support the implementation of the Birds and Habitats Directives, through targeting priority species and habitats, as well as Natura 2000 sites. In 2002 and 2008, 5 projects were funded which contained 'soil biodiversity' as a key word.

→ ADDRESSING THE IMPACTS OF CHEMICAL POLLUTION AND NEGATIVE MANAGEMENT PRACTICES ON SOIL BIODIVERSITY

The Plant Protection Products Directive

The Plant Protection Products Directive[67], adopted in 1991, concerns the authorisation, placing on the market, use, and control of plant protection products in commercial use within the EU. It was repealed in November 2009 by a Regulation concerning the placing of plant protection products on the market[68].

This Regulation will be complemented by a Thematic Strategy on the Sustainable Use of Pesticides (COM(2006) 372)(European Commission 2006) and the Directive establishing a framework for Community action to achieve the sustainable use of pesticides, adopted in October 2009[69], which address risks resulting from the actual use of pesticides (mainly plant protection products).

In this context, the newest development is the increasing attention for the structural aspects of ecosystems, i.e. biodiversity. This is further presented and discussed in a new opinion paper published by the European Food Safety Authority (EFSA) working group (EFSA 2009). In particular, in this opinion paper it is proposed that the unacceptability of the effects of the pesticide application should be assessed under

[66] Council Directive 79/409/EEC on the conservation of wild birds (the Birds Directive)

[67] Council Directive 91/414/EEC of 15 July 1991 concerning the placing of plant protection products on the market

[68] Regulation (EC) No 1107/2009 of the European Parliament and of the Council of 21 October 2009 concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC

[69] Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides

field conditions, the exposure scenarios should describe the exposure in that soil layer that is the habitat of the species and processes of interest: the Ecotoxicologically Relevant Concentration (ERC). The ERC is hence a function of space and time, and the exposure scenarios should describe the exposure in the soil layer that is the habitat of the species and processes of interest, i.e. the ERC. The ERC varies in space and time, and the dimensions are governed by the communities that are present in the various soil profiles in the different regions in Europe. Aiming at modelling ERC soil values for model species, the development of relevant exposure scenarios should be done using an ecoregion approach. The underlying concept relies on the fact that different ecoregions support different soil communities and different guilds that may play a different role in supporting relevant soil services (EFSA 2009). Therefore, this paper concludes that exposure assessments in soil could be refined based on a novel underlying concept using eco-region maps to define ecologically relevant exposure scenarios. This approach could indeed be very useful for taking into consideration regional specificities and how these might influence the effects of pesticide application. Nevertheless, establishing the ERC for different ecoregions would require considerable research efforts, as the current knowledge on the effects of pollutants on soils remains relatively limited, as indicated before. Indeed, most studies simply show the susceptibility of particular organisms to certain pollutants, and establishing these ERC would require taking into account a larger variety of species and their interactions at different spatial scales.

→ **ADDRESSING THE IMPACTS OF NEGATIVE MANAGEMENT PRACTICES**

Agricultural policies

Agriculture preserves many specific genes, species and habitats, both above and below ground. A **Biodiversity Action Plan for Agriculture** (European Commission 2001) promotes **environmentally-friendly and sustainable farming practices** and systems that benefit biodiversity directly or indirectly, through agri-environmental measures and compensatory allowances within the CAP (Common Agricultural Policy). In particular, **agro-environmental measures** offer opportunities for the enhancement of soil biodiversity and the build-up of soil organic matter, through the support to organic farming, low or no tillage, the protection and maintenance of terraces, limited pesticide use, integrated crop management, low intensity pasture systems, the use of certified compost, etc. Indeed, agricultural land use (e.g. arable and grasslands) can have significant positive (liming in grasslands or low levels of disturbance) and negative (ploughing, overuse of agrochemicals or organic wastes) impacts on different components of soil biodiversity. Therefore, policies promoting certain practices will have an important effect on the biodiversity of soils.

Cross compliance, a horizontal tool for both pillars, which is compulsory since 2005 (as established in the 2003 Regulation establishing common rules for direct support schemes under the common agricultural policy[70]), plays an important role in soil protection, conservation and/or improvement. Under cross compliance rules, the receipt of the Single Farm Payment and payments for eight rural development measures under Axis 2 is conditional on a farmer's compliance with a set of standards. First, the **statutory management requirements** (SMRs; listed in Annex III of the 2003 Regulation establishing common rules for direct support schemes) create synergies between the Direct Payments Scheme and the need to ensure compliance with a

[70] Council Regulation (EC) No 1782/2003 establishing common rules for direct support schemes under the common agricultural policy and establishing certain support schemes for farmers.

number of relevant EU environmental Directives, including the Nitrates Directive. Second, the requirement to keep agricultural land (whether in productive use or not) in **good agricultural and environmental conditions** (GAEC; listed in Annex IV of the 2003 Regulation establishing common rules for direct support schemes) aims to prevent land abandonment and ensure minimum maintenance of agricultural land (SoCo 2009). The elements of GAEC specifically target protection against soil erosion, maintenance or improvement of soil organic matter, and maintenance of a good soil structure. The fact that GAEC requirements are defined at national level enables Member States to address soil degradation processes flexibly according to national priorities and local needs.

Nevertheless, the impacts of cross-compliance on soil conservation were limited by the fact that the measure does not cover all European agricultural land: the majority of measures are applied only to a portion of agricultural land (e.g. set-aside land, parts of arable land), and certain land-use types (e.g. forestry or land used to cultivate some fruits and vegetables) are currently not included in the scheme. Moreover, the extent to which soil is protected by the identified measures depends on the level of implementation for these different measures, which can be highly variable.

For example, the GAEC standards depended on local conditions, including soil and land use. Thus each Member States has flexibility in deciding which standard to implement, and over what spatial scale (national or regional level), resulting in great variability in the measures implemented across EU-27. The Annex IV standards addressed by most MS are the standard of 'protection of permanent pasture' and 'avoiding the encroachment of unwanted vegetation on agricultural land', which belong to the GAEC issues 'minimum level of maintenance'. Significant benefits for soil biodiversity can be achieved through a shift from arable land to permanent pasture, which allows soil organic matter to be restored and the prevention of soil erosion from the permanent plant cover. A further standard addressed by many Member States is 'arable stubble management'. Prohibition of the burning of plant residuals on parcels after the harvest provides strong benefits for the improvement and growth of soil organic matter. While some Member States (Austria, France, Greece, Finland, Ireland, Italy, Spain, Cyprus, Slovenia, Luxembourg, and UK) adopted measures to deal with all the soils threats considered by cross-compliance, improvements are still possible and are necessary in these Member States , such as more detailed requirements for soil erosion measures in Greece and Italy, the introduction of crop rotation systems in Spain and UK (England), more clearly defined measures in Cyprus or the enhancement of standards addressing minimum level of maintenance in Slovenia. The standard of 'appropriate machinery use' in order to maintain soil structure has only been implemented by 11 Member States, although soil compaction is a widespread problem across Europe (Hudec 2007). In a few new Member States (Estonia, Lithuania, Slovakia and Latvia), GAEC standards place a strong emphasis on minimum level of maintenance, but additional measures are poorly designed (EU 2007 evaluation of soil protection in Member States). In contrast, certain Member States have introduced cross-compliance measures that are beyond the scope of Annex IV or have adopted a very detailed range of GAEC measures (e.g. France, the Netherlands, Spain, UK (Wales) and Germany).

The 2003 Regulation establishing common rules for direct support schemes has been substituted in 2009 by a new Regulation following the Health Check agreement in 2008[71]. According to Article 4 of the new 2009 Regulation, a farmer receiving direct

[71] Council Regulation (EC) No 73/2009 of 19 January 2009 establishing common rules for direct support schemes for farmers under the common agricultural policy and establishing certain support schemes for

payments shall respect the statutory management requirements listed in Annex II and the good agricultural and environmental condition referred to in Article 6, which establish that Member States shall ensure that all agricultural land, especially land which is no longer used for production purposes, is maintained in good agricultural and environmental condition. Member States shall define, at the national or regional level, minimum requirements for good agricultural and environmental conditions on the basis of the framework established in Annex III.

6. 1. 2. LEGISLATION WITH INDIRECT SOIL BIODIVERSITY LINKS

The European Union has adopted several policies dedicated to water protection, pollution prevention and waste management, which contribute to some extent to soil protection by addressing specific threats indirectly (e.g. nitrates, genetically modified crops, etc.), and which consequently have an indirect effect on soil diversity.

The following table summarises some of these policies and initiatives that were considered to be most relevant, their indirect impact on soil biodiversity and interaction with other policies.

farmers, amending Regulations (EC) No 1290/2005, (EC) No 247/2006, (EC) No 378/2007 and repealing Regulation (EC) No 1782/2003

Table 6-1: EU and legislation with an indirect impact on soil biodiversity

Policy / Programme	Description	Land Degradation Processes Targeted by the Policy /Programme	Impact on soil biodiversity
EU Forest Action Plan[72]	The Action Plan focuses on four main objectives: to improve long-term competitiveness; to improve and protect the environment; to contribute to the quality of life; and to foster coordination and communication. Eighteen key actions are proposed by the Commission to be implemented jointly with the Member States during the period of five years (2007– 2011).	Agricultural land abandonment; Desertification; Forest fires; Associated risks to: Soil erosion; Soil compaction; Declining soil biodiversity, fertility and organic matter content.	Soil biodiversity tends to be greater in forests (compared to grasslands) (SoCo 2009)
Water Framework Directive[73]	The implementation of the WFD is a priority in order to address mismanagement of water resources with the objectives of preventing and reducing pollution, promoting sustainable water use, protecting the aquatic environment, improving the status of aquatic ecosystems and mitigating the effects of floods and droughts	Water and soil contamination; Associated risks to soil erosion, soil compaction, declining soil biodiversity, fertility and organic matter content.	Stimulating various initiatives by MS to reduce diffuse pollution from agriculture, including soil run-off from arable land
European Climate Change Programme (ECCP)- Programmes I and II	During the ECCP I, the working group on agriculture identified and discussed about 60 measures having potential for GHG emissions mitigation, some of which relate to soil management and environmentally friendly practices in the agricultural sector to promote carbon sequestration.	Declinig of organic matter content.	Soil structure (composed of pedality and porosity) and soil organic matter (amount and distribution) are also the main factors influencing soil biodiversity. SOM in a warming climate is a major concern as soil is the largest terrestrial pool of carbon. Declines in SOM may have an important impact on soil biodiversity, which is closely related to it (i.e. soils with an adequate amount of organic carbon have a good structure).

[72] Communication from the Commission to the Council and the European Parliament on an EU Forest Action Plan(COM(2006) 302 final)
[73] Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy

Policy / Programme	Description	Land Degradation Processes Targeted by the Policy /Programme	Impact on soil biodiversity
Nitrates Directive[74]	Designed to protect the European Community's waters against nitrate pollution primarily arising from the application and storage of inorganic fertiliser and manure from agricultural sources.	Water and soil contamination	Shaping national legislation on manure storage and management, levels of inorganic fertiliser use and other aspects of farm management. This Directive is expected to have positive effects on local and diffuse soil pollution by nitrates (and phosphates).
Sewage Sludge Directive[75]	Regulates the use of sewage sludge on agricultural land, by limiting and restricting applications in such a way as to prevent harmful effects on soil, vegetation, animals and man. To this end, it prohibits the use of untreated sludge on agricultural land unless it is injected or incorporated into the soil.	Soil and water contamination; Declining soil biodiversity, fertility and organic matter content.	Sewage sludge can have mixed effects on soil biodiversity. It can increase the input of organic matter and nutrients and also increase the contaminants load of the soil. Sludge is potentially contaminated by a whole range of pollutants. Some of these can be broken down into harmless molecules by soil microorganisms, whereas others are persistent including heavy metals. This may result in increasing levels in the soil with subsequent risks for soil microorganisms, plants, fauna and human beings.
Waste framework Directive[76] (2006/12/EC)	Requires Member States to take the necessary measures to ensure that waste is recovered or disposed of without endangering human health and without using processes or methods which could harm the environment.	Contamination of land and water	

[74] Council Directive

[75] Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture

[76] Directive 2006/12/EC of the European Parliament and of the Council of 5 April 2006 on waste

Policy / Programme	Description	Land Degradation Processes Targeted by the Policy /Programme	Impact on soil biodiversity
Landfill Directive[77]	The Directive's objective is to prevent or reduce as far as possible negative effects on the environment from the landfilling of waste, by introducing stringent technical requirements for waste and landfills and preventing/reducing the adverse effects of the landfill of waste on the environment, in particular on surface water, groundwater, soil, air and human health	Contamination of land and water	
Regulation on organic production and labelling of organic products[78]	The goal of this new legal framework is to set a new course for the continued development of organic farming. Sustainable cultivation systems and a variety of high-quality products are the aim. In this process, even greater emphasis is to be placed in future on environmental protection, biodiversity and high standards of animal protection.	Associated risks to: Soil erosion; Soil compaction; Declining soil biodiversity, fertility and organic matter content.	Organic farming and soil tillage both have a positive effect on soil biodiversity, through enhancing the amount of carbon and reducing soil disturbance, respectively. However, the trade-off can be diminished yield and increased weed or disease incidence.

[77] Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste
[78] Council Regulation (EC) No 834/2007of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91

6.2. POLICIES IN MEMBER STATES

→ **SOIL BIODIVERSITY PROTECTION**

To date, less than a dozen Member States have developed specific legislation on soil protection, mainly related to pollution and clean-up. Otherwise, soil protection is mainly addressed in more general Environmental Codes and Acts. In any case, soil biodiversity is generally not addressed in soil protection related legislation. In some Member States, however, some progress has been made in considering soil biodiversity in relation to sustainable agriculture and nature restoration practices.

→ **MONITORING SOIL BIODIVERSITY**

Biological monitoring is in most cases not explicitly mentioned in soil protection laws. Nevertheless, a few Member States are progressing in this specific field.

The current approach in the Netherlands for site-specific ecological risk assessment of soil contamination is based on the estimation of effects from the presence of contaminants in soil and hazardous concentration values. However, legal authorities are qualified to use additional methods when the current approach does not provide a clear result. Trends in assessment methods are directed to the application of biological tests, like bioassays and biological field observations. For this framework the so-called TRIAD approach was adopted and transformed to fit the quantitative data of different assessment tools. The TRIAD comprises three elements: an assessment of risks from the presence of contaminants in the soil and biota (substance directed approach), an assessment of risks from the results of bioassays with samples from the site, and biological field observations.

The German government has also taken action within the past years for the protection of soil organisms by defining trigger values for selected chemicals and the ecological classification and assessment of soils (see previous section 5.2 on monitoring schemes).

In any case, soil biology monitoring is not mandatory or required in existing legislation in Member States.

6.3. CONCLUSIONS BARRIERS AND RECOMMENDATIONS

6. 3. 1. CONCLUSIONS

To date, no legislation or regulation exists that is specifically targeted at soil biodiversity, whether at international, EU, regional or national level. This reflects both the conspicuous lack of attention that has generally been paid to soil protection at the policy level (Giller 1996; Wolters 2001) and the lack of awareness of the value of soil biodiversity.

Soil biodiversity is neglected even amongst conservationists. Despite representing almost a fourth of the total biodiversity on earth, soil organisms represent only 1%

of the IUCN red-listed species, and only eight soil species have CITES[79] protection worldwide (Decaens, Jimenez et al. 2006): three scorpions, four tarantulas and one lucanid beetle. This is not because soil species are not endangered, but because their status is overlooked. A similar trend is noticeable in the identification of biodiversity hotspots, which focus more on aboveground diversity than on belowground biodiversity. Other soil species, such as bacteria, fungi and protozoans, as well as insects, earthworms, ants and termites, are completely overlooked.

The growing body of EU and national biodiversity and environmental legislation and regulations could thus offer an untapped potential for the sustainable (therefore long-term) protection of biodiversity, which could be expanded and developed to account for the specific needs of soil biodiversity. However managing aboveground biodiversity may not necessarily do much for the protection of soil biodiversity. For instance, soil biodiversity will only benefit from plant diversity when plant diversity promotes the diversity in habitats, water dynamics, microclimate and resource quantity and quality. Plant diversity effects therefore are species or trait specific. Similarly, little is known about the role of ecological corridors for soil biodiversity. Although their effectiveness could be supposed to be much lower than for some aboveground species (Rantalainen, Haimi et al. 2008), it is important to highlight again the current limited knowledge about the impacts, interactions and dynamics of the soil community at the landscape level, and therefore the need to focus future research at the larger scale and not only at the plot scale. Indeed, field margins may also improve the connectivity of the landscape for soil organisms.

In contrast, the management of soil communities could be the basis for the conservation of endangered plants and animals. Indeed, as we have seen, soil biodiversity directly affects aboveground plant diversity, and thus indirectly the rest of aboveground communities. Moreover, soil communities are essential to the provision of several regulating services, such as climate, water, erosion and disease regulation, which are main drivers of aboveground diversity. Therefore, developing policies suitable for soil biodiversity could have a much greater scope than soil biodiversity *per se*.

Given that soil biodiversity affects several other environmental areas, a European dimension to soil biodiversity protection is essential. Indeed, soil biodiversity can be affected by a number of existing policies, related to soil, water, agriculture, climate change and nature. For instance, different initiatives exist in several Member States promoting environmentally-friendly practices, mainly in the forestry and agricultural sectors, which could have a beneficial indirect impact on soil biodiversity, often due to the economical constraints they are facing. A European dimension would serve both as a catalyser to raise the awareness on the benefits of soil biodiversity and therefore of its protection, as well as an integrator, to ensure related policies and regulations are harmonised and do not conflict with each other. The EU dimension also ensures that soil degradation is prevented, rather than shifted to other areas with less stringent legislation.

[79] Convention in Internation Trade of endangered Species of Wild Fauna and Flora, is an international agreement between governments. Its aim is to ensure that international trade in specimens of wild animals and plants does not threaten their survival.

6. 3. 2. BARRIERS

Soil biodiversity remains poorly known and understood to decision-makers, farmers, conservationists, and the general public. This lack of awareness is one of the main reasons for the current neglect of soil biodiversity.

There is a need to improve the recognition of the multiple benefits of soil biodiversity. Indeed, ethical considerations like the intrinsic value of soil biodiversity alone are not sufficient reasons for the protection of soil. Instead, the anthropocentric point of view dominates, with a focus on the protection of the functional features of soil. This requires placing a value on the essential ecosystem services provided by soil organisms, and estimating how much of this value is due to soil biodiversity. While many advances have been made in recent years, this work is still under development (see Section 3.8).

The lack of conservation ecology approaches for soil biotas, beyond being a major shortcoming of soil science, may also be one of the main factors that hindered the development of appropriate legislations to protect soil biodiversity. The fact that the knowledge of the structure and functions of the soil community was still limited and the ecosystem so complex resulted in the practical exclusion of biological aspects from legislation.

Finally, many of the benefits provided by soil biodiversity tend to be imperceptible to the unknowing eye: soil biodiversity steers several processes from below and its effects may act over long time scales of several years. This may limit the uptake of practices that could improve soil biodiversity. For instance, farmers are driven essentially by economics and not by environmental concerns. Thus uptake of management practices that have a positive influence on soil biodiversity and improve or sustain land productivity over the long-term may require incentives.

6. 3. 3. RECOMMENDATIONS

Policies for soil biodiversity protection can act either directly, on the cause of soil biodiversity and services loss, or indirectly, on the impacts of this loss. Acting on the causes of soil biodiversity loss is the most sustainable option, leading to long-term solutions. However, before deciding on the best types of measures for soil biodiversity protection, two questions arise for the policy-maker:

- **Is all soil biodiversity necessary?** Indeed, given the extraordinary diversity of soil organisms and the fact that several organisms can perform the same functions, it could appear as though loss of some species would be of little consequence to the functions of ecosystems. However, as illustrated in Box 4, this is not the case. The specific roles of each soil organism are not yet fully understood, but it is clear that their diversity provides a form of insurance, increasing the resilience of soil systems. This is critical in the context of growing climate and land use change, which are the two main drivers of soil biodiversity loss (see Section 4.).
- **At what privileged scale should soil biodiversity protection be performed?** As we have seen, soil biodiversity is affected by a hierarchy of spatio-temporal processes, each of which is characteristic of a specific functional group (see Section 2.). This offers managers a clear framework, within which they can choose among direct action on the functional group

affected, or indirect action at higher spatio-temporal scales than that of the functional group affected. Moreover, the scale at which the driver of soil biodiversity loss acts also has to be considered: thus seasonal effects need not be addressed in the same fashion as long-terms trends in climate change.

As indicated before, so far there is not a single legislation dedicated to the protection of soil biodiversity. Furthermore, existing biodiversity legislation and initiatives (e.g. the Habitats Directive), do not fully recognise the importance of soil biodiversity and its contribution to the resilience of ecosystems and provision of several regulating services. This reflects the lack of awareness for soil biodiversity and its value, as well as the complexity of the subject.

To address this gap, and further promote soil biodiversity protection, one first step would consist in establishing the state of soil biodiversity and assessing the risks of soil biodiversity loss. This, on the other hand, would require the development of reliable indicators, so that long-term monitoring programmes can be set up. There exist a host of simple indicators that target a specific function or species group. Nevertheless, widely accepted reference sets of indicators, reference ecosystems and standardised sampling protocols are still missing and therefore further research would be necessary. The ENVASSO project already offers a set of minimum reference indicators of soil biodiversity and could serve as the basis for further improvements.

The **JRC** created in mid 2008 a **Biodiversity expert group** to provide advice and assistance regarding its scientific and technical activities in support to EU soil policy making and research[80]. Its **role in awareness raising could perhaps be further promoted** or its functions extended, so as to also coordinate further research related to the development of appropriated indicators and monitoring methods.

In this context, **the introduction of mandatory monitoring requirements could contribute**, as it has happened in other fields (e.g. the requirements for the monitoring of surface water status under the WFD), **to trigger the development of adequate indicators and monitoring methodologies**. Furthermore, this is key in order to improve awareness on the central role of soil biodiversity and for developing capacity-building among farmers to promote biological management.

In this regard, the proposal of a Soil Framework Directive could provide the legislative framework for introducing specific monitoring requirements. So far, the proposal of a Directive requires MS to identify and carry out an inventory of the sites which according to their assessment are posing a significant risk to human health and the environment, stemming from soil contamination. This would be complemented by the obligation for the seller or prospective buyer to provide a soil status report for any transaction of land where a potentially contaminating activity has taken or is taking place. Therefore, so far, biological aspects are not taken into consideration for determining the quality of soil, nor are there requirements to monitor the ecological status of soils. Therefore, the proposal could be revised to specify the need for monitoring, and require the inclusion of certain soil biodiversity parameters (see previous section on indicators). Much may

[80] Further information on this group available at: eusoils.jrc.ec.europa.eu/library/themes/biodiversity/wg.html

be learned from the assessment of the quality of aquatic ecosystems required under the WFD (in terms of the steps and the approach followed for implementation).

Currently, EU-wide monitoring of biodiversity occurs under the Habitats Directive and to assess progress towards achieving the EU Biodiversity action Plan (BAP), but as commented before, soil biodiversity is largely overlooked. **The Commission could consider the extension of the annexes of the Habitats Directive** to complete the so far limited list of soil-based habitats (i.e. mainly peat soils and bogs so far covered) that require special protection. Added to this, the list of species under Annex II could be revised and further extended, so as to better cover soil species. This could contribute, for example, to promote the consideration of soil taxa during impact studies necessary prior to the establishment of infrastructure (as required under the Environmental Impact Assessment Directive81). In France for example, the discovery of the beetle O. eremita, registered in annex II of the "Habitats" Directive, during entomological surveys prior to a road construction, justified the suspension of engineering works over more than 3 years (Decaens, Jimenez et al. 2006). **Further considering soil biodiversity under the Habitats Directive would contribute to increase awareness about its important role, but would perhaps not address soil biodiversity protection in a systematic and integrated manner**.

As seen before, several policies addressing specific threats to soil protection and therefore having an indirect effect on soil diversity are now in place (e.g. the Nitrates Directive, the Directive on the sustainable use of pesticides, the Regulation concerning the placing of plant protection products on the market, the Sewage Sludge Directive, etc.). In other to further improve the protection of soil biodiversity, another possibility would be to specifically require in these regulations and directives to take into consideration the impacts of the different threats on soil biodiversity, and in particular contamination (e.g. chemical substances, nutrients, etc.). To this end, the use of the concept of Ecotoxicologically Relevant Concentration (ERC), as proposed by the European Food Safety Authority (EFSA) could be most useful as is a function of space and time and allows taken into consideration the communities that are present in the various soil profiles in the different regions in Europe (i.e. use of ecoregion approach).

Furthermore, more attention should be given to developing and refining existing soil biodiversity and ecosystem management opportunities under different land uses and socio-economic conditions, and on integrating those strategies within the existing bodies of legislations that address soil management, such as the cross-compliance requirements established under the CAP (Common Agricultural Policy) since 2005. Cross-compliance includes so far minimum standards of good agricultural and environmental conditions related to the maintenance of soil organic matter levels (e.g. crop rotation, arable stubble management), the protection of soils against erosion, and the maintenance of carbon sinks (e.g., protection of permanent pasture). Agri-environmental measures provide support

[81] Directive 2003/35/EC of the European Parliament and of the Council of 26 May 2003 providing for public participation in respect of the drawing up of certain plans and programmes relating to the environment and amending with regard to public participation and access to justice Council Directives 85/337/EEC and 96/61/EC

for measures which go beyond the mandatory management practices (e.g. conservation agriculture). Further requirements could include the implementation of integrated soil biological management to emphasise the importance of protecting soil biodiversity. In this regard, it might worth mentioning that the **Mid-Term Assessment of Implementing the EU Biodiversity Action Plan (European Commission 2008)** that was published in December 2008 put forward the need to further improve the overall cross-sectoral policy integration, and adopt new legislation such as the proposed Soil Framework Directive.

Finally, given the differences among belowground and aboveground biodiversity, policies aimed at aboveground biodiversity may not do much for the protection of soil biodiversity. In contrast, **the management of soil communities could form the basis for the conservation of many endangered plants and animals**, as soil biota steer plant diversity and many of the regulating ecosystem services. **This aspect could be taken into account or highlighted in future biodiversity policies and initiatives.** For example, a new strategy for biodiversity protection post-2010 is expected in 2010, which could recognise and promote soil biodiversity protection by acknowledging the import role that soil biodiversity plays for the conservation of above ground biodiversity. The first step towards achieving an integrated conservation approach in this line was taken at the Athens Conference on Biodiversity Protection Beyond 2010 in April 2009. The Message from Athens highlights several important points, based on the objectives of the EU BAP. These include the need for a more integrated and coherent policy framework, more progress in integrating biodiversity protection into other sectoral policies, and coherent and coordinated spatial planning. The Athens conference was followed up by a high-level meeting entitled 'Visions for biodiversity beyond 2010 – People, ecosystem services and the climate crisis' in Sweden in September 2009. Several strategic principles were developed at this meeting to frame biodiversity protection post-2010, in conjunction with the message from Athens, including the need to ensure that the value of biodiversity is integrated into economic decision-making, based on the Economics of Ecosystems and Biodiversity (TEEB) study, and to manage ecosystems in order to allow species and habitats to be resilient to climate change and other environmental pressures. The priorities for biodiversity protection post-2010 would include the conclusions of both these conferences, which are in line with the need to promote soil biodiversity protection and highlight the need to focus on the main drivers of soil biodiversity loss, namely land use and climate change, in order to provide long-term sustainable solutions.

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7. RESEARCH NETWORKS

Soil biodiversity research centres			
Name		**Aim**	**Link**
Soil Biodiversity and Ecosystem Functioning Lab	Department of Biology at Colorado State University	Research on soil biodiversity and ecosystem functions in the following fields: Antarctic habitats, Grassland habitat and Global patterns.	www.rydberg.biology.colostate.edu/sites/walllab
AfNet	African Network for Soil Biology and Fertility	African network of the CSM-BGBD	www.tsbf.org/africa.html
BIOASSESS	The BIOdiversity ASSESSment tools project	Toolbox for assessing impacts of policies on biodiversity in the EU, and impacts of land-use change	www.nbu.ac.uk/bioassess.html
CONSIDER	Conservation of Soil Biodiversity	Assess the effects of agriculture and environmental policy with respect to the diversity of resources residing in soils 2003-2007	www.zi.ku.dk/consider/index.html
CSM-BGBD	Conservation and Sustainable Management of Below-Ground BioDiversity	Respond to the urgency to slow down losses in soil biodiversity and better assess the potential uses of soil biodiversity in ecosystem management and bioprospecting	www.tsbf.org/csm_bgbd.htm
DIVERSITAS	Special Target Area of Research 6: Soil and Sediment Biodiversity	Coordinate information on how soil and sediment species composition and community structure influence ecosystem functioning. DIVERSITAS will facilitate and coordinate the sharing of ongoing research and syntheses efforts among scientists	
GCTE	Global Change and Terrestrial Ecosystem	Predict the effects of changes in climate, atmospheric composition, and land use on ecosystems, including soils and biodiversity (Focus4 - biodiversity, land-use changes)	www.gcte.org

IUCN - Soil	IUCN Environmental Law Programme (ELP)	Improvement of environmental law and policy for the sustainable use of soils, particularly for the conservation of biodiversity	www.iucn.org/about/work/programmes/environmental_law/elp_work/elp_work_issues/elp_work_soil/index.cfm
SARNET	Tropical Soil Biology and Fertility Programme-South Asian Regional Network	South-Asian network of the CSM-BGBD	www.tsbf.org/asia.htm
Soil Biodiversity - NERC Thematic Programme	UK	Integrated research programme on the biological diversity of soil biota and the functional roles played by soil organisms (1997-2004). Online database	www.soilbio.nerc.ac.uk
SOWAP	SOil and WAter Protection	Assessment of the impacts of conservation tillage on the biodiversity above and below-ground, in Belgium, Hungary, and the UK	www.sowap.org/results/biodiversity.htm
Landscape research / Manaaki Whenua	Research centre in New Zealand	Research on soil biological and chemical interactions	www.landcareresearch.co.nz/research/research_details.asp?Research_Content_ID=85
SWCS	Soil and Water Conservation Society	SWCS is a non-profit scientific and educational organization that serves as an advocate for conservation professionals and for science-based conservation practice, programmes, and policy	www.swcs.org
evenor-tech			www.evenor-tech.com
Soil science research centers			
Name		**Aim**	**Link**
CIAT	Consortium for integrated soil management	Latin American network of the CSM-BGBD	www.tsbf.org/latin_america.htm
ENVASSO	ENVironmental ASsessment of Soil for mOnitoring	Holistic approach to soil protection through the robust and defendable selection of criteria, thresholds and indicators based on harmonised approaches to soil information collection, analysis and management	www.envasso.com

ISRIC	World Soil Information Centre	Hosts the World Data Centre for Soils, to serve the scientific community as custodian of global soil information	www.isric.org
GlobalSoilMap.net	Global soil mapping data	Consists of a consortium that aims to make a new digital soil map of the world using emerging technologies for soil mapping and predicting soil properties at fine resolution.	www.globalsoilmap.net/
Soil carbon fluxes and land-use changes	UK	Develop a system based on high-resolution spatial soils and land-use data coupled to a dynamic simulation model to predict carbon fluxes from soils resulting from land-use change.	www.rothamsted.bbsrc.ac.uk/aen/ukcarbon/
SOILSERVICE		Understand how economic production drivers will change current and future use of soil-related ecosystem services and how they affect the resilience and resistance of ecological-economic systems	www.kem.ekol.lu.se/soilservice.html
SOMNET	Soil Organic Matter Network	Facilitate scientific progress in the prediction the effects on Soil Organic Matter of the changes in land-use, agricultural practices, climate through modelling	www.rothamsted.ac.uk/aen/somnet/index.htm
STAMINA	STAbility and Mitigation of Arable systems in hilly landscapes	Assist decision-making for sustainable farming	www.rothamsted.bbsrc.ac.uk/aen/stamina/
Soil biodiversity research projects			
Name		**Aim**	**Link**
SOLOBIOMA I	Diversity of soil biota under anthropogenic influence	The aim is to assess the diversity of soil biota in the context of understanding ecosystem function and its relationship to human impact in forests and agroforestry systems in Paraná State, Brazil.	www.solobioma.ufpr.br/

The PROSOIL project	Aberystwyth University	The IBERS PROSOIL project has been developed to look at how typical grassland farming practices in both conventional and organic farming systems can affect soil health, which will be evaluated partially by monitoring earth worm populations	www.aber.ac.uk/en/ibers/research/prosoil/
LTSE	Long-term soil ecosystem studies	International resource centre for soil research project	www.ltse.env.duke.edu/
CSM-BGBD	Conservation and Sustainable Management of Below-Ground Biodiversity	The Project's main goal is to generate information and knowledge that can be used to better manage and conserve BGBD in tropical agricultural landscapes	www.bgbd.net/
DIANA	soil DIversity In Austrian NAtural forests	DIANA is a cluster of research projects supported by the Lebens Ministerium (Austrian Federal Ministry for Agriculture and Forestry, Environment and Water Management).	www.bfw.ac.at/rz/bfwcms.web?dok=5833
LAZBO		Long-Term Monitoring of Soil Physical and Biological Properties Pilot Project	www.bafu.admin.ch/boden/00972/00991/index.html/
CréBeo Project	CréBeo (Irish) = 'Living soil'	The primary objective of this project is to increase scientific knowledge of soil biodiversity in Ireland.	www.ucd.ie/crebeo/
Bioindicateurs de qualité des sols		Development of soil bioindicators to characterise soil degradations (either chemical degradation and agricultural practices)	National research programme managed by ADEME (French Agency for Environment and Energy Management) Contact : antonio.bispo@ademe.fr
ECOMIC-RQMS	Microbial biogeography	To characterise microbiological parameters and apply them to the French soil monitoring network (RQMS)	www.international.inra.fr/press/national_inventory_and_mapping_of_microbial_biodiversity_in_soils
RQMS Biodiv		To create an inventory of soil biodiversity in Brittany	www.sols-de-bretagne.fr/
Wisconsin Integrated Cropping System Trail Project	University of Wisconsin	Investigating soil health and biodiversity	www.cias.wisc.edu/wicst/research/coretrial/soil.htm

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