An integrated approach to above- and below-ground ecological monitoring for nature-based solutions

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Abstract

1. As the development of nature-based solutions (NbS) increases globally, it is important to ensure that projects meet the objective of benefiting biodiversity, alongside tackling societal challenges. However, most NbS projects do not directly monitor ecological outcomes, and those that do often focus on a limited set of metrics. It is therefore challenging to assess whether projects fulfil the aim of benefiting biodiversity.
2. We aimed to identify the most informative and feasible ecological metrics, both above- and below-ground, for monitoring ecological outcomes of NbS. We conducted a structured non-systematic literature review to identify possible biodiversity and soil health metrics, then developed a scoring system to rank these based on their informativeness and feasibility for monitoring.
3. Metrics were categorised into compositional, structural, and functional aspects of biodiversity, and biological, physical, and chemical aspects of soil health. We grouped biodiversity and soil health metrics into Tier 1 (the most informative and feasible metrics), Tier 2 (informative metrics with some limitations in scope or feasibility), and Future metrics (highly informative metrics which are currently less feasible to monitor). Tier 1 metrics collectively address multiple aspects of biodiversity and soil health and are currently the highest priority for NbS projects to assess. For biodiversity nine Tier 1, six Tier 2, and 15 Future metrics were identified, and for soil health 11 Tier 1, six Tier 2, and five Future metrics.
4. We identified existing standardised methodologies for monitoring the proposed metrics, noting that for many metrics standardised methodologies are not available and threshold or reference values for each metric are missing.
5. *Solution:* Our study provides practitioners with a framework for selecting optimum metrics for assessing above- and below-ground ecological outcomes of NbS relevant to the place in which they are being implemented. A definition of each metric and

standardised methodologies for collecting data are summarised, providing information to develop an ecological monitoring protocol for an NbS project. The information on each metric has been made freely available as a searchable database in an interactive online interface geared towards UK practitioners, but with wider applicability.

Keywords

Biodiversity, soil health, ecological monitoring, nature-based solutions

The need to monitor the ecological outcomes of NbS

Nature-based solutions (NbS) involve working with and enhancing nature to tackle societal challenges. Biodiversity should be at their core, underpinning the benefits they deliver and benefiting from the interventions (Seddon et al. 2020, 2021). Uptake of NbS is accelerating globally, as their ability to tackle societal challenges from climate change to food security is increasingly recognised (Seddon et al. 2019; Chausson et al. 2020; Donatti et al. 2022).

Nature underpins the delivery of ecosystem services through multiple pathways, from provision of supporting habitat to the presence of a particular species or functional group (Smith et al. 2017). These pathways are linked to biotic and abiotic attributes representing both above- and below-ground components of ecosystems at multiple scales, such as species richness, landscape diversity or geology (Smith et al. 2017). Above- and below-ground components of ecosystems should not be considered in isolation, as their reciprocal interactions influence ecosystem function and delivery of ecosystem services (Chomel et al. 2022). Soil health and quality concepts are therefore increasingly integrated into ecological restoration, as soil biota and processes significantly impact overall ecosystem health and the establishment of aboveground communities (Farrell et al. 2020; Young et al. 2005). Soil communities underpin many processes and attributes linked to key ecosystem services, such as erosion control, hydrological functions, and nutrient exchange (Bhaduri et al. 2022; Barrios 2007; Farrell et al. 2020). Maintaining soil health and biodiversity are also fundamental to achieving ecosystem stability and resilience in the face of perturbations such as climate change (Seddon, Turner, Berry, Chausson, & Girardin, 2019).

Despite the recognition that biodiversity and soil health underpin successful, resilient NbS that provide multiple benefits to people, benefits for biodiversity are often implicitly assumed rather than explicitly planned and monitored. For example, a review of 386 studies on NbS for climate

change adaptation found that only 34% reported on ecological outcomes (Chausson et al. 2020) and these often focussed on a limited set of metrics (Key et al. 2022). Empirical monitoring of the ecological outcomes of NbS projects allows objective assessment of success, providing an important crossover between science and policy (Mallette et al. 2022; Lovett et al. 2007). More effective monitoring strategies are therefore needed for biodiversity and soil health within NbS projects, to track their trajectory of change, confirm positive outcomes, highlight trade-offs, and assess efficacy of management (Farrell et al. 2020).

As part of a recent initiative to identify how to scale up the deployment of high-integrity NbS in the UK (Agile Initiative, n.d.), practitioners identified a need to build the evidence base on 'what works' through more effective and consistent monitoring, while also taking account of limits on time and resources. However, with a confusing array of possible indicators, each with different strengths and limitations, guidance is needed to help practitioners identify appropriate biodiversity and soil health metrics when developing monitoring strategies to meet their project and place-based objectives (Bünemann et al. 2018; Bhaduri et al. 2022; Knight et al. 2020; Noss 1990).

We addressed this challenge by developing a framework for selecting metrics to monitor biodiversity and soil health within NbS projects.

- We summarise a literature review identifying a range of possible biodiversity and soil health metrics across categories at multiple scales.
- We present a strategy for prioritising biodiversity and soil health metrics for monitoring NbS, based on metric informativeness and feasibility of monitoring.
- Using the UK as a case study, we identify existing protocols that can be used to monitor metrics and identify gaps where standardised methodologies are required.

- We provide a case study exploring the practical implementation of our monitoring framework.
- We highlight technological innovations that could simplify the monitoring process for practitioners.

A searchable database of the metrics and their characteristics is available in an interactive web platform (geared towards UK practitioners but with wider applicability) as part of the NbS Knowledge Hub developed through the Agile Initiative (https://nbshub.naturebasedsolutionsinitiative.org/monitoring-outcomes/).

A conceptual framework for biodiversity and soil health metrics

Effective monitoring programmes should use accepted, and ideally standardised, methods to ensure consistent and cost-effective collection of high-quality data (Lovett et al. 2007; Pocock et al. 2015). The planned data use and outputs should be considered throughout the planning process and aims should be clearly defined and maintain relevance into the future (Lovett et al. 2007; Pocock et al. 2015). Conservation activities frequently have an associated monitoring programme, but without an underpinning quantitative and scientific approach, their impact may be limited (Legg & Nagy 2006). Alongside characteristics that will ensure the scientific robustness of monitoring, consideration of the practicality and cost-effectiveness of monitoring are crucial considerations for practitioners (Wurtzebach & Schultz 2016; Cosović et al. 2020; Czúcz et al. 2021; Heink & Kowarik 2010).

Monitoring of biodiversity and soil health outcomes requires selection of appropriate indicators, i.e. measurable characteristics, that are widely representative of each area (Niemi & McDonald 2004). The complexity and multidimensionality of both biodiversity and soil health often results in monitoring a limited set of metrics (Key et al. 2022; Bünemann et al. 2018). One strategy for

identifying an informative set of metrics is to subdivide biodiversity and soil health into distinct concepts, so that relevant metrics for each aspect can be assessed (Niemi & McDonald 2004). For example, Noss established the biodiversity hierarchy, which partitions biodiversity into structural, compositional, and functional components (Fig. 1a), which can be further categorised by scale (gene to landscape) (Noss 1990; Niemi & McDonald 2004). Similarly, soil health can be subdivided into physical, chemical, and biological characteristics (Fig. 1b) (Jian et al. 2020; Stewart et al. 2018; Guo 2021). To achieve a comprehensive and representative understanding of ecological change, multiple complementary metrics should be assessed (Niemi & McDonald 2004; Knight et al. 2020; Czúcz et al. 2021).

Noss (1990) breaks biodiversity down into composition (i.e. the identity and variety of elements within a system e.g. species diversity), structure (i.e. the physical organisation of a system e.g. habitat complexity), and function (i.e. the processes acting within a system e.g. nutrient cycling). Noss' Framework has been proposed as the underpinning structure for other biodiversity and ecological monitoring approaches, such as monitoring biodiversity mitigation projects and assessing ecological integrity (Andreasen et al. 2001; Knight et al. 2020). This framing provides a systematic approach to developing monitoring programmes with a more comprehensive coverage of biodiversity. Most biodiversity monitoring currently fails to achieve this: biodiversity is often equated to species richness, and functional and structural aspects rarely assessed (Feld et al. 2009). Species richness inadequately reflects functional and compositional diversity, and therefore provides partial information on overall biodiversity (Lyashevska & Farnsworth 2012; Hines & Pereira 2021). An ideal monitoring approach would include metrics across the axes of composition, structure, and function, with representation at multiple scales (genetic, population, ecosystem, landscape) (Knight et al. 2020).

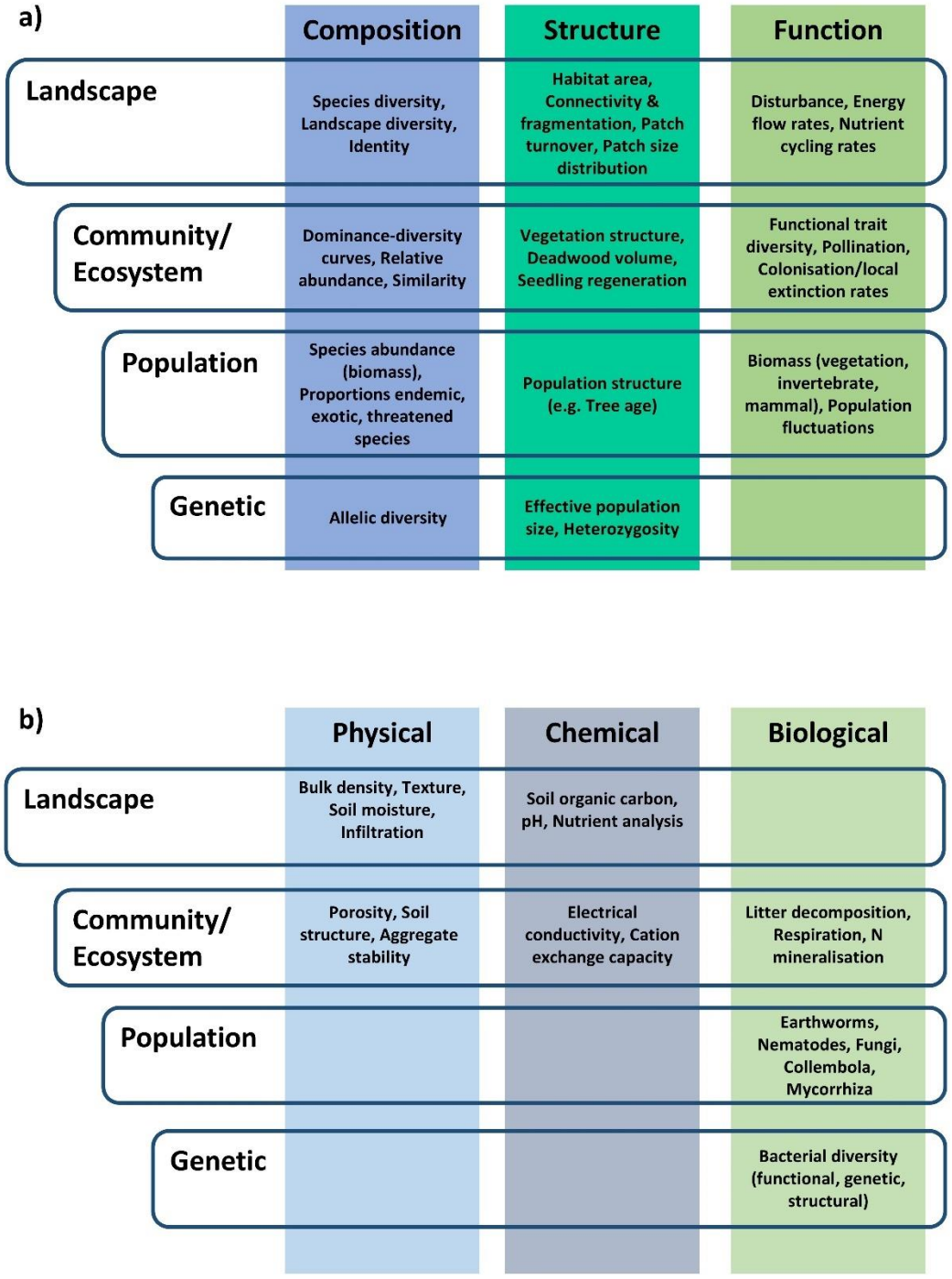

Figure 1 Metrics representing the primary attributes of (a) biodiversity (composition, structure, function), adapted from Noss (1990), and (b) soil health (physical, chemical, biological) at multiple scales (landscape, community/ecosystem, population, genetic). Some metrics apply to multiple axes and scales (see Tables S1 & S2).

Soil health is a similarly complex concept; soils are heterogeneous and vary depending on their environmental context and site history, which determines the functions they provide (Bünemann et al. 2018). Soil health indicators can be grouped into biological (properties relating to living organisms e.g. fungal diversity), physical (soil structural properties e.g. bulk density), and chemical indicators (properties linked to soil chemical composition e.g. pH). These three categories interact to maintain soil functions (Guo 2021). Typically, physical and chemical properties are easier to measure and interpret, with greater research and development required for biological indicators (Guo 2021). Coverage across different soil physical, chemical, and biological properties varies in soil monitoring programmes, and many approaches cover a limited set of indicators (Bünemann et al. 2018, Harris et al. 2023, Loveland & Thompson 2002). Similar to biodiversity assessment, effective soil health monitoring should cover a range of indicators to better represent soil health (Guo 2021).

Metric framework design and prioritisation

The compositional, structural, and functional axes of biodiversity (Fig. 1a) and the physical, chemical, and biological axes of soil health (Fig. 1b) provide a structure for identifying indicators to assess ecological outcomes of NbS. We carried out a structured non-systematic review of academic and grey literature, based on keyword searches, to identify possible metrics of biodiversity and soil health across each axis at different scales (see Supplementary Methods for full details). For each category of biodiversity, Noss defines metrics or groups of metrics, these were supplemented by non-systematic literature searches. Soil health metrics were identified using non-systematic literature searches of the academic and grey literature. We only considered metrics with a clear relationship with biodiversity or soil health.

For each metric, we extracted information on its informativeness (e.g. relevance, sensitivity, and applicability) and feasibility of monitoring (e.g. ease of sample collection, cost, and technical expertise needed) (Table 1). A scoring system was developed to objectively assess the metrics and group them into three categories: Tier 1, Tier 2, and Future, based on informativeness and feasibility (Fig. 2) (full details in Supplementary Methods). Tier 1 metrics had the highest informativeness scores and met a minimum feasibility score; we also aimed to ensure that the Tier 1 metrics collectively capture all axes of biodiversity and soil health at multiple scales (Tables S1 & S2). Tier 2 represents metrics that are useful and informative but to a lesser degree than Tier 1 indicators, or only apply to some ecosystem or soil types, and met the minimum feasibility score. Future metrics are assessed as being highly informative but currently not likely to be widely feasible for regular monitoring. We also collated information on technological innovations that could simplify or accelerate the data collection process in the future. It is anticipated that Future metrics could move into Tier 1 if technological improvements simplify the monitoring process or reduce costs, or as guidance on data collection improves.

Table 1 Scoring criteria for biodiversity and soil health metrics.

Informativeness	Feasibility
Relevance: How strong is the evidence that the metric is directly or indirectly relevant to biodiversity/soil health?	**Sample collection:** How straightforward is sample collection & analysis?
Information rich: How many metrics can be calculated from one data collection method? Can the metric be used as a surrogate for other metrics?	**Cost:** How expensive is data collection and analysis?
Sensitivity: How sensitive is the metric to management changes?	**Technical:** How much technical expertise or equipment is needed?
Functions/services: Are there clear links between the metric and ecosystem functions and derived services?	**Methodology:** Is there an existing standardised methodology available?
Applicability: Can the metric be applied across habitat types?	**Compatibility:** Is the methodology robust and repeatable?
Literature: How widely is the metric considered in the academic literature?	**Interpretation:** How easy are results to interpret?

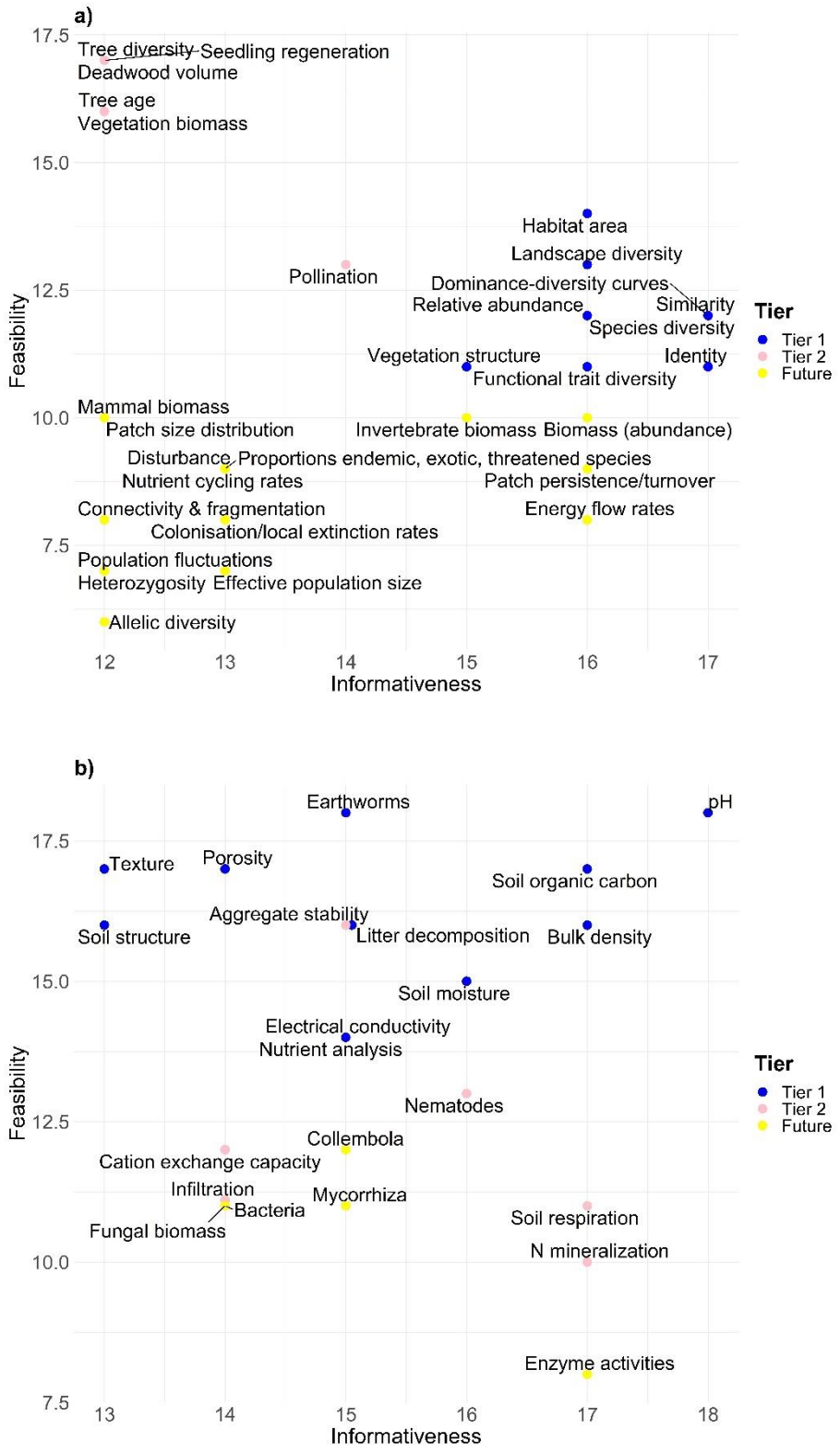

Figure 2 (a) Biodiversity and (b) soil health metric scores plotted on the axes of informativeness and feasibility.

Exploring the priority metrics identified

The highest ranked (see Supplementary Methods for scoring system) compositional biodiversity metrics included species diversity and functional diversity, as well as several less familiar metrics that provide insights on how composition influences function: dominance-diversity curves, identity, relative abundance, and similarity (Table S1). Dominance-diversity curves highlight changes over time in dominant and rare species that influence ecosystem functioning (Hillebrand et al. 2018). Identity and functional trait diversity capture the presence of traits that are likely to influence key ecological processes and therefore ecosystem services (Buckland et al. 2005; Hillebrand et al. 2018); they also have the advantage of simultaneously providing diversity and functional information. Habitat area, landscape diversity, and vegetation structure were the top ranked structural biodiversity metrics. Habitat area is relatively simple to measure and strongly determines overall biodiversity. Together with landscape diversity, this provides a proxy measure for the capacity of a landscape to support different species groups and can be measured at different scales that will be relevant to different groups (Maskell et al. 2019; Deane et al. 2020; Morelli et al. 2013; Dangerfield et al. 2003). The ease of monitoring vegetation structure varies with habitat type; at its simplest it can involve categorising variation in tree/shrub height or diameter within a woodland, and in grassland point-intercept and vegetation height variation methods can be applied (Table S3).

Tier 1 physical soil health metrics included bulk density, texture, soil moisture, porosity, and soil structure (Table S2). Bulk density, texture, and porosity provide information on soil structure which can be used to infer soil compaction and determine further critical properties related to the interaction of soil with air and water (Merrington et al. 2006; Cardoso et al. 2013; Schoenholtz et al. 2000). Bulk density is also needed to convert percentage nutrient or carbon values into volumetric measures. The highest ranked chemical soil health metrics were soil carbon, pH, nutrient analysis, and electrical conductivity. Soil carbon influences multiple soil functions such

as nutrient storage, water retention capacity, and aggregate stability, and strongly influences microbial activity (Cardoso et al. 2013). Nutrient content and type support the soil biotic community, which in turn determines nutrient cycling and decomposition (Merrington et al. 2006). Earthworm abundance and litter decomposition are the most easily monitored soil biological metrics. Earthworms act as a proxy for the wider soil ecological community and can also reflect soil physical structure and water dynamics (Pulleman et al. 2012; Griffiths et al. 2016). Earthworms are not found in some habitats, e.g. acidic and/or waterlogged soils; however, they were prioritised due to their ease of monitoring relative to other possible groups such as nematodes and potworms. Litter decomposition plays a key role in nutrient and carbon cycling, and rates provide insights into the microbial community (Guerra et al. 2021).

Standardised methodologies for collecting biodiversity and soil metrics in the UK

Once suitable ecological metrics have been selected, well-designed data collection methods are needed to ensure useful data is produced (Legg & Nagy 2006; Pocock et al. 2015; Lovett et al. 2007). Methods should be consistent through time, accurate, and follow accepted approaches, as the field design and methods determine future analyses, statistical rigour, and the questions that can be addressed (Lovett et al. 2007; Legg & Nagy 2006; Pocock et al. 2015). Standardisation of the data collection approach allows between-project comparisons or comparisons to datasets collected using consistently similar methods. If regional or national coordination of monitoring is possible, this also increases the potential for between-project comparisons, as well as allowing site-level comparisons to regional or national trends (Pocock et al. 2015).

To help guide practitioners on the best way of collecting data on metrics, we searched for suitable standardised data collection methodologies for each metric. In line with the wider initiative to provide tools and guidance for NbS practitioners (Agile Initiative, n.d.), we focused on

identifying methodologies that could be used in the UK, where there is a long history of biodiversity and soil monitoring. However, many of the methods will be more widely applicable.

For biodiversity metrics, standardised methodologies have most widely been developed to monitor specific species groups (Table S3). For example, in the UK there are methodologies for plants (National Plant Monitoring Scheme (NPMS 2019)), butterflies (UK Butterfly Monitoring Scheme (UKBMS 2021)) and birds (UK Breeding Bird Survey (BTO 2022)), and the UK Environmental Change Network (UKECN 2022) provides methods for carabids, spiders, moths, bats, frogs, crane flies, spittle bugs, and wild herbivores (rabbits & deer). Classification of habitat categories based on characteristic plant species and communities is also possible using the UK Habitats Classification (UKHab Ltd 2023). These species and habitat recording methodologies provide a basis for collecting the data that can be used to calculate several of the metrics we have identified as priorities for monitoring. However, in many cases an additional step which is not defined in the standardised methodology is required to calculate the final metric. For example, functional trait diversity could be calculated by linking trait information to species-level data collected during the species diversity surveys (Waldén et al. 2023). There are some examples of scientific papers that have assessed functional trait diversity and there are various databases with trait information for different species groups (e.g. Ecological Flora of Britain and Ireland http://ecoflora.org.uk/). Although this process has not been standardised, we included it in Tier 1 to represent the function axis of biodiversity and due to its high informativeness. There are also more developed standardised methodologies for a subset of biodiversity metrics relevant to woodland ecosystems (Table S3: Vegetation structure, Deadwood volume, Seedling regeneration, Tree age, Tree diversity, Vegetation biomass), which were developed for the UK National Forest Inventory (Forestry Commission 2016a) and provide information on the full process from sampling design and methodology through to calculation of the derived metrics.

The soil health concept has been extensively applied to farm systems (Griffiths et al. 2018) and consequently, there is reasonable coverage of methodologies targeting farmers for many identified metrics, which can usually be applied to non-farm habitats with little or no modification (Table S4). The Farm Carbon Toolkit, which was produced in the UK for farmers aiming to measure and understand their soil carbon stocks, provides information on sampling design (e.g. sample size, layout of samples, soil sampling depths) which can be applied to multiple other soil health metrics, as well as soil carbon (Farm Carbon Toolkit 2021). The Food and Agricultural Organisation of the United Nations has developed the Global Soil Doctors Programme which provides protocols for assessing many key soil health metrics (FAO 2020). Methodologies to assess the biological aspects of soil health are less well represented; some metrics have been assessed in scientific studies, however more work is required to develop widely applicable and robust standardised methodologies (Pulleman et al. 2012). Earthworms are one of the more accessible soil biological indicators and are the subject of the Earthworm Watch Citizen Science monitoring project in the UK and one of the indicators in the UK Centre for Ecology and Hydrology benchmark for soil health (Burton et al. 2024, Feeney et al. 2023). The UK's Agricultural and Horticultural Development Board provides a methodology for assessing earthworms using soil pits, which can be combined with the Farm Carbon Toolkit sample design recommendations (Stroud & Bennett 2018). However, other aspects of soil biodiversity are more complex to monitor, requiring greater expertise and with multiple potential methodologies available. For example, fungal biomass can be estimated using phospholipid fatty acid, ergosterol, or quantitative PCR analysis (Bünemann et al. 2018).

The way data is collected (i.e. sample design, replication, frequency) will determine its analysis (Ockendon et al. 2021). However, to gain useful information from the data collected, a point of comparison is needed (Pocock et al. 2015; McGlone et al. 2020). The gold standard study design is a Before-After-Control-Impact (BACI) design, which tracks the focal variables before and after

an intervention, with monitoring conducted simultaneously at a control site (Christie et al. 2019). However, a BACI design requires a relatively high investment of time and resources, and simpler designs are often used. For example, a Before-After (BA) design, which involves monitoring metrics before and after the interventions within the project site only (De Palma et al. 2018; Christie et al. 2019). However, the BA design makes a major assumption that the focal variables would not have changed without the intervention, and therefore does not account for variation resulting from other drivers (Christie et al. 2019, Wauchope et al. 2021). Ideally, projects would conduct monitoring at a control site, to understand whether the variable in question is improving or declining because of the project interventions (Wauchope et al. 2021). If projects collect data over an extended time-series, this offers the opportunity to assess metric trajectories over time (Wauchope et al. 2021). Ideally these trajectories of change would be compared to trends at comparator sites indicating a desirable status for that metric or to threshold values. For many of the metrics we assessed it was challenging to identify thresholds to indicate the direction of change, and it was clear that indication of positive or negative change would be context dependent (Tables S5 & S6).

Practical application of the framework – a case study

The final set of metrics selected for monitoring will be project-dependent; multiple dimensions of an ecosystem should be covered, but not every aspect needs to be assessed to understand the overall ecological response, direction of change, and whether project goals have been met (Andreasen et al. 2001). Our framework is available on an interactive web platform (https://nbshub.naturebasedsolutionsinitiative.org/monitoring-outcomes/) which allows users to filter metrics based on different criteria (Metric type, Aspect of biodiversity or soil health, Scale, Ecosystem, Cost, Technical Expertise, Standardised methodology) (Fig. 3). Once metrics have been selected, a sampling design and data collection plan will be developed for the project. Even when a standardised methodology is available, the precise sampling design will usually

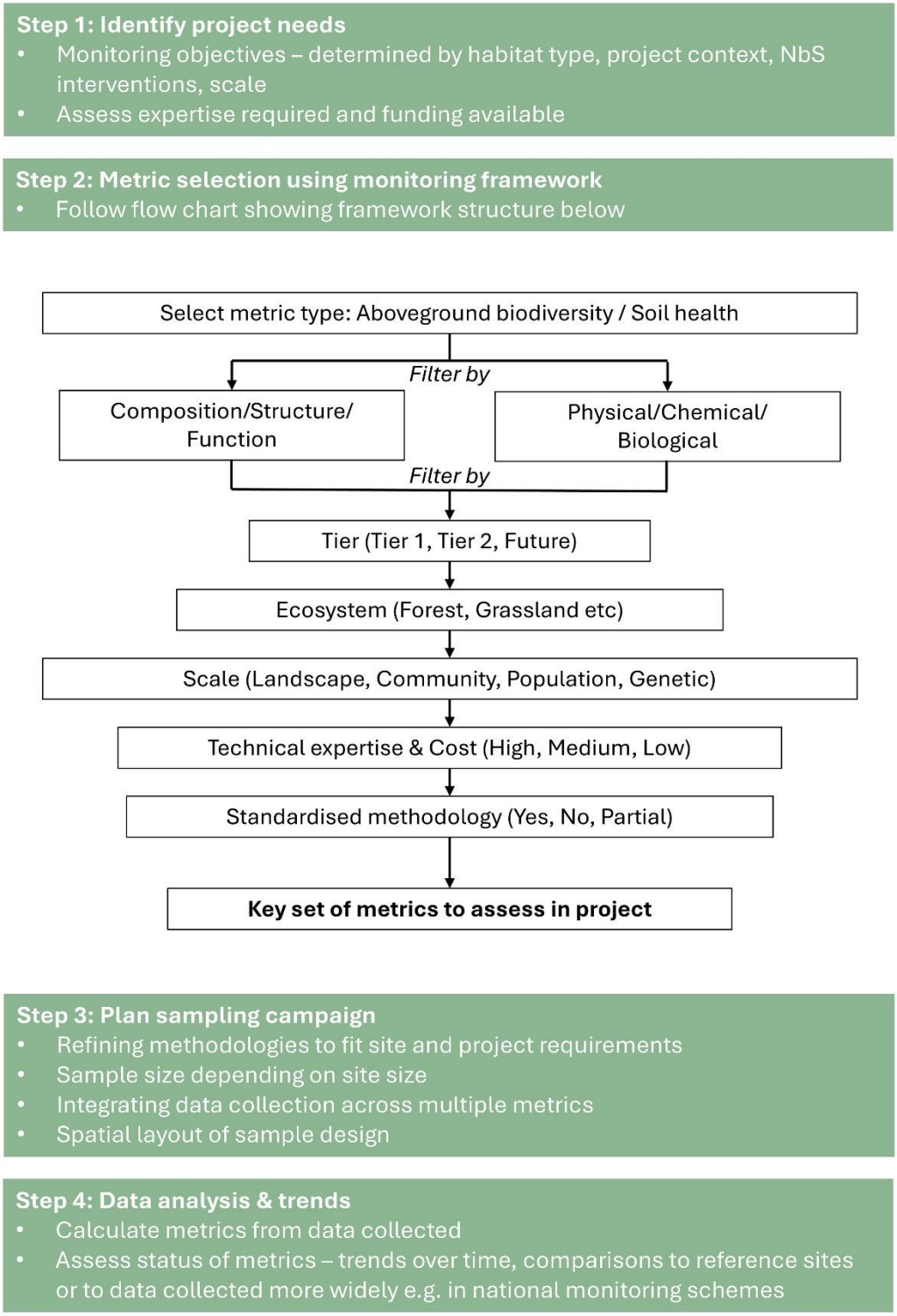

Figure 3 The steps involved in designing an ecological monitoring strategy for an NbS project. The flowchart in Step 2 captures the structure of the monitoring framework, available at https://nbshub.naturebasedsolutionsinitiative.org/monitoring-outcomes/.

need to be adapted to the layout and size of the NbS project site. A User Guide developed alongside the framework provides guidance on how to develop a monitoring plan for an NbS project (Warner et al., 2024). In this section, we illustrate the application of our monitoring approach using a theoretical example of an NbS project in the UK, following the steps outlined in Figure 3. Our hypothetical case study involves restoration of arable land to a mix of woodland, species rich grassland and wetland (Fig. 4), with the aims of sequestering carbon and slowing the entry of water to the adjacent river to reduce flood risk. We outline a set of metrics representing different aspects of soil health and biodiversity at multiple scales that could be selected to capture key aspects of this example project.

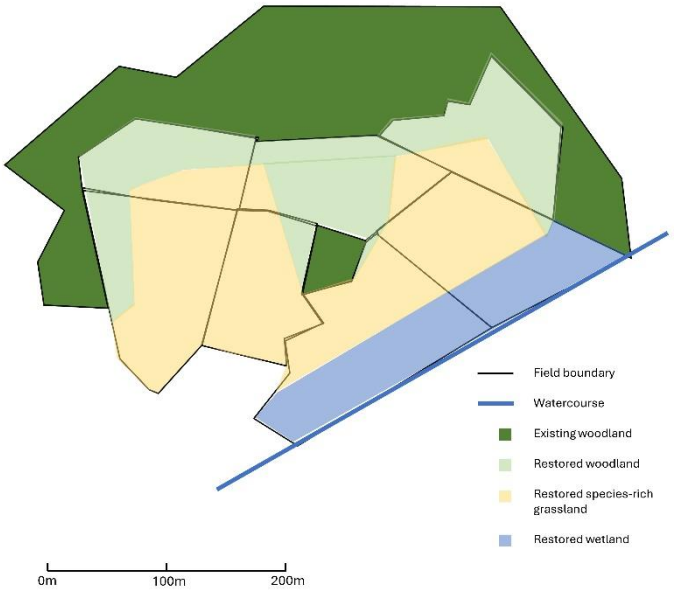

Figure 4 Schematic of a theoretical NbS project, showing the target habitats overlaid onto the existing field boundaries.

The project objectives require a method for tracking development of the target habitats, and ideally some direct biodiversity monitoring to assess changes in ecological communities in response to this. The carbon objectives of the project could be monitored by deriving carbon estimates from vegetation biomass and soil carbon monitoring. Finally, variables to track the recovery of the soil from past cultivation and assess whether its capacity to hold water is increasing would also be useful.

A starting point for biodiversity monitoring would be to monitor habitat development within the site to track the establishment of the target habitats over time, classifying habitat types using the UK Habitats Classification (UKHab Ltd 2023). This information can also be used to calculate landscape diversity and provides the potential to explore more complex metrics such as habitat connectivity and fragmentation, if this expertise was available in the project. Species level surveys representing different parts of the community, e.g. plants, carabids beetles, birds, could be used to calculate species diversity, and derive identity and relative abundance metrics. If reference habitats representing the target habitats exist nearby, similarity to the target habitat for each community of organisms could be calculated either using existing data for those sites (if available) or by conducting additional surveys. Vegetation structure could be monitored to assess the development of the habitat and biomass, for example by direct measurement of tree diameter at breast height, and above-ground carbon estimates can then be derived from the same dataset (Broughton et al. 2021).

For soil health, bulk density could be monitored to assess changes in soil compaction with the withdrawal of cultivation and to allow any soil carbon or nutrient analyses to be converted into volumetric measures (Merrington et al. 2006). Soil carbon would be assessed to evaluate uptake of soil carbon with the return of semi-natural habitats, especially as the soil carbon baseline would be expected to be low after a history of arable cultivation (Smith et al. 2020; Smith 2004).

In a post-arable site, if fertiliser has been used there could be a high nutrient load, which could limit the development of botanically diverse semi-natural habitats (Moeslund et al. 2023; Cramer et al. 2008). Nitrogen and phosphorus (nutrient analysis) monitoring would track changes in nutrient availability; depending on prior agricultural management the land may be suffering from nutrient enrichment or depletion, which can determine the trajectory of habitat restoration (McLauchlan 2006; Parkhurst et al. 2022; Cramer et al. 2008). Soil pH can provide additional context to other soil health measures and can also be used to track recovery of soil from agricultural activities (Cardoso et al. 2013). Given the focus on semi-natural habitat restoration for water retention, the porosity and infiltration of the soil will indicate the water-holding capacity of the soil and the recovery of desirable soil structure following cessation of disturbance from agricultural activities (Cleophas et al. 2022; Lipiec et al. 2006). At present, the most feasible biological indicator is earthworms (abundance/biomass/diversity), which is also positively correlated with water infiltration rates, as well as providing information on this important group of soil organisms (Griffiths et al. 2018).

Once the target metrics have been selected, the sampling design can be developed. Where possible, data should be collected at the same spatial locations for all metrics, so that the relationships between different metrics can be assessed. The habitat surveys conducted as part of the biodiversity monitoring approach provide the basis for developing the experimental design, as the sample replicates can then be stratified and allocated to each of the habitat areas. In the case of our theoretical study site, the area is also further subdivided into the individual former agricultural fields, which should also be used to guide the sampling design, as their unique management histories may influence the trajectory of above- and below-ground ecological components after restoration. For the soil health metrics, the Farm Carbon Toolkit offers advice on sample size and layout, recommending 5-15 samples per sampling unit, so in this case 5-15 samples in each habitat type-field combination (Farm Carbon Toolkit 2021). The same soil

samples can be used for the bulk density, soil carbon, soil nutrient and porosity analyses. The earthworm and infiltration sampling methodologies can be carried out at the same locations where the soil samples were collected. The species-level survey methodologies for all groups follow either a plot or transect approach. Our theoretical site is >10 ha in size, so we recommend a minimum of 3 replicates per sample subunit (habitat – field combination), following the guidance from the UK Plant Monitoring Scheme for plants, UK Environmental Change Network for carabid beetles, and UK Breeding Bird Survey for birds (BTO 2022; UKECN 2022; NPMS 2019). The standardised protocols for assessing vegetation structure in woodland and modified grassland protocol suggest slightly different plot shapes and sizes but data collection could still be centred on the same locations as the species diversity plots (Forestry Commission 2016b; Wood et al. 2012). An additional step to monitor changes in species identity would require the project to use functional trait databases to link relevant traits to the species-level data. For example, the Ecological Flora of Britain & Ireland (http://ecoflora.org.uk/) could be used together with trait data from previous studies e.g. (Spake et al. 2016) to link trait data to the species data collected. This adds an important additional layer of information on the functional element of biodiversity.

As with all monitoring programmes, sampling intensity will be balanced against practical constraints (Weiser et al. 2019). For many of the sampling approaches, alongside the time taken to collect samples and data in the field, further time and money is associated with processing samples collected (chemical analysis of soil samples, identifying specimens) and the subsequent data analysis (Mandelik et al. 2010). Future innovations that could reduce the financial or practical investment in any of the stages of monitoring therefore have the potential to increase the overall feasibility of monitoring.

Field data collection will be followed by analysis. Ideally metrics would be compared to known thresholds that indicate change in a positive or negative direction, however these are not available in many cases (Tables S5 & S6). Nationwide trends or data collected from reference sites offer potential alternative points of comparison.

Technological innovations to increase monitoring feasibility

Interest in technological advances that could simplify and reduce the costs of ecological monitoring is increasing. The most widely proposed innovations are remote sensing, acoustic monitoring, environmental DNA (eDNA), and artificial intelligence (AI) (Van Klink et al. 2024; Ford et al. 2024). We need to understand the extent to which these technologies could replace traditional ecological data collection methods, and the additional developments required for them to do so (Besson et al. 2022). During the literature review process, we identified emerging technologies which could enhance or replace traditional data collection methods (Tables S7 & S8).

Remote-sensed Earth Observation (EO) data has applications for quantifying taxonomic, structural, and functional biodiversity metrics at multiple scales (Lausch et al. 2016). Space and airborne EO sensors can detect a wide range of spectral signals (optical, thermal, radar) and information can also be captured using lasers (Lausch et al. 2016). A large focus of EO has been capturing plant spectral traits ranging from biochemical and biophysical, to functional and morphological (Frye et al. 2021; Lausch et al. 2016; Schweiger et al. 2018). Morphological, physiological, phenotypic, and activity traits can also be captured for some animal species, but this depends on characteristics such as body size and the resolution of the sensors (Lausch et al. 2016). These traits can be used as proxies for plant species and communities, ecological processes, and by extension wider aspects of the ecological community (Lausch et al. 2016). The accuracy of the relationship between the spectral traits and variables of interest is determined

by the sensor used, species characteristics, and assumptions used to fit the remote sensed data (Lausch et al. 2016). Habitat area, configuration, and diversity metrics can also be calculated from high resolution remote sensed imagery (Price et al. 2023; Sittaro et al. 2022). Structural data collected using LiDAR can be translated into vegetation structure and biomass metrics (Jucker et al. 2023; Broughton et al. 2022). The availability of space-collected data relevant to biodiversity monitoring is increasing, and structural and functional metrics are currently the most feasible to monitor (Skidmore & Pettorelli 2015; Pettorelli et al. 2016; Skidmore et al. 2021).

eDNA provides an approach for identifying the organisms present in an ecosystem by extracting DNA from an environmental sample such as water or soil (Bohmann et al. 2014). eDNA can often be a cost-effective method of gaining information on species present in a location and samples are collected relatively easily (Pereira et al. 2021; Bohmann et al. 2014). Sensitivity varies by taxon, but it can provide particular advances for cryptic or hard to identify species (Fediajevaite et al. 2021). eDNA analysis does not provide abundance data, so impact could be increased by combining eDNA surveys with traditional methods (Pereira et al. 2021; Deiner et al. 2017). The resolution of the resulting dataset depends on the completeness of reference databases which the sequence data is compared to, and these vary geographically and taxonomically (Keck et al. 2023). Processing eDNA samples, dataset curation, and analysis requires specialist skills; although commercial companies offer this service, cost can be a significant barrier (Larson et al. 2020). Directly metabarcoding plant or animal samples (e.g. a pitfall trap sample of many invertebrates) uses similar approaches to eDNA analysis and can be particularly useful when identifying taxonomically challenging groups e.g. invertebrates (Kirse et al. 2021).

Passive acoustic monitoring captures the soundscape generated by ecological communities and is relatively low-cost and easy-to-deploy (Ford et al. 2024). The data collected represents species that produce noise (e.g. bird vocalisations, cricket chirps, bees buzzing) and with post-

processing can provide estimates of species richness and occupancy models (Ford et al. 2024; Sethi et al. 2023). Generation of species richness metrics from acoustic data requires large amounts of training data for comparison and models can only detect vocalisations from well-characterised, common species (Sethi et al. 2023). There is also interest in the generation of overall soundscape metrics that, for example, can be used to compare restored ecosystems to a reference state (Sethi et al. 2020). These rely on complex modelling methods such as convolutional neural networks (Sethi et al. 2020). Whole-system metrics derived from overall soundscapes have been shown to correlate positively with biodiversity at a site, however, these relationships are not replicated across multiple sites, limiting their wide applicability (Sethi et al. 2023). Therefore, it is suggested that acoustic monitoring is conducted alongside traditional ecological monitoring (Sethi et al. 2023). Soundscape monitoring can also be applied to belowground communities, and a recent study found correlation between acoustic diversity and invertebrate abundance, but not richness (Robinson et al. 2023). Soil soundscape monitoring needs more development and refinement, as it currently sounds produced by living organisms cannot be distinguished from sounds generated by physical soil movement, and there are limited soil organism reference datasets (Metcalf et al. 2023; Robinson et al. 2023).

Many of the large, complex multi-dimensional datasets generated by remote sensing, eDNA, and acoustic monitoring require more sophisticated, technical, and time-consuming data-processing and analysis methods (Besson et al. 2022). Modelling approaches such as computer audition and vision packages, and machine learning could provide the final step of producing derived metrics in a fully automated biodiversity monitoring system (Besson et al. 2022). Large and properly labelled training datasets will be key to expanding automated monitoring using remote sensing, eDNA, and acoustic monitoring, for application across all ecosystems at multiple scales (Besson et al. 2022; van Klink et al. 2022).

Conclusions

An ecological monitoring approach integrating above- and below-ground ecological metrics is highly desirable in the context of NbS, as there are many interdependencies between above- and below-ground ecological processes (Farrell et al. 2020; Chomel et al. 2022). From a huge variety of possible metrics we have highlighted the most informative and feasible metrics to monitor, but emphasise that there are still practical limitations which limit monitoring of an optimum set of metrics. It is possible to harness and adapt existing standardised methodologies to develop an integrated approach for simultaneously collecting biodiversity and soil health metrics. However, there are still barriers to implementing workflows from data collection to metric calculation, including a lack of reference values or thresholds for many metrics, and in some cases a lack of standardised data collection methodologies. Emerging technologies have the potential to bypass some of the more technically demanding approaches to metric generation, however these are most likely to be used alongside traditional ecological monitoring methods. Ultimately, monitoring will always be project-dependent, and our framework provides high-level guidance on metric selection and data collection design that can be refined to meet project-specific goals.

Authors' contributions

EW conceived the idea, which was developed and refined with LMG, GAC, PS, ACS & NS. EW, LMG & GAC conducted the literature review and developed the metric prioritisation methodology. PS, ACS & NS provided feedback on the metric selection process and resulting set of metrics. DS designed and developed the online version of the monitoring framework. EW led the writing with input from all authors.

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Conflict of Interest

The authors declare no conflicts of interest.

Data availability statement

The monitoring framework is available at:

https://nbshub.naturebasedsolutionsinitiative.org/monitoring-tool/.

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An integrated approach to above- and below-ground ecological monitoring for nature-based solutions – Supplementary Methods

Detailed information on how the metrics were chosen and assessed

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Literature review

As a starting point for selecting biodiversity metrics we used Noss' Biodiversity Hierarchy (Noss 1990). Noss' Hierarchy organises biodiversity into three sub-categories: composition, structure, and function, aiming to capture the breadth and complexity of biodiversity (Figure 1). Composition metrics reflect the identity and variety of elements that comprise biodiversity and describe the species present and the communities that they form. Structure metrics represent the physical organisation of the system and describe the physical patterns that support species. Function metrics reflect the ecological and evolutionary processes that underpin functioning within an ecosystem, indicating the processes that result from interactions between species. These metrics apply at different scales (landscape, community, population, genetic). Landscape scale metrics apply at landscape scales (100m – km), encompassing a mosaic of habitat types, landforms, and land uses. Community scale metrics apply at ecosystem scales (cm – 100m), focussing on interacting organisms within a relatively homogenous area of habitat. Population scale metrics apply at the species level (mm – m), assessing population trends. Genetic metrics assess genetic variation and processes within species (nm – mm).

For each category of biodiversity at each scale, Noss defines metrics or groups of metrics. These were supplemented by literature searches on "biodiversity monitoring", "biodiversity metrics", "ecological indicators", "ecological integrity", and "ecological health", using Google Scholar and Web of Science. Results of the literature searches were ordered by relevance and suitable papers (reviews of biodiversity metrics or monitoring, or papers providing evidence on the relationship between a specific metric and biodiversity) were identified. A minimum of the first 50 papers identified during the search were assessed, as around this point the relevance of the papers would decline substantially. In some cases, additional papers were also identified by 'snowballing', i.e. using references within the papers generated by the primary literature searches. Grey literature relevant to biodiversity monitoring in the UK was also assessed, including: The Scottish Environment Strategy Initial Monitoring Framework (Scottish Government 2021), Field Studies Council – Monitoring and Indicators of UK Biodiversity Change (Burkmar 2017), Integrated ecological monitoring in Wales: the Glastir Monitoring and

Evaluation field survey (Wood et al. 2021), UK Countryside Survey – Technical Reports (UK Centre for Ecology & Hydrology 2010), Mapping the Species Data Pathway – connecting species data flows in England (UK Government 2021), UK Habitats Classification (UKHab Ltd 2023), Common Standards Monitoring – guidance documents for habitats and species (JNCC 2004), and Evaluating the Impact of Nature-based Solutions (Dumitru & Wendling 2021). This yielded a total of 71 indicators, which were further evaluated to select the minimum set of metrics as described in the section on Scoring the metrics (Table 1 in article text).

Soil health can be classified using a combination of physical, chemical, and biological indicators (Jian et al. 2020). Physical indicators reflect the physical structure of the soil, particularly related to solid particles and pores, they influence processes such as root growth, seedling emergence, infiltration, or movement of water within the soil profile. Chemical indicators reflect soil chemical composition and influence how soil properties change and react over time. Biological indicators reflect the living organisms within or connected to the soil, they are usually more dynamic than chemical and physical indicators, and likely to change more with time and management. These can be categorised by scale (landscape, community, population genetic) as described above for the biodiversity metrics.

Potential metrics of soil health were identified through literature searches of academic and grey literature on “soil health monitoring”, “soil health indicators”, “soil quality”, “soil ecosystem services”, “soil biological indicators”, “soil biodiversity”, “soil physical indicators”, “soil chemical indicators” and “soil health assessment”. A total of 118 indicators were obtained from the literature, these were then ranked by frequency and the percentage of papers and monitoring frameworks that considered each metric in their assessments was calculated. The indicators that were mentioned in at least 20% of the literature were shortlisted. Indicators that were considered as part of the minimum set of indicators required for a monitoring framework in grey literature, even if not mentioned in at least 20% of the academic literature, were also considered and combined with the shortlist. This yielded a total of 57 indicators, which were further evaluated to select the minimum set of metrics as described in the section on Scoring the metrics (Table 2).

The most relevant and useful metrics were identified by reviewing literature to assess evidence on the relationship between each metric and biodiversity or soil health. Information on pros, cons, and feasibility was also evaluated during the literature review process. We collated information on technological innovations that could simplify or accelerate the data collection process in the future.

Methodologies for data collection for both soil and biodiversity indicators were identified during the literature, grey literature, and web searches. We prioritised identification of existing standardised methodologies and monitoring schemes. This will allow integration with existing datasets, comparison between sites, and comparison to UK-wide trends. Taxonomic survey methods were identified using The UK National Biodiversity Network database of Wildlife Survey and Recording Schemes (https://nbn.org.uk/tools-and-resources/useful-websites/database-of-wildlife-surveys-and-recording-schemes/) and UK Environmental Change Network (https://ecn.ac.uk/measurements), which were mostly focussed on surveying specific taxonomic groups. Habitat-focussed survey schemes e.g. the UK Habitat Classification, UK Countryside Survey and the National Forest Inventory were also assessed. Soil health focused survey schemes that included standardised methodologies were identified, e.g. the UK Countryside Survey, AHDB soil health score card, FAO Soil Doctor Programme and Farm Carbon Toolkit.

Scoring the metrics

To help prioritise metric selection within an NbS project, we grouped the metrics into three Tiers. Tier 1 indicators cover a core set of the most informative metrics, and we aimed for this Tier to include broad coverage of the different axes of biodiversity and soil health. Tier 2 metrics build on the Tier 1 metrics and in some cases are applicable to some ecosystem types only. Future metrics are indicators that are highly informative but generally less feasible to collect, and/or require further testing and development.

To help to group the metrics into Tier 1, Tier 2 and Future, we developed a scoring system assessing informativeness and feasibility of data collection for each metric (Table 1 in article). The derived scores served as a guide for metric selection, and assessment of the suitability of each metric was also based on the literature review described above. Each metric was assigned 1, 2, or 3 points for each criterion; more points indicate better metric performance for that criterion. For example, 3 points for Relevance means there is strong evidence that the metric is relevant to biodiversity or 3 points for Cost means the metric is less expensive to monitor. The maximum possible score is 36 points.

For the biodiversity metrics, Tier 1 required an informativeness score ≥15, a feasibility score ≥12, and met the additional criteria of being applicable across all ecosystem types. After scoring, no metrics representing the function aspect of biodiversity were identified. Tier 2 metrics had an informativeness score 12-14 and a feasibility score ≥12; some Tier 2 metrics are applicable in some ecosystem types only. Future metrics scored <12 for feasibility and would enter Tier 1 in the future if informativeness scored ≥15 and Tier 2 if informativeness scored 12-14.

The biodiversity metrics Functional Trait Diversity and Identity were elevated from Future Metrics to Tier 1, despite a feasibility score of 11, to represent the Function aspect of

biodiversity within Tier 1. These two metrics lack a clearly defined framework for classifying species by functional traits, however we offer guidance for applying functional traits in different contexts, which can be tailored to meet project needs. Vegetation Structure is also an informative biodiversity metric but currently lacks a standardised and straightforward data collection approach in some ecosystem types, leading to a feasibility score of 11. However, we encourage practitioners to work with guidance provided to collect data on vegetation structure in their projects, as it provides structural information at a different scale to the other Tier 1 structure indicators, and have therefore included it in Tier 1.

For soil health, Tier 1 metrics scored ≥30 and Tier 2 metrics scored 25-30. All indicators scoring <30 points underwent further scrutiny, identifying two additional indicators that were included in Tier 1: Soil Structure and Electrical Conductivity. Soil Structure was included as it can add additional context to the other Tier 1 indicators and is relatively straightforward to monitor. Electrical Conductivity scored highly for informativeness, and although its feasibility score is lower, it can be analysed at the same time as pH to increase feasibility. Metrics that scored 25-30 were further analysed and if their lower score was due to lower feasibility, they were not included in Tier 2 and were categorised as Future metrics despite being informative. The Future metrics were all biological; their inclusion in this category may reflect that their relationship to soil health is not yet fully understood and requires further research, and they were also relatively expensive and require more expertise to monitor. We expect the soil health indicator rankings, particularly the Future metrics, to change over time in response to increasing availability of information on the relationship between the metrics and soil health.

Table 1 – The full set of biodiversity metrics derived from the literature review and assessed during the metric scoring process. A total of 71 indicators were identified.

Allelic diversity	Mammal biomass
Basal area	Metapopulation dynamics
Browsing	Multi-metric structure approaches
Canopy cover	Mutation rate
Canopy height	Outbreeding rate
Colonisation/local extinction rates	Patch dynamics
Connectivity	Patch persistence/turnover
Deadwood	Patchiness - patch size distribution
Debarking	Phenology
Deleterious recessives	Physiology
Density	Pollination
Dispersion (microdistribution)	Population fluctuations
Distribution	Population structure (age/sex ratio)
Disturbance processes	Predation
Disturbance rate	Presence rare alleles
Dominance-diversity curves	Proportions endemic/exotic/threatened
Dung density	Range (macrodistribution)
Effective population size	Rate of genetic drift
Energy flow rates	Recruitment
Erosion/geomorphic/hydrologic processes	Relative abundance
Fragmentation	Seedling regeneration
Functional trait diversity	Selection intensity
Gene flow	Similarity
Habitat evenness	Species diversity
Habitat proportions	Species richness
Herbivory	Structural elements (snags, logs etc)
Heterozygosity	Substrate/soil variables
Human intrusion rates/intensities	Topographical feature (cliffs etc)
Human land-use trends	Topography (slope/aspect)
Identity	Tree age
Inbreeding depression	Tree diversity
Invert biomass	Vegetation biomass
Karyotypic variants	Vegetation density
Landscape-diversity	Vegetation growth stage distribution
Life history	Vertical stratification
Life-form proportions	

Table 2 – The full set of soil health metrics derived from the literature review and assessed during the metric scoring process. A total of 57 indicators were identified.

Acari/mites (mesofauna)	Microbial biomass carbon
Available nitrogen	Microbial diversity
Available potassium	Microbial functional gene composition
Bacteria	Microbial N cycling processes (nitrification, denitrification)
Bacterial relative abundance	Mycorrhiza
Bulk density	N fixation/fixing bacteria
C mineralization	N mineralization
C:N ratio	Nematode communities
Cation exchange capacity (CEC)	Pathogens/parasites
Collembola (mesofauna)	Penetration resistance
Decomposition of organic matter	pH
Earthworm abundance, biomass and diversity (mesofauna)	Plant available phosphorus (P_2O_5)
Electrical conductivity	Porosity
Enchytraeids (mesofauna)	Root traits
Exchangeable magnesium, calcium, sodium, and potassium (Mg2+, Ca2+, Na+, K+)	Rooting zone
Foliar analysis (soil fertility)	Salinity, sodicity
Functional diversity	Soil cover
Fungi	Soil enzyme activities
Genetic diversity	Soil erosion
Heavy metals	Soil organic carbon
Infiltration	Soil respiration
Labile carbon fractions	Soil structure
Litter decomposition	Stream chemistry (soil loss/erosion)
Macroarthropods/macroinvertebrates (macrofauna)	Structural stability
Macronutrients	Texture (silt, clay and sand)
Metabolic quotient	Total nitrogen (TN)
Microarthropods (mites, nematodes)	Total phosphorus (TP)
Microbial activity	Water storage
Microbial biomass	

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An integrated approach to above- and below-ground ecological monitoring for nature-based solutions – Supplementary Tables

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Table S1 The selected biodiversity metrics grouped by Tier (Tier 1, Tier 2, Future). The axis of biodiversity (C = composition, S = structure, F = function) and scale (G = genetic, P = population, C = community, L = landscape) represented by each metric is shown by an X in the relevant columns. The metric summary provides information on the relationship between the metric and biodiversity.

Metric	Metric summary	Axis			Scale			
		C	S	F	G	P	C	L
Tier 1								
Dominance-diversity curves	Dominance-diversity curves summarise the abundance and evenness of species within a community, visually representing simple diversity metrics and showing patterns of competition and niche differentiation (Wilson 1991; Whittaker 1965). Changes in dominant and rare species can have important consequences for ecosystem functioning and are not captured by metrics such as species richness (Hillebrand et al. 2018). The x-axis of the curve displays the abundance rank, with the most abundant species ranked 1, the second most abundant 2 etc. The y-axis displays the relative abundance of the species. Species richness is summarised as the number of species ranked on the x-axis. Species evenness is summarised in the slope of the curve, the steeper the gradient the lower evenness.	C					C	
Functional trait diversity	Different species have different functional contributions to an ecosystem (Chiarucci et al. 2011; Botta-Dukát 2005). However, metrics such as taxonomic alpha and beta diversity often don't detect changes in the underlying functional roles of species in a community (Lelli et al. 2019).	C		F			C	L

	Functional diversity, rather than species numbers, strongly determines ecosystem functioning (Díaz & Cabido 2001; McGill et al. 2006; Botta-Dukát 2005; Flynn et al. 2011; Reiss et al. 2009). Turnover of species identity will have the greatest functional consequences for an ecosystem (Hillebrand et al. 2018; Buckland et al. 2005). Changes in functional diversity are assessed by linking functional traits to species data. Selection of functional traits for analysis will depend on the project and outcomes of interest. Assessing functional traits linked to ecosystem services is a useful approach (Waldén et al. 2023).							
Habitat area	The total area of habitat is simple to measure and is an important driver of overall biodiversity (Riva & Fahrig 2022; Deane et al. 2020). Maintenance of total habitat area is a key conservation target and habitat loss is a key driver of biodiversity declines (Fahrig et al. 2022; Hanski 2011). Habitat area can be tracked alongside other metrics such as **Landscape diversity**, **Patch size distribution**, **Connectivity and fragmentation**, to reflect the need for a range of habitats in a connected network. For species with specific habitat requirements, monitoring habitat availability can be a good proxy for species monitoring and has the advantage of providing information relevant to multiple species simultaneously (Bunce et al. 2013).		S					L
Identity	Identity considers the role of a species in an ecosystem. Species with particular traits can be monitored to provide functional information beyond richness and diversity metrics (Loreau et al. 2001). Species identity is monitored using functional traits (Cardinale et al. 2012). Identity can be monitored as the presence, abundance or diversity of a set of functional traits (such as morphological, ecophysiological or life-history characteristics) (Vandewalle et al. 2010). Selection of functional traits for analysis will depend on the project and outcomes of interest. Assessing functional traits linked to ecosystem services is a useful approach (Waldén et al. 2023).	C		F			C	L
Landscape diversity	Habitat heterogeneity at the landscape scale (compositional and configurational heterogeneity) has a positive relationship with many taxa (Maskell et al. 2019; Honnay et al. 2003). Spatial	C	S					L

	variation in habitats can maintain distinct communities by increasing beta-diversity within landscapes (Deane et al. 2020; Veech & Crist 2007) (see Species diversity for explanation of beta diversity). Different species groups respond to landscape diversity metrics at different scales. For example, landscape diversity at the c.250 m scale best explains diversity pattern in butterflies and birds (Krauss et al. 2003; Morelli et al. 2013). Different arthropod groups respond to landscape diversity metrics at different scales, with many groups showing dispersal 100s of metres from their focal habitat (Dangerfield et al. 2003). However, some beetle groups respond to habitat heterogeneity at 25 m scales (Dangerfield et al. 2003). Within habitat heterogeneity is captured in the metric Vegetation structure.							
Relative abundance	Changes in abundance, and identity, of species have greater functional consequences for an ecosystem than changes in simple metrics such as species richness (Hillebrand et al. 2018; Buckland et al. 2005). Trends in mean abundance can detect early signals of species decline and are less sensitive to demographic stochasticity (population fluctuations that occur by random chance as a result of births, deaths and migration) (Santini et al. 2017; Van Strien et al. 2012). It is easier to estimate abundance data at smaller scales (community, population) than at landscape scales (Chiarucci et al. 2011). Abundance data should be collected in a spatially explicit way (e.g. fixed area plots, defined density of sampling points per unit area) (Chase & Knight 2013).	C				P	C	L
Similarity	Similarity is a measure of species composition compared to a reference community. It is important to measure composition as well as diversity, because diversity metrics based on species richness and abundance can remain constant as species assemblage changes (Santini et al. 2017; Magurran et al. 2018; Magurran 2021). Similarity can be used to capture temporal turnover and, in projects with multiple habitats, spatial turnover (beta-diversity) within a project site (Santini et al. 2017; Schindler et al. 2008).	C					C	

	Spatial turnover can be linked to topographical and habitat metrics collected simultaneously. With small sample sizes there is a risk that some similarity metrics (e.g. based on Bray-Curtis similarity) may underestimate similarity (Hardersen & La Porta 2023).							
Species diversity (alpha, beta, gamma)	Species diversity captures the richness and relative abundance of species (Magurran 2021). Species diversity metrics can be applied at different scales. Alpha diversity is the species diversity within a focal site and applies at smaller (community) scales (Magurran 2021; Chiarucci et al. 2011). Beta diversity captures differences in species composition among sites, and is usually applied at larger scales e.g. to capture differences in species composition among habitat types within an area (Chiarucci et al. 2011). Gamma diversity captures overall species diversity at landscape scales (Anderson 2018). The species groups monitored provide information about different aspects of an ecosystem. Using 2-3 species groups dramatically increases the overall coverage of collective species responses (Larrieu et al. 2018). A synthesis of many biodiversity studies found that birds and plants were the best collective surrogate for unsurveyed species (Westgate et al. 2017). The species group chosen will provide information on different aspects of the overall ecological community. Plants are the most commonly assessed group and generally have higher correlation with other species groups (Burrascano et al. 2018). Butterflies respond to habitat conditions and the presence of their food plants in the area, beetles to habitat conditions and small-scale habitat heterogeneity (Burrascano et al. 2018; Ferris & Humphrey 1999). Birds have been linked to vegetation structure in forest systems and habitat diversity at larger scales (Ferris & Humphrey 1999). Species diversity monitoring at larger scales can return higher species richness as more environmental variation and greater numbers of rare species are captured across larger areas (Magurran 2021). However, thorough monitoring at large scales can be challenging and there is a chance that rare species won't be detected.	C					C	L
Vegetation structure	Structural complexity within habitats is an important determinant of biodiversity. Vegetation structure can be assessed from the habitat level (e.g. vegetation height diversity) to within-plant		S				C	

	architecture (e.g. branch density) (Langellotto & Denno 2004). Relationships between vegetation structure and biodiversity vary in different habitats and with different taxonomic groups. Invertebrates respond to changes at the habitat and within-plant scale, whereas patterns of diversity for organisms such as birds respond to habitat scale measures (Bradbury et al. 2005). In grassland there is a positive relationship between vegetation structure and invertebrate diversity (Langellotto & Denno 2004). The relationship between nesting birds and vegetation structure in grassland is more complex, and different features have different effects (Winter et al. 2005). In forests vegetation structure is a key determinant of habitat quality for bird species and can explain patterns of arthropod abundance (Shokirov et al. 2023; Halaj et al. 2000; Storch et al. 2023).							
Tier 2								
Deadwood volume	Standing deadwood volume reflects a broad range of species groups, deadwood specialists and species of conservation concern (Zeller et al. 2022; Gao et al. 2015; Evans et al. 2019; Hekkala et al. 2023; Storch et al. 2023). Deadwood also indicates stand structural development, accumulating as the forest develops and becomes more structurally complex. Deadwood has functional impacts: contributing to carbon and nutrient cycles, providing a substrate for natural regeneration, and fuelling fires (Ferris & Humphrey 1999). The rate of the deadwood accumulation in response to management changes depends on forest age. In young forest, the deadwood component will slowly accumulate. However, in mature forest, deadwood volume can increase fairly quickly in response to proactive interventions e.g. after ring-barking or with a reduction in management intensity (Kirby et al. 1998; Agnew & Rao 2014).		S				C	

	Deadwood requires little expertise to recognise and plot-based measurement of deadwood volume is fairly cost and time efficient to conduct (Cosović et al. 2020).							
Pollination	Biotic interactions structure ecosystems and underpin many ecosystem functions (Kaiser-Bunbury & Blüthgen 2015) (Kaiser-Bunbury et al. 2015). Interactions related to specific ecosystem functions include pollination, seed dispersal and predation, of these pollination is one of the most tractable to measure (Kaiser-Bunbury & Blüthgen 2015). It is relevant to measure pollination where this is a key functional goal of a project (Kaiser-Bunbury & Blüthgen 2015), for example grassland restoration in an agricultural landscape. However, it is possible to assess pollination in other ecosystem types, and pollination is key to maintaining plant richness, particularly rare species, underpinning overall biodiversity (Wei et al. 2021; Kral-O'brien et al. 2021). In some systems a small number of species maintain pollination functionality but as more sites are assessed, more species are required to provide the threshold level of pollination (Kremen 2018). Different pollinator groups have different sensitivities and high land-use intensity reduces overall pollinator abundance (Millard et al. 2021).			F			C	
Seedling regeneration	Natural seedling regeneration and its diversity is a key indicator of forest regeneration, persistence and resilience (Chazdon et al. 2023). In UK forests browsing impacts from species such as deer can inhibit seedling regeneration and establishment (Gullett et al. 2023). Tree species richness in the regeneration layer is positively related to overall richness of other taxa (Storch et al. 2023). Seedling regeneration also indirectly indicates the presence of seed dispersers, diversity of soil seedbanks and diversity of mature trees in the landscape (Chazdon & Guariguata 2016).			F			C	
Tree age	Tree age provides an insight into the overall status of the forest ecosystem: seedling regeneration and establishment to maturity determines ongoing survival of the forest and senescent trees often support important specialist biodiversity (Gao et al. 2014; Rainey & Holmes 2023).		S				C	

	Numbers of large living trees are correlated with species richness of multiple taxa, from birds to fungi (Zeller et al. 2022). Highest species richness overall across 29 taxonomic groups was found in old forest stands in a study using National Forest Inventory plots in Germany (Storch et al. 2023).							
Tree diversity	There is extensive evidence that tree diversity is a key driver of biodiversity and ecosystem functioning (Baeten et al. 2019; van der Plas et al. 2016; Ampoorter et al. 2020). For example, forests with a higher tree species richness have a higher richness of associated lichens and tree species richness in the regeneration layer is positively related to overall richness of other taxa (Storch et al. 2023). However, other studies have shown that tree species richness is a less important predicter of biodiversity than structural metrics such as vertical structure, large living trees, presence of tree microhabitats and proportion of gaps (Zeller et al. 2022). This may in part reflect that some natural forest types have a low natural level of tree species richness.	C	S				C	
Vegetation biomass	The direction of change in vegetation biomass provides different insights in ecosystem status, depending on the habitat. In grassland, an increase in vegetation biomass is often linked to declines in species richness, for example nutrient addition leads to grass dominance and a decline in species richness in chalk grassland (Willems et al. 1993). In woodland biomass can be used to track biotic (e.g. ash dieback) and physical (e.g. wind-throw disturbance) (Evans et al. 2019). Experiments have found a positive relationship between species diversity and productivity, reflecting niche complementarity, resource acquisition and utilisation efficiency, and leading to higher aboveground biomass in more diverse systems (Liang et al. 2015; Tilman et al. 2014). However, in natural ecosystems vegetation biomass is not always positively associated with species diversity (van der Plas 2019). At landscape and community scales, vegetation biomass is a measure of productivity, providing	C	S	F		P	C	L

	an insight into ecosystem function. Biomass recorded by species can be used to estimate species dominance and relative abundance.							
Future								
Biomass (measure of abundance)	Changes in abundance have greater functional consequences for an ecosystem than changes in simple metrics such as species richness (Hillebrand et al. 2018; Buckland et al. 2005). Collecting abundance data as biomass provides additional functional information, structural information, and accounts for size differences between organisms (Llopis-Belenguer et al. 2018; O'Connor et al. 2017). Trends in mean abundance can detect early signals of species decline and are less sensitive to demographic stochasticity (Santini et al. 2017; Van Strien et al. 2012). It is easier to measure abundance accurately at smaller scales (community, population), than at landscape scales (Chiarucci et al. 2011). Abundance data should be collected in a spatially explicit way (e.g. fixed area plots, defined density of sampling points per unit area) (Chase & Knight 2013).	C	S	F		P	C	
Energy flow rates	Ecosystem energetics provides information across an entire ecosystem, as all organisms are interlinked by energy flows through consumption and assimilation of resources, respiration, and biomass production (Buzhdygan et al. 2020). Higher biodiversity leads to more energy stored, greater energy flow, and higher community-energy-use efficiency across trophic networks (Buzhdygan et al. 2020). Energy fluxes are a good proxy for ecosystem functioning and allow understanding across diverse guilds and taxa in a unified manner (Barnes et al. 2014; Malhi et al. 2022).			F			C	L
Invertebrate biomass	Invertebrate biomass is a measure of abundance and can detect impacts of external pressures on invertebrates that are not detected by species richness (Robertson & Wentworth 2020; Vereecken et al. 2021). Abundance of invertebrates predicts ecosystem functioning at large scales and has stronger links to ecosystem service delivery than species richness or diversity (Weiss & Linde 2022; Woodcock et al. 2019; Winfree et al. 2015). Biomass provides greater insights into changes in invertebrate diversity than individual count abundance (Llopis-	C		F		P	C	

	Belenguer et al. 2018). Understanding changes in functionally important invertebrate assemblages is important given their links to ecosystem service delivery (Lamarre et al. 2020). Invertebrate biomass is influenced by the time of year of sampling, therefore comparison between projects and habitats is only possible with standardised sampling (Montgomery et al. 2020). Data collection in the UK is often biased towards pollinators and popular species such as butterflies, with less emphasis on other functional groups, so measuring invertebrate biomass across a broader range of groups is desirable (Robertson & Wentworth 2020).							
Patch persistence/turnover	Spatial and temporal landscape heterogeneity determines the structure and functioning of nature, spatial distribution of organisms and long-term population persistence (Johst et al. 2011; Pickett & Rogers 1997; Gravel et al. 2010; Bascompte et al. 2002). Turnover is an important consideration alongside patch quality, area and connectivity (Heinrichs et al. 2015; Fleishman et al. 2002). Many species are adapted to spatiotemporal heterogeneity, however an increasing speed of change could compromise this (Johst et al. 2011). In landscapes with higher habitat turnover, a greater number and connectivity of patches is required to maintain species populations (Johst et al. 2011). Compensation for patch loss by connectivity and patch creation depends on species characteristics (e.g. dispersal ability) and landscape characteristics (e.g. connectivity and patch turnover) (Johst et al. 2011).			F				L
Allelic diversity	Alleles are variants of the same gene. Different alleles can be associated with different traits (e.g. eye colour). The frequency that specific alleles occur in a population at two points in time can be used to determine the rate of genetic drift (Wang et al. 2016). Allele diversity can be used to identify genetic variants of interest for population persistence/fitness and can be sensitive to change on short timescales (Leroy et al. 2018).	C			G			
Colonisation/local extinction rates	Turnover indices reflect colonisation and extinction, they capture biodiversity changes that are not captured by species richness (Hillebrand et al. 2018). Turnover integrates information on species identity and abundance (Hillebrand et al. 2018). Spatial changes in habitat structure/composition and changes in external pressures can drive changes in colonisation and			F			C	

	extinction (Sirami et al. 2008). Colonisation and extinction impact the homogenisation and heterogeneity of communities, capturing the processes underlying changes in beta-diversity (Tatsumi et al. 2021). For rare species, lower probability of detection can make it harder to accurately determine whether they are occupying a site (Lindenmayer et al. 2020).							
Connectivity and fragmentation	Connectivity and Fragmentation are complementary metrics assessing the distribution of habitat within a landscape. There has been extensive debate on the role of spatial habitat configuration vs habitat loss on overall biodiversity (Fahrig et al. 2019; Fletcher et al. 2018). Habitat quality, loss, patch area, and connectivity have complex and interrelated effects on biodiversity (Wilson et al. 2016; Hanski 2011). Fragmentation and connectivity determine species movements and therefore influence the total availability of habitat to a species (Hanski 2011). Isolation of populations by fragmentation contributes to inbreeding and the accumulation of negative mutations in populations, reducing future population viability (Hanski 2011). Under a future changing climate, species' abilities to track their climatic envelopes will depend on habitat connectivity/fragmentation (Rudnick et al. 2012). Connectivity and fragmentation can be described by structural metrics of spatial habitat configuration, however the most informative metrics integrate biological data on the focal species (Kindlmann & Burel 2008; Calabrese & Fagan 2004).		S					L
Disturbance	Disturbance ranges from small to large events and shapes biodiversity at multiple levels of organisations, with consequences for ecosystem functioning (Dornelas 2010). Ecological theory suggests that biodiversity will be maximised at intermediate levels of disturbance, which links to conservation management intervention e.g. grazing, predation (Dornelas 2010). However, higher intensity human-driven disturbance can have a negative impact on ecological communities (Dornelas 2010).			F				L

	Disturbances cause temporary and localised shifts in demographic rates, influencing the numerical abundance of populations and relative abundance of species within communities (Dornelas 2010; Sousa 1984). Disturbance also affects ecosystem-level processes e.g. primary production, biomass accumulation, energetics, nutrient cycling (Sousa 1984). Physical processes (e.g. fires, storms, floods, winds, landslides, drought) and biological processes (e.g. predation, grazing, digging) cause disturbance (Sousa 1984). These disturbance processes can be monitored at medium to large scales.							
Effective population size	Effective population size is a key parameter that influences wildlife conservation and management decisions (Luikart et al. 2010). Effective population size is one of the most effective measures of genetic erosion and provides information on the rate of inbreeding and loss of genetic variation (Leroy et al. 2018; Hoban et al. 2020; Frankham 1995). Effective population size influences other important drivers of genetic diversity such as efficacy of mutation, selection and migration (Wang et al. 2016).		S		G			
Heterozygosity	Heterozygosity is a measure of genetic variation within a population and refers to individuals that carry two different alleles at a given locus (Leroy et al. 2018). Loss of heterozygosity is a driver of genetic erosion and is linked to genetic drift and inbreeding (Leroy et al. 2018; Gaggiotti et al. 2018). There is a direct relationship between heterozygosity and **Effective population size** (Wang et al. 2016). Reduced heterozygosity (i.e. genetic variation) can lead to reduced fitness and loss of adaptive potential (Leroy et al. 2018).		S		G			
Mammal biomass	Mammals have important influences on ecosystem processes: nutrient cycling, energy flow, top-down influences (e.g. predation), bottom-up influences (e.g. herbivory), seed dispersal (Lacher et al. 2019). Biomass provides an indication of abundance and whether species are declining or increasing (Damuth 2023). Biomass can give a more meaningful assessment of the impact of a species on an ecosystem, allowing comparison between species with different body sizes (Greenspoon et al. 2023). Species-specific monitoring may be relevant to certain project objectives. Small mammals are	C	S	F		P		

	an important prey resource and are seed predators and dispersers in woodlands (Sunyer et al. 2016). Large herbivores, particularly deer, can have a negative impact on woodland regeneration and can degrade peatland, and therefore numbers are often monitored (Putman et al. 2011).							
Nutrient cycling rates	Spatial variation in nutrient supply rates can influence the distribution of organisms (Gravel et al. 2010). Cycles of key nutrients underpin persistence of biodiversity, ecosystem functions and delivery of ecosystem services (Lavelle et al. 2005). Simplification of landscapes and habitat loss has disrupted natural nutrient cycles and agriculture can lead to nutrient oversupply and leakiness or nutrient depletion (Lavelle et al. 2005). Diversity of species and functional groups are required to maintain nutrient cycles (Chapin III et al. 2000; Lavelle et al. 2005). Alterations in nutrient cycling are observed from plot to landscape scales and are driven by small- to large-scale mechanisms (Lavelle et al. 2005).			F		P	C	L
Patch size distribution	Alongside total area of habitat, habitat configuration influences biodiversity at the landscape scale (Fahrig et al. 2022; Deane et al. 2020; Fletcher et al. 2018). Evidence is still equivocal on whether a Single Large or Several Small (SLOSS) patches support higher biodiversity (Deane et al. 2020). Multiple small patches at the landscape scale can increase beta diversity, increasing overall landscape-scale biodiversity (Deane et al. 2020). However, other evidence points to the importance of large habitat areas for maintaining biodiversity (Fletcher et al. 2018). The distribution of habitat area across multiple habitat types within a landscape is an important driver of biodiversity change (Proença & Pereira 2013).		S					L
Population fluctuations	Biodiversity stabilises ecosystems because different species respond to environmental perturbations in different ways, which stabilises overall ecosystem function (de Mazancourt et al. 2013; Tilman et al. 1996). Aggregation of population trends from multiple species can give rise to misleading overall trends due to random population fluctuations (Buschke 2021). The overall population trend from aggregate indices can mask a combination of increasing and declining populations (Buschke 2021). It is challenging to use a simple population metric to determine whether a species is at risk of			F		P		

	extinction, as this depends on the factors influencing population variability (Melbourne & Hastings 2008). In small populations random population fluctuations are an important driver of extinction risk, whereas environmental variability affects a greater range of population sizes (Melbourne & Hastings 2008). Population extinction risk at a given site is strongly influenced by spatial considerations such as connectivity and migration of source populations (Engen et al. 2002). Population size is a more important determinant of extinction risk than population trends, although trends can be useful in identifying endangered species (O'Grady et al. 2004).							
Proportions endemic, exotic, threatened species	The loss of species of conservation interest or endemics should be assigned higher value than the loss of common species (Buckland et al. 2005). Metrics such as species richness, evenness and diversity can be insensitive to the loss of rare species; for example, at smaller scales conversion from one habitat to another could cause loss of endemic species, but overall richness and evenness could still increase (Buckland et al. 2005). The rarity of species of conservation concern can provide additional challenges in effective monitoring, and monitoring objectives should be tailored to suit the attributes of the target species (Robinson et al. 2018; Lindenmayer et al. 2020). Accurate detection of population changes will require more intensive monitoring than conventional species surveys (Robinson et al. 2018; Martin et al. 2007). Nature-based Solutions projects are unlikely to support the entire population of a species, and monitoring will usually only capture presence, or with more intensive monitoring, population trends for a species within the project area. Monitoring of exotic species can be important where they are invasive and have a disproportionate negative effect on ecosystem functioning or native species populations (Bradshaw et al. 2016; Ehrenfeld 2010). Additionally, invasive species can increase species richness and functional/phylogenetic diversity, while having negative effects on native species (Santini et al. 2017).	C				P	C	L

Table S2 – The selected soil health metrics grouped by Tier (Tier 1, Tier 2, Future). The axis of soil health (P = physical, C = chemical, B = biological) and scale (G = genetic, P = population, C = community, L = landscape) represented by each metric is shown by an X in the relevant columns. The metric summary provides information on the relationship between the metric and soil health.

Metric	Metric summary	Axis			Scale			
		P	C	B	G	P	C	L
Tier 1								
Bulk Density	Bulk density constitutes a direct measurement of soil compaction (or loosening), serving as an essential tool for evaluating the total porosity. An adequate volume of pore space within the soil is essential for the sustainable use of soil resources, benefiting both productivity and environmental health (Merrington et al. 2006). Bulk density can capture the impacts of soil use and management on the dynamics of water/air relationships within the soil (Cardoso et al. 2013). Bulk density is the mass per unit volume, usually expressed as g/cm3 (Sparling et al. 2008; Merrington et al. 2006). Dry weight bulk density is the generally accepted measure of bulk density (Merrington et al. 2006). In order to ensure consistency and to obtain reliable data, it is important that sampling protocols for bulk density consider the timing, depth and specific location of sampling (Merrington et al. 2006).	P					C	L
Soil Organic Carbon	Soil organic carbon plays a pivotal role in influencing crucial functional processes within the soil, encompassing nutrient storage, particularly for nitrogen, water retention capacity, and the stability of soil aggregates. Furthermore, it exerts a substantial impact on microbial activity, thus establishing itself as an integral constituent of soil fertility (Cardoso et al. 2013).		C				C	L

	Soil organic carbon has been directly associated with several significant ecosystem services, including carbon sequestration, climate regulation, biomass production, the filtrating, buffering, and transformation of substances, as well as the preservation of genetic diversity within ecosystems (Guerra et al. 2021). The percentage of organic matter is determined by loss on ignition, based on the change in mass after a soil is exposed to high temperature (500 °C or 932°F) in a furnace.							
Texture (Silt, Clay and Sand)	Soil texture is extremely important to measure as it is intricately associated with a plethora of soil attributes and behaviours. The proportions of sand, silt, and clay within a soil matrix have considerable influence over various other critical soil properties, encompassing aeration, compaction, drainage characteristics, water-holding capacity and the decomposition of organic matter, among others (Merrington et al. 2006; Schoenholtz et al. 2000). Soil texture can also be a measure of the ecosystem structure or habitat extension (Guerra et al. 2021). It is worth noting that soil texture is not particularly responsive to alterations resulting from management practices (Cardoso et al. 2013). Consequently, it is advisable to incorporate soil texture assessment as an initial step in monitoring regimes (Stewart et al. 2018; Bongiorno 2020).	P						L
Soil Moisture	Soil moisture represents a key driver of vegetation characteristics, operating at both species and community levels. It has a substantial influence on crop yields in agriculture. The quantification of soil moisture content gives insights into critical aspects of soil dynamics, encompassing its water storage capacity, hydraulic properties, and chemical and biological activities (Le Roux et al. 2013; FAO 2020). Water availability within the soil is a pivotal determinant of microbial activity, and there is a well-established relationship between water availability and microbial activity. Constraints imposed by limited water resources can significantly impact soil microbial functionality and synthesis and mineralization of soil organic matter. This can have repercussions for biogeochemical cycles. It is	P					C	L

	important to note that distinct microbial groups within the soil exhibit different sensitivities to water restrictions, with bacteria exhibiting higher water requirements while fungi, benefiting from lower water thresholds, can readily explore air-filled pores (Cardoso et al. 2013).							
Porosity	Soil porosity pertains to the interstitial voids existing between soil particles and aggregates, serving as conduits through which water and air can move. These voids create essential habitats for the proliferation of roots and microorganisms (Nimmo 2013; Osman 2013). Porosity exerts significant influence over a multitude of critical soil processes, which depend on pore dimensions, shapes, and continuity (Pagliai et al. 2004). Management practices can alter the soil's pore structure, leading to modifications in its physical attributes. These transformations, in turn, can have significant ramifications for long-term sustainability and soil functionality (Bodner et al. 2013; de Andrade Bonetti et al. 2017; de Oliveira et al. 2021). Total porosity has indirect links with several critical ecosystem functions, including but not limited to, biomass production, filtration, buffering, transformation of substances within the soil matrix, safeguarding the genetic diversity within the soil's gene pool, maintenance of soil structural integrity and the regulation of soil hydrological processes (Guerra et al. 2021). The number, activity and biodiversity of micro-organisms and earthworms are also greatest in well aerated soils and they are able to decompose and cycle organic matter and nutrients more efficiently (FAO 2020).	P					C	L
Soil Structure	Soil structure refers to the arrangement of soil particles into distinct geometric patterns (Dexter, 1988). Soil structure, in conjunction with soil texture, collectively governs parameters such as total porosity, pore size and distribution, thereby influencing the retention and movement of water within the soil matrix. The complexity of soil structure arises from a confluence of diverse factors, including environmental conditions, soil management practices, mineral composition, textural attributes, the presence of soil organic matter, pedogenic (soil-forming) processes, microbial activity, and the prevailing moisture regime (Bronick & Lal 2005; Osman 2013; Pagliai et al. 2004).	P					C	L

	Soil structure has the capacity to serve as an indicator of ecosystem structure or habitat extent (Guerra et al. 2021).							
pH	Soil pH indicates the soil's acidity or alkalinity. Its inherent value is influenced by the chemical composition of the soil, but it can change due to both natural factors and agricultural activities. The soil's pH influences the accessibility of nutrients in the soil by plants, affects microbial activity, influences flora and fauna species diversity, and has been associated with leached soils (Cardoso et al. 2013; Merrington et al. 2006; Creamer et al. 2016; Gardi et al. 2009). Soil pH has direct associations with several critical ecological aspects, including but not limited to biomass production (including for food and fibre), filtering, buffering, transformation of substances within the soil matrix, and safeguarding genetic diversity within ecosystems. Furthermore, it exhibits indirect connections, influencing human health via impacts on the uptake of heavy metals into plants and the availability of micronutrients in food plants, impacting water quality, and playing a role in climate control (Guerra et al. 2021).		C				C	L
Nutrient Analysis (P,K,N)	Understanding the nitrogen supply stored in the SOM is fundamental to understanding how well the soil can support the microbial populations crucial for sustaining vital soil functions, including nutrient cycling, structural stability, water infiltration and storage, as well as the breakdown of organic residues, among other critical processes (Merrington et al. 2006; Creamer et al. 2016). Phosphorus, as an essential macronutrient, is required by all living organisms. Phosphorus is a crucial component of ATP (adenosine triphosphate), which is the primary energy carrier in living cells. This energy transfer is essential for various biochemical processes in plants, microbes, and other organisms in the soil (Merrington et al. 2006). Healthy phosphorus levels enhance the soil's ability to sustain diverse plant and microbial communities. On the other hand, unhealthy phosphorus levels may identify an environmental hazard (Allen et al. 2011).		C				C	L

	Potassium plays a significant role in influencing plant growth, species composition, and overall ecosystem function (Sardans & Peñuelas 2021).							
Electrical Conductivity	Measuring electrical conductivity (EC) in soils is crucial for evaluating soil health, providing valuable insights into the dynamics of water and soil quality, nutrient cycling, and the overall well-being of ecosystems (Allen et al. 2011). EC reflects the concentration of ions, including dissolved salts, in the soil solution. Elevated salt levels can have detrimental effects on plant growth, soil-water balance, and subsequently impact biological activity and nutrient cycling (Arias et al. 2005; Allen et al. 2011). Such salinity issues may arise naturally or result from inappropriate soil use and management practices (Arias et al. 2005). The assessment of EC is not only indicative of potential challenges but also serves as an early warning system for changes in the composition and functionality of soil microbial communities (Yang et al. 2020).		C				C	L
Earthworm abundance, biomass and diversity	Earthworms are sensitive to various management practices. Their presence not only signals a thriving ecosystem but also plays a crucial role in shaping it (Bispo et al. 2009; Griffiths et al. 2018; Pulleman et al. 2012). The increase in earthworm populations corresponds to a rise in biopores, enhancing macroporosity and concurrently reducing denitrification while promoting an increase in soil organic carbon (SOC) (Bispo et al. 2009; Griffiths et al. 2018; Pulleman et al. 2012). Earthworm abundances have also been positively correlated with water infiltration rates helping to counteract the effects of intense rain events on soil and plants (Griffiths et al. 2018). Earthworms are also sensitive to pollutants, making them indicators of environmental health (Pulleman et al. 2012). It is noteworthy that earthworms are not ubiquitous; coniferous forests, with their acidic soils, pose a			B		P	C	

	less-than-ideal environment for earthworms. The same is true in waterlogged soils. In such cases, it might be prudent to consider an alternative biological indicator like enchytraeids (potworms), a taxonomically related yet typically smaller group of worms (Bispo et al. 2009).							
Litter Decomposition (Litter bags)	Measuring litter decomposition allows for the comparison of decomposition rates in an ecosystem across different time periods (Arias et al. 2005). Litter decomposition is a key process in nutrient cycling, it can be linked to carbon cycling and microbial activity (Guerra et al. 2021; FAO 2020). The decomposed organic matter releases nutrients to the soil (Wagg et al. 2014). It is primarily driven by microbial activity and its measurement can provide insights into the health of the microbial community (FAO 2020). Changes in decomposition rates may signal disturbances in the soil ecosystem, such as pollution, compaction, or changes in land use (Wagg et al. 2014).			B			C	
Tier 2								
Aggregate Stability	Aggregates, formed through interactions among soil biota, plant communities, and soil mineral components, consist of multiple soil particles bound together. These structures play a pivotal role in various aspects of soil health, influencing water movement, storage, soil aeration, physical protection of soil organic matter (SOM), erosion prevention, root development, and microbial community activity (Arias et al. 2005). The vulnerability of soil aggregates to external forces is quantified through aggregate stability (Arias et al. 2005). Aggregate stability involves microorganisms producing glues, such as soil carbohydrates, which, along with fungal hyphae and fine roots, contribute to binding the aggregates. Soil aggregation integrates biological, chemical, and physical soil properties, and serves as a vital indicator of soil health (Stott 2019). The breakup of soil aggregates can result in oxidation, leading to a subsequent reduction in soil organic carbon (SOC). This also brings about a decrease in mean pore size, reducing plant-available	P					C	L

	water, plant growth, and microbial biomass and activity, through an increase in bulk density and a decrease in microporosity (Merrington et al. 2006).							
Infiltration	Soil water infiltration refers to the rate at which water enters the soil surface and moves through soil depth (Allen et al. 2011). Infiltration has a direct relevance to water retention, a key ecosystem service (Griffiths et al. 2016). Infiltration capacity greatly influences the soil's ability to store and provide water for plants. High infiltration rates play a crucial role in effective water storage, thereby reducing the risk of water runoff and minimizing the loss of topsoil and nutrients (Cleophas et al. 2022). Infiltration acts as a preventive measure against soil erosion, influences nutrient transport, and enhances the soil's resilience to extreme weather events, consequently lowering the risk of flooding (Haghnazari et al. 2015). The rate of infiltration is dynamic and can undergo substantial changes based on soil use, management practices, and over time. Recognizing its sensitivity to various factors, infiltration has been used as an indicator of soil health, especially in assessments evaluating the impacts of changes in land use (Arias et al. 2005; Allen et al. 2011).	P					C	L
Cation exchange capacity (CEC)	Soil Cation Exchange Capacity (CEC) is a measure of the soil's capability to retain and exchange positively charged ions (cations) within its structure. CEC plays a pivotal role in influencing nutrient availability, soil fertility, and water quality (UK Soil Observatory 2024; Arias et al. 2005; Lehmann et al. 2020). The principal cations involved in this exchange include calcium (Ca2+), magnesium (Mg2+), potassium (K+), and ammonium (NH4+) (UK Soil Observatory 2024). Soils with a higher clay content typically exhibit higher CEC due to the characteristics of clay particles, enabling them to effectively retain and exchange cations. This property endows these soils with the ability to retain more nutrients, mitigating the risk of nutrient runoff, and consequently, playing a crucial role in maintaining water quality (Agriculture & Horticulture Development Board 2024).		C				C	L

Soil respiration	Soil respiration is a process wherein soil microorganisms break down organic matter, releasing carbon dioxide (CO_2) into the atmosphere (Allen et al. 2011). It has been recognized as an indicator of ecosystem function and soil health, closely linked to carbon and nutrient cycling, regulation of CO_2 emissions, and microbial activity (Bispo et al. 2009; Griffiths et al. 2016; Arias et al. 2005). Additionally, soil respiration measurements have proven useful as indicators of pesticide and heavy metal toxicity (Nielsen & Winding 2002). Functioning as an overall indicator of microbial activity in the soil, soil respiration provides insights into the rate at which organic matter, encompassing both plant litter and soil organic matter, is undergoing decomposition (Allen et al. 2011). For more information on soil respiration you can follow the link from the South Dakota Soil Health Coalition: https://www.sdsoilhealthcoalition.org/technical-resources/biological-properties/soil-respiration/			B			C	
N mineralization	Organic nitrogen mineralization involves the microbial conversion of organic nitrogen into mineral forms, primarily ammonium (NH_4^+) and nitrate (NO_3^-), by a diverse array of soil microorganisms. This process reflects the turnover of organic material in the soil and the availability of indigenous nitrogen pools to plants (Nielsen & Winding 2002). It is significant for ecosystem services such as water quality, plant production, and climate control (Lehmann et al. 2020). Monitoring nitrogen mineralization provides valuable insights into the soil's capacity to supply nitrogen to plants, playing a crucial role in assessing soil fertility and overall nutrient cycling (Bünemann et al. 2018).			B			C	
Nematodes	Nematodes play a crucial role within the soil fauna community, engaging in interactions with various organisms such as bacteria, fungi, and microarthropods. Their involvement aids in essential soil processes, including nutrient cycling, decomposition, maintenance of plant health, and soil structure (Neher 2001).			B		P	C	

	The sensitivity of nematode populations to changes in environmental conditions (Bongiorno 2020), such as soil compaction, pollution, and moisture levels, allows them to serve as early indicators of disturbances in the soil ecosystem, offering valuable insights into soil health (Neher 2001). Occupying diverse trophic levels, from herbivores to bacterivores and fungivores, nematodes mirror the microbial and nutrient dynamics within the soil. The composition of nematode communities serves as a revealing indicator of nutrient cycling, microbial activity, and the overall structure of the soil food web (Neher 2001). Ubiquitous in nature, nematodes can be found in every habitat (Neher 2001). For more information on nematodes follow the AHDB knowledge library link: https://ahdb.org.uk/knowledge-library/soil-microfauna-nematodes.							
Future								
Genetic, Functional and Structural diversity of Bacteria	Bacteria play integral roles in soil ecosystems, influencing various soil processes. The structure and diversity of microbial communities are pivotal indicators of ecosystem changes resulting from alterations in land use and management practices. Recognized as early signals of soil ecosystem quality, microbial community diversity is quantified through the assessment of species richness and the proportional contribution of each species to the overall organism count. A diverse bacterial gene pool augments the soil's capacity for crucial functions, including nutrient cycling, organic matter decomposition, and the maintenance of overall ecological balance (Nielsen & Winding 2002; Pulleman et al. 2012; Trivedi et al. 2016).			B	G	P	C	
Fungal Biomass	More than 50% of the soil microbial biomass is composed of fungi (Vázquez et al. 2016). Fungi play crucial roles in nutrient cycling, organic matter decomposition, breaking down complex substances into simpler compounds that are available to plants and maintaining soil structure by forming mycelial networks that help bind soil particles (Nielsen & Winding 2002). Monitoring fungal			B	G	P	C	

	biomass provides insights into the health and functioning of the soil ecosystem and it is considered a sensitive measurement (Nielsen & Winding 2002; Bünemann et al. 2018).							
Enzyme activities (Soil enzymes)	Enzymes are integral to various metabolic processes of the soil, like decomposition of organic materials, impacting carbon sequestration, nutrient availability, soil productivity, and the global carbon cycle. As early and sensitive indicators of soil health changes, they are responsive to shifts in soil use and management, pollution and climate, and they represent the metabolic status of the soil microbial community (Cardoso et al. 2013; Stott 2019). They therefore serve as indicators for microbial activity, soil productivity, and the impact of pollutants. Changes in soil microbial activity, reflected in metabolic enzyme levels, can estimate ecosystem disturbance (Nielsen & Winding 2002; Cardoso et al. 2013; Stott 2019).			B			C	
Collembola	Collembola are ubiquitous, found in various habitats, making them suitable indicators for different ecosystems (Cicconardi et al. 2013). Collembola occupy various trophic levels within the soil food web, including herbivores, bacterivores, and fungivores. They provide insights into nutrient cycling, microbial activity, and the overall structure of the soil food web, and are sensitive to environmental changes (Li et al. 2022). Involved in organic matter decomposition, they primarily feed on fungal hyphae, playing a crucial role as facilitators of microbial succession during decomposition. They are also sensitive to physical soil degradation caused by various pressures, such as land-use intensity and unsustainable agricultural and forest practices (Bispo et al. 2009).			B	G	P	C	
Mycorrhiza	Mycorrhizal fungi establish symbiotic relationships with plant roots, enhancing the plant's nutrient uptake, particularly for essential elements like phosphorus. This symbiosis is crucial for robust plant growth and overall ecosystem productivity (Griffiths et al. 2016; Nielsen & Winding 2002). The hyphal networks formed by mycorrhizal fungi contribute to improved soil structure, enhancing soil aggregation, water retention, and resistance to erosion (Nielsen & Winding 2002).			B	G	P	C	

	For more information on mycorrhiza you can follow this link from the soil health academy: https://soilhealthacademy.org/blog/mycorrhizal-colonization/							

Table S3 Summary of standardised methodologies available for collecting data for each biodiversity metric. The methodology available column indicates whether a full methodology is available (Yes), a methodology covering part of the process from data collection to metric calculation is available (Partial), or no methodology is available (No). The methodology summary column summarises the data collection and metric calculation process.

Metric	Methodology available	Methodology summary
Dominance-diversity curves	Partial	Species diversity monitoring generates abundance and diversity data that can be used to calculate dominance-diversity curves (see **Species diversity**, **Relative abundance** for methodologies). The *goeveg* package in R contains the *racurve* function, for fitting Whittaker plots for community data (https://cran.r-project.org/web/packages/goeveg/goeveg.pdf).
Functional trait diversity	Partial	Data gathered during **Species diversity** surveys can be linked to functional traits. Common approaches for classifying functional traits include: • Plant traits: max height, habitat, flowering start, flowering duration, pollen vector (wind vs animal), seed dispersal agent (wind vs animal), seed weight, CSR (competitor, stress tolerant, ruderal) strategy, leaf persistence, life history, life form, sprout insulation, lateral spread, reproductive strategy, woodiness, light indication (Ellenberg values - shade vs light preference on a scale of 1-9) • Animal traits: body size, life form, trophic level, dispersal ability, habitat requirement, habitat specificity, temperature needs • Aboveground insects: trophic level, diet breadth, dispersal ability, voltinism (number of broods per year), body size • Birds: body size, trophic guild (granivore, insectivore, carnivore), dispersal ability, feeding strategy, nesting strategy, migration behaviour, dietary specialisation • Fungi: fruit body size, fruit body type, fruit volume, fruit thickness, tree host preference, dispersal vector (asexual spores, mycelial cores) • See Lelli et al. 2019, Moretti & Legg 2009, and Vandewalle et al. 2010 for more info

		Plant functional traits can be linked to ecosystem services. For examples, a study looking at functional traits in grassland ecosystems in Sweden assessed the following traits (Waldén et al. 2023): • Livestock production: leaf dry mass, specific leaf area, foraging value • Pollination: pollination syndrome (insect pollinated or not), flowering duration, nectar quantity, pollen quantity • Temperature regulation: height, specific leaf area, root architecture type (tap root, adventitious, fibrous), lifespan (longer-lived plants better for temp. reg.) • Water retention: specific leaf area, height, clonal lateral spread rate (non-clonal, <0.01 m/year, 0.01-0.25 m/year, >0.25 m/year), root architecture type • Cultural heritage: grassland specialist, mentions in traditional music • See Waldén et al. 2023, https://onlinelibrary.wiley.com/doi/full/10.1111/ele.14220 for more info Multiple online resources define functional traits for different species groups: • The Biological Records Centre website summarises relevant trait datasets for UK plants, invertebrates, birds and mammal https://www.brc.ac.uk/theme/species-traits-links-data-and-resources • Ecological Flora of Britain and Ireland http://ecoflora.org.uk/ • Ellenberg's indicator values for British plants http://nora.nerc.ac.uk/6411/1/ECOFACT2a.pdf (indicator values of plant tolerance to light, moisture, pH, nitrogen, salt) • TRY Global Plant Trait Database https://www.try-db.org/TryWeb/Home.php • Morphological trait database of European saproxylic beetles https://datadryad.org/stash/dataset/doi:10.5061/dryad.2fqz612p3 • Traits data for butterflies and macro-moths of Great Britain and Ireland https://esajournals.onlinelibrary.wiley.com/doi/10.1002/ecy.3670 • Sources of carabid beetle trait data in https://resjournals.onlinelibrary.wiley.com/doi/10.1111/icad.12348 and https://www.sciencedirect.com/science/article/pii/S0378112715005526#s0010 The fundiversity package can aid calculation of functional diversity indices in R (Grenié & Gruson 2023). Commonly used metrics include:

		• Mean trait value per community (average of trait values for a community, weighted by relative abundance) - dominant species have a disproportionate effect on ecosystem function (Roscher et al. 2012) • Rao's quadratic diversity (functional trait diversity)
Habitat area	Partial	Data collected during assessment of **Landscape diversity** can be used to derive information on the total area of semi-natural habitat and specific habitat types within a landscape. Projects will identify key relevant habitat types. Other projects may choose to focus on habitats of local importance or UK Priority Habitats: • England: https://www.gov.uk/government/publications/habitats-and-species-of-principal-importance-in-england • Scotland: https://www.nature.scot/scotlands-biodiversity/scottish-biodiversity-strategy-and-cop15/scottish-biodiversity-list • Wales: https://www.biodiversitywales.org.uk/Section-7 • Northern Ireland: https://www.daera-ni.gov.uk/articles/habitat-and-species-actions-plans Further patch level metrics can be calculated using FRAGSTATS, more information on metric selection and interpretation here: https://fragstats.org/index.php/background/patches-patchiness-levels-of-landscape-metrics and see **Patch size distribution**.
Identity	Partial	Data gathered during **Species diversity** surveys can be linked to functional traits, see **Functional trait diversity**. Presence, absence, abundance or diversity of specific functional traits relevant to a project can be monitored. See **Species diversity**, **Relative abundance**, **Biomass (measure of abundance)** for more details on metric calculation.
Landscape diversity	Partial	Habitats can be mapped using UK Habitats Classification (UKHab Ltd 2023) https://ukhab.org/. • Record habitat type and area for patches 400 sqm or larger (Maskell et al. 2019) • Calculate diversity metrics based on the UKHab Level 3 classifications (e.g. acid grassland, calcareous grassland, neutral grassland, dwarf shrub heath, coniferous woodland) • Simpson's diversity index is recommended for its more intuitive interpretation, it reflects the probability

		of habitat patches being in different habitat classes (higher value = higher diversity) • The FRAGSTATS software package (https://fragstats.org/) can be used to calculate Simpson's diversity index • More information on selecting and interpreting landscape diversity metrics can be found at https://fragstats.org/index.php/documentation There are many other metrics (Haines-Young & Chopping 1996; Nagendra & Gadgil 1999; Magurran 2004) under the umbrella of landscape diversity that can be calculated e.g.: • Shannon's diversity index = proportion of landscape occupied by a habitat type (has disadvantages, rare habitats are disproportionately represented) • Habitat proportion = % cover of specific habitat type within a focal area • Evenness measures (e.g. Simpson's evenness, Shannon's evenness) • Patch sizes (see **Patch size distribution**) • Measures of **Connectivity and fragmentation** (more complex analyses that integrate spatial habitat information with species dispersal data)
Relative abundance	Partial	Species abundance data is collected during some of the species-level surveys to obtain diversity metrics (see **Species diversity** for methodologies). Suitable standardised methods are available for trees (collected during woodland plant surveys, **Tree diversity** and **Seedling regeneration** surveys), carabid beetles, spiders, butterflies, moths, birds, bats, spittle bugs and crane flies, which all record the number of individuals in each group. Biomass can also be used to assess the abundance of individuals (see **Invertebrate biomass**, **Mammal biomass**, **Vegetation biomass**). • It can be also used to compare abundance between species groups and provides useful information when comparing species where individuals vary appreciably in size • Biomass can provide an abundance estimate for herbaceous plants It can be unfeasible to estimate abundance at large scales, using smaller representative sample units can

		increase the accuracy of detection, for example plants monitored within standardised quadrats (Buckland et al. 2005). Calculating metrics (based on Buckland et al. 2005): • Species-specific densities are calculated for each species = number of individuals per unit area • This can be used to calculate mean density per habitat or site • To calculate relative abundance over time, species-specific densities are divided by density at the initial time point, tracking increases or declines in abundance • If aggregating relative abundance across species, the geometric mean should be used; this allows overall trends of increases or declines to be detected (Buckland et al. 2005) • The geometric mean is calculated by averaging log(relative abundance) across species and taking the exponential
Similarity	Partial	Similarity metrics require data on species richness and abundance which can be obtained using methods for diversity metrics (see **Species diversity** for methodologies). They are typically calculated for each species group separately (e.g. for birds or plants). The Bray-Curtis index assesses the number of shared species between two communities and their relative abundances, and can be calculated using the *vegdist* function in the *vegan* package in R. The Bray-Curtis index ranges from 0 (no species in common) to 1 (two communities identical). Ordination methods can be used to visualise species assemblages in different communities. Ordination methods summarise information on species identity and relative abundance into a single value that captures the overall community composition of a site. A set of sites can be plotted on a graph, with sites that are more similar to each other plotted more closely together. A commonly used ordination method is non-metric multidimensional scaling (NMDS), which is based on Bray-Curtis similarity indices (*monoMDS* function in *vegan*). With NMDS, community composition in different sites can be assessed and analysis of similarities (*anosim* function in *vegan*) is used to test for differences using the *vegan* package in R. User guide for vegan available here https://cran.r-project.org/web/packages/vegan/vegan.pdf

		See https://rpubs.com/an-bui/vegan-cheat-sheet for information on the vegan package and examples of visualisation, including NMDS.
Species diversity (alpha, beta, gamma)	Partial	Step 1 - decide on species groups to monitor Using >2 species groups captures overall biodiversity more accurately. Species groups representing different parts of the ecosystem should be selected and the species listed below have monitoring protocols available to assess them: • Plants - have the highest number of correlations with other species groups • Carabid beetles - predatory so will reflect changes in their prey, respond to habitat changes differently to butterflies/moths • Spiders - generalist predators providing additional insights to carabids • Butterflies - respond to changes in vegetation abundance and quality, easy to recognise • Moths - good national data for comparison, reflect environmental change • Birds - reflect habitat structure and larger-scale habitat diversity, easy to recognise • Bats - sensitive to habitat change, roost site availability and invertebrate food sources • Frogs - ubiquitous predatory amphibian, feeding on invertebrates • Wild herbivores (Rabbits and Deer) - drivers of pant composition and structure, dropping counts to estimate relative abundance • Spittle bugs - xylem feeding invertebrates that are easy to spot and ecologically well-understood in the UK • Crane flies - soil-dwelling and important food source for other groups Step 2 - collect data • Plants - National Plant Monitoring Scheme - 5 x 5 m plots (or 10 x 10 m plots in woodland) - 3 plots per habitat type within 1 km square • Carabid beetles - UK Environmental Change Network (ECN) ground predators protocol https://ecn.ac.uk/measurements/terrestrial/i - 3 transects (10 traps per transect 10 m spaced), representing different vegetation types within central 9 ha representing site • Spiders - UK ECN spiders protocol https://ecn.ac.uk/measurements/terrestrial/i - 3 transects (10 traps per 100 m transect), representing different vegetation types within central 9 ha representing site

		• Butterflies - UK Butterfly Monitoring Scheme https://ukbms.org/ - 1-2 km transect representing habitats within 1 km square (subsections of transects) • Moths - UK ECN moths protocol https://ecn.ac.uk/measurements/terrestrial/i - 1 light trap in centre of 1 ha square • Birds - UK Breeding Bird Survey https://www.bto.org/our-science/projects/breeding-bird-survey - 2 transects per 1km square per site, one transect per half, habitat features recorded • Bats - UK ECN bats protocol https://ecn.ac.uk/measurements/terrestrial/b - 2 transects per 1km square per site, one transect per half, habitat features recorded • Frogs - UK ECN frogs protocol https://ecn.ac.uk/measurements/terrestrial/b - shallow ponds/ditches within site • Wild herbivores (Rabbits and Deer) https://ecn.ac.uk/measurements/terrestrial/b - UK ECN rabbits and deer protocol - 1-2 km transect representing habitats within site (subsections of transects) • Spittle bugs - UK ECN spittle bugs protocol https://ecn.ac.uk/measurements/terrestrial/i - 20 quadrats 0.25 m2 randomly in the central 1 ha of site • Crane flies - UK ECN crane flies protocol https://ecn.ac.uk/measurements/terrestrial/i - 50 x 40 m central area divided into 20 subplots, 10 cm diameter and 10 cm depth cores taken randomly from within subplots Sampling layout For sites <0.1 km (<10 ha) • Plants – 3 5 x 5 m plots per habitat type (or 10 x 10 m plots if in woodland) • Carabid beetles and spiders – 1-3 100 m transects (10 traps per transect) per habitat depending on size of site and habitat conformation • Moths – trap in centre of site (if >1 ha), traps attract moths from 30 m radius, so if there are large enough areas of distinct habitat consider multiple traps located in these • Spittle bugs – 20 0.25 m2 quadrats • Crane flies – 20 subplots in central 50x40 m area • Butterflies, bats, birds and wild herbivores – a modified transect approach might be possible – linking presence to smaller scale habitat heterogeneity will be harder

		For sites 0.1-1 km2 (10-100 ha) • Plants – 3 5 x 5 m plots per habitat type (or 10 x 10 m plots if in woodland) • Carabid beetles and spiders – 3 transects per habitat type (traps spaced every 10 m, aim for 10 traps = 100 m transects) • Moths – traps attract moths from 30 m radius, so if there are large enough areas of distinct habitat consider multiple traps located in these • Spittle bugs – 20 0.25 m2 quadrats • Crane flies – 20 subplots in central 50x40 m area • Bats and birds – in sites 1 km2 or close to 1 km2 2 c.1 km transects, in smaller sites a modified transect approach might be possible • Butterflies and Wild Herbivores - in sites 1 km2 or close to 1 km2 transects 1-2 km transect, in smaller sites a modified transect approach might be possible For sites >1 km2 (>100 ha) • Plants – for each 1 km square 3 5 x 5 m plots per habitat type (or 10 x 10 m plots if in woodland) • Carabid beetles and spiders – per 1 km square 3 transects per habitat type (traps spaced every 10 m, aim for 10 traps = 100 m transects) • Moths – traps attract moths from 30 m radius, so if there are large enough areas of distinct habitat consider multiple traps co-located with plant plots • Spittle bugs – 20 0.25 m2 quadrats per 1 km square • Crane flies – 20 subplots in central 50 x 40 m area per 1 km square • Bats and birds – for each 1 km square 2 transects (where large areas of one habitat are present sampling might be possible focussed on habitat type) • Butterflies and Wild Herbivores – for each 1 km square 1-2 km transect (where large areas of one habitat are present sampling might be possible focussed on habitat type might be possible) Step 3 - calculate diversity indices Point diversity • Calculated at the level of plot or trap

		• Simpson's diversity index (D) is recommended (Magurran 2004) • Simpson's diversity index can be calculated using the diversity function in the vegan package in R • Simpson's diversity index is often presented as the complement (1−D) • As the complement of Simpson's increases so does diversity • Can assess variation in point diversity by habitat type if multiple habitats present at site Alpha diversity • In sites <10 ha, calculated at the level of habitat for plants, carabid beetles, spiders, moths, crane flies if possible, calculated at the site level for bats, birds and butterflies • In sites 10-100 ha, calculated at the level of habitat for plants, carabid beetles, spiders, moths, crane flies, calculated at the site level for bats, birds and butterflies • In sites >100 ha, calculated per habitat for all species if possible • Simpson's diversity index (D) is recommended (Magurran 2004) • Simpson's diversity index can be calculated using the diversity function in the vegan package in R • Simpson's diversity index is often presented as the complement (1−D) • As the complement of Simpson's increases so does diversity Beta diversity • Between habitat diversity - likely only to be relevant for sites >10 ha, with multiple large areas of habitat • A simple metric of beta-diversity is Whittaker's beta diversity, which can be calculated using the betadiver function in the vegan package in R • Whittaker's beta diversity = (total site species richness)/(mean habitat species richness) • See also **Similarity** for measures of differences in the assemblages of species between habitats Gamma diversity • Total diversity at the site or landscape scale, calculated across all habitat types • Often calculated as species richness Other metrics that can be derived

		• Species richness • Species evenness metrics User guide for vegan available here https://cran.r-project.org/web/packages/vegan/vegan.pdf See https://rpubs.com/an-bui/vegan-cheat-sheet for information on the vegan package and examples of visualisation. See **Mammal biomass**, **Invertebrate biomass**, **Vegetation biomass** for more information on considerations relating to these groups.
Vegetation structure	No	Physical measurement of vegetation structure characteristics is possible with varying levels of guidance available depending on the habitat. However remote-sensing approaches such as LiDAR show promise for simplifying and standardising measurement of vegetation structure across habitats in the future. Forest: Collection of Vegetation structure data can be carried out alongside data collection for **Vegetation biomass**, **Tree diversity**, **Seedling regeneration**, **Tree Age**. The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record tree growth categories (Ditchburn et al. 2020). Young trees (seedlings and saplings), diameter at breast height (DBH) < 4 cm and mature trees, DBH > 4 cm, are recorded. • Forest is defined as having >20% canopy cover • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • The physical layering of the vegetation is recorded (tree canopy, shrub layer, field layer, ground layer) • Forest storeys (distinct forest layers) and their upper- and mid-crown heights are recorded • Within each plot DBH is recorded for all mature trees (DBH > 4 cm) + crown width and timber height for the dominant tree in each storey • Saplings (height > 50 cm, DBH < 4 cm) are recorded in a 2.52 m radius plot at the centre of each 0.01 ha plot

		• Seedlings (height < 50 cm, DBH < 4 cm) are recorded in a 1.78 m radius plot at the centre of each 0.01 ha plot • For saplings and seedlings browse class and origin (planted, regeneration, sucker) are also recorded The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • Recording vegetation layers - Chapter 9 Sub-Component Data • Recording forest storeys - Chapter 8 Components • Recording mature trees - Chapter 13 Tree Assessment Procedures • Recording young trees - Chapter 15 Young Tree Assessments Deriving metrics: • Vertical structural diversity: a simple metric is the number of forest layers, more complex is standard deviation of tree height (Mura et al. 2015) • Horizontal structural diversity: standard deviation of DBH (Mura et al. 2015) Shrub-dominated ecosystems (heathland, scrub): No standardised method available in the UK, but modification of protocols used elsewhere may be possible. E.g. A method of vegetation monitoring in shrubland is given in Wood et al. 2012, based on the BBird protocol developed by the Montana Cooperative Wildlife Research Unit (Martin et al. 1997) • Measurement should be taken at the peak of the growing season • Plots 50 m radius are dispersed throughout the habitat and should be surrounded by 100 m of the same habitat and 300 m from other sampling points • Within each 50 m radius plot are four 5 m radius subplots, one central, three at 0, 120 and 240 degree angles at random distances between 20 and 80 m from the centre • In each subplot, at the centre and 5 m N, S, E, W of the centre, a Wiens pole 12 m tall subdivided into 30 cm subsections is placed upright in the vegetation

		• The number of vegetation intersections in each subsection are recorded • At each 50 m sampling point 20 foliage-height tallies are generated • At each sampling point foliage height diversity can be calculated based on the number of hits in each subsection, this can be averaged across the 20 samples taken • Horizontal diversity can be calculated as the standard deviation of canopy height across all 16 sample points Herbaceous-dominated ecosystems (grassland, peatland, saltmarsh, wetland): No standardised method available in the UK, but modification of protocols used elsewhere may be possible. E.g. A method of vegetation monitoring in grassland is given in Wood et al. 2012, based on the BBird protocol developed by the Montana Cooperative Wildlife Research Unit (Martin et al. 1997) and Wien's 1969 method of assessing grassland vegetation structure. • Measurement should be taken at the peak of the growing season • Plots 50 m radius are dispersed throughout the habitat and should be surrounded by 100 m of the same habitat and 300 m from other sampling points • Within each 50 m radius plot are four 5 m radius subplots, one central, three at 0, 120 and 240 degree angles at random distances between 20 and 80 m • In each subplot, at the centre and 5 m N, S, E, W of the centre, a Wiens pole subdivided into 10 cm subsections is placed upright in the vegetation • The number of vegetation intersections in each subsection are recorded • At each 50 m sampling point 20 foliage-height tallies are generated • At each sampling point foliage height diversity can be calculated based on the number of hits in each subsection, this can be averaged across the 16 samples taken • Horizontal diversity can be calculated as the standard deviation of canopy height across all 20 sample points
Deadwood volume	Yes	The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record deadwood volume (Ditchburn et al. 2020). Standing dead trees, lying deadwood and stumps are recorded.

		• Forest is defined as having >20% canopy cover • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • Within each plot three 10m lying deadwood transects are established • The diameter of deadwood is measured wherever it crosses transect line • The length, breadth and circumference of each deadwood piece can also be measured to estimate deadwood volume • The decay class of each piece of deadwood is also assessed and allocated to one of five categories The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • How to set up deadwood transects and record deadwood - Chapter 16 Lying Deadwood Transects • Recording the decay class of the deadwood - Chapter 18 Decay Classes A detailed methodology on calculation of standing deadwood volume in m3 per ha is available from the National Forest Inventory team.
Pollination	Yes	A standardised methodology is available from the UP Pollinator Monitoring Scheme (UKPoMS) (https://ukpoms.org.uk/) • Site visits 4 times a year (mid-April to mid-May, June, July, Aug to mid-Sept) during suitable weather conditions • Pan traps are set for 6 hours during 0900-1700 • 3 traps (UV blue, yellow, white) are set up in 5 locations per km square - where possible these should be allocated in alignment with **Species diversity** data collection and in large sites with multiple habitats, stratified by habitat type • In short vegetation traps are placed on the ground, in vegetation >10 cm traps are supported on a stake • Traps are filled with water with a few drops of washing up liquid added • Local flower abundance is also recorded in the area surrounding the pan traps

		• Flower visitation in 50 x 50 cm plots is also recorded, targeting specific flowering plants at specific times of years Derived metrics include total abundance and diversity (see **Species diversity**). More advanced measures of network statistics are possible (Kaiser-Bunbury & Blüthgen 2015) e.g.: • Interaction diversity - higher interaction diversity = higher community stability • Interaction evenness - if a few species dominate and others are rare then interaction evenness will be low, but loss of rare species can increase evenness even as diversity declines • Partner diversity - diversity of interaction partners per species (pollinator species associated with each plant or plant species associated with each pollinator); high partner diversity increases overall resilience • Vulnerability and generality - mean diversity of interaction partners across species; high partner diversity increases overall resilience • Specialisation - dependency of species on few partners, implications for competition for resources and vulnerability to loss of partners • Modularity - identifying groups of species that share interactions more frequently within modules
Seedling regeneration	Yes	Seedling regeneration data can be collected at the same time as assessing **Vegetation biomass**, **Vegetation structure**, **Tree age**, **Tree diversity**. The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record tree growth categories (Ditchburn et al. 2020). Young trees (seedlings and saplings), diameter at breast height (DBH), < 4 cm are recorded. • Forest is defined as having >20% canopy cover. • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • Saplings (height > 50 cm, DBH < 4 cm) are recorded in a 2.52 m radius plot at the centre of each 0.01 ha plot • Seedlings (height < 50 cm, DBH < 4 cm) are recorded in a 1.78 m radius plot at the centre of each 0.01 ha

		plot The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • Recording young trees - Chapter 15 Young Tree Assessments
Tree age	Yes	Tree age data can be collected at the same time as assessing **Vegetation biomass**, **Vegetation structure**, **Seedling regeneration**, **Tree diversity**. The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record tree growth categories (Ditchburn et al. 2020). Young trees (seedlings and saplings), diameter at breast height (DBH) < 4 cm and mature trees, DBH > 4 cm, are recorded. • Forest is defined as having >20% canopy cover. • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • Within each plot DBH is recorded for all mature trees (DBH > 4 cm) • Saplings (height > 50 cm, DBH < 4 cm) are recorded in a 2.52 m radius plot at the centre of each 0.01 ha plot • Seedlings (height < 50 cm, DBH < 4 cm) are recorded in a 1.78 m radius plot at the centre of each 0.01 ha plot • All dead stems with DBH > 4 cm are recorded in the 2.52 m radius plot at the centre of each 0.01 ha plot The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • Recording mature trees - Chapter 13 Tree Assessment Procedures • Recording young trees - Chapter 15 Young Tree Assessments • Recording standing deadwood - Chapter 18 Decay Classes

		A methodology for categorising trees by growth stage (dependent on species) is given in Appendix 2: Field Survey Methodology of the Caledonian Pinewood Recovery Project https://treesforlife.org.uk/about-us/caledonian-pinewood-recovery/ (Rainey & Holmes 2023)
Tree diversity	Yes	Tree diversity data can be collected at the same time as assessing **Vegetation biomass**, **Vegetation structure**, **Seedling regeneration**, **Tree Age**. The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record tree growth categories (Ditchburn et al. 2020). Young trees (seedlings and saplings), diameter at breast height (DBH) < 4 cm and mature trees, DBH > 4 cm, are recorded. • Forest is defined as having >20% canopy cover • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • Within each plot DBH is recorded for all mature trees (DBH > 4 cm) • Saplings (height > 50 cm, DBH < 4 cm) are recorded in a 2.52 m radius plot at the centre of each 0.01 ha plot • Seedlings (height < 50 cm, DBH < 4 cm) are recorded in a 1.78 m radius plot at the centre of each 0.01 ha plot The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • Recording mature trees - Chapter 13 Tree Assessment Procedures • Recording young trees - Chapter 15 Young Tree Assessments Deriving metrics - see **Species diversity** for calculation of Simpson's diversity index.
Vegetation biomass	Partial	Collection of data on vegetation biomass can be carried out alongside data collection for **Vegetation structure**, **Tree diversity**, **Seedling regeneration**, **Tree Age**.

		Feeds into calculation of **Relative abundance**, **Dominance-diversity curves**, **Biomass (measure of abundance)**. *Forest* The UK National Forest Inventory (NFI) provides a standardised methodology to establish fixed area survey plots and record tree diameter at breast height (DBH) for all established trees (Forestry Commission 2020). • Forest is defined as having >20% canopy cover • 0.01 ha plots (5.64m radius) are established, the number of plots is determined by the size of the forest area • Within each plot DBH is recorded for all mature trees (DBH > 4 cm) Allometric equations can be used to convert DBH measurements into biomass: • Zianis et al. 2005 Biomass and stem volume equations for tree species in Europe • Muukkonen & Mäkipää 2006 Biomass Equations for European Trees: Addendum The NFI Survey Manual provides the methodology: https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/nfi-survey-manual-for-third-cycle-field-samples/ • How to allocate plots - Chapter 12 Plot Assessments • Recording mature trees - Chapter 13 Tree Assessment Procedures *Other ecosystems* • For ecosystems other than forest, standardised methods for assessing vegetation biomass are not available • Recommendations for sampling can be found in (Fahey & Knapp 2007) Principles and Standards for Measuring Primary Production (https://academic.oup.com/book/2502/) *Herbaceous-dominated ecosystems (grassland, peatland, wetland, saltmarsh)*

		• Destructive sampling of vegetation followed by drying can be used to estimate biomass • For grassland systems the Nutrient Network provides a method of assessing aboveground standing crop (see Core Sampling Methodology; Aboveground Standing Crop https://nutnet.org/exp_protocol) • Samples should be collected at or just after the peak of the growing season • The aboveground standing crop in two 10 × 100 cm strips is clipped at ground level (the precise location should be recorded to avoid resampling in subsequent years), sampling effort should follow the guidance for assessing plant diversity, but avoid harvesting biomass within the plant diversity plots • The standing crop is sorted into 1. previous year's dead material, 2. current year's bryophytes, 3. current year's graminoids (grasses, sedges, rushes), 4. current year's legumes, 5. current year's non-leguminous forbs, 6. current year's woody growth • Vegetation is dried at 60 oC for 48 hour and weighed to the nearest 0.01 g • Estimation of productivity in grazed systems is complicated - in actively grazed systems temporary exclosures can be randomly placed for 1-2 weeks and samples collected from these and areas outside the exclosures to calculated the consumption level, which can be added to residual biomass at the end of the season to estimate overall aboveground productivity *Shrub-dominated ecosystems (heathland, scrub)* • Estimation of productivity and biomass in shrub dominated systems requires a combination of destructive sampling, dimension analysis and extrapolation to unit area of landscape • Verra provide a methodology for estimating carbon stocks in non-tree pools: https://verra.org/methodologies/vmd0001-estimation-of-carbon-stocks-in-the-above-and-belowground-biomass-in-live-tree-and-non-tree-pools-cp-ab-v1-1/ • Circular sampling frames (radius dependent on the heterogeneity of the vegetation) • All vegetation inside the frame is cut at the base and removed and weighed to give the wet mass • One representative subsample of the cut material is dried to determine the dry mass and the wet:dry ratio is calculated • The wet:dry ratio is used to estimate the aboveground biomass
Biomass (measure of abundance)	Partial	See **Vegetation biomass**, **Invertebrate biomass** and **Mammal biomass** for methods.

		Biomass of species collected across multiple trophic levels can contribute to calculation of Energy flow rates.
Energy flow rates	No	Total energy flow is measured as the sum of all energy flows through all trophic compartments of an ecosystem (Buzhdygan et al. 2020). Flow and storage of energy is not necessarily correlated, therefore energy flows should be measured as well as standing biomass. Calculation of energy flows requires information on population density for all species, knowledge of diet/feeding behaviour, estimation of net primary productivity (Malhi et al. 2022). Buzhdygan et al. 2020 measured energy flows across multiple trophic levels in grassland by measuring: • Dry plant biomass in May and August - clipped aboveground biomass (0.2 x 0.5 m rectangle) and root biomass to 40 cm depth • Aboveground invertebrate fauna sampled from May to September by pitfall trap • Aboveground invertebrate fauna sampled by suction sampling in June and July • Aboveground invertebrates were sorted to order and into trophic groups (herbivore, carnivore, omnivore, decomposer) • Belowground mesofauna (5cm diameter, 5cm depth core) and macrofauna (22cm diameter, 10cm depth core) were sampled • Body mass of each species was estimated from the mean length of males and females using allometric equations • Microbial biomass was measured using substrate-induced respiration • Dry mass of surface litter from a 50 x 20 cm frame • Soil organic matter was estimated from soil carbon and bulk density • Energy flows were represented as dry mass per area per day See **Invertebrate biomass**, **Mammal biomass**, **Vegetation biomass**, **Species diversity** for relevant methodologies.
Invertebrate biomass	No	Biomass should be assessed at the Order level as a minimum, allowing trends for different groups to be followed.

		Biomass can be calculated as: • Dry biomass - can be influenced by preservation liquid/storage time • Fresh biomass - requires live trapping • Estimated fresh biomass using size-weight models (see allometric equations for carabids below) • Biomass of drained trap contents (e.g. pitfall trap contents drained on filter paper before weighing) Recommendations on standardised monitoring approaches for multiple methods and collection of relevant metadata are given in (Montgomery et al. 2021): Malaise trapping (adult semi-aquatic insects, non-lepidopteran pollinators, flies) • The Global Malaise Program is developing a malaise trapping network (https://biodiversitygenomics.net/projects/gmp) • Trapping should be carried out for a defined period each year • The placement of the trap and surrounding habitat will determine the efficacy of the trapping, so multiple traps in different microhabitats within a site are recommended • Malaise trapping generates large numbers of individuals, requiring a large investment of ID time Pitfall trapping (ground-dwelling beetles, ants) • For independent samples traps should be placed 25 m apart (other studies have suggested that 10m is sufficient) • Roofs over the pitfall traps prevent rain from diluting the samples • See Environmental Change Network (ECN) Ground Predators protocol: https://ecn.ac.uk/measurements/terrestrial/i and Species diversity Light trapping (adult semi-aquatic insects, ground-dwelling beetles, night-active moths, flies) • Insects are usually attracted from within 30 m of the trap • Microhabitat will determine the insects collected in the trap • Post-processing can be high, as high diversity and abundance of insects are often caught • See ECN Moths protocol: https://ecn.ac.uk/measurements/terrestrial/i and Species diversity

		Pan trapping (non-lepidopteran pollinators, flies) • See UK Pollinator Monitoring Scheme (UKPoMS): https://ukpoms.org.uk/ and Pollination • UKPoMS places 1 trap per square km, which is suitable for sampling across large geographic areas • The Bee Inventory Plot (USA) places 15 traps in an "X" with traps spaced 5m apart Beating sheet (leaf-chewing larvae, ants) • A square sheet 90 x 90 cm is used to catch insects that are knocked out of shrubs, trees, and, in some cases, ground cover • Some evidence suggests that three plants of the same species must be sampled to accurately estimate insect abundance Audio (singing insects) • Topography, habitat and the noise of other animals can influence the recording of invertebrate activity • Automated identification pipelines will be crucial to scaling acoustic monitoring Active visual surveys (adult semi-aquatic insects, non-lepidopteran pollinators, leaf-chewing larvae, dragonflies and damselflies, ants, butterflies) • Transects, point counts and area counts, the UK Butterfly Monitoring Scheme (UKBMS) is an example of a transect survey: https://ukbms.org/ • Visual surveys are only suitable for large easily identifiable invertebrates UKPoMS (https://ukpoms.org.uk/) • Site visits 4 times a year (mid-April to mid-May, June, July, Aug to mid-Sept) during suitable weather conditions • Pan traps are set for 6 hours during 0900-1700 • 3 traps (UV blue, yellow, white) are set up in 5 locations per km square • In short vegetation traps are placed on the ground, in vegetation >10 cm traps are supported on a stake • Traps are filled with water with a few drops of washing up liquid added • Local flower abundance is also recorded in the area surrounding the pan traps

		• Flower visitation in 50 x 50 cm is also recorded UK Butterfly Monitoring Scheme (UKBMS) (https://ukbms.org/) • Number and variety of species monitored • Transects take 45-60 minutes to walk and should be 1-2 km length • The transect represents the habitat variability within a site • The transect is divided into 15 sections and the habitat or management type in each is recorded • Butterflies 2.5 m either side of the transect and 5 m ahead are recorded ECN Ground Predators (https://ecn.ac.uk/sites/default/files/ECN/Protocols/IG.pdf) • 3 transects with 10 pitfall traps spaced 10 m apart • 7.5 cm diameter and 10 cm deep cups to make traps, covered with a chickenwire cage to stop small mammals from falling in and a plastic rain cover • Traps set out in first week of May, emptied fortnightly for 13 sampling periods until end of November Allometric equations convert carabid body length to biomass (Weiss & Linde 2022) • For carabids with body length <11.8 mm use Booij et al. 1994: ln(weight[g]) = -8.92804283 + 2.5554921*ln(size[mm]) • For carabids with body length >11.8 mm use Szysko 1983: log(weight[mg]) = -1.3 + 2.95*log(size[mm])
Patch persistence/turnover	Partial	A patch is a recognisable area of habitat with definable boundaries. The UK Habitats Classification can be used to define habitat types (https://ukhab.org/). Type and area of all habitat patches > 20 x20 m should be recorded, following the UK Countryside Survey approach https://countrysidesurvey.org.uk/science/habitats (Maskell et al. 2019). Linking patch dynamics to population dynamics requires technical modelling skills (e.g. (Johst et al. 2011)).
Allelic diversity	No	In the absence of mutation, selection and migration, changes in allele frequencies reflect genetic drift, and can also be used to calculate Effective population size and rate of genetic drift. Typically based on a

		set of two samples (e.g. DNA extracted from faeces, hair, blood, animal/plant tissue) from multiple individuals within a population taken at least one or more generations apart (Wang et al. 2016). As with calculation of Effective population size, estimation requires the collection and analysis of genetic data that requires a high level of technical expertise.
Colonisation/local extinction rates	Partial	Local colonisation and extinction metrics can be derived from data on species richness and abundance which can be obtained using methods for the richness, abundance and diversity metrics (see Invertebrate biomass, Mammal biomass, Vegetation biomass, Species diversity for methodologies). However, for rare species, there will be lower confidence that these methods accurately represent species occupancy. Community survey data in the form of site x species matrices is required to calculate these metrics. Estimates of biodiversity turnover, rates of extinction and immigration are reported as a closed range of values between 0 and 1 (Hillebrand et al. 2018). • Richness-based species exchange ratio (SERr) = (immigrations + extinctions)/total number of species. SER measures gross change in species composition - where 0 means all species persist and 1 means all species are exchanged. • A more robust approach would consider changes in proportional abundances over time. Proportional abundance species exchange ratio (SERa), where 0 means identity and dominance structure unchanged and 1 means all species replaced. (Complements Wishart's similarity ratio and Simpson's diversity index).
Connectivity and fragmentation	No	Data collected during assessment of Landscape diversity can be used to derive structural connectivity and fragmentation metrics. The FRAGSTATS software package (https://fragstats.org/) can be used to calculate metrics such as: • Mean nearest neighbour distance - the distance from a focal patch to its nearest neighbouring patch • Edge density - the ratio of habitat edge to area • Contagion - degree of aggregation at the landscape scale, inversely related to edge density, i.e. contagion is high if a single habitat type occupies a large percentage of the landscape • Proximity index - size and proximity of all patches with edges within certain radius of focal patch • Perimeter-area fractal dimension - relationship between patch area and patch perimeter, can be calculated at the habitat type or landscape level

		• Patch density - number of patches per unit area, allowing comparisons between different sized landscapes • Clumpiness index - frequency with which different pairs of patch types appear side-by-side, scaled by proportion of focal habitat class within an area More information on selecting and interpreting metrics can be found at https://fragstats.org/index.php/documentation. Functional metrics of connectivity and fragmentation that integrate biological information of the focal species with structural metrics require more bespoke, specialist modelling approaches.
Disturbance	No	Disturbance can be quantified as (Sousa 1984; Moloney & Levin 1996): • Area of disturbance • Magnitude of disturbance (intensity/severity) • Frequency of disturbance (random point frequency/regional frequency) • Predictability (variance in mean time between disturbances) • Turnover rate Large-scale disturbance is captured in metrics of landscape-scale habitat dynamics: Patch persistence/turnover, Landscape diversity. Medium-scale disturbance could be captured through assessment of metrics such as: herbivory, predation, patch dynamics (fine-scale disturbance), browsing/debarking, dung density. Disturbance events are discrete in time and space and are usually temporary and localised. Changes in demographic rates as a consequence of disturbance can be captured by changes in abundance (Relative abundance, Mammal biomass, Invertebrate biomass, Vegetation biomass) and diversity (Species diversity) (see also Population fluctuations).

		Disturbance effects on ecosystem processes can be captured by monitoring: Energy flow rates, Nutrient cycling rates, Vegetation biomass.
Effective population size	No	The process involved in establishing effective population size (Ne) is methodologically complex and requires specialist expertise that is likely to be beyond the reach of the average Nature-based Solutions project. Ne is relevant in projects that target the conservation of a specific species and will have an impact on a distinct population. An outline of the process involved in calculating Ne is given below. There are two main approaches to estimating effective population size (Wang et al. 2016): • Prediction from demographic parameters e.g. census size, variance of reproductive success • Prediction from genetic properties e.g. changes in allele frequency (an allele is a variant of a gene, in some cases different alleles are associated with different traits e.g. for different eye colours) and linkage disequilibrium (association between alleles at different loci) Linkage disequilibrium is a widely used and well-evaluated measure, and can be calculated from samples (e.g. DNA extracted from faeces, hair, blood, animal/plant tissue) from multiple individuals within a population taken at a single point in time: • Simple to calculate from samples of multilocus genotypes (alleles at multiple loci that are transmitted together from a parent to an offspring) • Can be used to track population trajectories on a yearly basis • Risk of inaccuracies with small populations and non-random mating/population structure Change in allele frequencies also tracks Ne: • Reflects genetic drift • Mutation, selection and migration can also influence genetic drift, which can complicate interpretation • Estimates are based on two samples of a set of 10-20 microsatellites (small pieces of repeating DNA that serve as markers), taken at least one generation apart Demographic parameters:

		• Require data on census sizes, variances of progeny numbers, type of mating system and other demographic data • These data are rarely available and the increasing availability of genetic data makes genetic estimation the leading approach Additional related metrics can be calculated: • Nei inbreeding effective population size (loss of Heterozygosity, doesn't change until inbreeding accumulated, influenced by number of parents) • Nev variance effective size = rate of genetic drift (change in allele frequencies through time, early detection, influenced by number of offspring) Effective population size, Allelic diversity and rate of genetic drift are three interrelated concepts and metrics that give slightly different pieces information on the underlying genetic properties of populations.
Heterozygosity	No	As with calculation of Effective population size and Allelic diversity, estimation requires the collection and analysis of genetic data that requires a high level of technical expertise. Calculation of heterozygosity requires a high marker density (markers across multiple loci, to give an overall assessment of heterozygosity within an individual's genome) but can be assessed with a low sample size (<100 individuals within a population sampled e.g. DNA extracted from blood or animal/plant tissue) (Leroy et al. 2018). Metrics include: • Runs of homozygosity (F metric) - provide information on the history of inbreeding in a population • Change in heterozygosity (He) - rate of loss of heterozygosity is a measure of genetic drift • Effective number of alleles (Ae) - probability that two randomly chosen genes in a population are identical by descent (reflects the expected frequency of homozygous individuals in a population)
Mammal biomass	Partial	Biomass can be estimated from the number of individuals and their average body mass (Damuth 2023). Standardised methods of estimating numbers of small mammals and deer are outlined below. Camera trapping can be deployed to support and complement these direct methods (Smart et al. 2004).

		The Mammal Society National Woodland Small Rodent Survey (Flowerdew et al. 2004): • 7x7 grid, 2 traps per point to survey 0.81 ha area May-June and November-December For monitoring deer populations (Smart et al. 2004): • Direct counts are only possible in open habitats (see Deer Initiative Vantage Point Counting Guide below) • Indirect methods e.g. Faecal Standing Crop (FSC) and Faecal Accumulation Rate (FAR) can be used in woodland, FSC is more accurate but requires the decay rate to be determined (see Smart et al. 2004 for method) • FSC - randomly placed plots stratified by habitat type - converted into estimates of deer population size and density (based on defecation rates and dung persistence rates) UK Environmental Change Network Rabbits and Deer protocol https://ecn.ac.uk/measurements/terrestrial/b The Deer Initiative Dung Counting Best Practice Guide, method for Faecal Accumulation Rate: https://www.thedeerinitiative.co.uk/uploads/guides/175.pdf Method for Faecal Standing Crop in Smart et al. 2003 Monitoring woodland deer populations in the UK: an imprecise science https://onlinelibrary.wiley.com/doi/10.1046/j.0305-1838.2003.00026.x The Deer Initiative Vantage Point Counting Best Practice Guide https://www.thedeerinitiative.co.uk/uploads/guides/107.pdf
Nutrient cycling rates	Partial?	Indicators at different scales have been suggested (Lavelle et al. 2005): • Micro - soils (maturity index), diversity in microfoodweb communities (microfauna, mesofauna) • Ecosystem - physical structure of soils/sediments, bioindicators of soil quality, nutrient balance at plot level, decomposition rates, invertebrates as bioindicators of soil quality, distribution of organic matter in particle size fractions, chemical analysis of plant material, soil texture, soil porosity, soil aggregation

		• Landscape - monitoring nutrient concentrations along soil profiles, landscape composition, eutrophication, erosion rates See Soil Health metrics: Texture, Soil Structure, Nutrient analysis, Electrical conductivity, Earthworm abundance, biomass and diversity, Litter decomposition, Cation exchange capacity, Soil respiration, N mineralisation More complex modelling approaches have been used to couple Net Primary Productivity and stoichiometry of leaves, wood, and fine root tissue for quantification of flows of nutrients (Malhi et al. 2021).
Patch size distribution	No	Data collected during assessment of Landscape diversity can be used to derive additional informative habitat-based metrics: • % of area in large patches, defined based on target species' habitat needs and project scale, for example minimum patch size needed for species persistence • Size class distribution; area brackets will be defined based on project scale (e.g. White et al. 2017, 42,000 ha forest landscape had patch brackets: >10, 10-50, 50-100, 100-500, >500 ha) • Mean habitat patch size per 1 km square (Maskell et al. 2019) Patch-level metrics can be calculated using FRAGSTATS. More information on interpretation and selection of metrics available here: https://fragstats.org/index.php/background/patches-patchiness-levels-of-landscape-metrics
Population fluctuations	No	Two of the most accepted variability metrics are (Heath 2006): • Standard deviation of log transformed abundances = sd[log(N)] • Coefficient of variation (CV) is the ratio of the standard deviation to the mean of abundance per year; CV = (sd of abundance)/(mean of abundance) (Franzen et al. 2013) Sampling design and grain (resolution) and extent (scope) of sampling determine the scale at which a dataset is relevant. Data collected at the site level can give an idea of population fluctuations for a species

		within a site only if it is reasonable to expect that reproductive and foraging activities are strongly influenced by the site. (Heath 2006) proposes a new metric - population variability (PV) - which quantifies the variability of the average percent difference between all combinations of observed abundances. PV is expected to be more accurate at estimating long term variability even when data is collected on short timescales. See metrics on Relative/absolute abundance, Vegetation biomass, Mammal biomass, Invertebrate biomass for methods on estimating species abundance, these can be tracked over multiple years to monitor population trends.
Proportions endemic, exotic, threatened species	No	In projects with a specific species of interest (of conservation concern or exotic), approaches to species monitoring discussed elsewhere can be used (see Invertebrate biomass, Mammal biomass, Vegetation biomass, Species diversity for methodologies). However, additional consideration at the design stage will be needed to account for issues such as low detectability of rare species. Considerations will be species-specific but should broadly cover: • The variability or dynamics of the species/system • Location of monitoring sites to capture species with low and variable occupancy • Quantity of data required to estimate change for low abundance species • Type of data required to estimate population change Species-level data from other surveys can be classified using data on rarity/conservation status for UK species, however this approach may underrepresent trends in species of conservation concern (Robinson et al. 2018). Data sources available that can be used to classify the conservation status and rarity of species in the UK: • Priority species for England (https://www.gov.uk/government/publications/habitats-and-species-of-principal-importance-in-england) • Priority species for Scotland (https://www.nature.scot/scotlands-biodiversity/scottish-biodiversity-strategy-and-cop15/scottish-biodiversity-list)

		• Priority species for Wales (https://www.biodiversitywales.org.uk/Section-7) • Priority species for Northern Ireland (https://www.daera-ni.gov.uk/articles/northern-ireland-priority-species) • The JNCC lists further species-level conservation classifications for UK taxa such as Nationally Scarce/Rare species, Birds of Conservation Concern (Red/Amber list), National Red lists (based on IUCN criteria): https://hub.jncc.gov.uk/assets/478f7160-967b-4366-acdf-8941fd33850b

Table S4 – Summary of standardised methodologies available for collecting data for each soil health metric. The methodology available column indicates whether a full methodology is available (Yes), a methodology covering part of the process from data collection to metric calculation is available (Partial), or no methodology is available (No). The methodology summary column summarises the data collection and metric calculation process.

Metric	Methodology available	Methodology summary
Bulk Density	Yes	The FAO Soil Doctor guide (FAO 2020) provides a standardized methodology for soil bulk density: • Intact soil cores are collected and the known volume of soil in the cores is dried at 105 °C and weighed. • This method uses a metal core of a known volume, which represents the volume of the soil for the purpose of the calculation of the soil bulk density. The core should be of cylindrical shape to allow for easy determination of its volume. • The core sample is pushed into the soil to the desired depth and then gently removed without altering the contents of the core. After obtaining the soil using the core, the soil weight is measured, and using the known volume of the core (an estimation of soil volume), soil bulk density can be determined (FAO 2020). Detailed methodology on the core method and calculation of soil bulk density can be found at Soil Testing Methods from FAO pg. 16-19- https://www.fao.org/3/ca2796en/CA2796EN.pdf Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf (Farm Carbon Toolkit 2021)
Soil Organic Carbon	Yes	To determine soil organic carbon by loss of ignition: • A sample is dried at 105°C to remove all water. • The sample is weighed. • The sample is then ashed (for weight loss on ignition) for two hours at 500°C, and the percent of mass lost is calculated after weighing again. • The % loss on ignition (LOI) is converted to % organic matter (OM) using the following equation: % OM = (% LOI *

		0.7) - 0.23 (Moebius-Clune et al. 2016) • In the LOI method some laboratories can use different ignition temperature, so it is recommended to use the same laboratory and keep record of the temperature used by the laboratory. Detailed information on how to monitor soil organic carbon can be found at: https://farmcarbontoolkit.org.uk/2021/11/11/new-guide-on-monitoring-soil-carbon/ (Farm Carbon Toolkit 2021). The guide includes: • How to sample within fields? • At what depths should samples be taken? • How often should you repeat your sampling? • How to collect and prepare your samples? • What are the options for the lab analyses?
Texture (Silt, Clay and Sand)	Partial	The FAO Soil Doctor guide (FAO 2020) provides a standardized methodology for determining soil texture by hand on site: The ribbon method: Take a sample of soil and remove the > 2 mm fraction (gravel — see below, roots, organic material) by sieving or by hand. The sample should be sufficient to fit comfortably into the palm of your hand. Moisten the soil with a little water and knead it into a ball. Continue to work the ball, adding more soil and water if necessary, until the soil no longer sticks to your fingers and there is no apparent change in plasticity (usually 1 – 2 minutes). Using a clean, moistened hand, place the ball between your thumb and forefinger and slide your thumb across the soil (shearing) to extrude a ribbon. Soils with high clay content are further categorised by moulding the bolus into rods. If the rods fracture the soil is assigned a texture grade lighter than a medium clay. This method has been adapted from (McDonald & Isbell 2009). Gravel (particles > 2 mm) is removed from the soil prior to texturing because it does not contribute to chemical and some physical properties of soils.

		Detailed methodology on how to measure soil texture on site can be found at Soil Testing Methods from FAO pg. 4-15 https://www.fao.org/3/ca2796en/CA2796EN.pdf Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf Note: To know the exact percentages of sand, clay and silt it is necessary to send the sample to a laboratory.
Soil Moisture	Partial	The oven-dry method is the standard laboratory method for determining the water content of soil. The FAO Soil Doctor guide provides a standardized methodology for soil moisture, for calculating the gravimetric water content with high accuracy: • Using the normal procedure for soil sampling, obtain a soil sample from a desired depth. • Place the soil samples in sealable sample paper bags, and label the samples • Measure the weight of the field wet soil • Oven dry the soil for about 24 hours at 105 degrees Celsius or air dry the soil for 7 days • After drying, weigh the soil in order to obtain the dry weight of that soil • The difference between the wet and the dry weight is the amount of water in the soil Detailed methodology on determining soil moisture can be found at Soil Testing Methods from FAO (FAO 2020) pg. 24-34 -https://www.fao.org/3/ca2796en/CA2796EN.pdf Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit (Farm Carbon Toolkit 2021): https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Porosity	Yes	Porosity can easily be calculated if the bulk density and the particle density of the soil are known with no need for extra sampling, with the following formula: Porosity %=(1-(Bulk density)/(Particle density))×100

		Particle density of mineral soils is relatively consistent and is often considered to be around 2.65 g/cm^3 If particle density of organic soils is unknown, to calculate the bulk density you can use a range from 0.1 to 0.3 g/cm^3 To do a visual assessment of porosity you can refer to FAO Soil Doctor guide (FAO 2020) pg. 72-73 - https://www.fao.org/3/ca2796en/CA2796EN.pdf
Soil Structure	Yes	Soil structure can be measured through Visual Evaluation of Soil Structure (VESS): Dig out one spade-sized block of soil (depth approximately 30 cm) Cut down on three sides and then lever the block out, leaving one side undisturbed If the soil block falls apart easily, dig out one block and then a second next to it to assess Lay the soil block on a plastic sheet or tray If the structure is uniform, assess the block as a whole If there are two or more horizontal layers of differing structure, identify the layer with the poorest structure (the limiting layer) Record the depth of this limiting layer and carry out the rest of the assessment on this layer Assign a score following instructions on the below leaflet from AHDB A detailed guide on visual evaluation of soil structure on site can be found at the AHDB website https://ahdb.org.uk/knowledge-library/how-to-assess-soil-structure
pH	Yes	Soil pH is measured on a scale of 0 to 14, with pH level below 7 being acidic while pH level above seven is alkaline (or basic). A pH of 7 is considered neutral (neither acidic nor alkaline) (FAO 2020). Soil pH is usually measured by glass electrode in a slurry of 1 part by weight of soil to 2.5 parts water. Measure soil pH in a weak salt (0.01–0.1 M) solution of calcium or potassium chloride, rather than just water, to reduce the influence of varying electrolyte concentration because it is then small relative to the total salt concentration in solution. The use of CaCl2 has been recommended for a range of practical and interpretive reasons (Merrington et al. 2006; Schofield & Taylor 1955).

		Normally pH measurements are conducted in a laboratory, but they could also be measured on-site if necessary equipment is available. Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Nutrient Analysis (P,K,N)	Partial	Nutrient analysis should be carried out by a laboratory as it requires a certain level of expertise. The following methods are the most commonly used for measuring nitrogen: Dumas combustion method Kjeldahl method For more information on the Kjeldahl method follow this link https://www.fao.org/3/cb3642en/cb3642en.pdf For more information on the Dumas combustion method follow this link :https://www.fao.org/3/cb3646en/cb3646en.pdf The Olsen P method is a widely used technique for measuring available phosphorus (P) in soil. For more information on Olsen P method follow https://www.fao.org/publications/card/en/c/CB3644EN/ Potassium can be extracted from soil using ammonium nitrate. For more information on the procedure follow the link https://www.geog.cam.ac.uk/facilities/laboratories/techniques/potassium/#:~:text=Standardisation%20is%20unnecessary.-,Method,shaking%20machine%20for%2030%20mins. Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Electrical Conductivity	Partial	**Handheld EC meter:** A handheld EC meter is a simple and easy-to-use tool that measures the EC value of soil extract. The meter is equipped with a probe that is inserted into the soil extract, and the reading is displayed on a

		digital screen. **Laboratory analysis:** Soil samples can be sent to a laboratory for analysis, where the EC value is measured using a conductivity meter. This method provides accurate results, but it can be time-consuming and expensive. **Soluble salt test strips:** Soluble salt test strips are paper strips that are dipped into a soil extract. The color of the strip changes based on the EC value of the soil extract, and the result can be compared to a color chart to determine the EC value. **Electrical resistivity imaging (ERI):** ERI is a geophysical method that uses electrodes to measure the electrical resistivity of soil. By measuring the resistivity, the EC value of the soil can be estimated. **Soil moisture sensor:** Soil moisture sensors are commonly used in smart agriculture to help farmers in real-time and remotely monitor soil EC and ensure optimal plant growth. (taken from https://www.seeedstudio.com/blog/2022/07/15/soil-electrical-conductivity/)
Earthworm abundance, biomass and diversity	Yes	Step-by-step guide from the AHDB knowledge library: • Dig out a soil pit (20cm x 20cm x 20cm) and place soil on mat • Hand-sort the soil, placing each whole earthworm into the pot • Count and record the total number of earthworms • Separate earthworms into adults and juveniles • Return juveniles to the soil pit • Count and record the number of each type of adult earthworm • Return earthworms to the soil pit and backfill with soil • Repeat steps 1–7, until 10 soil pits per field have been assessed Detailed information on how to count earthworms, with photographs and a video can be found following this link: https://ahdb.org.uk/knowledge-library/how-to-count-earthworms

		Detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Litter Decomposition (Litter bags)	Yes	The approach involves measuring the reduction in mass of a specific organic material. A predetermined mass of organic material (like a tea bag) is buried in the field for a specified duration and later dug up to calculate how much weight is lost during the field incubation. Detailed methodology on measuring litter decomposition using teabags can be found at Soil Testing Methods from FAO pg. 24-34 -https://www.fao.org/3/ca2796en/CA2796EN.pdf
Aggregate Stability	No?	The easiest way to measure aggregate stability on-site is to use the Emerson test: • Select 3 air-dry aggregates, 5–10 mm diameter. • Place 75 mL deionised water in a container. Place the 3 aggregates in the container of water, spaced equally around the side. Do not stir or otherwise disturb. • Record the time placed in the water. After 2 hours and 20 hours, assess aggregate behaviour according to the following: - Record whether the aggregate has disintegrated. If there has been no disintegration, record if there has been any swelling of the aggregate. - If the aggregate has dispersed, note the degree of dispersion. For more detailed information watch the following video and read the following instructions from the New South Wales Government: https://youtu.be/xuXFQDzXNQU?si=0Lmosq1UTjwOwW1E https://www.environment.nsw.gov.au/resources/soils/testmethods/eat.pdf
Infiltration	No	The drainpipe test is a cheap and simple way to measure the infiltration of water into soil: -Drive the pipe halfway into the ground using a hammer, to leave 10 cm standing above the ground. -Pour in water (approximately 800 ml) to a depth of 10 cm.

		-Start the stopwatch immediately and measure the time taken for the water to drain into the soil. -Repeat at several locations. For detailed method and a video tutorial please follow this guide from AHDB: https://ahdb.org.uk/knowledge-library/water-infiltration-test
Cation exchange capacity (CEC)	Partial	Different testing facilities employ diverse techniques to assess Cation Exchange Capacity (CEC), and outcomes may vary based on the specific soil fraction subjected to measurement. CEC is conventionally expressed in meq/100 g. The determination of cation exchange capacity (CEC) and exchangeable bases (calcium, magnesium, potassium and sodium) in soil using 1N ammonium acetate at pH 7 is used as the standard methodology by FAO, although different methods can be used depending on the different soil types. For detailed instructions see: https://www.fao.org/3/cc1200en/cc1200en.pdf To collect samples for sending to a laboratory, detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Soil respiration	Yes?	There are different methods to measure soil respiration. It can be measured by capturing and quantifying carbon dioxide (CO_2) released from a re-wetted sample of air-dried soil held in an airtight jar for 4 days. Greater CO_2 release is indicative of a larger, more active soil microbial community (Moebius-Clune et al. 2016). It can also be measured by determining oxygen consumption (Bispo et al. 2009). Some commercial laboratories offer the CO_2 burst test and can provide guidance. For these tests, soil is dried and re-wetted to measure a burst of biological activity. One of the most commonly used is the Solvita test (Soil Association). For a step-by-step guide on the Solvita test follow the link to Soil Association on page 12. https://www.soilassociation.org/media/8179/martinwood.pdf See also the detailed protocol from the Cornell Soil Health Laboratory Comprehensive Assessment of Soil Health (Moebius-Clune et al. 2016): https://cpb-us-e1.wpmucdn.com/blogs.cornell.edu/dist/f/5772/files/2015/03/CASH-

		Standard-Operating-Procedures-030217final-u8hmwf.pdf To collect samples for sending to a laboratory, follow detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf Test it yourself: DIY basal soil respiration tests kits are available where soil is tested in its natural state. It takes just 24 hrs to complete with minimal equipment. Soil should be moist but not waterlogged, and at a temperature of 18-24ºC, when carrying out the test. Follow the link for more information: https://ahdb.org.uk/knowledge-library/soil-respiration Follow this link from the soil association for a video tutorial: https://youtu.be/BqNCWJAodBI?si=BFHa4nN7YagnVEJ8
N mineralization	No	To measure N-mineralization soil samples must be sent to a laboratory. N-mineralization is measured by the accumulation of NH4 + in soil slurry under aerobic conditions over a period of several weeks. Anaerobic incubation is sometimes preferred because there is less microbial immobilisation under anaerobic conditions and nitrification is inhibited (Nielsen & Winding 2002). Systematic sampling ensures that the entire site being sampled is well represented by the individual samples. The analytical variability between laboratories can be controlled by analysing all samples by one specific method within one specific laboratory as done in the Dutch Soil Monitoring Programme (Nielsen & Winding 2002). To collect samples for sending to a laboratory, detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Nematodes	Yes?	• Specialist required for morphological identification. It is expected that 2 samples can be analysed per day. • Recent advances in rapid DNA sequencing (metabarcoding) could provide a practical and affordable way for

		laboratories to determine the composition of the soil nematode community (Bongiorno 2020). For more information on methods on nematode community analysis follow: https://ahdb.org.uk/knowledge-library/nematodes-as-biological-indicators-of-soil-health Soil samples can be sent to a laboratory for extraction, counts and identification into taxonomic groups or functional groups based on their characteristics, behaviour, and ecological roles. To collect samples for sending to a laboratory, detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf
Genetic, Functional and Structural diversity of Bacteria	Partial	Genetic diversity in bacteria is commonly explored through the analysis of 16S rDNA genes, which are universally present and exhibit variations in base composition among species (Nielsen & Winding 2002). To assess structural diversity, phospholipid fatty acids (PLFAs), stable components of microbial cell walls, are extracted from soil samples and subsequently analysed by gas chromatography. This method enables the identification and quantification of specific PLFAs, providing sensitive and reproducible measurements. It characterizes the numerically dominant segment of soil microbial communities without the need for cultivation (Nielsen & Winding 2002). Functional diversity is evaluated by examining carbon utilization patterns using the BIOLOGTM assay. In this process, a soil extract is incubated with up to 95 different carbon sources in a microtiter plate, and microbial activity is indicated by a redox dye. Specifically selected sets of carbon sources are employed to study soil microbial communities. The assay yields a qualitative physiological profile of potential functions within the microbial community, and variations in these profiles can be analysed using multivariate statistics (Nielsen & Winding 2002). Systematic sampling ensures that the entire site being sampled is well represented by the individual samples.

		The analytical variability between laboratories can be controlled by analysing all samples by one specific method within one specific laboratory as done in the Dutch Soil Monitoring Programme (Nielsen & Winding 2002).
Fungal Biomass	Partial	Methods to measure fungal biomass include: • Phospholipid Fatty Acid (PLFA) Analysis: This method identifies and quantifies specific fatty acids associated with fungal membranes (Nielsen & Winding 2002; Bünemann et al. 2018). • Ergosterol Analysis: Ergosterol is a sterol found in fungal cell membranes, and its concentration can be used as an indicator of fungal biomass (Nielsen & Winding 2002; Bünemann et al. 2018). • Quantitative PCR (qPCR): This molecular biology technique quantifies fungal DNA in soil samples (Bünemann et al. 2018).
Enzyme activities (Soil enzymes)	No	The three most common enzymes for use are— • N-acetyl-β-D-glucosaminidase (NAG) that is involved in both the C- and N-Cycle • Phosphomonoesterases (acid/alkaline phosphatase; Pase) is involved in the P-cycle • Arylsulfatase (AS) that is involved in the S-cycle See appendix 4 of Stott 2019, Recommended Soil Health Indicators and Associated Laboratory Procedures, for detailed procedure.
Collembola	Yes	Described by ISO: Global standards for trusted goods and services: Soil samples are collected in the field using a split corer. Soil cores are placed in plastic tubes (or plastic bags) and transported to the laboratory. Afterwards, Collembola and Acarida are rapidly (within a few days) extracted by behavioural methods, using a MacFadyen apparatus, and preserved for future identification. Preparation techniques are also described. Finally, abundance values can be recalculated related to area (usually 1 m2), volume or weight (usually 1 kg) ISO 23611-2:2006 - Soil quality — Sampling of soil invertebrates. https://www.iso.org/obp/ui/en/#iso:std:37027:en
Mycorrhiza	Partial	Soil samples need to be sent to a laboratory, where they can perform DNA extraction, PCR, and sequencing to identify mycorrhiza (Gorzelak et al. 2012). To maintain an equitable portrayal of soil fungi, it is crucial to minimize the inclusion of root fragments in soil samples. Failure to do so may lead to an overestimation of the impact of fungi associated with a prevalent plant species in your ecosystem (Gorzelak et al. 2012).

		To collect samples for sending to a laboratory, detailed information on soil sampling such as where to take the samples from, how many, the best time to sample, and depth of sampling, can be found at the Farm Carbon Toolkit: https://farmcarbontoolkit.org.uk/wp-content/uploads/2021/09/Soil-Sampling_-What-To-Expect.pdf

Table S5 – A summary of data thresholds or directions of change in each metric indicating a positive outcome for biodiversity.

Metric	Metric threshold or direction of change
Dominance-diversity curves	Targets on changes in relationship between species evenness and abundance of specific species will depend on the project aims (e.g. dominance relationships between target species for different habitats).
Functional trait diversity	Functional trait objectives will depend on the project aims, for example see summary on linking plant functional traits to ecosystem services in Methodology summary.
Habitat area	Projects will aim to increase the total area of semi-natural habitat and increase the area of key relevant habitats. Information on the core habitat size needed to sustain breeding populations of key species can be used to set targets.
Identity	See **Functional trait diversity**
Landscape diversity	An increase in Simpson's diversity index indicates an increase in landscape diversity. The desired landscape diversity will depend on the project aims - heterogeneous mosaics vs aims to restore, create or manage a specific habitat type. Landscape diversity can be measured alongside **Habitat area** and **Patch size distribution** to capture the need for large patches of core habitat alongside diversity.
Relative abundance	Targets on changes in abundance of specific species will depend on the project aims (e.g. target species that are characteristic of a certain habitat).
Similarity	Changes in species assemblage can be tracked over time, with project specific targets based on assemblages associated with target habitats. Comparisons can also be made to nearby sites with the target habitat type.
Species diversity (alpha, beta, gamma)	Generally increases in species diversity could be seen as positive, however the species composition underlying change in these metrics should be considered and increases in target species (e.g. species characteristic of target habitats) are important. Where data has been collected following standardised UK-wide methods (e.g. UK Breeding Bird Survey), species-level trends over time can be compared to national trends.

	Increases due to the presence of invasive or non-native species may not be desirable.
Vegetation structure	Higher structural diversity is usually associated with higher biodiversity.
Deadwood volume	The UK National Forest Inventory defines three categories of standing and lying deadwood: Favourable: >= 80 m3 per ha Intermediate: >= 20 and < 80 m3 per ha Unfavourable: 0 - <20 m3 per ha Taken from: The NFI Woodland Ecological Condition Scoring Methodology https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/what-our-woodlands-and-tree-cover-outside-woodlands-are-like-today-nfi-inventory-reports-and-woodland-map-reports/nfi-woodland-ecological-condition/
Pollination	Generally, higher pollinator diversity is desirable. See notes on advanced measures of network statistics in Methodology summary.
Seedling regeneration	In forest ecosystems high diversity and abundance of native seedlings is desirable. Their transition to later growth stages will determine the long-term persistence of the forest habitat.
Tree age	A healthy forest ecosystem will have a diverse age structure, with trees represented from seedlings through to senescent trees.
Tree diversity	Generally, increases in species diversity could be seen as positive; however the species composition underlying change in these metrics should be considered. The desired tree diversity will also depend on the target woodland habitat. Increases in target species (e.g. species characteristic of target habitats) are important, whereas increases due to the presence of invasive and/or non-native species may be undesirable.
Vegetation biomass	The desired direction of change will depend on project objectives and the ecosystem type. In woody systems, particularly projects aiming to sequester carbon, an increase in vegetation biomass is often desirable. In grassland systems, increasing biomass over time may indicate nutrient enrichment and can be correlated with a loss of plant diversity.
Allelic diversity	The desired allelic diversity will be species/project specific. Generally, a higher allelic diversity is better.
Biomass (measure of abundance)	The desired direction of change will depend on project objectives and the ecosystem type.

	In woody systems, particularly projects aiming to sequester carbon, an increase in vegetation biomass is often desirable. In grassland systems, increasing vegetation biomass over time may indicate nutrient enrichment and can be correlated with a loss of plant diversity. Increasing invertebrate biomass, particularly of functionally important species is generally likely to be desirable, but in agricultural systems an increase in pest species biomass is undesirable. Changes in mammal biomass will depend on the objectives of the project, for example in woody systems it is important that numbers of herbivorous mammals don't prevent tree and shrub regeneration.
Colonisation/local extinction rates	Thresholds and direction of change will be context dependent, with desirable loss/gain of species depending on the project aims.
Connectivity and fragmentation	The ideal level of connectivity will be context and species dependent, influenced by the habitat matrix and species' dispersal ability/behaviour.
Disturbance	Thresholds not clearly defined, but disturbance is generally beneficial at an intermediate level. A desirable threshold and direction of change will be project specific.
Effective population size	The desired effective population size will be species/project specific. A higher effective population size is better, reflecting greater underlying genetic diversity.
Energy flow rates	Thresholds and direction of change will be context dependent. Future research in this area may lead to thresholds being defined.
Heterozygosity	The desired heterozygosity will be species/project specific. Generally, a higher level of heterozygosity is better.
Invertebrate biomass	Increasing invertebrate biomass, particularly of functionally important species is generally likely to be desirable, but in agricultural systems an increase in pest species biomass is undesirable.
Mammal biomass	For deer densities thresholds for impacts on habitats are given in Putman et al. 2011, but should be interpreted alongside direct monitoring of habitat impacts: For woodland regeneration - 4-5 large deer or <25 roe deer per 100 ha. Open habitats suffer light/moderate impacts at 7-8 red deer per 100 ha.
Nutrient cycling rates	Mechanisms between landscape features, biodiversity and nutrient cycles are not well understood. Thresholds for degradation are not well established (Lavelle et al. 2005).

Patch persistence/turnover	Thresholds not clearly defined, but spatial/temporal heterogeneity is generally beneficial at an intermediate level, though high levels could reflect increasing landscape fragmentation. A desirable threshold and direction of change will be project specific.
Patch size distribution	The ideal patch size distribution will be project/species specific.
Population fluctuations	Generally a persistent downward trend is undesirable.
Proportions endemic, exotic, threatened species	Generally increasing populations of endemic/threatened species are desirable and so are declining populations of exotic species with negative effects on an ecosystem.

Table S6 – A summary of data thresholds or directions of change in each metric indicating a positive outcome for soil health.

Metric	Metric threshold or direction of change
Bulk Density	In general lower bulk density is considered good for soil health as high bulk density can indicate compaction. Thresholds will depend on the type of habitat, vegetation cover, soil type. In general in the UK a low bulk density can range between 0.3-1.3 g/cm3 as benchmarked by the UKCEH (Feeney et al. 2023).
Soil Organic Carbon	In general higher soil organic carbon is considered good for soil health as it can provide nutrition for microbes and release nutrients for plant growth, as well as improve soil structure and water-holding capacities. Thresholds will depend on the type of habitat, vegetation cover and soil type. In general, in the UK it is difficult to define a specific threshold as it can range from 5 % in light coarse-textured soils in arable land to 95 % in wetlands with carbon-rich soils, as benchmarked by the UKCEH (Feeney et al. 2023).
Texture (Silt, Clay and Sand)	Not applicable
Soil Moisture	Thresholds are context (habitat, land use, etc) dependent.
Porosity	Thresholds are context (habitat, land use, vegetation type, soil type, etc) dependent. For example, well-drained soils with higher porosity may be suitable for grasslands, while wetland habitats might benefit from soils with lower porosity to retain more water.
Soil Structure	Score can be good moderate and poor.
pH	Thresholds are context (habitat, vegetation type, soil type, etc) dependent. DEFRA have produced targeted pH for arable and grassland soils for light sand soils, medium soils, deep clay soils, deep clay silty soils, organic soils (10-25% SOM) and peats (25%+ SOM) (Available at: https://www.gov.uk/government/publications/landspreading-how-to-manage-soil-health/landspreading-how-to-manage-soil-health Feeney et al. 2023 have also benchmarked pH for some habitat types and soil types in the UK.
Nutrient Analysis (P,K,N)	Nmin in agricultural soils - In agricultural soils, critical limits for total N or available N (the mineral N content), related to specific soil functions, are difficult to define. Organic layer of forest soils - Based on the relationship between N leaching and the C/N ratio in the organic layer of forests, C/N

	ratios of around 25 (between 20 and 30) are considered critical, with a very high N retention fraction and thus limited leaching risk at a C/N ratio above 30, while the N retention fraction is low and the leaching risk is high at a C/N ratio below 20; in between there is strong variation. A C/N ratio below a value of 25 is often suggested as a threshold value for increased leaching. (Table 3.2., (European Environment Agency 2023)). N Concentrations in air and water – • NH_3 in air: 1-3 mg NH3/m3 (Cape et al. 2009) • N in soil solution: leakage from forests: 1mg N/l (de Vries et al. 2007) • N in soil solution: impacts on forests: 1-5 mg N/l (de Vries et al. 2007) • NO_3 in groundwater: 50 mg NO_3/l- (WHO, 2011) • N in surface water: 1.0-2.5mg N/ (Camargo & Alonso 2006) P in agricultural soils • above a target level below which crop yield is limited (Mallarino & Blackmer 1992); • below a critical level above which P leaching and run-off is significantly enhanced (e.g., Li et al. 2011). (See Bai et al, 2013 figure from European Environment Agency 2023). Build up and maintenance approach (Li et al. 2011) • not be made in soils with available soil P levels above the change point (threshold) for P leaching. • equal the P withdrawal in harvested crops, if: o available soil P > target level for crop yield. o available soil P < critical level for P leaching. • equal the P withdrawal in harvested crop plus an additional amount of P fertiliser, to build up available soil P to the required level, if: o available soil P < critical level for crop yield (Li et al. 2011). Critical Limits for dissolved P and Soil P saturation in agricultural soils • The critical P saturation index (PSI) is mostly around 0.15, i.e. 15% (12.5-17.5%) of the concentration of (Al+Fe)ox, based on data for the Netherlands (Schoumans & Chardon 2015) and Canada (Beauchemin & Simard 1999). • The critical value is expressed as 25-35% of the P sorption capacity, which in turn is calculated as 0.5×(Al+Fe)ox for sandy soils and non calcareous clay soils. Critical limits for N/P ratio in organic layer of forest soils • N/P ratio in organic layer >18 (coniferous forests) and N/P ratio in organic layer >25 (deciduous forests).

Electrical Conductivity	New South Wales (NSW) Government - A productive soil's conductivity should be below 0.15 dS/m (decisiemens per metre).
Earthworm abundance, biomass and diversity	According to the AHDB, 16 or more earthworms as well as the presence of different ecological groups such as epigeic, endogeic and anecic, per soil pit indicates the soil is in good condition.
Litter Decomposition (Litter bags)	Not applicable
Aggregate Stability	A Mean Weight Diameter (MWD) of 2 mm is regarded as optimal for both productivity and environmental well-being. A decline in MWD below 2 mm, or even reaching 1 mm, is seen more as an environmental concern (Merrington et al. 2006). There is a consensus that the trend in aggregate stability aligns with the "more-is-better" principle (Stott 2019), as the higher proportion of stable aggregates with a MWD of around 2mm can be associated with environmental well-being.
Infiltration	In well-structured soil water moves faster down the soil profile. For soil in good health, the water should drain away within 2 to 5 minutes for light or medium soils. A heavy clay soil with poor structure could take 20 minutes or longer (AHDB, water infiltration test)
Cation exchange capacity (CEC)	Typical CEC values: CEC (meq/100g) Very low 0-10 Slightly low 11-15 Normal range 16-40 High > 40 Very low nutrient holding capacity indicating sandy soils with little or no clay or organic matter. Nutrients will be easily leached and foliar applied nutrients are strongly recommended.

	Slightly low nutrient holding capacity indicating a more loamy mineral soil. Leaching may still be a problem and therefore foliar applications should be considered. Adequate to high nutrient holding capacity indicating soils with increasing clay content. Very high level normally found in very heavy soils with a high clay content or soils with a high organic matter level. Nutrients can be bound very tightly to the soil particles and availability can be restricted. From https://www.yara.co.uk/siteassets/crop-nutrition/media/uk/uk-analytical/understanding-cation-exchange-capacity.pdf
Soil respiration	The AHDB: CO2-C benchmarks for UK mineral soils under grassland suggest a respiration rate > 180mg/kg to be soil with good microbial activity. CO2-C benchmarks for UK mineral soils under cropland suggest a respiration rate of >135 mg/kg to be indicative of good microbial activity
N mineralization	Not applicable
Nematodes	Not applicable
Genetic, Functional and Structural diversity of Bacteria	When no reference value is available, the initial measurements may be the best reference value for future measurements. Measuring soil parameters in a specific soil system over time rather than in comparison with other systems is recommended as a dynamic assessment approach (Nielsen & Winding 2002)
Fungal Biomass	Fungal biomass can vary spatially and temporally based on factors like soil type, land management practices, and environmental conditions (Bünemann et al. 2018).
Enzyme activities (Soil enzymes)	It is generally agreed in the literature that higher EAs are present in healthier soils, as they are necessary for improved nutrient cycling in the soil (thus following the more-is-better model) (Stott 2019).
Collembola	Not applicable
Mycorrhiza	The prevailing assumption is that a higher abundance and richness of mycorrhizae correlate with greater benefits for plant growth. However, it's crucial to recognize that the relationships between mycorrhizae and plants are dynamic, context-dependent, and

	span the spectrum between mutualism and parasitism. Root colonization, a key aspect of this interaction, can undergo changes over weekly time scales (Fierer et al. 2021).

Table S7 – A summary of technological innovations that could support biodiversity monitoring in the future.

Metric	**Technological innovations**
Dominance-diversity curves	• Earth Observation sensors generate high resolution imagery, which can be used to identify larger species (Lausch et al. 2016). • Advances in machine learning are increasing the efficiency of classifying and processing camera trap imagery (Tabak et al. 2019). • Sequencing methods have shown some promise in detecting invertebrate biomass, with mitogenomic sequencing achieving higher accuracy than metabarcoding. However detection is still poor for low abundance species (Bista et al. 2018).
Functional trait diversity	• Plant spectral data has shown potential to be used as a proxy for plant functional diversity (Frye et al. 2021; Lausch et al. 2016; Schweiger et al. 2018).
Habitat area	• In Switzerland, high resolution (1m) aerial imagery (land cover, Landsat, LiDAR) was used to classify habitats with high accuracy, although rarer and finer-scale habitats were less accurate, highlighting the need for ground-truthing (Price et al. 2023). • Machine-learning approaches were used to classify Natura 2000 Habitat Types in Germany (Sittaro et al. 2022). • Other examples of classifying habitat types using Earth Observation data in Lausch et al. 2016.
Identity	• Plant spectral data can provide information on the functional identity of plant species (Frye et al. 2021; Lausch et al. 2016; Schweiger et al. 2018).
Landscape diversity	• In Switzerland, high resolution (1m) aerial imagery (land cover, Landsat, LiDAR) was used to classify habitats with high accuracy, although rarer and finer-scale habitats were less accurate, highlighting the need for ground-truthing (Price et al. 2023). • Machine-learning approaches were used to classify Natura 2000 Habitat Types in Germany (Sittaro et al. 2022). • Other examples of classifying habitat types using Earth Observation data in Lausch et al. 2016.
Relative abundance	• Earth Observation sensors generate high resolution imagery, which can be used to identify larger (Lausch et al. 2016). • Advances in machine learning are increasing the efficiency of classifying and processing camera trap imagery (Tabak et al. 2019). • Sequencing methods have shown some promise in detecting invertebrate biomass, with mitogenomic sequencing

	achieving higher accuracy than metabarcoding. However detection is still poor for low abundance species (Bista et al. 2018).
Similarity	
Species diversity (alpha, beta, gamma)	• Environmental DNA (eDNA) can facilitate accurate and efficient species surveys that could replace traditional surveys. Sensitivity varies depending on the focal taxon (Fediajevaite et al. 2021; Deiner et al. 2017). • For bulk invertebrate samples metabarcoding can be used to identify all species present. The detection of all species in a sample can be improved by sorting taxa by size (Elbrecht et al. 2017).
Vegetation structure	• LiDAR can measure 3D vegetation structure and cluster analysis of lidar-derived habitat variables can be used to classify vegetation structure into different classes (Guo et al. 2017; Bradbury et al. 2005). • For understorey measures, e.g. understorey vegetation complexity or vegetation volume, terrestrial LiDAR is better (Shokirov et al. 2023). • A Canopy Height Model can be calculated by subtracting a LiDAR-derived Digital Terrain Model from a Digital Surface Model. Vegetation extent and height for scrub and trees can be derived from the Canopy Height Model (Broughton et al. 2022). Individual tree and shrub extents can also be derived using the ForestTools R package (Broughton et al. 2022). This can also be used to separate out trees and shrubs from other woody scrub extent e.g. brambles (Broughton et al. 2022). • Image texture characteristics from remote-sensed data captures vegetation structure in grassland, savannah and woodland habitats (Wood et al. 2012). • Tree height and crown area can be obtained using terrestrial and airborne laser scanning (Lines et al. 2022).
Deadwood volume	• Terrestrial laser scanning could be used to estimate deadwood volume in the future (Yrttimaa et al. 2019).
Pollination	• Radio frequency ID (insects tagged and tracked) and radar (insects tagged and tracked, large-scale weather radars) for detecting and tracking insects (Barlow & O'Neill 2020). • Automated visual and audio monitoring and classification by machine learning for invertebrate ID (Barlow & O'Neill 2020). • Vision motion software for monitoring fine-scale pollinator behaviours (Barlow & O'Neill 2020).
Seedling regeneration	

Tree age	• LiDAR can measure 3D vegetation structure and cluster analysis of lidar-derived habitat variables can be used to classify vegetation structure into different classes (Guo et al. 2017; Bradbury et al. 2005).
Tree diversity	•Hyperspectral imagery and LiDAR data collected using UAVs and machine learning algorithms have shown promise for identifying tree species. However, these models have been developed and parameterised at specific sites (Zhong et al. 2022; Onishi & Ise 2021). • 3D laser scanning has also been combined with deep learning for tree species ID, however there are still challenges to this approach (Seidel et al. 2021; Lines et al. 2022).
Vegetation biomass	• Most remote-sensing methods for estimating vegetation biomass focus on forest ecosystems, but there are also some examples of plant biomass estimation in wetlands (Lausch et al. 2016). • LiDAR sensors on airplanes or UAVs can be used to generate 3D canopy height models. Metrics such as canopy height, canopy cover, basal area, tree density and aboveground biomass can be derived from these (Jucker et al. 2023). • Spaceborne LiDAR data (captured by NASA GEDI) can also capture fine-scale variation in habitat 3D structure (Jucker et al. 2023). • Global equations linking tree height and crown size to aboveground biomass show promise for converting remote-sensed data to biomass (Jucker et al. 2017). • Terrestrial laser scanning addresses uncertainties in estimating aboveground biomass using allometric equations or Earth Observation methods (Demol et al. 2022; Calders et al. 2022).
Allelic diversity	• New computational approaches will bring advancements in analysing multilocus genetic data (Luikart et al. 2010). • As SNP (Single Nucleotide Polymorphism - a type of genetic marker) discovery and genotyping costs decline many species will soon have 100s of SNPs available (Wang et al. 2016).
Biomass (measure of abundance)	• LiDAR sensors on airplanes or UAVs and spaceborne LiDAR can be used to generate 3D canopy height models. Metrics such as aboveground vegetation biomass can be derived from these (Jucker et al. 2023). • Global equations linking tree height and crown size to aboveground biomass show promise for converting remote-sensed data to biomass (Jucker et al. 2017). • Terrestrial laser scanning addresses uncertainties in estimating aboveground biomass using allometric equations or Earth Observation methods (Demol et al. 2022; Calders et al. 2022). • Automated image-recognition can recognise invertebrates and the area of the specimen in the image used for biomass estimation (Ärje et al. 2020).

	• Metabarcoding of bulk invertebrate samples is less reliable for determining abundance of individuals (Bista et al. 2018). • Earth Observation sensors generate high resolution imagery, which can be used to identify large mammal species. (Lausch et al. 2016). • eDNA could provide a reliable alternative to camera trapping for identifying all species present within a project (Leempoel et al. 2020). • Advances in machine learning are increasing the efficiency of classifying and processing camera trap imagery (Tabak et al. 2019).
Colonisation/local extinction rates	
Connectivity and fragmentation	• In Switzerland, high resolution (1m) aerial imagery (land cover, Landsat, LiDAR) was used to classify habitats with high accuracy, although rarer and finer-scale habitats were less accurate, highlighting the need for ground-truthing (Price et al. 2023). • Machine-learning approaches were used to classify Natura 2000 Habitat Types in Germany (Sittaro et al. 2022). • Other examples of classifying habitat types using Earth Observation data in Lausch et al. 2016.
Disturbance	• Earth Observation can measure intermediate-scale disturbance that is challenging to capture using field plots e.g. forest disturbance (Anderson 2018; Masek et al. 2013; Wolter & White 2002). • Biological effects of inundation, fire disturbance, and pest and disease outbreaks can be captured by remote sensing (Skidmore et al. 2021).
Effective population size	• Improvements in genetic and statistical methods complement or improve on traditional population census methods. Combining genotyping 100-1000s of loci with new computational approaches will bring greatest advancement in estimating Ne (Luikart et al. 2010). • As SNP (Single Nucleotide Polymorphism - a type of genetic marker) discovery and genotyping costs decline many species will soon have 100s of SNPs available, which will improve estimation of N_e. This will be coupled to improved statistical methods and computer efficiency (Wang et al. 2016).
Energy flow rates	See **Invertebrate biomass**, **Mammal biomass**, **Vegetation biomass**.
Heterozygosity	

Invertebrate biomass	• Automated image-recognition can recognise invertebrates and the area of the specimen in the image can then be used for biomass estimation (Ärje et al. 2020). • Metabarcoding has mainly been used for species richness assessments and is currently less reliable for determining abundance of individuals (Bista et al. 2018).
Mammal biomass	• Earth Observation sensors generate high resolution imagery, which can be used to identify larger species (Lausch et al. 2016). • Advances in machine learning are increasing the efficiency of classifying and processing camera trap imagery (Tabak et al. 2019).
Nutrient cycling rates	• Remote sensing, such as hyperspectral remote sensing of canopy leaf nutrients, can contribute to monitoring nutrient cycling (Reddy 2021).
Patch persistence/turnover	• High spatial resolution (1m) aerial imagery and machine learning shows promise for mapping habitats with broadly good accuracy, but performance is lower with fine-scale, linear and rare habitats (Price et al. 2023; Sittaro et al. 2022).
Patch size distribution	• In Switzerland, high resolution (1m) aerial imagery (land cover, Landsat, LiDAR) was used to classify habitats with high accuracy, although rarer and finer-scale habitats were less accurate, highlighting the need for ground-truthing (Price et al. 2023). • Machine-learning approaches were used to classify Natura 2000 Habitat Types in Germany (Sittaro et al. 2022). • Other examples of classifying habitat types using Earth Observation data in Lausch et al. 2016.
Population fluctuations	NA
Proportions endemic, exotic, threatened species	• Threatened species monitoring can become expensive and logistically unfeasible. Advances in drones, eDNA and data-processing of camera trap imagery are promising innovations for enhancing the process (Robinson et al. 2018). • eDNA, remote sensing, chemical ecology and internet-based citizen science could facilitate early detection and monitoring of invasive species (Larson et al. 2020).

Table S8 – A summary of technological innovations that could support biodiversity monitoring in the future.

Metric	Technological innovations
Bulk Density	• Thermo–time domain reflectometry (thermo-TDR) technique for monitoring bulk density in situ is non-destructive and able to capture spatial and temporal variations. Less labour intensive than sampling methods. More expensive and less accurate than laboratory methods (Liu et al. 2008). • The radiation method is non-destructive, allowing for the measurement of soil bulk density without altering the physical structure of the soil. It can be a fast method which reduces the time and labour required for soil characterization. The measurements can be taken directly in the field, making it a practical method for on-site evaluations. Its limitations rely on the complex methodology and the accuracy of measurements decreases with soil depth, the need for an experienced operator (Al-Shammary et al. 2018). • Digital electrochemical system for measuring and recording dry and wet for soil at different soil depths and textures, remotely (Al-Shammary et al. 2018).
Soil Organic Carbon	• Near-Infrared Spectroscopy (NIRS) can be employed for soil organic carbon measurement by examining the relationship between soil spectral reflectance and the spectral characteristics of soil organic matter. This approach has demonstrated efficiency and cost-effectiveness. Calibration of NIRS models necessitates the utilization of a well-representative dataset of soil samples, accompanied by corresponding laboratory measurements. The calibration process, while effective, can be time-consuming and demands expertise to ensure precise predictions. NIR models developed for a specific region may not be universally applicable to other regions due to variations in soil properties, climate, and vegetation (Long et al. 2023).
Texture (Silt, Clay and Sand)	• NA
Soil Moisture	• The cosmic-ray neutron sensing (CRNS) is a promising technique for estimating soil moisture at an intermediate scale in a passive and non-invasive way (Li et al. 2024). • The development of smart soil sensors using IoT (Internet-of-Things)-based systems for analysing and monitoring soil nutrients in agriculture is a promising tool for monitoring soil health. These systems utilize a network of digital sensors that can provide real-time measurements. They are capable of quickly determining the nutrient content of the soil, including nitrogen, phosphorus, and potassium levels. The network of sensors can provide real-time measurements of other soil properties such as moisture, pH, electrical

	conductivity, and other plant properties at the same time as providing a NPK (nitrogen, phosphorus, and potassium) content in the soil (Pyingkodi et al. 2022; Ramson et al. 2021; Soetedjo & Hendriarianti 2023).
Porosity	• Electrical Resistivity Tomography (ERT) can measure how much space there is between soil particles by looking at how easily an electrical signal passes through them. The size of the particles affects this, and ERT helps us understand soil structure without digging. ERT is a promising technology for assessing soil porosity, soil structure, water movement, and overall soil health. Advances in instrumentation and data interpretation methods may further enhance its applicability. At present the accuracy of ERT results can be affected by the presence of metallic objects or infrastructure in the subsurface, which can cause distortions in the electrical signal, it has limited resolution at greater depths. ERT is time-consuming, as it requires the collection of multiple measurements along a survey line or grid, which can limit its applicability for large-scale investigations (Abd Malik et al. 2023; Pereira et al. 2023).
Soil Structure	• Electrical Resistivity Tomography (ERT) can measure how much space there is between soil particles by looking at how easily an electrical signal passes through them. The size of the particles affects this, and ERT helps us understand soil structure without digging. ERT is a promising technology for assessing soil porosity, soil structure, water movement, and overall soil health. Advances in instrumentation and data interpretation methods may further enhance its applicability. At present the accuracy of ERT results can be affected by the presence of metallic objects or infrastructure in the subsurface, which can cause distortions in the electrical signal, it has limited resolution at greater depths. ERT is time-consuming, as it requires the collection of multiple measurements along a survey line or grid, which can limit its applicability for large-scale investigations (Abd Malik et al. 2023; Pereira et al. 2023).
pH	• The development of smart soil sensors using IoT (Internet-of-Things)-based systems for analysing and monitoring soil nutrients in agriculture is a promising tool for monitoring soil health. These systems utilize a network of digital sensors that can provide real-time measurements. They are capable of quickly determining the nutrient content of the soil, including nitrogen, phosphorus, and potassium levels. The network of sensors can provide real-time measurements of other soil properties such as moisture, pH, electrical conductivity, and other plant properties at the same time as providing a NPK (nitrogen, phosphorus, and potassium) content in the soil (Pyingkodi et al. 2022; Ramson et al. 2021; Soetedjo & Hendriarianti 2023). • Conductimetric pH sensor – A standard conductimetric sensor consists of two identical electrodes, between which a sensing layer is deposited (Kumar et al. 2015; Khan et al. 2017). Also utilises polymers to help gathering of pH data.

	• Ion selective field effect transistor (ISFET) – ISFET is a modification of the normal field effect transistor used in many amplifier circuits. In the ISFET, the metal gate, which is normally used as input, is replaced by an ion-sensitive membrane, the measured solution, and a reference electrode Thus, an ISFET combines in one device a sensing surface and a signal amplifier which produces a high current, low impedance output and allows the use of connecting cables without excessive shielding (Kumar et al. 2015; Khan et al. 2017; Yin et al. 2021).
Nutrient Analysis (P,K,N)	• The development of smart soil sensors using IoT (Internet-of-Things)-based systems for analysing and monitoring soil nutrients in agriculture is a promising tool for monitoring soil health. These systems utilize a network of digital sensors that can provide real-time measurements. They are capable of quickly determining the nutrient content of the soil, including nitrogen, phosphorus, and potassium levels. The network of sensors can provide real-time measurements of other soil properties such as moisture, pH, electrical conductivity, and other plant properties at the same time as providing a NPK (nitrogen, phosphorus, and potassium) content in the soil (Pyingkodi et al. 2022; Ramson et al. 2021; Soetedjo & Hendriarianti 2023). • Ion selective field effect transistor (ISFET) – ISFET is a modification of the normal field effect transistor used in many amplifier circuits. In the ISFET, the metal gate, which is normally used as input, is replaced by an ion-sensitive membrane, the measured solution, and a reference electrode Thus, an ISFET combines in one device a sensing surface and a signal amplifier which produces a high current, low impedance output and allows the use of connecting cables without excessive shielding (Kumar et al. 2015; Khan et al. 2017; Yin et al. 2021). • Conductimetric pH sensor – A standard conductimetric sensor consists of two identical electrodes, between which a sensing layer is deposited (Kumar et al. 2015; Khan et al. 2017). Also utilises polymers to help gathering of pH data.
Electrical Conductivity	• The development of smart soil sensors using IoT (Internet-of-Things)-based systems for analysing and monitoring soil nutrients in agriculture is a promising tool for monitoring soil health. These systems utilize a network of digital sensors that can provide real-time measurements. They are capable of quickly determining the nutrient content of the soil, including nitrogen, phosphorus, and potassium levels. The network of sensors can provide real-time measurements of other soil properties such as moisture, pH, electrical conductivity, and other plant properties at the same time as providing a NPK (nitrogen, phosphorus, and potassium) content in the soil (Pyingkodi et al. 2022; Ramson et al. 2021; Soetedjo & Hendriarianti 2023).
Earthworm abundance, biomass and diversity	• Proximal soil sensing could be used in the future as it has been correlated with earthworm abundances in agricultural soils and could be used for modelling earthworm abundances at the field scale. It has the advantages of a rapid and large-scale data collection, allowing for repeated measurements over time without

	disrupting the soil ecosystem, reducing the need for labour-intensive manual sampling and sorting (Lardo et al. 2012; Schirrmann et al. 2016). • Future work is being done on image analysis methods, although is still in developmental stages. • Paper by Phillips et al. 2021 showcases current landscape of global earthworm analysis. • Ecological acoustic survey methods, also known as "ecoacoustics," have gained popularity as a non-invasive and efficient method for collecting biodiversity data (Robinson et al. 2023).
Litter Decomposition (Litter bags)	• NA
Aggregate Stability	• Laser diffraction method shows promising agreement with the wet sieving method, whilst having a faster processing time and lower variability of results (Gyawali & Stewart 2019; Kubát et al. 2022).
Infiltration	• NA
Cation exchange capacity (CEC)	• NA
Soil respiration	• Soil microbial fuel cell (SMFC) is an innovative device initially created to produce electricity by harnessing the power of microorganisms from organic matter in soil. SMFC-based biosensors are emerging as a promising avenue for conducting real-time and swift monitoring of soil respiration to quantify microbial activity. In contrast to traditional biosensors, SMFC-based biosensors offer distinct benefits, including cost-effectiveness, straightforward design, in-situ capability, and sustained self-powering for long-term monitoring. These features make SMFC-based biosensors an appealing choice for extended, on-site assessments of soil (Abbas et al. 2022; Jin et al. 2020).
N mineralization	• NA
Nematodes	• Recent advances in rapid DNA sequencing (metabarcoding) could provide a practical and affordable way for laboratories to determine the composition of the soil nematode community (Bongiorno 2020; Kawanobe et al. 2021).
Genetic, Functional and Structural diversity of Bacteria	• Environmental DNA metabarcoding from soil samples can be used to identify bacteria present in the soil, allowing diversity metrics to be derived (Guerrieri et al. 2021). • Substrate-utilisation systems e.g. Biolog-ECO microplates can provide proxies for bacterial functional diversity, by generating profiles of soil microbe utilisation of different carbon sources (Yang et al. 2013).
Fungal Biomass	• NA

Enzyme activities (Soil enzymes)	• The cosmic-ray neutron sensing (CRNS) is a promising technique for estimating soil moisture at an intermediate scale in a passive and non-invasive way (Li et al. 2024).
Collembola	• DNA sequencing (metabarcoding) allows for rapid and high-throughput species identification from environmental DNA, bypassing the need for time-consuming morphological identification, especially in species-rich ecosystems (Basset et al. 2022).
Mycorrhiza	• Advances in molecular biology, such as DNA metabarcoding and relative qPCR, allow for the identification and quantification of mycorrhizal fungi in soil samples (Bodenhausen et al. 2021).

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