Science of the Total Environment xxx (2017) xxx–xxx

# Measuring soil sustainability via soil resilience

Marie Ludwig [a,*], Paul Wilmes [b], Stefan Schrader [a]

[a] Johann Heinrich von Thünen-Intitute, Federal Research Institute for Rural Areas, Forestry and Fisheries, Institute of Biodiversity, Bundesallee 50, D-38116 Braunschweig, Germany
[b] Luxembourg Centre for Systems Biomedicine, University of Luxembourg, Esch-sur-Alzette, Luxembourg

## HIGHLIGHTS

- Calculating soil ecosystem thresholds is the current challenge.
- Soil resilience can be used as a measure for soil management sustainability.
- Soil resilience can be calculated from the response diversity by multi-omic markers.

## GRAPHICAL ABSTRACT
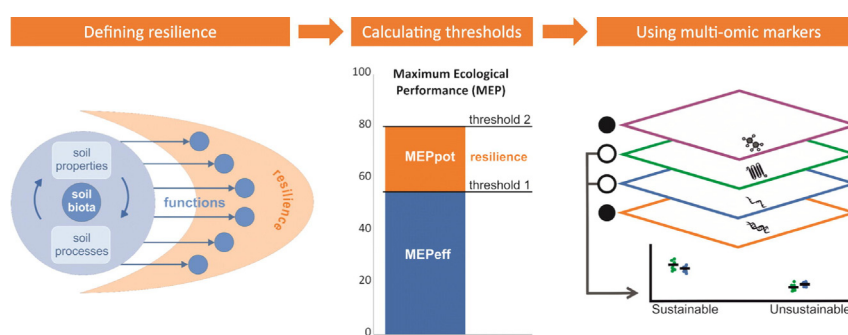


## ARTICLE INFO

## ABSTRACT

Soils are the nexus of water, energy and food, which illustrates the need for a holistic approach in sustainable soil management. The present study therefore aimed at identifying a bioindicator for the evaluation of soil management sustainability in a cross-disciplinary approach between soil science and multi-omics research. For this purpose we first discuss the remaining problems and challenges of evaluating sustainability and consequently suggest one measurable bioindicator for soil management sustainability. In this concept, we define soil sustainability as the maintenance of soil functional integrity. The potential to recover functional and structural integrity after a disturbance is generally defined as resilience. This potential is a product of the past and the present soil management, and at the same time prospect of possible soil responses to future disturbances. Additionally, it is correlated with the multiple soil functions and hence reflecting the multifunctionality of the soil system. Consequently, resilience can serve as a bioindicator for soil sustainability. The measurable part of soil resilience is the response diversity, calculated from the systematic contrasting of multi-omic markers for genetic potential and functional activity, and referred to as potential Maximum Ecological Performance (MEPpot) in this study. Calculating MEPpot will allow to determine the thresholds of resistance and resilience and potential tipping points for a regime shift towards irreversible or permanent unfavorable soil states for each individual soil considered. The calculation of such ecosystem thresholds is to our opinion the current global cross-disciplinary challenge.

## 1. Sustainability. Are we asking the right questions?

The Great Acceleration describes the human-driven acceleration of the global change in the second half of the 20th Century. Since 1950, this has led to a fundamental shift in the functioning of the Earth System, turning the Holocene, the most stable period in the planet's natural history, into the Anthropocene (Steffen et al. 2015). Given that

\* Corresponding author.
E-mail address: marie.ludwig@thuenen.de (M. Ludwig).

planetary boundaries are set, the challenge of the new geological era is to remain within a safe space for humanity by avoiding human activities causing unacceptable environmental change by overstepping boundaries which may represent tipping points associated with irreversible change (Rockström et al. 2009). Three of nine identified planetary boundaries have already been overstepped including the decline in global biodiversity. The overstepping of additional boundaries is expected. Especially, because the recent industrialized forms of agriculture increase environmental degradation, which results in the irreversible overstepping of the identified thresholds (Rockström et al. 2009). And these forms of agriculture are expanding globally. The forecasted increase in world population of up 9.7 billion in 2050 (UN 2015) will require an additional food production of 60–100% as well as energy and clean water demands of additional 100% and at least 55% compared to today (IRENA 2015; Valin et al. 2014). Soils are the nexus of water, energy and food (Biggs et al. 2015; Jónsson et al. 2016), which illustrates the need for a holistic approach in sustainable soil management (McCormick and Kapustka 2016; Weigelt et al. 2014). The challenge facing a growing population is the increased and intensified use of the ecosystem services provided by soils (Blum 2005). In order to guarantee sustainable agriculture, soil management needs to simultaneously include different aspects on the multiple soil functions and the ecosystem services linked to this nexus (Baveye et al. 2016). To meet the requirements of such a holistic perspective it is necessary to find relevant bioindicators. Bioindicators are measurable proxies for environmental end points that are in themselves too complex to assess or too difficult to interpret in terms of ecological significance (Pulleman et al. 2012). Indicators for agro-ecosystems, either biological, physical or chemical, provide information on the state, trends and the seriousness of the situation by complex interactions between agriculture and environment (COM, 2000). The search for relevant bioindicators for arable soils has led to a huge output of studies on the national and international level over the course of the past decades, currently fueled by the specification of the UN Sustainable Development Goals (SDGs) towards an integration of more soil related indicators (IASS 2015; Keesstra et al., 2015; Jónsson et al. 2016).

### 1.1. Why has soil bioindication not yet been successful?

Until now, indicators for soil management have mostly focused on the physical, chemical and biological aspects of soil (Jónsson et al. 2016). A detailed overview of the existing indicators can be found in comprehensive reviews (Bastida et al. 2008; Cluzeau et al. 2012; Havlicek 2012; Pulleman et al. 2012; Ritz et al. 2009) and as output of the latest international initiatives for evaluating soil monitoring on the European level, ENVASSO (Bispo et al. 2009) and EcoFINDERS (Faber et al. 2013; Griffiths et al. 2016; Stone et al., 2016). However, no standardized soil sustainability indicator has yet been established on the national or international level. The two main approaches used to establish sustainability indicators are the development of a single, composite index or the development of an indicator set (Zhou and Ang 2008). This seems necessary because a single indicator is not believed to be able to give a full picture of a complex system (Nourry 2008). Too many indicators, however, make the data collection and processing difficult to handle at a reasonable cost and time. But too few indicators may miss out crucially important developments (Bossel 2001). Hence, the set of indicators needs to be reduced to a relevant choice, which is no trivial task. Usually, studies reduce due to a statistical process, resulting in a minimal data set (MDS) consisting of a subset of variables that confers a maximum of discriminatory information (Askari and Holden 2015). The other way to create such a subset can be the result of a process in which experts reduce variables personally (e.g. via the Delphi survey technique) due to their best professional judgement (BPJ) (Rutgers et al. 2012). Both procedures, however, do not automatically prevent indicators from overlapping in their informational content and, thus, the problem of autocorrelation can negatively impact the

ability to discern or predict. Having correlated indicators in an indicator set may bias the weighting and evaluation of the whole indicator. Prevention of autocorrelation becomes more complicated due to the fact that indicators can happen to be no real measures but estimates. These "guesstimates" can be derived indirectly from other parameters, like pH-values derived from land-use or soil texture derived from soil type. However, they enter the set or index like true values, which makes their connection with reality very unclear, abstract and subjective (Baveye 2017). Additionally, indicators strongly depend on scale. That has a number of consequences, including unpredictable cross-scale dynamics. Correlations identified between bioindicators and parameters at one scale may not exist at another (Baveye 2017), and upscaling of measurements or estimates is not straightforward (Baveye et al. 2016). Another problem with bioindication in general is the calibration of the results. To evaluate an indicator, this first of all requires reference values for comparison and contextualisation (Aspetti et al. 2010; Bastida et al. 2008), otherwise no real interpretation is possible. Overall, most current approaches reflect natural dimensions including physical, chemical and biological parameters of soil (Jónsson et al. 2016). However, these approaches usually lack social dimensions (Jónsson et al. 2016) which are highly relevant, especially for production ecosystems. Nevertheless, promising approaches can be found, which try to come over at least some of these problems. Rutgers et al. (2012) calculated a maximum ecological potential of arable soils by comparing them to a representative reference soil sample (equal to 100%). Rüdisser et al. (2015) base the estimation of their so-called biological soil-quality index (BSQ) on soil microarthropods in relation to land use and land cover (LULC) data in order to conduct a state-wide sustainability assessment. And Weathers et al. (2016) and McCormick and Kapustka (2016) illustrate the need for holism and cross-disciplinary collaboration in sustainability research and encourage ecologists to engage in increasingly important new interfaces between disciplines. The scope of the present study is therefore to identify a measurable bioindicator for the evaluation of soil management sustainability in a cross-disciplinary approach while accounting for the lessons learned from the literature so far. For this purpose, we will discuss the remaining problems and challenges of this task and consequently suggest one possible bioindicator for soil management sustainability.

As apparent from a comprehensive literature survey, for development of such an indicator, there is a need for a strong focus on function. This focus however entails two fundamental problems that need to be solved first: The definition problem and the evaluation problem.

Box 1
Soil function or service?

> The term soil service is derived from the common term ecosystem service and is currently preferentially used as a surrogate for the term soil function in many studies. However, there is no general recommendation of which term should be used in soil science since both bring along difficulties: While soil '*service*' has a strong anthropocentric connotation and only seems to include aspects of soil processes that serve the needs of human populations, soil '*function*' is more ambiguous, philosophically laden and its use in the soil science literature is utterly confusing (Baveye et al. 2016).
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> Hence, for this study we will use both terms. We prefer the term *soil function* as a single component of [soil] *functioning*, referring to the sum of processes that sustain the [soil] system. Depending on the context, we will additionally use the term *ecosystem services*, as the ensemble of soil functions specifically beneficial for humankind.

## 1.2. The definition problem. Or what is soil sustainability?

Although it is a frequently used term, there has been an ongoing debate over the definition of sustainability in general for the last thirty years. Since the definition by the Brundtland Commission (1987) of sustainable development as able "to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs", there have evolved over 100 different definitions till the mid-1990s (Büchs 2003). Abbott and Murphy (2007) defined soil sustainability based on the Brundtland idea as "soil management that meets the needs of the present without compromising the ability of future generations to meet their own needs from that soil". The early discussion about sustainability in general had an almost exclusive anthropocentric focus: Increasing the economic performance and energy efficiency of the system (McMichael et al. 2003). This is significantly reflected in the whole terminology of the research field that evolved in this context, resulting in the frequent use of the term soil service instead of soil function (see Box 1).

Especially regarding production ecosystems the debate about what are sustainable practices is ongoing and strongly contrasting (Rist et al. 2014; Wezel et al. 2013). This seems to be foremost fueled by the "Paradox of Sustainability" (Gunderson and Pritchard 2002), that can be outlined as short-term efficiency vs. long-term sustainability. Most definitions of sustainability can be summed up by the catch phrase "reduce, reuse and recycle" (Walker et al. 2012), i.e. a focus on the efficient use of resources. But efficiency in general means the optimization of only a narrow range of values and a particular set of interests. So being efficient, in a narrow sense, leads to elimination of redundancies, by keeping only those things that are directly and immediately beneficial (Walker et al. 2012). This sets the system on a trajectory leading to drastic losses in resilience, which are the major cause of unsustainability (McCormick and Kapustka 2016; Walker et al. 2012). Another reason for the failure to achieve a collective vision of how to attain sustainability lies in the limitations and disjunction between disciplines (McMichael et al. 2003). Because "nature doesn't do disciplines", and neither should people when considering complex system problems (Weathers et al. 2016). No system - whether social or ecological, or to be more accurate, no social-ecological-system – has one sustainable "optimal" state (Walker et al. 2012). Hence to define sustainability, we need a holistic concept that allows the identification of bioindicators to measure and evaluate sustainability in its varying states. To avoid confusion and ambiguity regarding the use of the term sustainability in this study, we refer to it as the maintenance of soil functional integrity. However, to fully define sustainable soil management via soil functioning, this directly leads to the second problem connected to the evaluation of soil functioning – the multifunctionality effect.

## 1.3. The evaluation problem. Or what is the multifunctionality effect?

Most studies focused on soil functions (Box 1) only provide a limited view regarding the environmental context. This is for example due to their anthropocentric or monetary perspective. Additionally, most approaches that try to evaluate the significance of a certain soil function are mono-functional approaches whereby they only consider one soil function at a time, which is not representative for, or even distorts, the significance of the bioindicator in the total context of multiple functions. The significance of e.g. biodiversity is different at every level of the ecosystem service hierarchy: as a regulator of underpinning ecosystem processes, as a final ecosystem service and as a good that is subject to valuation, whether economic or otherwise (Mace et al. 2012). That reflects the multilayered relationship of bioindicators with functions/services, which implies conflicts for the accounting of its significance. A recent attempt to do so through modeling the multifunctionality of soils by including multiple and interlinked soil functions simultaneously - as opposed to the common way considering just single functions – has shown to strongly increase the significance of soil biodiversity as a

driver for each considered function (Bradford et al. 2014). That means, to identify the real drivers of soil functioning or to measure and evaluate trade-offs and redundancy among soil functions, the approach has to appreciate the interdependence of functions as a key property (Baveye et al. 2016). This is why the scope of this study was to find a suitable bioindicator for soil functional integrity, which is relevant although there is a strong interdependence of functions, to indicate the sustainability of soil management practices from the past, present and future.

## 1.4. What is the link between functioning and biodiversity?

There is confusion in the current discussion and a lack of understanding on how biodiversity is explicitly linked to soil functions and derived ecosystem services. Many studies state that soil biodiversity "underpins" the ecosystem services provided by soils (Mace et al. 2012; Robinson et al. 2014; Smith et al. 2015). However, these studies lack a detailed explanation or data validation. In general, the significance of biodiversity for ecosystem services is assumed, while ranging between the extreme viewpoints of 'biodiversity and ecosystem services are the same thing' (therefore the terms are used almost synonymously) and 'biodiversity is one ecosystem service itself' (Mace et al. 2012). And soil management ignoring this tight coupling of soil biodiversity and soil functioning is thought to cause non-linear losses of soil ecosystem services (Birge et al. 2016). Nevertheless, it is extremely problematic to demonstrate a causal link between soil biodiversity and ecosystem services due to several theoretical problems (What measure should be used to express biodiversity? Does the concept of species make sense for soil bacteria or archea at all, given the extent of genetic material transfer among them?) and operational problems (How to adequately extract DNA from soils? How to design experiments to demonstrate this causal link given all the interconnected side effects of soil structure, biochemical composition and hydrologic regime on soil biodiversity?) (Baveye et al. 2016). Against this background, it becomes even more important to define the term soil biodiversity to which these viewpoints refer to. In general, biodiversity is defined as the variability among living organisms from all sources, including diversity within species, between species and of ecosystems (CBD 1992; Mace et al. 2012).

Species richness alone could be shown to not matter for ecosystem functioning, unless species differ in their properties (traits) (Norberg 2004). This is also reflected in the idiosyncratic nature of the relationship between biodiversity and function that was exemplary shown by soil carbon cycling (Nielsen et al. 2011). Regarding species that simply means "some matter more". However, biodiversity effects on ecosystem functioning are not due to the individual's affiliation to a species or genus, but to its range of traits and hence its genetic potential for ecological performance, i.e. in response to ecosystem disturbances (Gunderson et al. 2012). Therefore, it is the so-called response diversity, which matters.

Response diversity is defined as "the range of reactions to environmental change among species contributing to the same ecosystem function" (Elmqvist et al. 2003). Functional diversity means species are functionally dissimilar and therefore complementary. If these species now get substantially reduced or lost, but are at the same time replaced by their functional redundant analogues, this is response diversity. Both functional diversity and response diversity are significant for ecosystems. But focusing on response diversity helps increase the effectiveness of ecosystem management (Elmqvist et al. 2003).

But how can we examine the response diversity of a soil? Baveye (2017) rightly poses the question "Beyond all the 'guesstimates', how do we get real data?" Latest studies highlight the promising progress by molecular technologies to open the "black box" of soil biodiversity, potentially allowing a rather trait-based than a taxon-based approach, to understand the role of the different aspects of soil biodiversity in driving soil functionality (Jansson and Baker 2016; Smith et al. 2015). We think, the current challenge is to use this approach to identify the relevant bioindicator to measure soil management sustainability.

## 1.5. Soil sustainability indicator – How to measure the immeasurable?

We earlier defined soil sustainability as the maintenance of soil functional integrity. Ecosystems with a high response diversity increase the likelihood for renewal and reorganization into a desired state after disturbance (Chapin Iii et al., 2000; Elmqvist et al. 2003), which makes response diversity the guarantor of functional integrity. Once measured, it could visualize the soil ecosystems' inherent potential to recover its functional and structural integrity after a disturbance, in general defined as resilience (Lal 1997; Seybold et al. 1999). This potential is a product of the past and the present soil management, and allows at the same time prognosis of possible soil responses to future disturbances. Soil resilience, measured by the response diversity of the soil, could therefore be used as a bioindicator for soil management sustainability.

## 2. Resilience. Or the two ways of thinking about stability

### 2.1. Resilience as a concept of stability

The concept of resilience has emerged from many different fields. It has, for instance, longer roots in psychology than in ecology (Olsson et al. 2015). There it describes a personal trait, although it is most commonly understood as a process – a dynamic process of positive adaption to significant threat, adversity, trauma, tragedy or stress. For ecosystems, two very different definitions of resilience exist. Both deal with aspects of system stability. The first defines this stability over the attributes efficiency, control, constancy and predictability, and is referred to in the literature as engineering resilience (Holling 1996). According to Pimm (1984), in this concept resilience is determined by the time necessary for a system to return to an equilibrium state after a disturbance. The second definition characterizes stability by the attributes of persistence, adaptiveness, variability and unpredictability. It is referred to as ecological resilience and emphasizes the dynamic features of an ecosystem concept with no single equilibrium state (Holling 1973). To be precise, engineering resilience is maintaining the efficiency of function, while ecological resilience is maintaining the existence of function. These two contrasting aspects of system stability have fundamentally different consequences for understanding, evaluating and managing complexity and change (Gunderson and Holling 2002), especially regarding a multifunctional system like soil.

### 2.2. Resilience as a meta-function of soil

Looking at sustainability requires an emphasis on the second definition of resilience, since it describes the amount of disturbance that can be sustained before a significant change in ecosystem functioning occurs. Therefore, resilience is not a single parameter but more likely a holistic meta-function (Fig. 1) of a community, of a soil or of a whole ecosystem, derived from all its single properties, in interplay with the ongoing processes, driven by biota interactions. It is a measure of the (pre-)adaptive potential to cope with future disturbances while at the same time representing the multifunctionality of the system. Resilience is a product of the past and the present as well as a prospect of the system's future. Hence it reflects all important indication levels for evaluating the soil state: it is derived from the soil's [management] past as part of the soil memory, representing the soil's present status over the affectedness under the given pressures and disturbances as well as providing possible recommendations for future improvement of holistic soil management by evaluating the intrinsic adaptive potential. Hence, in many concepts resilience is seen as a crucial part of sustainability or even equated with the term of sustainability (Marchese et al., 2017). Therefore, soil resilience can be used as an appropriate measure of soil management sustainability.
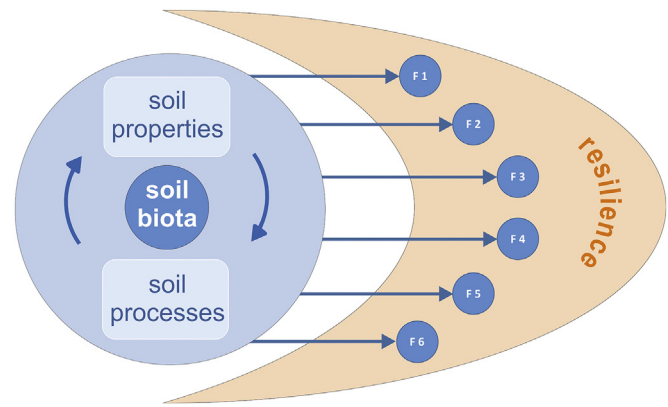


**Fig. 1.** Resilience as a meta-function of soil. Soil biota are a main driver of the interplay of soil properties with soil processes that build the natural capital of soils. Derived from that interplay there are various soil functions (schematically shown as functions F1 to F6), jointly reflected in the functional and in the response diversity of the soil, which represents the resilience of the whole soil system.

### 2.3. Resilience as a third dimension of ecosystem functioning

Ecosystems have no single equilibrium (Scheffer et al. 2001). They are dynamic systems in a constant change, like endless loops consisting of phases of exploitation (r), conservation (K), structural collapse and release ($\Omega$) as well as reorganization ($\alpha$) (Fig. 2). In the conceptual idea of an adaptive cycle (Angeler et al. 2015; Gunderson and Holling 2002), change is neither continuous and gradual nor consistently chaotic. Change is more probably episodic, with periods of slow accumulation of natural capital (such as biomass, biostructures or nutrients), punctuated by sudden releases and reorganization of those biological legacies (Franklin et al. 2000; Franklin and MacMahon 2000). These legacies differentiate the adaptive cycle concept from the classical theory of ecological succession.

There are five main theories in traditional ecosystem succession research: the monoclimax theory (Clements 1916), the polyclimax theory (Tansley 1939), the polyclimatic climax theory (Tuxen 1933), the climax pattern theory (Whittaker 1951) and the site climax theory (Dyksterhuis 1949). All these concepts are seen as being controlled by two functions: *exploitation*, which describes the rapid colonization of recently disturbed areas by pioneers; and *conservation*, which is the following period of slow accumulation and storage of material and energy (see frontloop of Fig. 2).
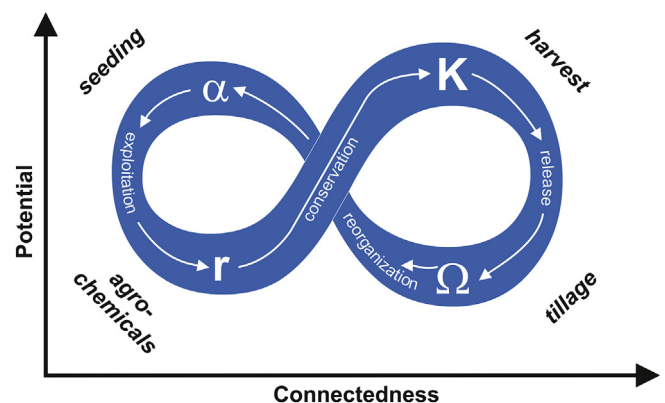


**Fig. 2.** The adaptive cycle (modified after Gunderson and Holling 2002) consisting of the phases exploitation ($\alpha \rightarrow r$), conservation ($r \rightarrow K$), structural collapse and release ($K \rightarrow \Omega$) and reorganization ($\Omega \rightarrow \alpha$). For soils on the field scale cycling is driven by the seasonal and annual development as well as by soil management events.

In ecology, the pioneer species of the exploitation period are characterized as r-strategists, while the accumulation of natural capital is seen as driven by the settling of K-strategists. These two strategists represent opponent dispersal- and reproduction ecotypes (MacArthur and Wilson 1967; Pearl 1927) and are used as eponyms for these two periods of ecosystem development (Fig. 2). In this strategic change, the ecosystem experiences the shift from species who adapt to external variability (r-strategists) to species who control variability (K-strategists). Thus the system is gaining internal control (connectedness) by exerting over external control by functionally shifting species. With this increase in connectedness, i.e. the specialization and networking of species and overall the system's degree of order, the increased functional efficiency rises at the same time as the system's potential (Fig. 2). This entails the growth of natural capital: from resources and biota, that form the interplay of soil properties and processes (Fig. 1) to soil functions or soil ecosystem services, respectively.

Up to the point where the system peaks in potential, connectedness and accumulated natural capital, the classical theories and the adaptive cycle follow exactly the same pattern. This can be seen for all natural ecosystems, but also for systems managed by humans like arable soils this pattern of development can be found. Relating the frontloop of the adaptive cycle concept to an agro-ecosystem on field scale (Fig. 2), the annual season may start with crop seeding ($\alpha$) where the soil system has high potential for a diverse plant succession as well as active, abundant and diverse soil biota. This potential decreases in the experimental phase of the system ($\alpha \rightarrow r$) because chemical management practices exclusively promote the cultivated crop. The farmer controls the crop development by the specific application of agro-chemicals such as pesticides and fertilizers preventing other plants to develop. After tillering and stem elongation the crop stand is established (r), the phase of accumulation ($r \rightarrow K$) follows and this is roughly characterized by booting, inflorescence emergence, flowering, fruit development and ripening. The accumulation phase ends with senescent grains in case of cereals (K), which are then ready for harvest. At this stage the agro-ecosystem shows its highest grade of connectedness regarding its seasonal development. In the traditional view, this equilibrium is called climax state of the ecosystem. In the concept of the adaptive cycle, it is leading into conservation which ends with structural collapse ($K \rightarrow \Omega$, see Fig. 2).

### 2.3.1. Structural collapse is necessary for renewal of system resilience

The distinction of the adaptive cycle to these theories is the backloop of the cycle, which is triggered by a structural collapse. This collapse is caused by an increased vulnerability of the system as sources of novelty are eliminated. In conjunction, functional and response diversity ($=$ redundancy) are reduced within and across scales (Biggs et al. 2012; Gunderson and Pritchard 2002) due to increased system efficiency. This is the case after a certain time of conservation. Therefore, ecosystems sooner or later reach a point where a structural collapse happens naturally, mostly triggered by a natural disturbance. Natural disturbances usually are *pulse* disturbances with a characteristic magnitude and frequency, while human activities tend to transform some pulse disturbances into *press* or chronic disturbances (Bengtsson et al. 2002). However, there are also anthropogenic pulse disturbances. Managed soils experience the structural collapse mostly triggered by pulse disturbances due to the two main management practices at the end of the conservation period (K), which interrupt any further development of the system: (i) crop harvest resetting the system to an initial state of any plant succession; (ii) soil tillage as heavy mechanical impact disrupting soil structure (turning the topsoil in case of ploughing) along with declining soil biota (Fig. 2). Both pulse disturbances cause a temporary collapse of the system ($K \rightarrow \Omega$) initializing the backloop of the adaptive cycle.

This structural collapse can be regarded as the creative destruction of the established structures which releases accumulated matter and energy from their bond or sequestered and controlled state. The elimination

of structuring species or processes causes an ecosystem to reorganize (Gunderson and Pritchard 2002). Reorganization is always linked to structures that already exist, but which are now fragmented. By their new arrangement these biotic legacies form a source of novelty that leads into a period of experimentation. This turning point ($\Omega$) between the release and the reorganization (Fig. 2) is the source for renewal of system resilience. It is a highly unpredictable but also highly potential period, due to the variety of legacy effects. The variety of biotic legacies strongly depends on the soil history. This makes soil resilience, which is formed mainly by legacies, also a proxy for soil memory (Bengtsson et al. 2002). For managed soils, the backloop is also a period of system restoration ($\Omega \rightarrow \alpha$) letting the soil recover to gain potential and loosening connectedness for establishing and managing the following crop. This phase is characterized by recovery of soil and biota diversity. Biological and physical processes cause soil to become restructured: (i) ecosystem engineers like earthworms form new aggregates and macropores in the soil profile; (ii) freezing and thawing cycles as well as swelling and shrinking events restructure soil due to significant changes in temperature and precipitation, respectively. For managed systems this backloop turns with seeding (Fig. 2), which decreases the potential as consequence of the anthropogenic selection and promotion of just a certain crop/community composition, also due to application of agro-chemicals. For natural ecosystems the period of reorganization turns with the establishment of pioneer species, which then trigger the frontloop.

### 2.3.2. Legacy effects are the drivers of within and across-scale cycling

Legacy effects and self-organization are the engine of the cycling within one system, but can also influence the dynamics of other systems, even on higher or lower scale levels. This is explained by the concept of panarchy, illustrating the interdependence of dynamics across scales (Fig. 3). A panarchy can be regarded as a nested set of adaptive cycles (Gunderson and Holling 2002). Each one is self-organized, but its dynamics are linked across scales (from local, to regional, to global) making dynamics at one scale depend on those at other scales (Allen et al. 2014; Angeler et al. 2016). The cross-scale effects can be either
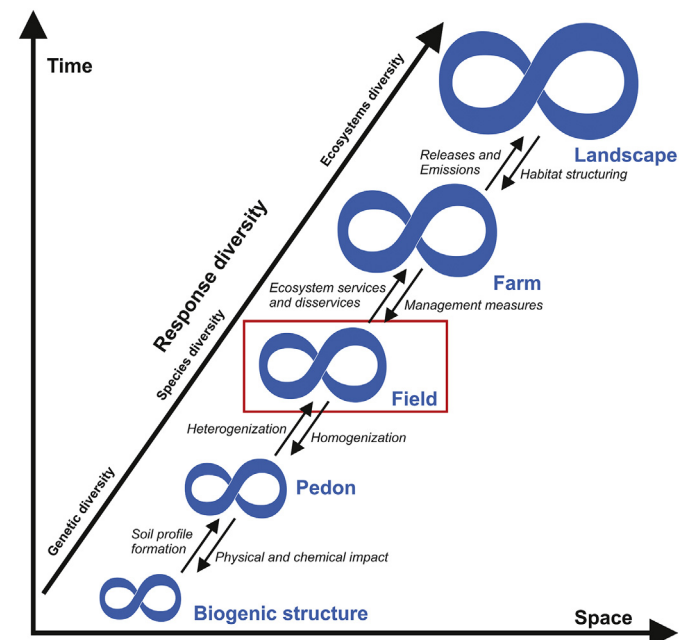


**Fig. 3.** Cross-scale effects in the panarchy of an agro-ecosystem (modified after Gunderson and Holling 2002). Starting from the field scale, a managed soil is interlinked with related structural units of soil and biodiversity in increasing and decreasing scales of time and space under land use. Soil units presented as adaptive cycles (see Fig. 2). Arrows indicate examples of legacy effects from nature and management between soil structural units across scales.

negative or positive, depending on the legacies that accumulate and are exchanged between the scales.

For an agroecosystem, the field scale is interlinked with other related structural units of soil and soil biodiversity in increasing and decreasing scales of time and space (Fig. 3). Following the panarchy concept legacies affect slower cycles (farm, landscape) on larger scales and faster cycles (pedon, biogenic structure) on smaller scales. Agricultural management in field is aimed at homogenizing pedons for cultivating a crop stand as uniform as possible. Homogenization of pedons has physical and chemical impact on biogenic structures: disrupting of biogenic soil aggregates and pores which may change nutrient cycling. In turn, biogenic structures essentially contribute to soil profile formation resulting in structurally diverse pedons. That leads to more heterogeneity within a field.

The following farm scale is mainly driven economically by fields' input and output. On one hand, a farm benefits from ecosystem services by soil biota regarding for instance regulation of the water balance, aeration and promotion of soil fertility. On the other hand, a farm suffers from ecosystem disservices like root infestation by soil-borne pathogens and pests. These outputs are controlled by farm management practices. Again, on the next scale several farms shape a landscape, which is then called a cultivated landscape. Farms' releases like agrochemicals via drainage and draining ditches and farms' emissions like climatic relevant trace gases via livestock husbandry may lead to environmental stress within the landscape (press disturbance). A high grade of diversification enhances the aesthetic value of a cultivated landscape and provides manifold habitat structures for beneficial organisms in farms.

Within these cascading dependencies across scales different levels of the biodiversity pool dominate. On a smaller scale, genetic diversity of soil biota is the main source of response diversity, driving soil processes. On a larger scale, species diversity becomes responsible for response diversity, which ensures soil functioning and the provision of ecosystem services. In both cases redundancy allows for functional stabilization and makes the system more robust against adverse impact. This is similar to the relevance of ecosystem diversity on the large scale. To evaluate the state of the soil ecosystem, we need to introduce one additional dimension: soil resilience as a third dimension of soil functioning.

### 2.3.3. Resilience determines thresholds of ecosystem functioning

To build resilience, a system needs the interplay between stabilizing and destabilizing forces. If these forces are out of balance and cannot make the system circle in its natural feedback loops anymore, regime shifts will occur (Meadows and Wright, 2008). Regime shifts occur when a system's resilience threshold is crossed and the processes responsible for system functioning change and create new self-organized structures (Allen et al. 2014). While structural collapse can be a natural phenomenon within the adaptive cycle, leading to renewal and reorganization of the regime, a functional in addition to structural collapse triggers a complete regime shift. These regime shifts usually result from a combination of a shock (pulse disturbance such as a large rainstorm) and gradual changes in slow variables (press disturbance such as nutrient depletion) that erode the strength of the dominant feedbacks. When a critical threshold is crossed, a different set of feedbacks becomes dominant, and the system reorganizes, often abruptly, into a new regime with a different characteristic structure, behavior, and set of ecosystem services (Biggs et al. 2012). These tipping points (Lenton et al. 2008; Wall, 2007) in an adaptive cycle can also have effects on the next higher and slower cycle, leading to a cascading change across scales (Gunderson et al. 2012; Gunderson et al. 2002).

Opposing legacies are drivers for and links between cycles across scales. However, extreme pulse disturbance as a single event and/or press disturbance as a gradually cumulated multiple event may lead the system into a so-called poverty trap or a rigidity trap (Fig. 4). For soils, this can be due to misuse leading in a highly degraded or depleted soil (poverty trap) of low resilience, connectedness and potential, or leading to a state where the soil is not a self-organized system anymore

(rigidity trap) due to high-performance cultivars forced to stay in a state of high potential and connectedness and artificial resilience by external input (fertilizer, pesticides, genetically modified material, etc.) An example for a poverty trap is a highly degraded soil due to repeated and severe compaction by means of heavy machinery. Within the adaptive cycle it occurs in the backloop $\Omega \rightarrow \alpha$ and prevents recovery of soil and soil biota diversity as well as restructuring after collapsing from harvest and tillage ($K \rightarrow \Omega$). Now at this tipping point a critical threshold of resilience is crossed and the system leaves the current cycle for another one characterized by a different set of boundaries and feedbacks. In this example, a regime shift happens from cultivated field to a set-aside field no longer under production. An example for a rigidity trap is a monocultural soil management that causes diversity loss and poor genetic variability of a high-performance cultivar. Within the adaptive cycle it concerns the frontloop ($r \rightarrow K$) and results in loss of adaptive potential against for instance environmental stressors like extreme climate events or loss of power to resist diseases and pests. Here a regime shift will be discarding the old cultivar and breeding a new and more adaptive one.

## 3. Soil sustainability indicator – How to measure and to manage

Summing up, to derive a tool to measure sustainability, we need to determine the two system thresholds of resilience: the threshold between resistance and resilience, and the threshold between resilience and a regime shift (tipping point) as responses to a disturbance of the ecosystem (Fig. 5). These two response states can also be characterized as maximum ecological performance of a soil, divided in the part of effective (MEPeff) and potential (MEPpot) maximum ecological performance (Fig. 5). In order to derive MEPpot and MEPeff, the biomolecular information content of soil can be harnessed. More specifically, MEPpot is directly related to the genetic potential encoded in the soil biota whereas MEPeff can be derived from their functional activity. The systematic contrasting of genetic potential and functional activity will allow for calculating MEPpot as a measure of response diversity and therefore to determine the thresholds of resistance and resilience for the individual considered soil.

In this context, integrated multi-omic analyses hold great promise as a high-resolution tool to calculate thresholds within soil. More
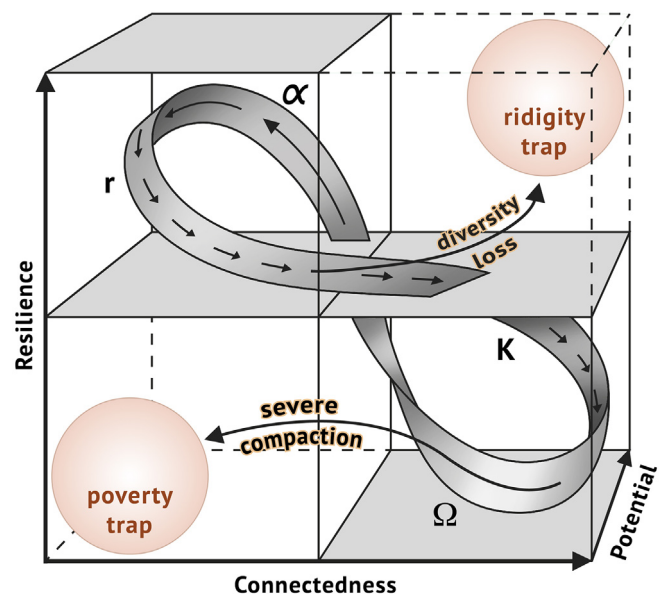


**Fig. 4.** Resilience as a third dimension to ecosystem functioning following the adaptive cycle (see Fig. 2) (modified after Gunderson and Holling 2002). If thresholds (tipping points) of resilience are overstepped by disturbances, the ecosystem experiences a regime shift towards an impoverished state in which it can be trapped. For soils, this can be due to misuse leading in a poverty or in a rigidity trap.
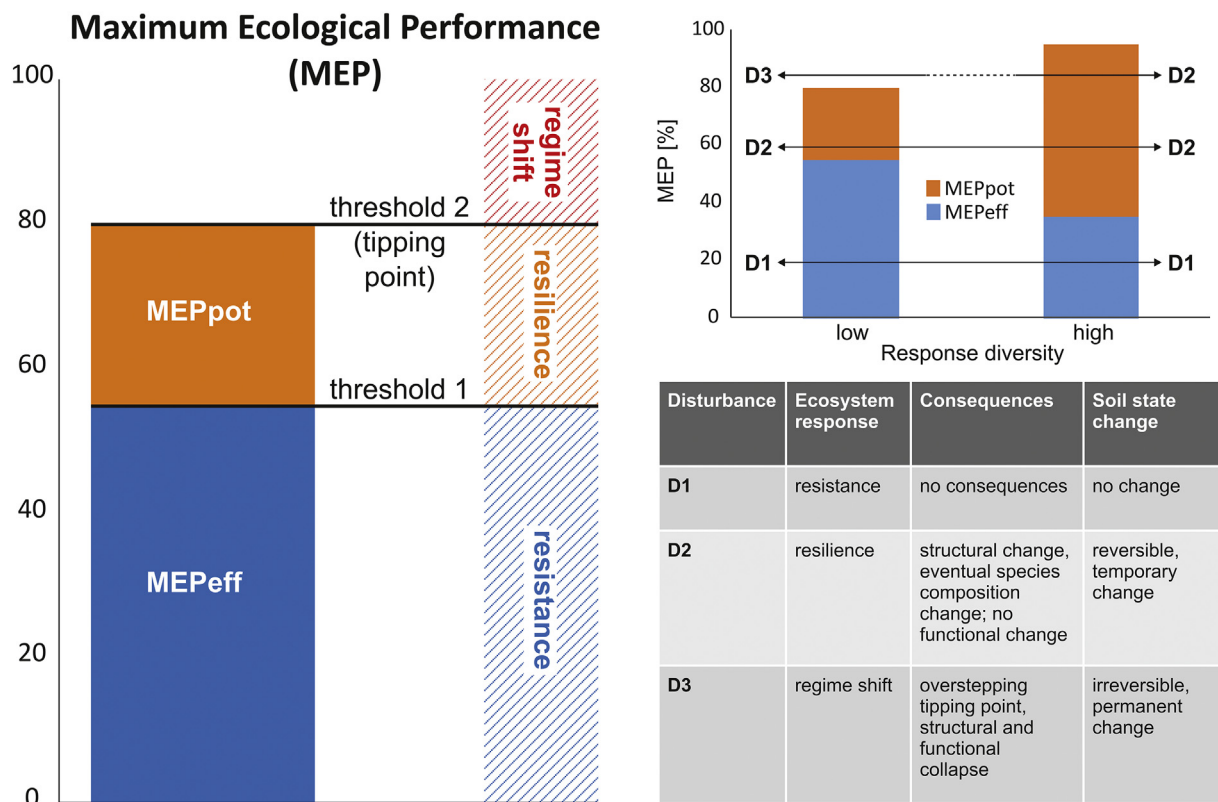
**Fig. 5.** The Maximum Ecological Performance (MEP) of a soil ecosystem with 2 exemplary thresholds that differentiate the ecosystem response to disturbances: resistance, resilience or regime shift (see table with definitions for orders of disturbances D1 to D3) and the differing soil ecosystem response to the same disturbances due to low and high potential MEP (MEPpot) to effective MEP (MEPeff) ratio.

specifically, by integrating metagenomic, metatranscriptomic, metaproteomic and metabolomic data, these approaches allow the bridging of genetic and species diversity, functional potential and actual phenotypic traits. Metagenomic analyses have already allowed unprecedented insights into microbial community-wide responses to soil perturbation, for example to the thawing of permafrost (Yergeau et al., 2010; Mackelprang et al., 2011; Mondav et al., 2014), contamination (Sutton et al., 2013), burning (Oliver et al. 2015), tillage (Souza et al. 2015) or flooding (Argiroff et al. 2017) of soils. By also including the functional omic dimensions, multi-omic analyses allow the resolution of key functional processes which correlate well with biogeochemical process rates, e.g. methanogenesis (Hultman et al., 2015). The multi-omic read-outs thereby represent a good predictor of soil functioning and may be an essential source for multi-factorial markers of key processes which underpin soil services (Fig. 6).

Using the multi-omic approach for soil assessment, a number of key considerations have to be taken into account. First, given the inherent heterogeneity of soil and its constituent microbiota, systematic measurements across all omic levels are essential to allow coherent data integration and deconvolution of significant signals from the multi-omic data. This implies that all biomolecular fractions, i.e. DNA, RNA, proteins and metabolites, have to be obtained from single samples to avoid inconsistent coverage of the different omic levels due to variation introduced from subsampling of the heterogeneous sample material (Roume et al., 2013). Following biomolecular isolation and dedicated high-throughput measurements, the complementarity of the multi-level omic data has to be exploited to allow comprehensive data usage and systematic interrogation of the different levels for the identification of the most informative markers (Fig. 6). More specifically, integrated analysis of the multi-omic data allows enhanced data usage (Narayanasamy et al., 2016), which in turn allows the systematic identification of discriminatory features across the different omic levels, ranging from assessments of structural and functional diversity to the

identification of key functions (Roume et al., 2015; Heintz-Buschart et al., 2016). For the development of such multi-omic markers which represent the different levels of biomolecular information, systematic studies of soils ranging from those with low to high resilience will be necessary to identify markers with high sensitivity and specificity analogous to what has recently been demonstrated in the context of the human gut microbiome and disease (Heintz-Buschart et al., 2016).

Adaptive management has become a prominent concept in natural resource management (Rist et al. 2012) because it is the appropriate scheme when there is uncertainty regarding response to management, but an ability to manage (Allen et al. 2011). An adaptive management
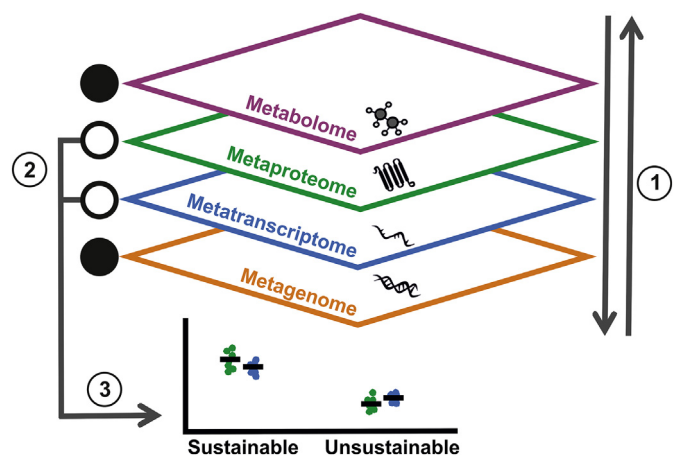


**Fig. 6.** The identification of informative multi-omic markers to assess soil sustainability: (1) Integration of the multi-omic data across the different biomolecular levels; (2) Identification of specific features of interest; (3) Assessment of discriminatory features to assess soil sustainability.

cycle begins with explicit conceptual models about the system at hand, and addresses the management sustainability with variables that can be tested through monitoring. The monitored values have afterwards to be evaluated and the management practices adjusted, before the new management cycle starts. This contrasts with trial and error management, in which management is only adjusting when an error occurs, and a lack of error is already interpreted as successful management (Birge et al. 2016). Adaptive management promotes learning about the system and can reconcile long- and short-term management priorities (Rist et al. 2012). This is especially suited for the soil system, which operates at multiple scales across space and time and is known for nonlinear responses to management (Birge et al. 2016). However, the success of an adaptive soil management depends on the relevance of the soil variables monitored – it requires a parameter that serves as a significant bioindicator, meaning it works on a meta-level, represents the meta-community of the soil and covers all relevant information levels like the past (soil memory), the present (soil state), and the future (soil sustainability prognosis and accordingly recommendations for adjusted management practices). Soil resilience reconciles all these information levels. In general, resilience is known to be responsive to management practices but subject to uncertainties and unpredictability because of its multi-scale and multifunctional source system (Williams 2011). Therefore, adaptive management is recommendable to manage soil sustainability. For measuring soil resilience via the response diversity (MEPpot), multi-omic markers would be particularly valuable. Also the repeated monitoring procedure and the evaluation of the adaptive management scheme would favour working with markers as to be obtainable from the multi-omics approach.

## 4. Conclusions

We need a new understanding and evaluating of the soil ecosystem to manage its complexity and change. Sustainable soil management was yet defined as the efficient use of resources. But this approach is paradox and counterproductive, since short-term efficiency is inhibiting long term sustainability by reducing ecological resilience. Soils represent ecosystems that are moving targets with multiple possible outcomes, being inherently uncertain and unpredictable. Anthropogenic management creates additional press and pulse disturbances for the soil ecosystem. This increases uncertainty. Soils therefore need an adaptive management that leaves opportunity for positive legacy effects (e.g. novelties, redundancy) and time for self-organized restoration periods that both contribute to soil resilience. Resilience is promoted to be a boundary concept to integrate the social and natural dimensions of sustainability (Olsson et al. 2015). The multifunctionality of the soil system makes it difficult to assess the true significance for each single soil function. Bioindicators for monitoring and managing soil sustainability require understanding not only of the components of the systems, but their dynamic interactions over temporal and spatial scales (Deutsch et al. 2002; Gunderson and Holling 2002). As a meta-function of soil, resilience incorporates the multifunctionality of the system. Therefore, it is a promising parameter when it comes to measuring sustainability of soil management, since it reflects both its highly interlinked ecological and social components while its significance for ecosystem functioning is derived from all levels of the functional hierarchy across scales. To be more precise, resilience derives from functional redundancy within scales and functional reinforcement across scales. The measurable part of the soil resilience is the redundancy within scales. It is represented by the response diversity of the soil. Response diversity can be calculated as the Maximum Ecological Performance from the systematic contrasting of genetic potential and functional activity via multi-omic markers. This allows to identify the soil ecosystem thresholds of resistance and resilience and potential tipping points for a regime shift towards irreversible or permanent unfavorable soil states (e.g. rigidity or poverty traps). This will also be of increasing interest due to the emerging global efforts to integrate more soil indicators into the UN

Sustainable Development Goals (SDGs) as currently postulated throughout the international community (IASS 2015; Keesstra et al., 2015; Jónsson et al. 2016). We assume, that arable soils (e.g. in comparison to forest soils) on the one hand have a high ecological resilience, because they are disturbance-trained (frequent structural collapses, mechanical and chemical shocks due to tillage, harvest, agro-chemical applications) Arable soils therefore have a higher response diversity than forest soils (Szoboszlay et al. 2017). On the other hand, especially agroecosystems experience the efficiency of production which is only supporting what is directly and immediately beneficial, leading to drastic losses in soil resilience (Walker et al. 2012). Hence, to find management solutions for challenges linked to the nexus of water, energy and food, we need holistic approaches that explicitly draft meta-functions like resilience to any sustainability goal (McCormick and Kapustka 2016). We particularly need measures to monitor this part of arable soil resilience and adjust our adaptive management accordingly, to sustain the soil's functional integrity. Using the multi-omics approach in this context enables to resolve higher-level features, i.e. resilience, along with very specific and discriminatory features like key functions and organisms that have a disproportionate effect on soil functioning and are therefore vital. The concept presented discusses resilience as a measurable bioindicator for soil management sustainability by identification of soil ecosystem thresholds. The calculation of such thresholds is to our opinion the current global cross-disciplinary challenge, involving an evaluation that never stops asking the right questions.

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