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Considerations for the use of soil ecological classification and assessment concepts in soil protection

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Abstract

In comparison with other environmental compartments, the protection of soil was identified relatively late as an important issue. Even later, policy makers and the general public considered the ecological functions of the soil. However, recently, the central role of these functions, as well as the importance of the soil organism community, has become obvious in several countries in Europe and in the European Union itself. In this contribution, basic considerations referring to use of ecological endpoints for the classification and assessment of soils are discussed. The article begins with two goals of protection: to protect soil so that it can function and to protect soil as a habitat for organisms necessary for these functions. Afterward, the quality of soil from is discussed from a biological point of view. Finally, on the basis of experience from limnology and botany, recommendations are made for monitoring programs. In other articles of this series, details of the entire approach to ecological classification and assessment are outlined.
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1. Introduction

In the discussion of environmental protection, soil was identified as a compartment of its own long after the recognition of water and air (von Lersner, 1982). There are several reasons for this underrepresentation of the "third compartment", soil.

Scientifically, soil is often described as part of a larger system (e.g., as one small layer from a geological or geomorphological point of view or as the substrate for vegetation units). In addition, up to now soil science has often defined its topic very, restrictively, i.e., taking ecological aspects only slightly into account.

In public opinion, air and water are more important to humans than soil, because they are used directly for breathing and drinking. Soil, on the other hand, only indirectly influences human life as a substrate for the production of food.

Legally, soil usually is not owned by society as are air and (most) water bodies, it belongs to individuals who have their own private rights and interests in using the soil.

Consequently, the ecological classification and assessment of terrestrial field sites, in theoretical approaches as well as in practical work like landscape planning, nature (site and soil) conservation, and ecotoxicology (from laboratory tests to monitoring methods), clearly lags behind limnology and related sciences including aquatic ecotoxicology.

Before discussing why and how soils should be protected, soil must be defined to avoid misunderstandings.

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Soil: The upper layer of the earth crust composed of mineral parts, organic substance, water, air and living matter (International Organisation for Standardisation, (ISO), 1996).

This definition definitively includes soil organisms as an integral part of soil. The term "living matter" can be specified as "organisms" or even better "plants, microorganisms, animals and their interactions (i.e., functions)" as done, for example, in the German Plant Protection Law (PflSchG, 1998). This is not a trivial statement as can be seen from the following definition published by the Soil Science Society of America.

Soil quality: the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Doran and Parkin, 1994; Karlen et al., 1997).

"Within ecosystem boundaries" means that there are differences between soils, partly due to soil use. The definition given above focuses on soil functions and benefit to humanity. This viewpoint may be assumed only as long as an intact environment and the sustainable use of its resources are acceptable as a political aim by society. However, for reasons of clarity, it is preferable to state clearly that the ecosystem itself should be maintained, even if there is no direct and immediate advantage for humans. Most obviously, the argument for conservation of biodiversity because of its intrinsic value lacking here should be added. Soil organisms are worthy of protection as are all other organisms: that they are small, taxonomically difficult, and problematic to monitor should not devalue their status (Dunger, 1994). However, it is correct that there are no absolute quality criteria, and the quality of each soil must be determined relative to natural variability in site properties, e.g., soil type, soil use, and climate.

In fact, soil can have many functions. Most specifically, a number of processes occur in the soil, such as degradation of organic matter, nutrient cycles (carbon, nitrogen, sulfur, etc.), and degradation of pollutants (very important for groundwater quality). Moreover, soil plays an important role in hydrology; it carries the buildings and infrastructure and it is the substratum for agriculture and nature. And last but not least, the soil contains a huge amount of organisms of high biodiversity that interact closely with the above-ground organisms. They have a great impact on the processes mentioned, and as a consequence, on plant growth and the presence and abundance of pathogens in agricultural crops. In particular, the last function, i.e., soil as a habitat for organisms, is covered in this article.

For these reasons, environmental policy is increasingly being focused on sustainable use of the soil. That means that current use of soil should meet the needs of the present without compromising the ability of future generations to meet their own needs, nor compromising the use of soil elsewhere. One outcome of this political shift is the recent implementation of soil protection laws or ordinances in various European countries (e.g., Germany: BbodSchG, 1998). In addition, the European Union addressed this topic officially for the first time in 2001 although it has been organizing discussions on soil protection since 1999 (Moll, 1999; European Union (EU), 2002; cf. Römbke et al., 2005).

Within the EU, OECD, FAO, and other European countries, there are initiatives to agree on an ecological characterization of soil. An actual ecological characterization of soil is in harmony with the Biodiversity Programme of the UNEP World Conservation Monitoring Centre and with the Clearing House Mechanism (CHM) of the Convention on Biological Diversity (CBD). Such projects are in fact relevant to the Biological Inventory Systems Component of The Man and the Biosphere (MAB) Programme, and fit the COP4/1 decision regarding the development of the GBIF "to allow people in all countries to share biological diversity information and to provide access to critical authority files." Observing the development of the latest Johannesburg Summit (Rio 10+), we believe that, in time, the expected results can be fully incorporated into the UNESCO strategies (Breure et al., 2002a), notably DIVERSITAS. But what are we trying to conserve? Biological conservation has probably become the easiest target for governments, UN agencies, and development organizations. As this question has engendered the greatest amount of agreement (we saw an entire international convention devoted to it), it may become "misleading" due to its omnipresence. This discussion takes place in the context of an ongoing and dramatic loss of soils in Europe as well as globally. For example, by 1985, 12.5% of the area of Germany has been devoted to human activities like building houses or traffic purposes (Blume, 1990). Thus, soil is scarce, especially in countries like The Netherlands, where there is no place to keep space vacant. Good stewardship makes demands on the sustainability of soil management, so that (over)exploitation of the soil is not being shifted toward future generations. In The Netherlands, soil use changes relatively often: agricultural land becomes nature area, agriculture is extensified, and new nature area is planned. In comparison to other European countries, where some (more or less) pristine soil exists and where land use changes occur less frequently, this situation even increases the pressure on the functioning and resilience of the soil.

2. Protection of soil as a habitat for soil organisms

The overall goal in soil protection is to maintain a "good soil quality," meaning that certain soil functions

should be fulfilled. Among those possible, we focus here on the ecological functions of the soil (also called life support functions, being essential to both environmental health (Schouten et al., 1999) and human health (cf. Limburg et al., 2002), in particular on the soil as a habitat for soil organisms. This function can be defined so that the structure (i.e., biodiversity of plants, microorganisms, and animals) and function (i.e., processes in soil depending on the interactions between these organisms and their environment) of the soil biocoenosis are not impacted. Usually, this function is best fulfilled in soils that are "intact, undisturbed and natural," a situation rarely found in Europe. In other words, it is implicitly accepted that humans influence soil ecosystems.

Therefore, looking at the function of soil as a habitat for soil organisms, it is clear that the protection goal in detail depends on the use of that soil: For what purpose is the soil used? And what is the function of the soil now and in the future? The specific soil biocoenosis required in national parks will be different from that one required at a site used for agriculture. For example, the German Soil Protection Law has, on one hand, maintenance of nature as a standard when assessing the habitat function in protected areas, and, on the other hand, does not assess nature at all when looking at industrial areas.

Of course, any assessment of the protection goal "soil as a habitat for soil organisms" requires detailed knowledge of the structure and function of the soil biocoenosis in all ecosystems to be assessed. In other words, research on soil biology and ecology (including taxonomy) is prerequisite to the assessment of this function. In addition, this underlines the point that soil organisms are an integral part of the soil and not just "visitors," as can be read until today in pedological papers (e.g., Auerswald, 1998). However, it is also clear that organisms cannot change the soil at a certain site completely but simply within the restrictions given by the geological, pedological, and climatic conditions at this site. Therefore, soil organisms are influenced by soil properties but the same is true vice versa (e.g., Jungerius et al., 1995). The best example is the humus layer in forests, which strongly depends on the activity of earthworms and/or other soil animal groups, but at the same time provides a habitat for these animals due to its form. For many widespread annelid species, in fact, critical moisture and pH optima are well known (Graefe, 1998; Graefe and Beylich, 2003; Jänsch and Römbke, 2003).

3. Soil organisms

Before discussing the criteria and requirements for soil ecological classification and assessment concepts, the organisms themselves as well as the processes they control or perform are briefly summarized. In this context it is important to distinguish between the structure and the function of the soil biocoenosis or community on the ecosystem level, which can be defined as follows (Odum, 1985; Schäfer and Tischler, 1983):

Structure: composition of the soil biocoenosis (biodiversity), traditionally described on the species level (i.e., abundance, biomass, diversity, and dominance).

Function: Biologically determined processes in the soil ecosystems, based on the interaction of its different components (i.e., nutrient cycling, community respiration, or, most prominently, organic matter breakdown).

3.1. The structure of the soil biocoenosis

Soil biota are thought to harbor a large part of the world's biodiversity and to govern processes that are regarded as globally important components in the cycling of organic matter, energy, and nutrients (e.g., Griffiths et al., 2000). Rough estimates of the soil biodiversity indicate several thousands of invertebrate species apart from the largely unknown microbial and protozoan diversity (e.g., 1500–1800 invertebrate species in a German beech forest (Weidemann, 1986)). By far the most dominant groups of soil organisms, in terms of numbers and biomass, are microbial organisms, i.e., bacteria and fungi. Estimates of the number microbial species (genotypes) in the soil range from 10^4 to 10^5 per gram (Torsvik et al., 1990; Dykhuizen, 1998). Besides these organisms, soil ecosystems generally contain a large variety of animals, like protozoa (bacterivores, omnivores, predators), nematodes (bacterivores, fungivores, omnivores, herbivores, predators), micro-arthropods such as mites (bacterivores, fungivores, predators), and collembola (fungivores, predators) and enchytraeids and earthworms (both being mainly saprophagous). In addition, a large number of macrofauna species (mainly arthropods like beetles, spiders, diplopods, and chilopods as well as snails) are living in the uppermost soil layers, the soil surface and the litter layer.

When discussing the possibility of using soil organisms as indicators of soil quality, one important difference between microorganisms and animals in the soil must be mentioned: Usually, when studying microbes (in particular bacteria), their functions (e.g., soil respiration or activity of a certain enzyme) are measured, while the methods for studying the structure of the microbial community (e.g., species or physiological groups) have not been available until recently. This situation is in the process of changing, as elaborate biochemical and genetical methods have been developed. For soil animals the situation is just the opposite: Here the main organizational level studied is species (or, in the case of very difficult species groups, a slightly higher taxonomical level like genus or family); i.e., the structure of the community, not its function is

measured. This difference must be kept in mind when using soil organisms in soil quality classification and assessment.

Anthropogenic activities clearly influence soil biota, most strongly in industrial or urban areas where only very few species (if at all) can survive. Modern agricultural practices characterized by high levels of inorganic fertilizer additions, use of pesticides, and soil tillage are known to affect the diversity of the soil community in fundamental ways, leading to the extinction of various groups of organisms. Development of more sustainable forms of agriculture may counteract these negative effects (Brussaard et al., 1988). Also, soil diversity in natural ecosystems is increasingly gets under pressure, because of water management regimes, acidification, and large infrastructural projects. There are also strong indications that the development of new natural ecosystems is seriously hampered by "inadequate" soil biodiversity (Van der Putten et al., 2001).

The importance of soil biodiversity becomes obvious when looking at such activities as the Global Taxonomy Initiative (GTI-CBD), the main goal of which is the development of a global taxonomic information system for animals, plants, fungi, and microorganisms. Within this framework, it is noteworthy that Wall et al. (2001b, p. 115), in their Summary of Research Priorities for Soil and Sediment Ecosystems, claim the *highest* priority for the development of "a strategy to increase the number of taxonomists working with below-surface invertebrates, as there are presently very few of these specialists globally."

3.2. Functions of the soil organisms

With its large diversity and complexity the soil community has a strong impact on soil processes and the way in which these processes may vary in time and space. Most noteworthy are:

decomposition of organic matter and, thus, regulation of the cycling of nutrients;
fixation of nitrogen from the air, making it available for plants;
degradation of anthropogenic compounds like pesticides;
stabilization of soil aggregates, specifically through the building of clay–humus complexes;
improvement of soil porosity as a result of burrowing activities;
influencing of soil pH by nitrification and denitrification, resulting in mobility changes of heavy metals;
influencing heavy metal mobility under different redox conditions (e.g., in the sulfur cycle, especially important in areas with fluctuating water tables);
being prey for many above-ground organisms.

As these processes also determine nutrient availability for uptake by plants, the below-ground decomposer food web interactions also influence above-ground primary productivity and carbon sequestration (Scheu, 2001; Tscharntke and Hawkins, 2001). In fact, plant productivity appears to be increased by reduced turnover of the microbial biomass due to stabilized carbon content and soil pH. Vascular plants are known to be extremely sensitive to microbial symbionts. Mutualism in new environments is key to competitiveness (and successful migration due to human activities and global change) for most plants, as microbial symbionts are required to induce N fixation.

Soil organisms are assumed to be directly responsible for soil ecosystem processes, especially the decomposition of soil organic matter and the cycling of nutrients (Bardgett and Chan, 1999). For example, the soil biomass is known to process more than 100,000 kg of fresh organic material each year per hectare (25 cm top soil layer) in many agricultural systems. This processing includes the decomposition of dead organic matter by microbes, as well as consumption and production in the soil community food web (Hunt et al., 1987; De Ruiter et al., 1993, 1998). The soil food web is defined as the structure and interactions across and between the communities of soil-living organisms, which are linked by conversions of energy and nutrients as one organism eats another. Therefore, most food web models merely provide a way to connect the dynamics of populations to the dynamics in ecological pathways within the cycling of matter, energy, and nutrients (Pignatti, 1994). The food web models have to use the observed abundance of the various groups of soil organisms as input variables. By incorporating the multivariate habitat response relationships for the occurrence of organisms, they aim at assessing the effects of (changes in) abiotic and management conditions in scenario studies. Although this approach enables food web models to provide a reliable way to analyze the dynamics of soil populations in the context of soil community structure as a whole, these models do not unravel the key problem of sustainability as an effect of environmental change in terms of stability and biodiversity. From this point of view, the most important aspect remains *the reference*.

3.3. The interaction between structure and function

From the viewpoint of functional biodiversity, an indicator of soil life support functions is best based on the measurement of processes. However, soil processes fluctuate strongly in time and space. Establishing a mean annual value for a process requires an intensive sampling program, and is difficult to establish on a national scale. Therefore, it is more practical to use the species composition, aggregated in functional groups, as an indicator for processes. The relationship between

species composition and ecosystem functioning is difficult to quantify. When species disappear, others may become more dominant and take over a link in the overall process. It is possible that a process will continue while species composition has changed or degraded. So the preservation of biodiversity cannot be guaranteed by measuring process values. Processes are too general or insensitive as an early warning indicator.

However, when discussing the species composition in soil, one has to remember that due to the extreme heterogeneity (both spatially and temporally) of soil, the number of species is often very large (Bardgett, 2002). Therefore, it is often very difficult to identify simple and predictable relationships between the diversity and function of soils. Soil functions seem to depend mainly on the complexity of biotic interactions (e.g., between the components of the food web). However, in some cases, individual species (often defined as key species or ecosystem engineers) can dominate the function of the whole system (e.g., large anecic earthworms in many nonacidic soils (Lavelle et al., 1997) or enchytraeid worms in acid forest soils (Laakso and Setälä, 1999). Therefore, it is important to distinguish between species when assessing the habitat function of the soil.

In general, heavy pollution or disturbances select for a few resistant (tolerant) species. In such a situation, the ecological basis for processes has become very narrow. When the resistant species disappear, a process stops and the life support function is permanently affected. The indicator system for life support functions of the soil is based on the following hypothesis: The threat of vital soil processes can be expressed by comparing the number of species in functional groups of a certain area with its reference (undisturbed locations). A process is assumed to continue to exist with fewer species, in which case the risk of instability and uncontrolled fluctuations will increase.

The assessment of any actual interaction between structure and function in the field suffers for the reference choice. Human use of the landscape has brought about profound changes above and below ground, and has even provoked global changes. In this context, knowledge of structures and functional processes becomes increasingly relevant. If the food web perspective is added to the previous characters, the most immediately perceptible, sensitive elements for the characterization of a (soil) community are obtained: species composition, food web structure, and rhythmic courses (individual phenology). Rhythmic courses are universal components of the natural variability in living organisms (and between trophic levels), making it difficult to identify cause–effect relationships between stress (e.g., pollutants from remote sources) and measurable physiological processes. Soil communities are complex, open systems with exchange of matter and energy controlled by feedback loops (cf. Odum, 1971). The presence of these loops enables the ecosystem to react unpredictably to environmental stress. Dynamic models may provide unique possibilities to assess the ecological risk aside from our perception (i.e., "expert judgment"), due to a categorization of ecological processes aside from a static reference.

Although in Central Europe plenty of attention has been given to the effects on above-ground vegetation induced by humans during recent history (e.g., Mucina et al., 1993; Zechmeister and Moser, 2001), as well as during the last 2500 years (long-term studies by Küster, 1988, and Vera, 2000), still very little is known about human-induced cascade effects in below-ground systems. The anthropogenic, ruderal species carefully described by these authors for the upper vegetation still have no analog taxa for the soil. This confirms that a tremendous lack of knowledge of soil organisms, albeit sufficient evidence of environmental impact on the composition and abundance of soil biota at different trophic levels, exists (Wall et al., 2001a), and confirms the extension to which species are "redundant". Rapport et al. (1998) suggest resilience (measured in terms of a system's capacity to maintain both structure and functions under environmental stress), organization (diversity of interactions within a system, mainly the loops described before), and vigor (productivity) as indicators of ecosystem health. Also, their approach requires a careful selected reference for any local scale assessment, as the concepts of resilience, structure, and vigor (processes) rely entirely on dynamic trends.

3.4. Synergy and plant community composition

One of the main monitoring approaches in terrestrial ecosystems is vegetation science. Field surveys (quantitative *relevés* according to empirical scales of plant cover and dominance) are supposed to provide a valuable tool to assess the vegetation in space and time. Several empirical scales have been developed for a quantitative study of the plant cover: the most common are the Braun-Blanquet scale in Europe (Braun-Blanquet, 1926) and the Domin scale in the United Kingdom (Dahl and Hadac, 1941). Aside from the kind of scale used to estimate the cover of *all individuals of a given species* within the area investigated, the surface of a plot varies with the kind of vegetation (i.e., the landscape structure). Rodwell et al. (2000) suggest 2×2 m for short, herbaceous vegetation (e.g., dwarfshrub heath), 4×4 m for taller herb communities (specie-rich), 10×10 m for very tall herb communities (species poor), 50×50 m for shrubs and woodland (both tree canopy and understory). Riparian vegetation, or linear vegetation (ditches), has to be sampled in 10-m-long strips (open vegetation) or in 30-m- long strips (if shrubs and/or trees occur). Alas, phytosociology implies a certain degree of autocorrelation in space between environments in close

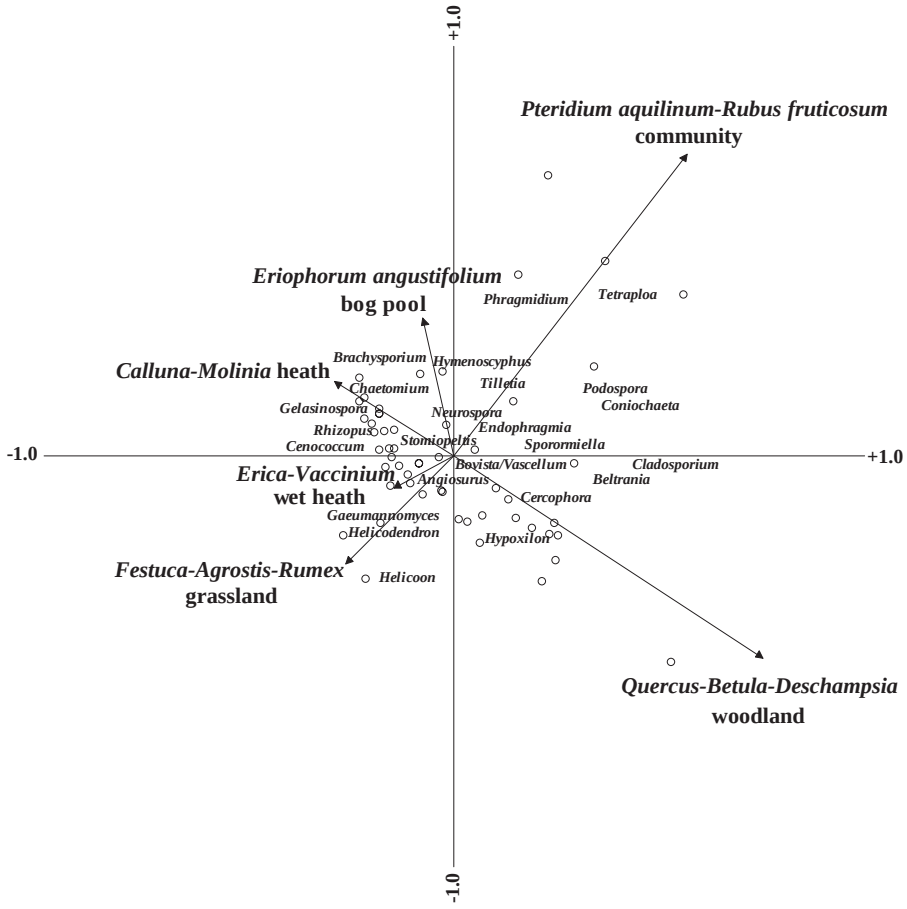


Fig. 1. Redundancy analysis of the litter in two Dutch heathlands, showing the above-ground vegetation trends, in boldface (on the right the disturbed types), and the fungal response to eutrophication, in italics (data from Mulder et al., 2003). Note the dominance of human-induced stressors on the right, shrublands as response to grazing/mowing (*Rubus*) and fire (*Pteridium*), and (coppice) woodlands as response to failed management (cf. Mulder and Janssen, 1999).

proximity. Thus, it may become extremely difficult to reach a level where these different scales in classification are comparable. Furthermore, as climate and edaphic properties determine the ability of a given taxon to become established or even competitive (selection), a careful study on the actual abiotic factors within each area investigated is required. Ellenberg et al. (1992) introduced, for the ecological ranking of *sites,* seven independent indicator values for abiotic factors, the so-called Ellenberg values. However, this autecological assessment of the environmental sensitivity of a given taxon (vascular plant, mosses, lichens, or even fungi) cannot be taken as a synecological characterization of one community. Although discussion on plant communities is open and more alive than ever (e.g., Diekmann, 2003), a systematically arranged inventory of vegetation units and coenoformations (ecophysiological grouping *sensu* Passarge, 1993, 1996) still relies on ''expert judgment'' and has to be seen merely as abstract units. A literature survey would point out that vegetation history, succession, and reference (''climax'') are hot items, and provide several historical examples of redundancy versus specialization. Representing different trophic levels within a soil ecosystem, each of these layers (fungi, algae, mosses, and higher plants) reacts independently to environmental stressors (Fig. 1). Furthermore, within a given layer (*synusiae sensu* Barkman, 1978) some more sensitive taxa can either be enhanced or threatened by a given stress factor. The complexity of this phenomenon strongly affects the final resilience of the ecosystem as a whole, and causes shifts in both structure and processes.

4. Lessons learned from aquatic ecological classification and assessment systems

Before debating how to derive references (policy targets for ecological soil quality), it might be helpful to get a general overview of the requirements for setting up a system of quality criteria for soil. For this reason, it is helpful to study how quality criteria in other environmental compartments are established. The setup to gain to policy targets for soil ecosystems might show

an analogy with activities that have been performed in the framework of ecology and environmental and nature protection policy.

It should be noted here that references are descriptions of ecosystems that may be used for identification of an indicator. To identify an indicator ruler, one may have a negative reference (e.g., description of a disturbed side) and a positive reference (e.g., description of an undisturbed site). The policy goal might be something inbetween. These references may be geographical, historical, or theoretical.

4.1. Overview on systems for the aquatic compartment

Different users and different water quality managers make completely different demands on ecological assessment systems, in particular the following:

- to use them for pure quality assessment (descriptive systems in which the measured situation is compared in relation to a chosen reference);
- to see trends and to examine the effectiveness of (regional) policy (diagnostic systems which might give insight to the user into the controlling variables that might explain the measured situation);
- to predict the effects of management measures (predictive systems that may estimate the proposed effects of policy and/or management measures).

International and national governments are mostly interested in a description of the quality and in trends, possibly in relation to chosen references. Regional governments also want to estimate the effectiveness of management measures. Short-term (seasonal) and long-term (more years or even decades) surveys of the trophic state of lakes are hot items for both researchers and governments (e.g., Harris, 1986). For instance, Sechi and Lugliè (1996) showed the extent to which the trophic state of the Sardinian reservoirs (16 artificial lakes and 1 natural lake) changed. In comparison to the standard measurements of total phosphorus (P_{tot}), chlorophyll ***a*** is a much more reliable tool. Fig. 2 shows how a much higher trophism can be obtained from chlorophyll ***a*** as compared with P_{tot}, as chlorophyll ***a*** is much more representative in all the lakes dominated by Cyanophyceae plankton. Such a different sensitivity of measured proxies and the independent responses of species to the elimination of phosphorus from sources in rural as urban areas require sophisticated systems, like those developed to assess the ecological quality of main rivers (Statzner et al., 2001).

Other characteristics of aquatic assessment systems are that they work on different levels of regional scale (e.g., catchment area, subsystem, measuring point), different levels of policy and management measures, and different levels of biological/taxonomic organization (e.g., species, populations, or communities). In any case different groups of organisms are taken into account such as mesofauna, macrophytes, phytoplankton, fishes, and birds. Parallel to the biological endpoints, environmental parameters such as morphology, hydrology, and environmental quality (also fresh/saltwater) are taken into account. Assessment systems may be built on the basis of literature, expert knowledge, and directed monitoring. From the knowledge obtained, steering variables and processes are inferred. When the assessment systems are sufficiently reliable, and the effect of steering variables is well known, sometimes application of a smaller monitoring effort may be possible, with a decrease in monitoring efforts.

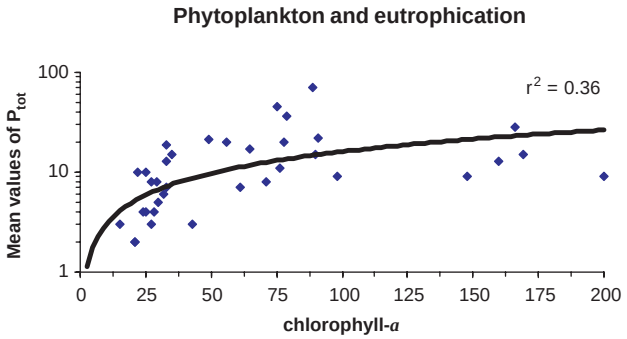


Fig. 2. Semilogarithmic plot of the values (based on seasonal samplings) of chlorophyll ***a*** and total phosphorus (both in $mg\,m^{-3}$). The trophic situation on the basis of this biochemical proxy is much worse than is inferred from P_{tot} (raw data from Sechi and Lugliè, 1996).

Assessment systems for ecosystems are based on extensive monitoring programs. The description of references in The Netherlands is partly inferred from literature data (cf. Handboek Natuurdoeltypen (Bal et al., 1995)), ecological databanks, expert knowledge, and the use of prognostic models. Later, specific research into missing knowledge on causal relationships may be performed. For aquatic ecosystems in The Netherlands, it is thought that about 100 reference descriptions are necessary (see also STOWA/RIZA/RIVM/IPO, 2001). One of the most elaborated and widely used systems is described in detail in the following section.

4.2. RIVPACS and related systems

River InVertebrate Prediction And Classification System (RIVPACS) was developed by the IFE (Institute of Freshwater Ecology, UK) to assess the ecological quality of rivers. As summarized by Wright (2000), this approach allows one to make site-specific predictions of the macroinvertebrate fauna to be expected in the absence of major environmental stress (see Fig. 3 for a general scheme). The expected fauna is derived by RIVPACS using 12 environmental characteristics (e.g., geographical coordinates, water properties, and climatic

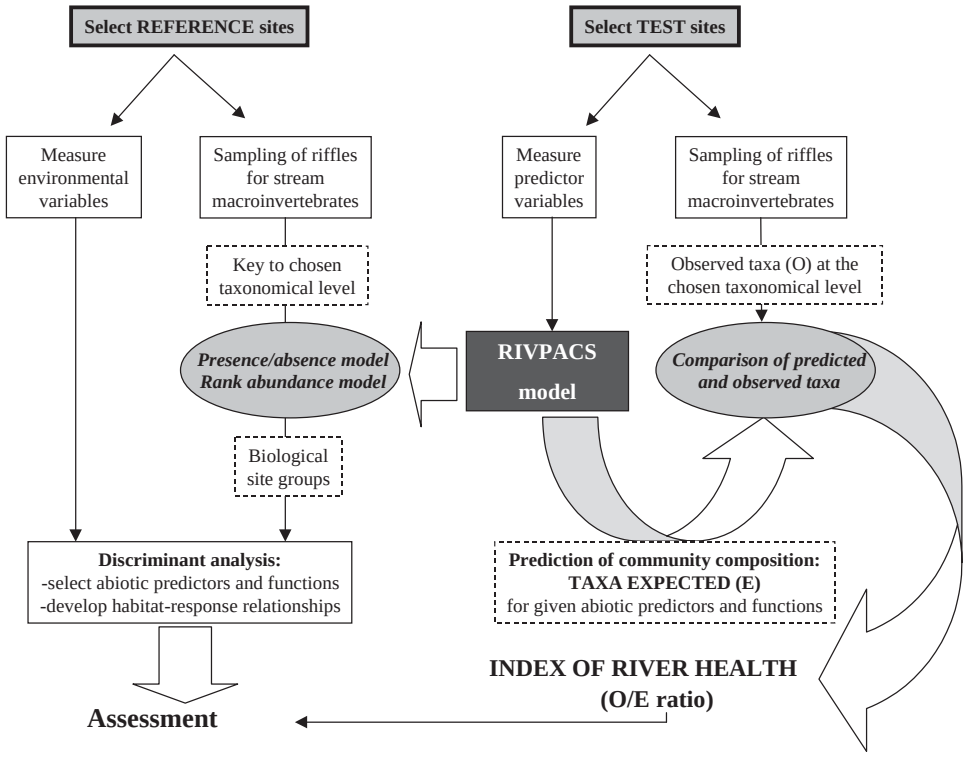


Fig. 3. Schematic drawing of the RIVPACS assessment system. For an explanation, see text.

factors). The ecological evaluation is then made by comparing the fauna observed at the site with the expected fauna. RIVPACS also includes a site classification based on the macroinvertebrate fauna of the reference sites sampled since 1977 (in the first phase 168 sites; now, in phase III, 614 sites all over Great Britain). New sites, judged by their fauna to be of high ecological quality, may be allocated to classification groups within the fixed RIVPACS classification. At the moment, the freshwater streams of Great Britain are classified in 35 groups having similar macroinvertebrate communities, which can be partitioned into four major blocks representing different size classes of streams and regions.

From a practical point of view, different taxonomic levels could be used as measurement endpoints. In accordance with other indicator systems used in Great Britain, 22 major taxonomic groups of macroinvertebrates, containing 117 families with 637 species have been identified, based on samplings at the 614 reference sites. In addition to presence/absence data, biological indices as well as abundance data can be used for evaluation of the sites. Of course, all sampling is done using standardized methods.

Systems similar or comparable to RIVPACS were developed in Australia and Canada (respectively, AUSRIVAS in Davies, 2000, and BEAST in Reynoldson et al., 2000). In particular, in the latter system, the difference between the observed biocoenosis and expected can be further investigated by using ecotoxicological tests to identify cause–effect relationships. Besides this, the test results allow to classify the investigated sediment on a simplified scale as nontoxic, potentially toxic, or toxic. However, in its current form, RIVPACS is not able to predict the changes in the invertebrate biocoenosis after the impact of an anthropogenic stress factor (De Pauw, 2000). The information from all these systems is currently an integral part of the assessment of the ecological quality of freshwaters performed by governmental authorities in several states (UK, Canada, Australia). Other aquatic classification systems are still under development, e.g., a Dutch approach called EKOO, which is based on the description of very different water bodies grouped into 42 "cenotypes" (Verdonschot and Nijboer, 2000). An assessment system similar to RIVPACS (maybe including features of BEAST) needs to be established for soil (cf. Breure et al., 2005).

5. Basic requirements for soil ecological classification and assessment systems

It is important to distinguish between the classification of soil communities (and, therefore, soils) and their assessment (two-step approach). In other words, in the first step, the huge diversity of soil organism communities and soil types is classified so that this diversity can

be handled from a practical point of view. In the second step, the result of the classification of a certain site to a specific class has to be assessed, and a decision made as to whether the soil quality at this site, with respect to the habitat function, is "good" or "bad." In addition, not only the use of the soil, but also the needs of the person or institution performing the classification and assessment have to be considered: Any ecological assessment system for the soil must result in a system that makes such a differentiation possible. A local area manager needs different information than a province authority, and a farmer requires different information than a forester.

To facilitate the process of risk assessment and ecological quality assessment of soil at a specific site, a decision support systems (DSS) is under development in The Netherlands. The DSS can be used for determining remediation goals and the urgency of remediation, for spatial planning of land use at a site, and for developing measures to cope with soil contamination within the soil management system. The basic framework for site-specific risk assessment takes three steps into account: (1) recognition of the actual or desired land use; (2) specific ecological characteristics of the site in relation to the land use; and (3) the risk assessment itself. The DSS consists of a framework according to a triad approach based on the integrated analysis of three different site-specific sources of information (chemical, ecotoxicological, and ecological information), so, for example, estimation of risks based on the presence of the pollutants by measuring the chemical concentration of pollutants, by determining the toxicity in samples from the site via bioassay testing, and by making ecological field observations.

Insight into site-specific ecological risks can be obtained by evaluating the bioavailability of the chemicals and by incorporating the results of site-specific biological information, such as bioassay testing in ecological field observations, into the risk and quality assessment. The three site-specific information sources according to the "triad" approach improve the reliability of the assessment because conceptual uncertainties in assessment techniques are efficiently eliminated as different types of information are used (Rutgers et al., 2000).

5.1. What is good soil quality?

Independent of which soil function has to be assessed, it is hard to answer the question which criteria have to be met to characterize a soil as "healthy" or "qualitatively good" (Lancaster, 2000). On the basis of present knowledge it might only be possible to state in general terms whether a certain quality is higher or lower that a certain fixed, or chosen, "count down point." In fact, this question cannot be answered objectively, and social choices have to be made. Policy makers have to perform this process of choice. The outcome should be based on ecological insights and social motives. In general terms, sustainable use of a soil implies a minimal shift of negative effects to the future and the surroundings. Herewith it is important to consider integrally the physical, chemical, and biological soil characteristics and to take technical, social, economic, ethical, and even esthetical factors into account. Most of those factors share many characteristics and are linked to dynamic ecosystem processes. Ecological characterization of soil may provide a measure of the extent of sustainability of soil use.

On ecological and social grounds, one may argue that a managed soil may be classified as being use "sustainable," and is supposed to be able to perform the ecological functions desired for the specific soil use even if it is disturbed by a minimal amount of chemicals and energy input (now and in the future). For agricultural ecosystems, this might mean that the situation before 1950 might provide references for "good"-quality ecosystems, analogous to the approach in nature policy (cf. Van der Meijden et al., 1989). Another possibility is to qualify the soil quality of biological farms as "good" or "desirable," or the situation in a nature area with a corresponding type of soil. Other options are theoretical references, inferred from a desired functioning of the soil ecosystem in relation to C and N mineralization capacity, stability of the soil food web, and/or diversity within functional groups of soil organisms as a measure of the resilience of the soil ecosystem (Breure et al., 2002b). For application of the DSS to determine the location-specific ecological risk assessment as described above, specific quality criteria need to be inferred from the type of land use desired.

5.2. Criteria for the use of organisms as indicators of soil quality

It is clear from the results obtained so far that the organisms in soil can be identified and quantified sensitively and reproducibly and that environmental stress and soil use influence soil life. When ecological quality assessment of a soil has to indicate the extent of sustainability of the soil, data on the presence and abundance of species have to be coupled to, or correlated with, ecosystem processes, and they have to provide an integration of physical, chemical, and biological parameters.

There it will be shown that soil organisms have been successfully used in ecological classification and assessment systems (e.g., the Dutch Biological Indicator of Soil Quality (BISQ) system or the German Soil Biological Site Classification (BBSK) system). These indicator systems are supposed to provide an integral

view of the ecological state of the soil relative to a desired or optimal quality.

5.3. The use of reference sites for the definition of quality standards

In pollution gradients, it is possible to use a local and clean control. However, in many cases such a reference location is not available. In general, the "value" of an indicator is dependent not only on stress factors, but also on soil type, land use, and vegetation. Therefore, references may be inferred from a large number of observations, as has been done in aquatic assessment systems like RIVPACS (cf. Section 4.2). The choice of a desired reference is a political one. For a specific combination of soil type and land use a reference may be the average value of 20 analogous locations within the Dutch National Soil Monitoring Network, or the average of 20 biological farms of a given management type (biodynamic systems) on a certain type of soil (grasslands on sand). Soils with a very high or a very low indicator value should then be further investigated.

A major question is how many references should be described to cover the different combinations of land use and soil type. For aquatic systems in the United Kingdom, more than 600 reference sites are described. In The Netherlands it is thought that about 100 references need to be described to cover the different aquatic ecosystem types. Monitoring of changes in soil quality with time may reduce the importance of subjective reference values. Such changes may be interpreted more easily than "snapshots." Spatially extensive and long-term monitoring programs may not seem ideal, but they are probably the most realistic way to derive objective information about spatial and temporal differences within, and human impact on, ecosystem quality.

5.4. Practical considerations for monitoring programs

All ecological assessment systems are based on extensive monitoring activities and a huge amount of monitoring data. The possibilities for the setup of a monitoring system for the soil differ very much from the possibilities for aquatic ecosystems, which are usually owned by the government (or, comparatively, organizations such as the "polder-boards" in The Netherlands). It is relatively easy to make management and monitoring arrangements with such owners and managers. And therefore this has been done for decades. For soil the situation is different. Especially outside nature areas, there are a large number of landowners with different interests. To come to a systematic monitoring on a scale comparable with monitoring aquatic ecosystems, a management framework must be created.

Farmers manage a great part of the land in most of the European countries. They are used to having their land investigated to determine soil quality so that they can be adviced on fertilization, cultivation, and the presence of pathogenic organisms in the soil. For them, it will not be unusual to make arrangements for ecological monitoring of the soil within a soil management framework. As for aquatic ecosystems, the monitoring data must be incorporated into a national database. Clear arrangements are necessary to determine ecological soil quality on a detailed level, as is usual in water quality frameworks. At the moment, there is no general framework and a description and the notion of a common interest is missing, on both the national and the European levels.

As far as necessity and development of ideas and concepts are concerned, it is the proper time for Europe (or at least individual countries or regions) to develop a handbook of soil reference types. For the time being, a quality system, equal in detail to that used for aquatic ecosystems, seems a utopian goal. It may be expected that the first rough versions of reference descriptions will be produced in the coming years. The first attempts to reach that goal are in progress. A better framework of soil managers may be a valuable addition to reach that goal (e.g., in The Netherlands (Schouten et al., 1999), in the United Kingdom (Weeks et al., 1997), or in Germany (Römbke et al., 2000, 2002)). Permanent plots for long-term monitoring have been usefully introduced in vegetation science (Austin, 1981), which is also strongly recommended as reference sites covering different soils in different European regions. In fact, such a network of permanent plots does already exist in some countries (e.g., the Bodendauerbeobachtungsflächen in Germany), but up to now they have been used for other purposes in forestry and nature protection (Barth et al., 2000).

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