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The development and use of soil quality indicators for assessing the role of soil in environmental interactions

Science Report SC030265

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Lead Author:
Graham Merrington

Contributing Authors:
Sam Fishwick, Environment Agency
Declan Barraclough, Environment Agency
Jane Morris, Environment Agency
Neil Preedy, Environment Agency
Tatiana Boucard, Environment Agency
Malcolm Reeve, Land Research Associates Ltd
Pete Smith, University of Aberdeen
Changming Fang, University of Aberdeen

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Graham Merrington, Evenlode House, Wallingford, Oxon.

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Steve Killeen

Steve Killeen

Head of Science

Executive Summary

Healthy soils are vital to a sustainable environment. They store carbon, produce food and timber, filter water and support wildlife and the urban and rural landscapes. They also preserve records of ecological and cultural past. However, there are increasing signs that the condition of soils has been neglected and that soil loss and damage may not be recoverable.

In 2004, the Environment Agency published a report on the state of soils in England and Wales. The report found that there was not enough good quality information on these soils. Without this information, it is not possible to develop effective policies to protect and improve soil quality. Hence the need for tools to determine the state of our national soils and develop ways of monitoring them.

Selecting indicators for soil quality monitoring is a potentially onerous task. Indicators need to be meaningful and easy to interpret with respect to the state of soils under a range of potential pressures. Furthermore, an indicator needs to respond directly and promptly to perturbation; any change it reflects should be understandable in terms of the desired condition for that soil and/or land use. Other criteria when selecting indicators includes potential costs, the need to use existing data, practicality and simplicity.

This project builds on the findings of the previous Environment Agency-led project, *Identification and development of a set of national indicators for soil quality* (Loveland *et al*., 2002). The 2002 project concluded that soil quality indicators (SQIs) should be based on soil function, with the most important functions being those of environmental interaction, food and fibre production, building platform, support for ecological habitat and biodiversity, the provision of raw materials and protection of cultural heritage. From a choice of 67, the report identified nine key functional soil quality indicators.

In order to further these findings, a consortium was set up with members from the Environment Agency, Countryside Council for Wales (CCW), Department for Environment, Food and Rural Affairs (Defra), Forestry Commission, Scottish Executive, English Heritage, National Assembly of Wales, Scottish National Heritage and the Scottish Environmental Protection Agency. Through the auspices of the consortium the function-based approach was adopted and the members were charged with leading on specific soil functions which fell within their remit. Defra was chosen to lead on food and fibre production and English Heritage on protection of cultural heritage. The Environment Agency was charged with establishing SQIs for the function of environmental interaction which could then be used for national soil monitoring. Examples of important soil interactions with the environment include the filtering of groundwater and carbon gas exchanges with the atmosphere.

This report assessed indicators identified in the first project, and others not yet identified, in terms of their relevance to, sensitivity to and discrimination of soil quality. The report's objectives were to:

- Develop a methodology for challenging the indicators, which is transparent, auditable and through which uncertainties and expert judgement can be incorporated.
- Establish procedures for using the indicators depending on different requirements for information, for example for policy questions at a national scale or more localised, contextualised issues of concern.
- Test the selected indicators using medium to long-term scenarios, such as climate or land use change, using extrapolated datasets to establish a minimum dataset (MDS) - the minimum number of indicators needed for policy decisions and national-scale assessments.
- Select trigger values or workable ranges for those indicators chosen for the MDS, above or below which change would be critical in terms of the soil's fitness for a specific use. Using these triggers, soils sampled at different time intervals could be compared and the changes interpreted.
- Gain a broad consensus across the UK soil science community of the suitability of potential indicators for soil monitoring in the UK.

The Environment Agency used a consensus-based approach to achieve these objectives, seeking consortium and expert peer review and feedback throughout the project. One outcome was the creation of a 'challenge process' which was used by our staff to draft reports on eleven potential indicators selected from the first project, published evidence from previous soil quality schemes and expert advice. The reports aimed to assess the relevance, interpretation, measurability and cost of the chosen indicators. The reports were reviewed by a group of expert peer reviewers and consortium members, where suggestions and recommended changes were then incorporated into revised drafts (Section 2.3). The reports formed the basis for discussion at a two-day technical workshop with a broader group of experts, including academics, consultants, practitioners and policy makers from the UK soil science community. This process was in part based on the work undertaken by Sparling *et al*. (2003) who had conducted a similar exercise in selecting indicators for soil quality monitoring in New Zealand.

This project established a tiered, risk-based procedure for users to select indicators for soil sampling and monitoring based on identified levels of risk, in line with the UK government's guidance for environmental risk assessment (DETR *et al.*, 2000). At the first level of this decision-based framework, the MDS provides generic information on the state of soils, identifying dominant trends and areas at risk of deterioration in function. Increasing levels of risk, identified via the exceedance of trigger values or knowledge of a risk specific to land use and soil types (for example, cropping on a specific soil type may routinely cause soil loss and affect surface water quality), guides the user through further levels of decision-making. The MDS also includes information on site characteristics, land use history, soil classification and profile description.

Key soil facets in terms of the function of environmental interaction are given in Figure 2.5, in the report along with indicators selected for the MDS by the technical workshop and an indication of their relevance. Summary reports of group findings from the workshop are given in Appendix II, along with the delegate list, delegate pack and ground rules for the meeting.

Biological indicators of soil quality that could be incorporated into an MDS are not yet well developed and are at present treated largely as research tools. A review of these tools is currently being carried out under a Defra-funded project, *SP0529: SQID: Soil quality indicators - developing biological indicators*.

Trigger values or workable ranges for each of the MDS indicators were initially established using baselines or ranges of values for soil types linked to land uses, from which significant changes could be identified. It is important to stress that such trigger values are for use within the tiered assessment and are therefore relatively conservative. Further, the values of the triggers are to be viewed in the context of interpretative information also collected at the site on soil type, land use and so on. The triggers are to be used to build up 'weight of evidence' to determine if there is a potential issue with the quality of the soil in relation to the function of environmental interaction. Some SQIs were relatively straightforward to define, such as pH for crop types and major soil groups (*cf*. MAFF, 1988). However, for semi-natural ecosystems this has been especially onerous, due primarily to the paucity of data for these systems and the relative insensitivity of the methods for deriving indicators in the MDS (e.g. Olsen P). Nevertheless, from the technical workshop trigger tables for some indicators were developed, whereas for others, such as soil organic carbon (SOC), alternative approaches were suggested including the use of reference sites or 'no increase values' under specific site conditions.

To summarise, this project established:

- a process for assessing the suitability of soil quality indicators (SQIs) to measure the soil function of environmental interaction for a national soil monitoring network
- a tiered, risk-based approach to using these indicators which caters for the broad soil interests of agencies and stakeholders;
- agreement on a minimum dataset (MDS) of soil quality indicators for the soil function of environmental interaction that is soil organic carbon, total nitrogen, Olsen P, available and total copper (Cu), nickel (Ni) and zinc (Zn), bulk density and pH;
- approaches to determine critical changes in these indicators to trigger further action
- agreement from technical representatives and the UK Soil Indicators Consortium on approaches and findings.

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1 Introduction

1.1 Why have indicators of soil quality?

Healthy soils are vital to a sustainable environment. They store carbon, produce food and timber, filter water and support wildlife and landscapes. However, there are increasing signs that the condition of soils has been neglected and that soils may not recover following loss and damage.

In 2004, the Environment Agency published a report on the state of soils in England and Wales. The report found that there was not enough good quality information on these soils. Without this information, it is not possible to design effective policies to protect and improve soil quality. Hence the need to develop tools to assess the state of UK soils and establish ways of monitoring them.

There is a considerable amount of international literature on the derivation and use of soil quality indicators (SQIs). Many of the concepts and much of the philosophical thinking behind the use and derivation of these indicators was discussed in a previous Environment Agency-led project, *Identification and development of a set of national indicators for soil quality* (Loveland *et al*., 2002). Nevertheless, a significant amount of work has been undertaken since the completion of this report.

The 2002 project, which this report builds on, recommended that SQIs should be based on soil function, with the most important functions being those of environmental interaction, food and fibre production, providing a building platform, support for ecological habitat and biodiversity, the provision of raw materials and protection of cultural heritage.

In the first phase of the 2002 project, 67 potential indicators for all the functions were identified and subsequently classified on the basis of these soil functions. Classification fell into three broad groupings:

- those for which there was significant data and experience in interpretation;
- those for which there was less data, but which could nevertheless be interpreted with a good measure of confidence;
- those for which there was little data or which represented emerging science, but which might make useful indicators if more research were undertaken.

A consortium was then set up with members from the Environment Agency, Countryside Council for Wales (CCW), Department for Environment, Food and Rural Affairs (Defra), Forestry Commission, Scottish Executive, English Heritage, National Assembly of Wales, Scottish National Heritage and the Scottish Environmental Protection Agency. The consortium adopted the function-based approach and members were charged with testing indicators that fell under their remit (e.g. 23 of the 67 concerned environmental interaction), by applying them, or the concepts behind them, to a wide range of soil and land use systems. Defra was chosen to lead on food and fibre production and English Heritage on protection of cultural heritage.

The Environment Agency was charged with establishing SQIs for the function of environmental interaction, which could then be used for national soil monitoring.

The practical use of SQIs as a quantitative tool to measure and assess soil quality in regard to a specific soil function has been undertaken in a range of contexts at a wide variety of scales (Motta *et al*., 2002; DeClerk *et al*., 2003; Kenney *et al*., 2003; Seybold *et al*., 2003). However, common to all is the use of a framework or procedure within which indicators are used. A framework sets out the purpose of the indicator and its correct interpretation in the context of soil quality, whilst also providing an auditable pathway through which soil management decisions can be made. SQIs are not just useful for assessing the state or condition of soil; they can also help shape soil and land use policies.

This project builds on approaches recommended in the 2002 report and refines the test of usefulness of potential indicators in assessing the soil function of environmental interaction.

A glossary of terms used in this report is provided in Section 5.

1.2 Workable ranges or 'trigger' values for soil quality indicators

In order for selected soil quality indicators to be of use in providing direction and momentum for setting soil policy it is imperative that the quantitative information they provide can be interpreted within the context of the driver for undertaking the soil sampling. Therefore, in order to identify an effect of land management practices or environmental stressor through the use of a SQI an interpretative framework needs to be established. Interpretation of change in an indicator and a consistent understanding of what change means, especially in relation to direction and magnitude within this framework is most often effected through the use of 'trigger values' (Arshad and Martin, 2002), indexes (MAFF, 1988), thresholds (Page-Dumroese *et al*., 2000) and target ranges (Lilburne *et al*., 2004). All offer a means of interpreting soil measurements; they may be quantitative, semi-quantitative or even qualitative (such as a description of a state or condition), but they are usually contentious (Sojka and Upchurch, 1999; *cf*. Sparling *et al*., 2003).

The contentious nature of triggers lies in the fact that values vary with soil type, land use and whether production or environmental considerations dominate. Further, it was clearly identified from the first phase of this work (Loveland *et al*., 2002) and by others (Gough and Marrs, 1990; Sparling *et al*., 2003a; Stevenson, 2004) that there is a lack of scientific information on likely trigger values for indicators for the soil function of environmental interaction. Indicators of agricultural status of soils are almost exclusively chemical and often have well-established trigger values that reflect a soils state in regard to production. The application of such triggers to soils not under intensive agriculture, such as those under semi-natural habitats, which have often developed in conditions of low nutrient availability, is clearly inappropriate (Goodwin *et al*., 1998; Walker *et al*., 2001).

Furthermore, trigger values established for intensively-farmed soils may not be relevant for other types of production systems. In a review of soil quality for UK forestry, Moffat (2003) suggested that the interpretation of traditional agricultural indicators is particularly difficult within a forestry context as their relationships to ecosystem function and productivity is not always know. Moffat further highlighted the very limited number of soil indicators which consistently relate to tree growth, in contrast to agriculture, soil fertility is not a frequent explanatory factor of growth.

The setting of trigger values is also exacerbated by a lack of data on the effect of change on a soil quality indicator, and the fact that changes in many indicators follow a curvilinear pattern, with no clear, critical or step-wise alteration in content or form (Loveland and Webb, 2003). In the absence of clear triggers and a dearth of incontrovertible quantitative ecological evidence, the setting of soil quality indicators often require the input of ‘expert judgement’. However, the use of such judgement is in itself contentious, conjuring ideas of the use of arbitrary anecdotal evidence or best guesses (*cf*. Sparling *et al*., 2003). Further it is imperative to avoid the 'consensus in error' syndrome that can plague committees of experts when egos outweigh knowledge (Gufstafson *et al*., 1973). Nevertheless, evidenced-based expert judgement is widely used in a range of regulatory fora in Europe including Existing Substances Regulations (Directive 98/8/EC) and the Water Framework Directive (2000/60EC); as long as it is used in a transparent, rules-based way, it can provide a technically justifiable approach.

Expert judgement has proved effective in setting trigger values for SQIs as part of an evidence-based approach in New Zealand (Sparling *et al*., 2003a), where an expert group used three methods to derive a minimum trigger level for a soil organic carbon (SOC) indicator. These methods were:

- using national soil baseline data to calculate median and lower quartile values for SOC for each soil order;
- modelling SOC loss with the CENTURY model, with the lower SOC limit set at a value to permit recovery to 80 per cent of the original SOC level within a 25 year period;
- expert judgement used in a workshop format using a Modified Delphi Technique, to derive response curves for changes in SOC (in Mg/m^3) against soil quality (up to 100 per cent) as shown in Figure 1.1. This final approach involved individual scientists drawing curves which were then overlaid with curves of other scientists and, through discussion, modified if individuals agreed.

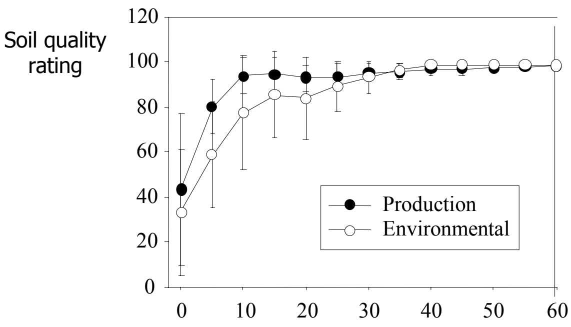


Figure 1.1: The derivation of response curves using expert judgement

The New Zealand group found that if production and environmental uses conflicted, as could be the case for indicators such as Olsen P, the curves could be combined by taking the minimum from each. An example of a single, conservative interpretation produced by combining minima from response curves is shown in Figure 1.2; this example could represent Olsen P, where more might be better for production but less for environmental quality.

Of the three methods listed above, the New Zealand group found that the modelling approach was preferred but was hampered by a lack of data. Thus, expert judgement and baseline data were used to set provisional triggers for carbon content for a limited number of soil type and land use combinations.

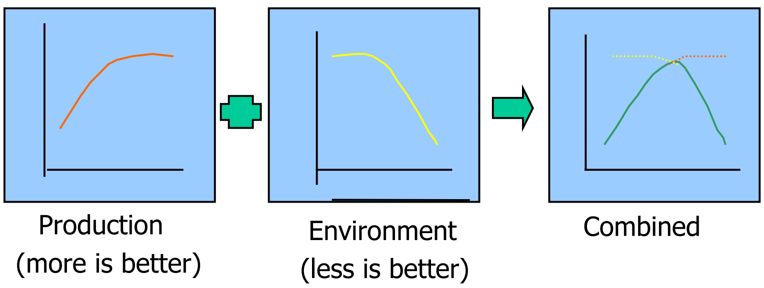


Figure 1.2: The combining of minima from response curves

The 2002 project (Loveland *et al*., 2002) highlighted the need for trigger or limit values. However, establishing numerical limits would require an extensive literature search, in many cases with supporting further research - a goal outside of the project's remit but an objective of this report. In this report, trigger values are defined as values or a range of values above or below which a level of change is understood to be critical in terms of the soil's fitness for a specific use. The values of triggers are to be viewed in the context of interpretative information also collected at the site on soil type, land use and so on. The triggers are to be used to build up 'weight of evidence' to determine if there is a potential issue with the quality of the soil in relation to the function of environmental interaction.

1.3 The Environment Agency's approach

The Environment Agency adopted a consensus-based approach to test the chosen indicators, which were selected for their quantitative capacity and relevance to policy issues, where their relevance would also be understood by a lay audience.

The approach, outlined in subsequent sections, was built upon knowledge and understanding from similar work undertaken internationally (Sparling *et al*., 2003) along with the experience of the UK soil science community. This approach, involving internal and external experts, ensured that the disparate needs of the consortium as well as those of the Environment Agency were met.

2 Approach and results

2.1 The challenge – testing the indicators

The previous Environment Agency-led project (Loveland *et al*., 2002) identified 67 potential soil quality indicators and recommended a minimum set of nine for national soil monitoring purposes. However, to ensure that this selection of nine was appropriate, the full list of 67 was tested by applying them, or the concepts behind them, to a wide range of soil and land use systems.

'Challenge' in the context of this project has been considered from the scientific and technical perspective and has meant the judging of indicators against a number of key criteria as outlined by Barraclough (*Pers. Comm*. 2004) in a document to the UK Soils Consortium. The following is an edited extract from that document and highlights the criteria for challenge:

Relevance

The indicator must be relevant to the function of environmental interaction and it must be interpretable in quantitative terms as an indicator of soil quality and the temporal changes in soil quality.

Allied to this is the issue of clarity. It must be clear what interpretation can or cannot be placed on an indicator.

It may be useful to consider indicators as direct or indirect indicators of a soil function. Thus, a catchment hydrograph is an indirect indicator of rainfall interception and storage by soils, but changes in soil water storage following rainfall is a direct indicator.

Sensitivity, discrimination and signal-to-noise ratio

1. Soil properties are notoriously spatially variable: 50% is not unusual as the standard errors of the mean of many typical soil parameters. Against this, many soil parameters change only slowly with time. Thus long term monitoring must attempt to discriminate long term trends from "noisy" backgrounds.
2. In selecting indicators, we need to consider the probability of detecting significant changes over the sampling intervals, Thus, for example, if a parameter is likely to change by 5% between samplings, and the 95% confidence limits of the measured mean are equivalent to 50% of the mean, it will be many years before a significant change is detected.
3. This leads to the idea of the *undetected change.* Indicators should be evaluated against the time span over which significant changes will go *undetected*; and whether such changes, once detected, are already irreversible. Ideally the time over which a change is undetected is minimised. These aspects are easily determined using simple statistical procedures.
4. We should not adopt indicators which, because of significant variability (due either to actual spatial or temporal variability or to sampling and measurement errors), are unlikely to detect change over reasonable time intervals.

Practicability

1. How practicable is a potential indicator? Are there robust, proven methods for its measurement? Are such methods in the pipeline? Or will they need considerable development? In the latter case, there would need to very strong reasons to include an indicator which would require significant further development.
2. Where such reasons do exist, possibly because the indicator furnishes information unavailable in any other way, the project should support such further development.

Efficiency and cost

1. We should seek to maximise the use of automatic methods including sensors, remote sensing and automatic data retrieval. Potential indicators should be examined against the need to minimise cost and maximise efficiency.
2. Allied to this is the general consideration of cost. Potential indicators must be assessed against the likely cost of populating them over 5, 10 and 20 years.

Integrative indicators

1. Wherever possible we should be looking for integrative indicators. These are indicators which effectively integrate the information from a number of subsidiary indicators. One example is the catchment hydrograph, which reflects the average hydrology of the soils in the catchment.
2. However, integrative indicators should only be adopted where they can be interpreted in terms of one of the key soil functions. In the case of catchment hydrographs, for example, it is still difficult to extract quantitative information on soil hydrology from what is a very smeared picture.

Of the 67 potential indicators identified, only a limited number were found to be relevant to the soil's interaction with the environment. These were divided into chemical, biological, physical and 'other' categories listed below:

Chemical:

- Soil organic carbon
- Top soil pH;
- cation exchange capacity (CEC) to 1 m depth;
- anion adsorption capacity in topsoil;
- base saturation;
- concentration of potential pollutant elements/organic micropollutants (POPs).

Biological:

- SOC;
- microbial biomass carbon/SOC;
- soil biomass;
- Biolog score
- DNA-based microbial diversity index and enzyme assays.

Physical:

- integrated air capacity to 1 m depth;
- number of locations with erosion features;
- Soil organic carbon;
- topsoil surface condition;
- aggregate stability;
- toil bulk density;
- topsoil plastic limit to a depth of 1 m;
- time to ponding;
- water dispersible clay.

Other:

- catchment hydrograph;
- surface water turbidity;
- biological status of rivers with and without sewage treatment works;
- number of eutrophication incidents per year.

Fully testing all of the above indicators would be prohibitive in terms of time and resources. Equally, only some of the indicators would be likely to provide information to inform high-level policy decisions and are understandable to a lay audience. Indeed, arguably the only practical and pragmatic approach to be taken is to select a minimum number of potential indicators that provide the appropriate level of policy information and to challenge those.

The New Zealand group (Sparling *et al*., 2003), from an initial list of over 20, selected the following seven indicators:

- total carbon
- total nitrogen
- pH (as measured in water)
- Olsen P (phosphorus)
- mineralizable nitrogen
- bulk density
- macro and total porosity.

For this report, eleven potential indicators were selected for challenge on the basis of the first project (Loveland *et al*., 2002), published evidence from previous soil quality schemes (Pankhurst *et al*., 1994; Doran and Jones, 1996; Schipper and Sparling, 2000; Seybold *et al*., 2003; Sparling *et al*., 2003;) and selected expert advice (Chambers, Scholefield, Parkinson, Harrod, Reeve, *Pers. Comm*.). The eleven chosen for the working list (which could be added to or reduced during the project) were:

- SOC (reported as soil organic matter or SOM)
- pH (water and $CaCl_2$)
- Olsen P
- total N
- bulk density
- catchment hydrograph
- aggregate stability
- heavy metals
- organic micropollutants
- mineralisable N
- macroporosity.

It is important to note that the selection of these eleven is by no means exhaustive or definitive, indicators selected here may not be on the final MDS list and similarly, indicators (following the review process or the Technical Workshop) not selected could be placed on the list.

Figure 2.1 illustrates the consensus-based approach with which these indicators were challenged. Reports for each indicator were produced by Environment Agency staff for internal experts and consortium members to discuss and comment on. The reports were not produced to be definitive guides to a particular soil property, but to illustrate salient points for deliberation and discussion. Each report contained the following information on the potential indicator:

- a brief background, including how will the indicator relate to the soil function of environmental interaction and what any changes might mean;
- information on rates of change, what else might be affected and an indication of inherent variability – that is, whether it would be possible to detect change in a reasonable time period or whether, by the time change was detected, it would be too late;
- a selection of trigger values linked to soil type and land use;
- likely integration with other national monitoring schemes;
- cost, suitability for broad-scale monitoring and ease of undertaking analysis by a range of laboratories;
- conclusion and recommendations if possible.

A group of five selected soil science experts from a range of backgrounds were used to peer review the reports and truth the challenge process. These selected experts were chosen because of their breadth of soil science expertise and practical knowledge and understanding of soil monitoring, requisites thought necessary to ensure delivery of useable outputs for a soil monitoring scheme. The selected experts were:

- Brian Chambers, Senior Principal Research Scientist, ADAS
- Rob Parkinson, School of Biological Sciences, University of Plymouth
- David Scholefield, Principal Research Scientist, SEES, IGER
- Tim Harrod, retired from Soil Survey and Land Research Centre
- Malcolm Reeve, Soil Consultant, Land Research Associates Ltd

The process of internal challenge is detailed in Section 2.4, and took the form of a mini workshop to discuss the reports with the selected experts, following the receipt of written comments to focus discussion on principal areas of concern (Appendix I).

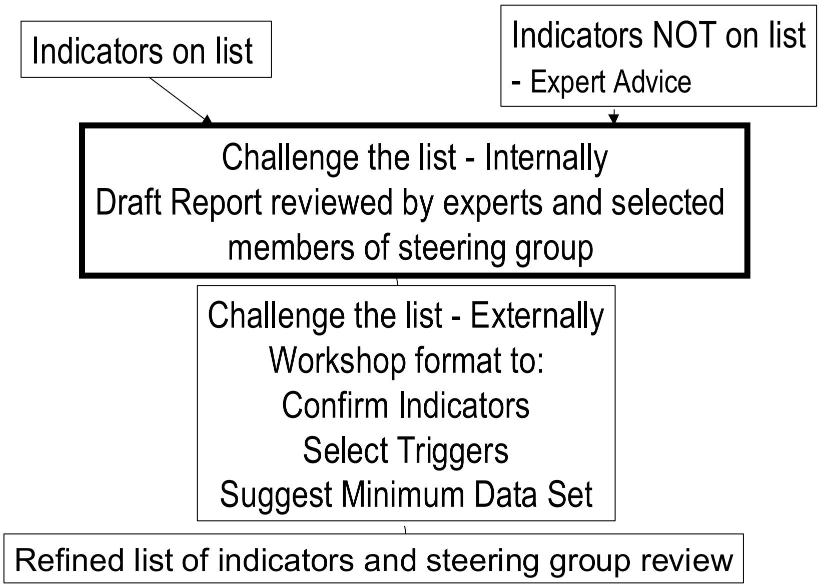


Figure 2.1: Diagram of the challenge process for the potential SQIs

Following incorporation of comments and suggestions from the selected experts and members of the consortium, the reports were redrafted (Section 2.3) and passed on to an external audience for comment and discussion in the technical workshop.

As explained previously, the use of expert judgement is potentially contentious; nevertheless, it is used to tackle questions for which there is insufficient or mixed evidence, and is broadly accepted in regulatory fora across Europe. Therefore, the use of experts may be practically seen as a method with which to reduce uncertainty in the face of having to make a decision. Therefore it is highly relevant for this project where quantitative evidence by which to support decisions is often mixed or does not exist.

A previous Environment Agency project (P6-020/2 – *A framework for environmental and human health standards*) examined decisions taken in deriving environmental quality standards and questioned the use of expert judgement, suggesting the following guidance:

- Where knowledge is incomplete, other approaches may be adopted using personal experience based on an understanding of certain scientific processes

and relationships. However, it is imperative that the personal experience is substantiated with evidence.
- Decisions must be auditable and the decision-making processes transparent.
- Rule out as many points of possible contention prior to the meeting. Potentially time-consuming and will-sapping discussions to establish definitions should be identified in advance and avoided where possible.

This project followed the above guidance, which is largely in line with the work undertaken by the New Zealand group (*cf*. Sparling and Tarbotton, 2000; Lilburne *et al*., 2004). The structure and outcomes of the technical workshop, which dominated the second stage of the process shown in Figure 2.1, are outlined in Section 2.5.

2.2 The tiered approach

The use of a MDS is often criticised because of its use of a limited number of indicators and its justification in terms of the wider soil data needs of a broad range of interest groups. However, the use of a MDS can be supported through the use of an interpretative framework in which other SQIs are not excluded, but can be used for specific purposes at different, lower tiers than the MDS. Such an approach is advocated for this project and has a number of advantages over single tier schemes, the primary one being that levels of risk can be balanced against the effort involved.

This approach is not new. Indeed, recently European working groups on soils have suggested three levels for a soil monitoring strategy:

- Level 1: basic monitoring;
- Level 2: reference sites;
- Level 3; special sites (for example, sites with unusual climatic conditions or localised issues).

A modification of this approach is suggested here, whereby Tier 1 represents broad-scale monitoring with data collected using the MDS across the UK. Tier 2 is more specific monitoring, using a more targeted set of indicators for sites identified through the use of the triggers or through the risk-based approach outlined below. This type of sampling could, for example, be spatially defined by a 'gross' soil type and land use combination resulting in specific threat of capping and monitored through the use of 'time to ponding' indicator. Tier 3 monitoring refines the information collected to answer localised issues. The type and magnitude of the issue would dictate the type of monitoring carried out.

At Tier 1, generic information is collected on the state of UK soils to identify headline trends and areas at risk. At this level, existing data on soil properties can be used to broaden the dataset and enable early trend analysis to be undertaken. Transition from Tier 1 to Tier 2 and then Tier 3 occurs through the identification of increasing levels of risk, such as the exceedance of a trigger or the identification of a scenario that presents an obvious risk. Changing management practices or land use may alter the level of risk, where sampling at the site(s) could revert to Tier 1 or Tier 3 (as shown in Figure 2.2). The MDS is also collected at Tier 2 and 3 (in addition to any extra/alternative sampling) and includes site characteristics, land use history, soil classification and a profile description.

Tier 3 sites represent highly localised issues or threats requiring localised management responses. These are likely to be significantly fewer in number and scale compared with Tier 2 sites.

In summary, progression through the levels allows further refinement, enabling local or geographically limited issues to be tackled using a range of indicators. This tiered, risk-assessment approach ensures that resources are focussed and cost is minimised (Figure 2.2).

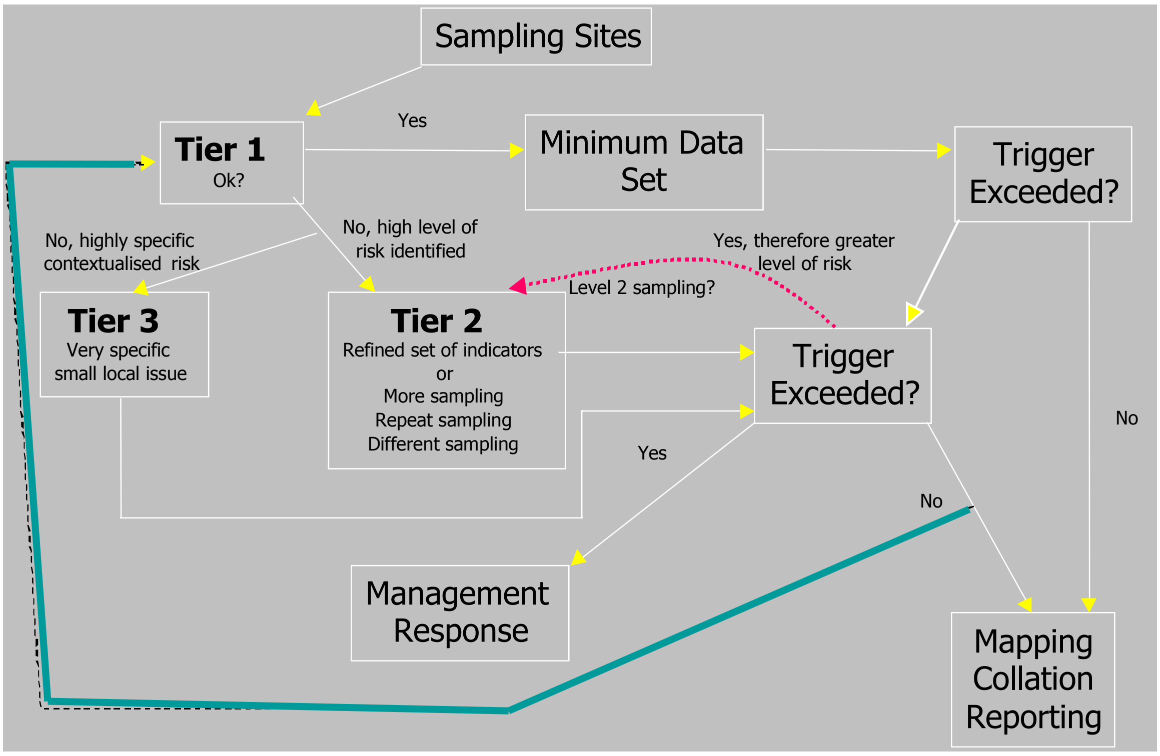


Figure 2.2: A tiered, risk-based approach to using soil quality indicators

For Tier 1 sites, the MDS collection will be carried out at a specified location, possibly every five years (Loveland *et al*., 2002) - the required sampling programme and road-testing of indicators have yet to be undertaken by the consortium. The identification of a land use/soil type at potential risk in regard to the function of environmental interaction may result in a transition from one Tier to the next. This may be identified on a gross scale through the use of previous soil monitoring data, risk maps and local knowledge, but it will be necessary to establish what is required before moving onto the next tier. For example, consider the following questions:

- Does the information from Tier 1 or the risk identification process need confirmation – i.e., what is the level of uncertainty?
- If more testing is needed, will this require more sampling (such as a greater number of samples from the same area) for the same or different indicators?

Guidance will need to be developed to help the user navigate between tiers. Setting standard operating procedures for each directional arrow shown in Figure 2.2 will ensure that practices are consistent, clear and accountable.

Figure 2.3 shows how the risk assessment might affect sampling practices at different tiers. At Tier 1, sampling is expected to be carried out, initially, every five years – or at a frequency to be determined through subsequent work. This sampling is likely to continue at regular intervals, but be reviewed after 30 years. Over this time period, trends can be identified and refinements to the strategy undertaken.

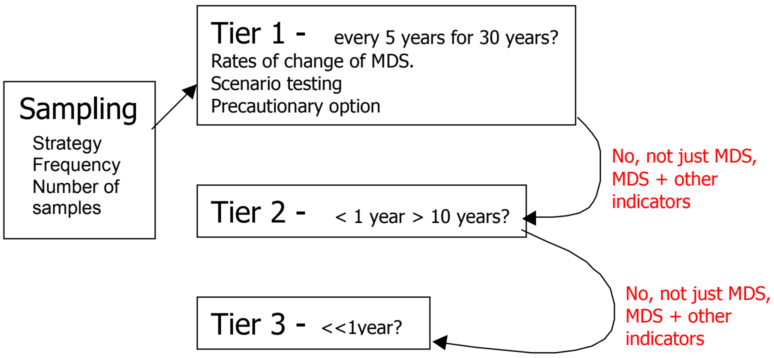


Figure 2.3: The potential influence of risk assessment on soil sampling practices

However, if triggers have been exceeded or greater levels of risk identified, then sampling frequency may change. This is particularly likely to be the case at Tiers 2 and 3 where seasonal biological effects may be pronounced and sampling confined to specific time periods. At Tier 3, some indicators may require monitoring several times in a single year in line with the development or life cycles of biological species.

2.3 Establishing the minimum dataset

The internal reports given in this section have focussed on the type of indicators found in Tier 1 that might be included in the MDS; that is, the relatively few headline indicators that are sampled extensively across the monitoring network. The primary purpose of these reports was to provide a discussion piece for Environment Agency experts at a technical meeting in Bath (described in Section 2.4) and for consortium members to comment on. The reports, presented below in sections 2.3.1 to 2.3.10, cover the eleven potential indicators listed in Section 2.1. The reports incorporate comments and views from the selected experts and Consortium members and are ‘stand alone’ and fully referenced.

The feasibility of detecting temporal changes in SOC (one of the potential indicators) is discussed in an accompanying report by Smith and Fang for the Environment Agency, presented in Section 2.3.11

2.3.1 Soil reaction as measured by pH

The use of a minimum dataset (MDS) is arguably the most practical way to establish a national soil monitoring network (Doran and Parkin, 1996; Arshad and Martin, 2002). Soil quality indicators in the minimum dataset are to be used broadly to assess relatively high level trends in soil quality and risks or threats to sustainable land management practices. Further, their use should provide a basis for evidence-based decision making and policy development for soils.

Soil quality indicators, including those in the minimum dataset, need to be readily interpretable, relatively precise and not readily estimated from other indicators (Schipper and Sparling, 2000; Seybold *et al*., 2003).

Background

The inclusion of a measure of soil reaction in the minimum dataset seems a sensible one. Indeed, there are few, if any, scientific papers or monitoring schemes looking at the physical, chemical and biological status of soils that do not measure pH in water or calcium chloride. Reasons include the significant influence pH has upon many soil processes including nutrient availability, biogeochemical cycling, contaminant sorption, structural stability and biological activity. Furthermore, soil acidity has a significant influence upon drainage water quality/composition and subsequently on local surface and lake water quality.

Acidification is a natural soil process, the rate of which can be altered by anthropogenic activity. The extent to which a soil becomes acidic naturally depends upon climatic conditions and inputs from vegetation, the atmosphere, microbial populations and the soil's propensity to resist these acidic inputs. Anthropogenic-induced changes in soil pH are many, including atmospheric inputs from the burning of fossil fuels, but especially agricultural activities, particularly liming, but also fertilizer management, irrigation and organic waste recycling. An in-depth coverage of the causes of anthropogenic acidification of soils is not warranted here, and is provided in great depth elsewhere (Reuss and Johnson, 1986; Rengel, 2003).

Loading of nutrients and also acidity to soils from atmospheric sources has decreased over the last 20 years, and this decline is expected to continue (Makela-Kurtto and Sippola, 2002; CEH, 2004). These findings concur with other studies in Europe, but results are very much soil and land use specific (Adamson *et al*., 1996; Webb *et al*., 2001).

The key concern for this paper is the use of pH as a soil quality indicator of the function 'environmental interaction'. More specifically, the aims are to challenge the use of pH as a soil quality indicator: what does change in soil pH mean? What triggers in soil pH exist? Is it possible to integrate pH with other indicators? How much does pH measurement cost and what is the sensitivity and likely variability of the method used?

It is important to stress that while the measurement of pH is standardized, in the MDS (as with the other suggested indicators) its interpretation would not be, depending on major soil group and land use. This is an acknowledgement that different soil conditions are desirable for different land uses – that is, the soil condition is fit for that particular land use (Schipper and Sparling, 2000).

What do changes mean?

A buffered system is one that changes little in pH on the addition of acid or base, and it may be considered that soils behave as weak acids (Figure 2.3.1.1). This facet of soils to buffer pH change is influenced by a number of soil properties including clay, organic matter and carbonate content.

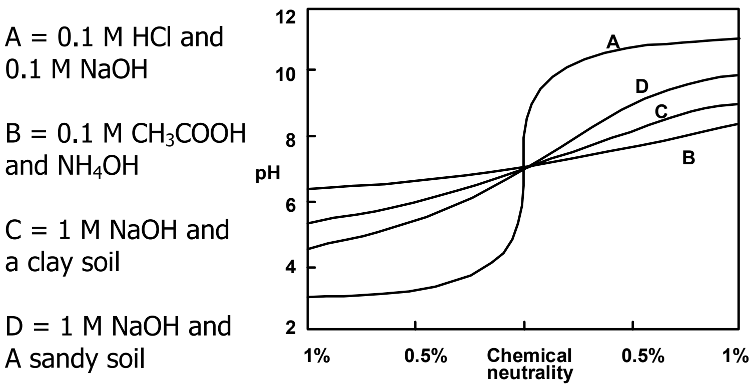


Figure 2.3.1.1: Illustration of soil behaving as a weak acid

Therefore, because soils are buffered, relatively rapid or drastic changes in pH are often only associated with extreme events in anthropogenically modified areas, such as at mine sites (Moore and Luoma, 1990) or ill-conceived agricultural practices on soils with a significant propensity for pH change, such as acid sulphate or saline soils (McTainsh and Broughton, 1993; Young, 2000). That is not to suggest that small changes in soil pH are not significant. Indeed, as pH is a logarithmic measure (i.e. pH 6 is 10 times more acid than pH 7), small changes represent relatively large differences in hydrogen ion activity. Relatively small changes in soil pH in poorly buffered soils may have dramatic effects upon nutrient and trace metal availability leading to crop yield reduction (Tisdale *et al.*, 1993), potentially toxic concentrations of non-essential elements, being released into soil solution (Christensen, 1984) and detrimental effects upon surface drainage and groundwater quality (Reuss and Johnson, 1986).

Acidification

Adverse factors associated with low soil pH are rarely ever associated with the direct harmful effects of H^+, but more often the indirect effects of elevated H^+ activities (McBride, 1994). These indirect effects may include toxic solution concentrations of Al, Fe and Mn and in certain soils, elevated concentrations of other trace elements to potentially toxic levels or indeed plant deficiences (such as Mo). There is also a significant effect of accelerated acidification on soil organic matter amounts, composition and cycling. In part, this is due to changes in the microbial population and community structure brought about by soil pH change (Table 2.3.1.1). Under acidic conditions (pH <5) fungi tend to dominate with reduced competition from bacteria and actinomycetes and there is a commensurate change in bacterially mediated processes such as nitrification or certain functions of nutrient cycling (optimum pH 6.5-8). This reduction in microbial activity may also reduce pesticide efficacy and degradation rates, and potentially increase prevalence of disease and pests (Smith and Doran, 1996).

Table 2.3.1.1: Optimum pH values for microbial groups and processes (Smith and Doran, 1996)

Group-Process	pH range	Optimum
Bacteria	5-9	7
Nitrification-denitrification	6-8	6.5-8
NH_3 inhibition and NO_2 oxidation	>8	-
S-Oxidizers	1-8	2-6
Actinomycetes	6.5-9.5	8
Fungi	2-7	5
Blue green bacteria	6-9	>7
Protozoa	5-8	>7

Crop growth and subsequent harvesting and removal promotes the loss of basic cations from soils. This removal of bases increases soil acidification and, in the temperate climate of the UK, can be balanced by the addition of liming materials. This balancing of soil pH is obviously not undertaken on soils in natural and semi-natural habitats affected by aerially deposited acidification, a stimulus for the work on critical loads in the 80's (DoE, 1991) that continues today (CEH, 2004). Naturally, soil pH values are mostly in the range of 3.5 – 8.5, but it has long been recognised that, in terms of crop production, there are favourable soil pH conditions to maintain a satisfactory nutrient supply (Truog, 1946). This recognition has led to the production of tables of arable crop sensitivity highlighting soil pH values below which there are likely to be adverse effects upon yield (MAFF, 1981). Indeed, from previous national or state surveys of soil pH values from New Zealand, Finland, Canada and the UK, arable cropped soils tend to show the least change through time of any land use, being between 5.4 and 7.0 (as measured in water). This reflects the need to control soil pH to ensure maximum yield (Skinner and Todd, 1998; Schipper and Sparling, 2000; Kenney *et al*., 2002; Makela-Kurtto, and Sippola, 2002). Similarly, De Clerk *et al*. (2003) noted that, in a survey of 125 agricultural soils in California with paired samples spanning 60 years on a range of land uses, the pH values of cropped soils changed very little compared with other land uses (6.9 – 7.1).

While arable soils show limited changes in reaction over extended periods, permanent grasslands in England and Wales have been observed to show a decrease in pH (from 5.7 to 5.4 in 25 years) (Webb *et al*., 2001). The soils under permanent grassland are often more poorly buffered, and importantly, no rough grazing land was included in this particular survey (Representative Soil Sampling Scheme or RSSS), suggesting a greater spatial extent of soil acidification (Skinner and Todd, 1998). For grazing livestock a soil pH of 6 is recommended (MAFF, 1981), but the significant reduction over time observed is thought to have been exacerbated by a decline in the practice of liming (Skinner and Todd, 1998). Also much grassland is on small, economically marginal farms, remote from lime sources in the upland fringes.

Acidic soils are base poor and deficiencies of base cations, such as Ca, Mg and K, may also be manifested in plant tissue deficiencies. A reduction in base saturation and an increase of Al species in solution often also accompany an increase in soil acidity. For example, Adamson *et al*. (1996) noted that for an A horizon of an upland soil in the UK, a reduction in soil pH of 0.6 (5.6 to 5.0) in 20 years had an accompanying reduction in base saturation of 11% (35.5 to 24.2). Yet, direct effects of these changes in UK soils are rarely measured, but often noted by authors as being 'cause for concern' (Skinner and Todd, 1998) or having 'important implications' (Adamson *et al*., 1996). These affirmations are based on the well-founded view that poorly buffered soils are those that are at greatest risk of degradation and long-term damage. In the UK, soils with the smallest critical loads (those sensitive to acidity – see below) dominate in North and West Britain, while the South and East are dominated by soils with larger critical loads (Hornung *et al*., 1995).

However, examples do exist of dramatic cause and effects in both the Nordic countries and central Europe, where acid deposition on poorly buffered soils has led to 'die back' of vegetation, removal of base cations and soil structural decline (Bunce, 1993).

Yet, naturally acidic, poorly buffered soils support unique flora and fauna that have adapted to low pH and nutrient poor conditions (Koptsik *et al*., 2001). Lowering of the soil pH further (and also increasing nutrient content or availability) can significantly damage these ecosystems and reduce biodiversity (*cf*. Wilson *et al*., 2001). It is certain that the affect of anthropogenic acidification on soils produces conditions that fall outside the envelope within which many plants and fauna of a region have evolved. Roem *et al*. (2002) demonstrated that a reduction in pH of a sand soil from 4.3 to 3.8 over a five-year period significantly reduced plant species diversity and seedling germination of native heathland species. The acidification of the soil increased Al solution concentrations and enabled the dominance of *Culluna vulgaris* and *Molinia caerulea*. In a study of Environmentally Sensitive Areas in England, Critchley *et al.* (2002) attempted to establish relationships between plant communities and a range of soil variables at 38 sites. Species richness was lowest ($< 20\ m^{-2}$) at soil pH values of less than 5 and greater than 8; these sites were also characterised by plant species that had a relatively higher tolerance to resource-limited conditions.

The critical loads approach has been applied to broad habitats and is a quantitative estimate of exposure of those habitats to a pollutant, in this case acidity, below which no significant harmful effects are thought to occur. Exceedances are calculated by establishing the acidifiying contribution from aerially deposited sulphur and nitrogen compounds and subtracting the critical load for that habitat, calculated from empirical or steady state mass balance methods (CEH, 2004a). This method applied in the UK shows that 72.6% of key habitats, including acid grasslands, dwarf shrub heath and montane, have their acidity critical loads exceeded, with accumulated exceedance for acidity greatest in England for broadleaved woodland.

The critical load concept does provide a steady-state measure of soil acidity, but it is thought to be limited in terms of protecting the quality of complex ecosystems, because the critical load is not an indicator in itself – although the exceedance of the critical load may be. Further, the dynamic aspects of soil acidity, including recovery, are not taken into account, leading to the view that critical load exceedances are underestimates of effects upon soil quality (Schmieman, *et al*., 2002).

In relation to ‘environmental interaction’, the composition of drainage waters from soils greatly influences stream and lake water quality within catchments. Increased acidity can promote elevated concentrations of Al in leachate compared to those of Ca and Mg in poorly buffered soils, often in upland catchments (McBride, 1994). Under normal conditions in acid soils, Al is mobile primarily as a complex (inorganic or organic), and is precipitated in the B horizon. But with increasing levels of acidity, the Al^{3+} ion dominates which is not readily precipitated in the B horizon and is leached through the profile to surface drainage channels (Reuss and Johnson, 1986). This low-pH, often cation-rich discharge can have detrimental effects upon the ecology of poorly buffered drainage channels and lakes. Fish and other gilled organisms are highly sensitive to elevated soluble Al concentrations (Bunce, 1993).

Rates and magnitudes of changes in soil pH

The degree of soil pH buffering is particularly important in the identification of long term trends in the affects of increased acidity. On soils with elevated clay contents (<40%) and so a greater buffering capacity, it has been noted that differences in land management systems can not be readily differentiated through the use of soil pH (Seybold *et al*., 2003). Further, for peaty or organic rich soils, changes in pH of up to 0.2 units are thought to be insignificant with regard to ecological damage (Hornung *et al*., 1995).

The critical loads work has demonstrated the importance of climate and vegetation as well as land management and much work has centred on UK uplands and also acid lowlands (CEH, 2004). Yet, buffering against pH change is finite and as soil buffering decreases more rapid changes in soil pH will occur over shorter durations. Rates and magnitudes of soil pH change in agricultural systems are relatively well documented from field and trial plot experiments probably due, as already mentioned, to ensure optimum pH for crop growth. Relatively long-term changes in soil pH under permanent grassland were monitored in a liming trial at Rothamsted (silty clay loam) and Woburn (sandy loam) and modeled by Goulding *et al*. (1989). The data from the control (0 application) treatment showed a rate of acidification of only 0.5 and 1 pH units for twenty years of the trial for the silty clay loam and sandy loam, respectively. However, this model does not incorporate the acidifying effects of atmospheric deposition, or chloride or sulphur addition with the fertilizer, due to the view that while important on non-cultivated soils, the impact was insignificant compared to fertilizer N, crop type and rainfall. It is thought that soil pH change is relatively slow, and is one of the more stable physico-chemical soil parameters measured. Further, in an agricultural context, yearly sampling or greater can be sufficient in terms of frequency depending on the threats or risks, and time of sampling during the year is not crucially important (Sarrontanio *et al*., 1996).

Rates of change of soil pH are not well documented for semi-natural habitats and information is limited on the sensitivity and rates of change in these ecosystems. The critical loads work (CEH, 2004a) does highlight the likely risks associated with certain broad habitat groups, but the effects of relatively small changes in pH upon soil fauna or community structure or function are not readily available.

The CEH (2001) project analysed 1067 soil samples for pH; changes were significant and detected over 30 years, yet the magnitude of the changes, with regard to soils now falling outside their originally assigned soil pH class, have yet to be determined. The range of pH units which the soils fall into also varies with soil major group, being relatively narrow for peats and podzolics (<1.5 units) and significantly wider for gleys and lithomorphic soils (>1.5-2).

In part, the magnitude of pH change is reflected in the methods used to measure soil pH, their sensitivity and reproducibility. This is covered in Section 3 of the main report.

Selection of triggers for pH

In attempting to answer the question of what is a significant change in soil pH, there is a need to define baselines or ranges of soil pH values for soil types linked to land uses. This has been undertaken by the New Zealand group, and the provisional quality classes for six major soil and land use combinations are given below in Table 2.3.1.2 (Sparling *et al*., 2003). The range of pH values for cropping are deliberately wide as an acknowledgement of the need of crops for differing pH conditions.

Table 2.3.1.2: Trigger values for topsoil pH in water from Sparling *et al*. (2003)

Pasture on all soils except organic	4	5	5.5	6.3	6.6	8.5
Pasture on organic soils	4	4.5	5	6	7.0	
Cropping and horticulture on all soils except organic	4	5	5.5	7.2	7.6	8.5
Cropping and horticulture on organic soils	4	4.5	5	7	7.6	
Forestry on all soils except organic		3.5	4	7	7.6	
Forestry on organic			Exclusion *			
	Very Acid	**Slightly Acid**	**Optimal**	**Sub-optimal**		**Very Alkaline**

Figures in shade represent upper and lower limits.
* This combination is unlikely in practice due to windthrow

From Table 2.3.1.2 it is clear that, from an agricultural perspective, it would be relatively straightforward to define values for soil pH for crop types and major soil groups in the UK. However, from an environmental interaction perspective this presents greater difficulty in the determination of triggers (*cf*. MAFF, 1988). Nevertheless, it could still be undertaken. Indeed, by consideration of the functions that are required for the soil to perform and the connectivity between the indicator and those functions, it is possible to produce a table based on broad soil types. An example is given in Table 2.3.1.3 and has been populated with potential trigger values derived from a range of sources (MAFF, 1988; 1993; Alloway, 1995; ECI, 2003). The habitats selected are those from the Biodiversity Action Plan that are likely to be affected by pH.

Table 2.3.1.3: Potential trigger values for topsoil pH for the soil function of environmental interaction

Function	Soil type		
	Mineral	Peaty	Calcareous
Metal retention	<6	<5.5	
Microbial function/ biofiltering	<5	<4.5	
Gaseous emissions			
Habitat support table			
Calcareous grassland			<7
Mesotrophic grassland	<5>7	<5>7	
Acid grassland	>5	>5	
Dwarf shrub heath	>4.5	>5	
Biomass production			
Arable & horticultural	<6.5	<5.8	
Improved grassland	<6	<5.3	

Integration with other national monitoring schemes

Soil acidity as measured by pH is often undertaken in either a weak salt solution, such as calcium chloride, or water. The CEH (2001) and New Zealand surveys all use water, but there is National Soil Research Institute and RSSS (Table 2.3.1.4) data available for pH determined using calcium chloride.

The use of historical datasets on soil pH may be particularly useful, allowing hindcasting and scenario testing to be undertaken. Although for soil pH, archived and re-tested soils have been shown to vary significantly (<52% n = 100) from the original dataset (Adamson *et al*., 1996), a facet thought to be attributable to the drying of soil prior to undertaking the procedure. But, if soils are treated in the exactly the same way, there is thought to be little difference due to storage (*cf*. De Clerk *et al*., 2003). This does highlight the importance of detailed methodological information accompanying historical datasets if they are to be useful.

A further point of relevance with regard to integration and interpretation of pH measurements is that nearly all are measured in topsoils (0-15 cm, 0-7.5 cm). This is appropriate for agronomic purposes, but deeper measures may be required for environmental relevance.

Cost and suitability for broad-scale monitoring

The cost of measuring pH is relatively low, and is a routine determinand for many analytical laboratories, especially in relation to soil fertility (Giltrap and Hewitt, 2004). Further, it is the most practical and efficient way of measuring acidity in soils.

Laboratories often quote costs for soil pH determination of £2-6 (National Laboratory Service, Environment Agency, *Pers. Comm.*). From other national soil monitoring surveys, the greatest costs are often not associated with the analysis but with site identification, sampling and description (Schipper and Sparling, 2000).

Values of soil pH are often reported to the nearest 0.1 of a pH unit (National Laboratory Service, Environment Agency, *Pers. Comm.*), but measured to 0.01. Further, it is suggested that diurnal fluctuations of up to 1 unit may occur in soils, indicating that it is not necessarily meaningful to report pH measurements more precisely than to the nearest 0.1-0.2 division of a unit (Alloway, 1995). This variation and low time stability can also be used as an argument against the use of a measure of soil pH as a soil quality indicator (De Clerk *et al*., 2003). Spatial dependence of soil pH varies across soil types and land use, with some showing significantly less variation (cropping) than others (e.g. forest plantation) (Giltrap and Hewitt, 2004).

Soil pH is often measured routinely in research and commercial laboratories in either water or a weak salt solution. Because the electrolyte concentration of soils in the field varies with such factors as rainfall, evaporation and fertilizer application, the values obtained for pH will also vary with time. The measurement of soil pH in a weak salt (0.01–0.1 M) solution of calcium or potassium chloride, rather than just water, is an attempt to reduce the influence of this varying electrolyte concentration because it is small relative to the total salt concentration in solution. The use of $CaCl_2$ has been recommended for a range of practical and interpretive reasons (Schofield, 1955; Archer *et al*., 2003):

- The ionic strength and the concentration of calcium and chloride are the same order as those commonly found in many soil solutions, and consequently there is little exchange of H^+; the values for pH obtained are probably quite close to those actually prevailing in soils in the field at their ‘natural’ water content.
- Use of $CaCl_2$ avoids the drift in pH measurement due to leakage of KCl from the electrode that affects measurement made in water.
- The soils flocculate and settle readily in $CaCl_2$ solution, and thus errors arising from the liquid junction potential can be minimised by placing the calomel electrode in the clear supernatant solution.

It should be noted that whether the concentration of 0.01 M is most appropriate for all soils is still a matter for debate (Dolling and Ritchie, 1985).

Other reasons for the use of $CaCl_2$ over water for the measurement of pH are that the former gives smaller values (there can be up to a pH unit difference between water and $CaCl_2$) with less variation, and measures more truly the pH at soil particle and root surface, being less affected by seasonal fluctuations in the electrolyte concentrations of solutions (Wild, 1988; Archer *et al*., 2003). However, the variation in the mean of pH measurements by both methods from the RSSS database from two sampling periods show that there is no significant difference in variation with method (Table 2.3.1.4).

Table 2.3.1.4: Data from RSSS across all soil types and land uses

	pH (water) 2002, n = 225	pH (water) 1969, n = 402	pH ($CaCl_2$) 2002, n = 225	pH ($CaCl_2$) 1969, n = 402
Mean	6.61	6.68	6.00	6.12
Standard deviation	0.89	0.82	0.93	0.88
95th Percentile	8.05	7.9	7.45	7.435
50th Percentile	6.45	6.7	5.875	6.1
25th Percentile	5.85	6	5.25	5.3
5th Percentile	5.4	5.4	4.7	4.8

If soil pH is to be an effective soil quality indicator in a minimum dataset, it is important that through the pH measurement it is possible to distinguish between (broad classes of) land use practices on the same soil group; further, that intra-soil variability is less than these differences between land uses. However, it is perfectly reasonable to consider that there would be a greater difference between soil groups than between land use practice within soil groups (*cf*. Schipper and Sparling, 2000).

The depth at which soil is taken in order to measure pH for agronomic reporting is routinely between 0-15 cm for arable land (or grassland if it is to be ploughed out) and 0-7.5 cm for grassland (MAFF, 1993). In order to best account for environmental interaction it is suggested that the two sampling depths 0-15 cm and 0-7.5 cm be undertaken. The 0-15 cm will provide continuity with existing data bases and 0–7.5 cm is to cover grasslands (MAFF, 1988), but also semi-natural habitats where biological activity would be expected to be greatest at this depth (*cf*. Critchley *et al*., 2002).

The variation in pH caused by soil heterogeneity, analytical uncertainties and seasonal perturbations, must not be so great as to mask or obfuscate trends and interpretation (*cf*. Smith and Doran, 1996). Indicators with large coefficients of variation are unlikely to be able to detect significant changes (especially if the differences in means are small) unless many, many samples are collected. The New Zealand group observed a coefficient of variation in pH (water) across 29 sites of 2.3%, compared with 48% for unsaturated hydraulic conductivity (Schipper and Sparling, 2000).

Conclusion and recommendation

The measurement of soil pH has been undertaken in many national and state soil surveys in order to answer broad, relatively high level questions on the state, condition and suitability for use of soil (Seybold *et al*., 2003). As a use-dependent soil quality indicator, soil pH has been shown to be both relatively sensitive to change and practical to measure and interpret (Smith and Doran, 1996). Further, the communication of this information to non-specialists to improve awareness and understanding of how soils are changing and soil processes is not onerous.

Due to the regular and wide spread use of soil pH as a measure of reaction, the collation of data for a baseline for major soil groups or types should be relatively straightforward. In an agricultural context the derivation of triggers for soil pH for UK soils should not be a complicated task. However, the threats and risks posed by relatively minor soil pH changes in other more sensitive habitats are not so well documented, meaning that the derivation of triggers may rely more upon expert judgement. The use of pH as an indicator for the soil function of environmental interaction is due to its explicit relevance to soil biological activity, potential metal toxicity (especially Al) and water quality, nutrient behaviour and flora and fauna species diversity.

It is recommended to use soil pH as a measure of soil reaction in a minimum dataset, further to undertake the measurement in water to maximise use, continuity and comparison with previously measured values and datasets.

References

Adamson, J.K., Rowland, A.P., Scott, W.A. and Hornung, M. (1996) Changes in soil acidity and related variables over 25 years in the North Pennine Uplands, UK. *Soil Use and Man.*, 12, 55-61.

Alloway, B.J. (1995) *Heavy metals in soils*. 2nd edition. Blackie Academic and Professional, London.

Archer, J., Oliver, M. and Baxter, S. (2003) *Review of the Representative Soil Sampling Scheme (RSSS)*. Final report for Defra.

Arshad, M.A. and Martin, S. (2002) Identifying critical limits for soil quality indicators in agro-ecosystems. *Agriculture, Ecosystems and Environment*, 88, 153-160.

Bunce, N.J. (1993) *Introduction to Environmental Chemistry*. Wuerz, Winnipeg, Canada.

CEH (2001). *MASQ: Monitoring and Assessing Soil Quality in Great Britain*. Countryside Survey Module 6: Soils and Pollution, Rep. No. P5-053/01. Environment Agency.

CEH (2004) *Update to: The status of UK critical loads, critical loads methods, data and maps.* February. Centre for Ecology and Hyrdrology, Monks Wood, Huntingdon.

CEH (2004a) *Addendum: The status of UK critical load exceedances.* April. CEH, Monks Wood, Huntingdon.

Christensen, T.H. 1984. Cd soil sorption at low concentrations: 1. Effect of time, Cd load, pH, and Ca. Water, Air, Soil Poll. 21:105-114.

Critchley, C.N.R., Chambers, B.J., Fowbert, J.A., Bhogal, A., Rose, S.C. and Sanderson, R.A. (2002) Plant species richness, functional type and soil properties of grasslands and allied vegetation in English Environmentally Sensitive Areas. *Grass and Forage Science*, 57, 82-92.

De Clerk, F., Singer, M.J. and Lindert, P. (2003) A 60 year history of California soil quality using paired samples. *Geoderma*, 114, 215-230.

DoE (1991) *Acid Rain – Critical and Target Load Maps for the UK*. Air Quality Division, Department of the Environment.

Dolling and Ritchie (1985) 23, 309) Aust. J. Soil Res

Doran, J.W. and Parkin, T.B. (1996) Quantitative indicators of soil quality: a minimum dataset. In: *Methods for assessing soil quality*. Eds J.W. Doran and A.J. Jones. SSSA Special Publication Number 49, Madison, Wisconsin. pp25-37.

ECI European Copper Industry (2003) Draft report on *Effects Assessment to the Terrestrial Compartment*, June 2003.

Giltrap, D.J. and Hewitt, A.E. (2004) Spatial variability of soil quality indicators in New Zealand soils and land uses. *New Zealand Journal of Agricultural Research*, 47, 167-177.
Goulding, K.W.T., McGrath, S.P. and Johnston, A.E. (1989) Predicting the lime requirement of soils under permanent grasslands and arable crops. *Soil Use and Management*, 5, 54-58.
Hornung, M., Bull, K.R., Cresser, M., Hall, J.R., Langan, S.J., Loveland, P. and Smith, C. (1995) An empirical map of critical loads of acidity for soils in Great Britain. *Environ. Poll.*, 90, 301-310.
Kenney, E.A., Hall, J.W. and Wang, C. (2002) Temporal trands in soil properties at a soil qulaity indicator benchmark site in the Lower Fraser Valley, British Columbia. *Can. J. Soil Sci.*, 82, 499-509.
Koptsik, S., Berezina, N. and Livantsova, S. (2001) Effects of natural soil acidification on biodiversity in boreal forest ecosystems. *Water, Air, and Soil Pollution*, 130, 1025-1030.
MAFF (1981) *Lime and liming*. Ministry of Agriculture, Fisheries and Food Reference Book 35. HMSO, London.
MAFF (1988) *Fertiliser Recommendations*. Reference Book 209. HMSO, London.
MAFF (1993) *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Potentially Toxic Elements*. HMSO, London.
Makela-Kurtto, R. and Sippola, J. (2002) Monitoring of Finnish arable land: changes in soil quality between 1987 and 1998. *Agricultural and Food Science in Finland*, 11, 273-284.
McBride, M. B. (1994) *Environmental Chemistry of Soils*. Oxford University Press Inc. UK.
McTainsh, G. and Broughton, W.C. (1993) *Land degradation processes in Australia*. Longman Cheshire, Melbourne.
Moore, N.J. and Luoma, S.N. (1990) Hazardous wastes from large-scale metal extraction. *Environ. Sci and Technol.*, 24, 1278-1285.
Rengel, Z. (2003) *Handbook of Soil Acidity*. Marcel Dekker, New York.
Reuss, J.O. and Johnson, D.W. (1986) *Acid deposition and the acidification of soils and waters*. Springer-Verlag, New York.
Roem, W.J., Klees, H. and Berendse, F. (2002) Effects of nutrient addition and acidification on plant species diversity and seed germination in heathland. *J. App. Ecol.*, 39, 937-948.
Sarrantonio, M., Doran, J.W., Liebig, M.A. and Halvorson, J.J. (1996) On-farm assessment of soil quality and health. In: *Methods for assessing soil quality*. Eds J.W. Doran and A.J. Jones. SSSA Special Publication Number 49, Madison, Wisconsin. pp83-105.
Schipper, L.A. and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses. *Soil Sci. Soc. Am. J.*, 64, 300-311.
Schofield (1955) J. Soil Sci., 6, 137.
Schmieman, E., de Vries, W., Hordijk, L., Kroeze, C., Posch, M., Reinds, G.J., and van Ierland, E. (2002) Dynamic cost-effective reduction strategies for acidification in Europe: An application to Ireland and the United Kingdom. *Environmental Modeling and Assessment*, 7, 163-178.
Seybold, C.A., Grossman, R.B. and Pierce, F.J. (2003) On-site assessment of use-dependent soil properties in Michigan. *Commun. Soil Sci. Plant Anal.*, 34, 765-780.

Skinner, R.J. and Todd, A.T. (1998) Twenty-five years of monitoring pH and nutrient status of soils in England and Wales. *Soil Use and Management*, 14, 162-169.
Smith, J.L. and Doran, J.W. (1996) Measurement and use of pH and electrical conductivity for soil quality analysis. In: *Methods for assessing soil quality*. Eds J.W. Doran and A.J. Jones. SSSA Special Publication Number 49, Madison, Wisconsin. pp169-185.
Sparling, G, Lilburne, L and Vojvodic-Vukovic, M (2003). *Provisional target for soil quality indicators in New Zealand*. Landcare Research, Palmerston North, New Zealand.
Tisdale, S.L., Nelson, W.L., Beaton, J.D., and Halvin, J.L. (1993) *Soil Fertility and Fertilizers*, 5th edn. Prentice Hall, New Jersey, USA.
Truog (1946) *Soil Sci. Soc. Am. Proc*., 11, 206.
Webb, J., P.J., L., Chambers, B.J., Mitchell, R., and Garwood, T. (2001) The impact of modern farming practices on soil fertility and quality in England and Wales. *Journal of Agricultural Science*, 137, 127-138.
Wild, A. (1988) *Russell's Soil Conditions and Plant Growth*. 11th Edition. Longman group UK Ltd, Harlow.
Wilson, S.M., Pyatt, D.G., Malcolm, D.C. and Connolly, T. (2001) The use of ground vegetation and humus type as indicators of soil nutrient regime for an ecological site classification of British forests. *Forest Ecology and Management*, 140, 101-116.
Young, A. (2000) *Environmental Change in Australia since 1788*. 2nd Edition. Oxford University Press, UK.

2.3.2 Aggregate stability

Background

Aggregates are soil particles that are composed of smaller soil particles, which may themselves be aggregates. Aggregates range in size from microns to millimeters (Figure 2.3.2.1). A measure of the mean aggregate size that is present in a soil is termed Mean Weight Diameter (MWD) and a MWD greater than 2 mm is optimal for soil sustainability, both in terms of production and environmental well being (Sparling *et al*., 2003). The stability of aggregates and the pore spaces between them affects the movement and storage of water, aeration, erosion susceptibility, microbial activity and the growth of crops. Thus, good aggregate stability is important to a wide range of physical and bio-chemical processes in natural and agricultural environments. (Amezketa, 1999).

Micro aggregates <2 μm diameter
(clays attached to organic molecules by polyvalent cations)

↓

Micro aggregates <250 μm diameter
(aggregation of micro aggregates <2 μm diameter)

↓

Macro aggregates >250 μm diameter
(aggregation of micro aggregates <250 μm diameter)

↓

Peds mm to cm in diameter
(bonding of macro aggregates)

Figure 2.3.2.1: Hierarchical order of soil aggregation (Amezketa, 1999)

Aggregates are quasi-permanent units of soil structure that exist in the soil despite seasonal modification by weathering and the action of shorter-term disruptive forces such as swelling, impact of raindrops, mechanical overloading (compaction and sheer) and creep (flow of water under its own weight) (Matkin and Smart, 1987). A myriad of processes continuously act together in the aggregation and stabilization process. Some primary factors are clay flocculation, wet-dry/freeze-thaw cycles, the compressive and drying action of roots and the action of earthworms. Inorganic stabilizing agents include mainly clays, cations such as Ca^{2+}, Fe^{3+}, and Al^{3+}, oxides and hydroxides of Fe and Al and Ca and Mg carbonates. Organic stabilizing agents can be considered in three main groups (Amezketa, 1999; Sollins *et al*., 1996):

(1) transient binding agents are decomposed rapidly by microorganisms and include microbial and plant derived polysaccharides;
(2) temporary binding agents are roots, hyphae and some fungi;
(3) persistent binding agents consist of resistant aromatic humic material associated with metal cations and strongly sorbed polymers, which are derived from the resistant fragments of roots, hyphae, bacteria cells and colonies.

If aggregate stability and MWD deteriorate there is a greater risk of capping, overland flow and erosion of surface soil. Breakup of soil aggregates can lead to oxidation and subsequent decrease in SOC. Other effects include a reduction in mean pore size and less plant-available water, decrease in plant growth, and decrease in microbial biomass and activity.

Links with other measurable soil parameters

Deterioration of aggregate stability will lead to a decrease in MWD, an increase in bulk density and a decrease in macroporosity. Apart from these physical soil properties, aggregate stability also has mutual links with chemical and biological soil parameters.

Increased SOC levels, particularly light fraction SOC from manure amendments or crop residues, are known to improve aggregate stability, MWD and overall soil structure (Loveland and Webb, 2003; Webb *et al*., 2003; Tripathy and Singh, 2004). However, Amezketa (1999) noted that aggregate stabilisation from SOC is concerned with macro aggregates (>250 μm), with SOC acting as a deflocculent on micro aggregates. Clay mineralogy is another important aggregating factor. The physiochemical effect of different clays affects aggregation because of the variation in specific surface area, Cation Exchange Capacity (CEC) and consequently, high physiochemical interaction capacity. Some clay mineralogies are more sensitive to chemical dispersion because surface irregularity and smaller edge to face attraction forces, such as illite. (Amezketa, 1999). Soil pH also acts as a primary factor controlling clay dispersion/flocculation (Amezketa, 1999).

Biological indicators, such as nutrient mineralization, are also influenced by a decline in aggregate stability (Webb *et al*., 2003). In particular, disaggregation of macro-aggregates reduces macroporosity, which in turn causes an associated decrease in microbial biomass and activity (Sparling *et al*., 2000; Neves *et al*., 2003). However, any form of soil perturbation that results in a short-term increase in soil aeration actually increases oxidative microbial activity and leads to a reduction in SOC (Chantigny, 2003). The decline of SOC through the oxidation process is further enhanced when disaggregation allows microbes to access SOC previously bound within inaccessible micropores (Sollins *et al.* 1996; Martens *et al*., 2003). Obviously the loss of SOC has long-term destabilising implications for soil aggregates. Microbial activity is only checked when aggregate stability and macroporosity have deteriorated to the extent that remaining pore spaces are too small to be accessible to bacteria (Sollins *et al*., 1996).

Pressures on soil aggregate stability

Mechanical factors (tillage and poaching) have obvious implications for soil disaggregation, especially in conditions of high soil moisture. However, tillage in particular affects aggregate stability by:

- redistribution of organic matter and microbial activity
- soil solution effects
- soil fauna effects.

Tillage breaks down aggregates through mechanical stress. This exposes SOC to microbes, with the SOC subsequently mineralized and lost. Progressive deepening of tillage can dilute topsoil SOC by mixing with subsoil horizons low in organic matter (Amezketa, 1999). Tillage effects can be countered to a degree by long-term and regular manure amendments, aiding greater longevity of macro-aggregates (Williams, 2004). Conservation tillage also conserves SOC, microbial biomass and general soil structure (Amezketa, 1999).

Usefulness of aggregate stability as an indicator of soil quality

The aim of the aggregate stability test is to give a reliable description and ranking of the behaviour of soils under the effect of different environmental conditions and managements (Amezketa, 1999). To be included in a MDS of soil quality indicators for the UK, a measure of aggregate stability would need to demonstrate interpretable variation between and within soil and land use groupings. However, there is much confusion in methodology and terminology used to determine and present aggregate stability findings in the literature. With no single method accepted for the measure of aggregate stability, it has not and still is not included in standard soil assessments. To date, most aggregate stability testing has generally only been applied to arable soils.

Numerous methods have been used to determine aggregate stability with varying success (notably variations in MWD, water stable aggregation (WSA) which is the weight of stable aggregates per kg of soil, dispersivity test and raindrop). These methodologies assess ability to resist slaking (the compression of air trapped by infiltrating water air) or rainsplash but do not account for mechanical strength or biochemical factors. Variations of the wet sieving method (which involves saturation and shaking of aggregates >4 mm over sieves of varying aperture down to 0.5 mm) used to derive MWD or WSA are the most commonly used approaches. Full descriptions of the various methodologies are described in ADAS (1977), Amezketa (1999) and Matkin and Smart (1987). The use of different methodologies and variations on methodologies has made comparison of data difficult (Box). For the same reason it has also been difficult to obtain a consistent correlation between aggregate stability and other important soil factors such as erodibility and crusting potential (Amezketa, 1999).

Although the current shortcomings in aggregate stability methodology are many, if a single aggregate stability method was decided upon it might be worthy of consideration for the MDS, not least because the quasi-permanent nature of soil aggregates may be a more sensitive indicator of soil physical quality than the more ephemeral properties of soil bulk density or macroporosity. Other advantages of aggregate stability are that it is usable for stony soils and, like the collection of bulk density samples that do not have to remain intact, the collection of soil for aggregate stability tests does not require great care in sample collection. Importantly, if a single, reliable method of aggregate stability was adopted for the MDS it could provide an extremely useful indicator of erosive and crusting potential of the soil, which is otherwise measured in the field using resource-intensive rainfall simulation apparatus.

Sparling *et al*. (2003) found that MWD aggregate stability is a useful measure of soil physical condition under cropping and horticulture and may be a more useful indicator of soil physical quality than macroporosity in cultivated or stony soils.

The reason for the existence of so many different methods to measure aggregate stability may be explained by:

1. the existence of different mechanisms that produce destabilization such as:
    a) slaking (of macro aggregates >250 µm);
    b) time-dependent clay dispersion (of microaggregates <2 µm) by chemical processes;
    c) clay swelling;

2. the different aggregate size (micro to macroaggregates) and spatial (lab to field) scales at which stability is determined;

3. methodological and analytical differences, such as:
    a) conditions of sample collection;
    b) pre-treatment of sampled soils that determine the structure, size and moisture contents of aggregates treated;
    c) treatments applied to the aggregates, such as the wetting technique, rate of wetting, time of shaking the sample and raindrop impact, which present different degrees of disruptive energy applied;
    d) measurement of disaggregation and dispersion, which is often not clearly distinguished from the treatment;
    e) expression of the result in terms of stability parameter. The most commonly used parameter for stability of macroaggregates is water stable aggregates. In reality, a useful measure of aggregate stability should not really discriminate between macro and microaggregates as both are important in sustaining soil structure.

Terminology is another source of confusion. Tests to determine the stability of macroaggregates are often termed aggregate stability tests, whereas tests to determine the stability of microaggregates are termed clay dispersion tests (Amezketa, 1999).

They found that macro aggregate stability varies greatly under different cropping systems in decreasing order of virgin > grass and forage crops > rotation agriculture > mono-agriculture > fallow (Amezketa, 1999). Shulka *et al.* (2003) found a no-till plus manure treatment to have a positive effect on MWD and WSA aggregate stability under long term (25 years) research into a range of tillage treatments on corn. However, it is recommended here that neither WSA nor MWD are included in the MDS. These resource-intensive methods are particularly destructive to the aggregates and results do not correlate well with more simple tests of aggregate stability (ADAS, 1982; Matkin and Smart, 1987). Also, the outcomes of these methods, in particular, are strongly influenced by seasonality; the reliability of results decreases with storage time (immediate testing on sample collection produces the most reliable results); and, if stored, results are strongly affected by prepared moisture content of the aggregates on testing (Matkin and Smart, 1987). Importantly, the single type of disintergrating force represented by the method is not universally applicable to all soils (ADAS, 1977).

Instead, it is recommended that the more simple and resource-friendly dispersion ratio (DR) is used. The DR method measures the proportion of total silt and clay which is dispersed as a result of slaking and mild applied forces and is most useful for assessing field situations where damaged and undamaged areas occur within the same soil type. It is universally applicable and can be used to measure the general stability of soils under different management conditions (ADAS, 1977). The DR is calculated by expressing the ratio of silt plus clay contents obtained by mild dispersion in distilled water (M) and drastic dispersion in sodium hexametaphosphate solution (D) where $DR = M/D*100$. However, it is noted that the practical significance of the results must be interpreted in relation to soil type, drainage, climate and field experience.

The chosen scale and level of statistical interpretation, whether national, regional or local, will dictate the importance attached to the location and timing of sample collection. In terms of sampling location, one main difficulty in measuring soil physical parameters is that they are spatially heterogeneous. For example, processes of compaction and disaggregation induced by machinery or livestock do not necessarily have a uniform effect over the whole topsoil area and volume. Topographic location (such as position on hillslope) also has important implications on soil physical properties due local variation in clay and organic matter content (Bissonnaise *et al*., 2002). Once a sampling site has been selected, as for bulk density measurements, samples for aggregate stability should be taken in the spring when processes of freezing/thawing and clay swelling have occurred and the soil is at its maximum field volume. The sampling depth is also an important consideration. As previously mentioned, aggregate stability has important implications for soil crusting and associated erosion problems. For this reason it is recommended that soil from the top 0-5 cm of the soil is most relevant for aggregate stability determination. It would be sensible if this was supported by a second measure from the middle of the topsoil.

What does change mean?

A change in aggregate stability over time represents a change in the fundamental factors affecting soil structure. There are many factors that may impact on structural stability, but some of the primary factors will include a change in crop type, a change in management practice, a change in soil pH or a change in SOC. More indirectly, a change in aggregate stability will infer a change in other physical parameters such as macroporosity, hydraulic conductivity and water release curve and might also infer a change in chemical and biological parameters such as SOC and mineralizable N, respectively.

What are the rates of change?

Apart from influencing factors such as climate, we would expect aggregate stability to remain unchanged in non-managed systems. However, aggregate stability will respond to perturbation by livestock and cultivation. It will also change due to natural amelioration processes that occur with time after perturbation, such as consolidation of loose aggregates, wetting and drying and freeze-thaw cycles, worm activity and root activity (Amezketa, 1999; Shulka *et al*., 2004).

Measurements of aggregate stability will be strongly influenced by timing in relation to livestock grazing, cultivation, soil moisture conditions, grazing/cultivation history and depth of topsoil sampled. Consideration of these factors is critical in terms of interpreting long-term soil quality trends against background noise

Is change measurable?

Measurable change has been demonstrated for long-term experiments (25 years) in tilled systems (e.g. Amezketa, 1999; Shulka *et al*., 2003, 2004).

Trigger values for aggregate stability

In previous attempts to define sustainability thresholds for aggregate stability Sparling *et al*., (2003) reported that a MWD of 2 mm is considered to be optimum in terms of productivity and environmental well being. A deterioration of MWD to below 2 mm, or even 1 mm, was considered to be more an environmental than a productivity concern.

After assessment of available literature, Shukla *et al.* (2004) defined sustainability descriptors (productivity and environmental factors) for silt-loam cropping systems in the USA. The sustainability index was based on the method developed by Lal (1993) for tropical soils. The index of Shulka *et al.* (2004) is for both WSA and MWD and is shown in Table 2.3.2.1.

Table 2.3.2.1: Water stable aggregate and mean weight diameter described in terms of a sustainability index for tilled, silt-loam soils in the USA (Shulka *et al*., 2004)

Sustainability	WSA ($g\ kg^{-1}$)	MWD (mm)
Highly sustainable	>700	>2.5
Sustainable	450-700	2-2.5
Sustainable with high management input	250-450	1-2
Sustainable with another land use	150-250	0.5-1
Unsustainable	<150	<0.5

However, in terms of the simple and universally applicable DR method for the interpretation of DR, values are shown in Table 2.3.2.2. Although there are seven broad stability categories, the index is sufficient to indicate improving or deteriorating trends within each category. By considering the functions that are required for the soil to perform and also the connectivity between the indicator and those functions, it is possible to produce a table based on broad soil types. This has been attempted in Table 2.3.2.3. It should be noted that there is no differentiating between broad soil types in the DR because consideration of soil type is integrated within the DR method.

Table 2.3.2.2: Dispersion ratios and corresponding interpretation

Dispersion ratio	Interpretation
<5	Very stable
6-10 stable	Stable
11-15	Fairly stable
16-25	Somewhat unstable
26-30	Unstable
>31	Very unstable

Table 2.3.2.3: Dispersion ratio triggers that indicate problematic conditions for different habitat types.

Function	Soil type		
	Mineral	Peaty	Calcareous
Surface crusting		>11	
Compaction		>15	
Decreased porosity		>15	
Habitat support table			
Calcareous grassland		>11	
Mesotrophic grassland		>11	
Acid grassland		>11	
Dwarf shrub heath		>11	
Biomass production			
Arable & horticultural		>15	
Improved grassland		>15	

Integration of aggregate stability with other national monitoring schemes

There are currently no nationally applicable datasets relating to aggregate stability across all land use and soil types, as most aggregate stability experimentation to date has focused on arable systems. However, if use of a robust, consistent and interpretable method such as the DR method was agreed then the measure would add value to existing national soil monitoring schemes.

Cost and suitability for broad-scale monitoring

It is not possible to predict the potential use of aggregate stability in a broad-scale monitoring scheme as there are currently no existing datasets to assess its usefulness across different land and soil groups. The wet sieving methodology would incur considerably more cost than bulk density or macroporosity tests. The simple DR method is quick, cheap and has universal application.

Conclusion and recommendation

It is recommended that the DR method is accepted as a national indicator for soil physical quality and for soil aggregate stability in particular. It could possibly be a more sensitive indicator of long-term change than either macroporosity or bulk density, depending on its relative ability to resist rapid temporal changes. A measure of aggregate stability would also offer a better assessment of soil physical quality in stony soils.

However, as with all soil quality indicators, the sampling protocol must be carefully designed to answer questions that will be posed with the future dataset. On a national or regional scale, collection of sufficient data will enable broad-scale changes in soil quality. However, at the local scale, the sampling protocol must be sufficiently careful and robust to minimise variation within and across sampling sites.

References

ADAS (1977) *Techniques for measuring soil physical properties*. MAFF, London.

Amezketa, E. (1999) Soil aggregate stability: A review. *Journal of Sustainable Agriculture*, 14, 83-151.

Bissonnaise, Y.E, Cros-Cayot, S and Gascuel-Odoux, C. (2002) Topographic dependence on aggregate stability, overland flow and sediment transport. *Agronomie*, 22, 489-501.

Chantigny, M.H. (2003) Dissolved and water extractable organic matter in soils: a review on the influence of land use and management practices. *Geoderma*, 113, 357-380.

Lal, R. (1993) Tillage effects on soil degradation, soil resilience, soil quality, and sustainability – introduction. *Soil and Tillage Research*, 27, 1-8.

Loveland, P and Webb, J. (2003) Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research*, 70, 1-18.

Martens, D.A. (2000) Management and crop residue influence soil aggregate stability. *J. Environ. Qual*., 29, 723-727.

Matkin, E.A and Smart, P. (1987) A comparison of tests of soil structural stability. *Journal of Soil Science*, 38, 123-135.

Neves, C.S.V.J, Feller, C, Guimaraes, M.F, Medina, C.C, Tavares, J and Fortier, M. (2003) Soil bulk density and porosity of homogeneous morphological units identified by the Cropping Profile Method in clayey Oxisols in Brazil. *Soil and Tillage Research*, 71, 109-119.

Shukla, M.K, Lal, R and Ebinger, M. (2003) Tillage effects on physical and hydrological properties of a typic aggriquoll in central Ohio. *Soil Science*, 168, 802-811.

Shukla, M.K, Lal, R and Ebinger, M. (2004) Soil quality indicators for the north apalachian experimental watersheds in Coshocton, Ohio. *Soil Science*, 169, 195-205.
Sollins, P, Homannn, P and Caldwell, B.A. (1996) Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74, 65-105.
Sparling, G.P, Shepherd, T.G and Schipper, L.A. (2000) Topsoil characteristics of three contrasting New Zealand soils under four long-term land uses. *New Zealand Journal of Agricultural Research*, 43, 569-583.
Sparling, G, Lilburne, L and Vojvodic-Vukovic, M. (2003) *Provisional target for soil quality indicators in New Zealand*. Landcare Research, Palmerston North, New Zealand.
Tripathy, R and Singh, A.K. (2004) Effect of water and nitrogen management on aggregate size and carbon enrichment of soil in rice-wheat cropping systems. *Journal of Plant Nutrition and Soil Science*, 167, 216-228.
Webb, J, Bellamy, P, Loveland, P.J and Goodlass, G. (2003) Crop residue returns and equilibrium soil organic carbon in England and Wales. *Soil Science Society of America Journal*, 67, 928-936.
Williams, J.D. (2004) Effects of long term winter wheat, summer fallow residue and nutrient management on field hydrology for a silt loam in north-central Ohio. *Soil and Tillage Research*, 75, 109-119.

2.3.3 Soil bulk density

Background

Soil bulk density (BD) is the single most useful parameter of soil physical structure. It is a direct measure of soil compaction (or loosening) and is essential to assess total available pore space within a soil (that is, total porosity) (Hernanz *et al*., 2000). Bulk density is an excellent measure of a most important contemporary form of soil degradation: that which occurs due to ill-timed cultivation, trafficking and stocking. Packing density (PD) - a derivative of BD calculated as PD = BD*0.009(% clay) - is a better parameter than BD for comparison of physical structure between different soils and, based on derived statistics, affords an excellent indirect estimation of soil porosity (Hall *et al*., 1977).

A good level of volumetric pore space is fundamental for sustainable soil use, both in terms of productivity and environmental well being. The pores hold water and air, which are essential for all functions that support biomass production, and they also allow water and air to move through the soil. There are a myriad of problems associated with severely compacted soils: loss of macropores (and subsequent increase in micropore volume); less plant-available water; poor aeration; slow drainage; reduced infiltration and water storage; increased overland flow; increased topsoil erosion; retarded root development; and reduced herbage yield (Chen *et al*., 1998; Hernanz *et al*., 2000; Neves *et al*., 2003). At the other extreme, soils with low bulk density and strength, although open-textured and porous, are susceptible to erosion, poor water retention and oxidation of soil organic matter (and associated loss of SOC) (Sparling *et al.*, 2003).

Links with other measurable soil parameters

Total pore space (the volumetric proportion of all micro- (<60 μm) and macro- (>60 μm) pores in the soil) typically comprises about half a soil's volume. Bulk density accounts for much of the variance in soil total porosity, with clay and organic matter content accounting for much of the remaining variance (Hall *et al*., 1977). If total porosity is reduced, either through direct compaction or through mechanical, chemical or biological breakdown of soil aggregates, bulk density will increase. Bulk density is an important measure in its own right, but it is also critical for the determination of a wide spectrum of soil physical attributes, e.g. hydraulic conductivity, total porosity, macroporosity, solute diffusion/transport and assessments of soil strength for crop production potential (Chen *et al*., 1998). It is, however, an insensitive measurement of pore size distribution and pore continuity parameters, which are critical factors affecting soil hydrology and biochemistry (Hernanz *et al*., 2000). Indeed, changes in pore size distribution and continuity (e.g. through tillage) can vary greatly over time while measurements of bulk density remain constant.

Bulk density is also a useful indicator of change for other soil parameters. For example, an increase in bulk density might indicate a decrease in soil aggregate stability and aggregate size, particularly due to the breakdown of soil macro aggregates (>60 μm) (Amezketa, 1999). Apart from physical soil properties, bulk density also has mutual links with chemical and biological soil parameters. Indeed, for all soil groups in England and Wales, Loveland and Webb (2003) found that increased SOC levels improved soil structure by decreasing bulk density, improving aggregate stability, increasing pore size and an increasing the proportion of air-filled pore space. Similarly, Chen *et al.* (1998) found that soil clay content and soil organic matter (SOM) were primary factors affecting topsoil bulk density (0-30 cm) for a range of cultivated soils. In terms of biological indicators, such as mineralizable nitrogen, any decline in soil structure, particularly macropore volume, is also frequently associated with decreases in microbial biomass and activity (Sparling *et al*., 2000; Neves *et al*., 2003). For example, it was reported by Sollins *et al.* (1996) that 95% of the pore spaces in a degraded silt-loam were too small to be accessible to bacteria.

Pressures on soil bulk density

For arable systems in the UK, where up to 90% of a field may be crossed by wheelings in the course of a single year due to ploughing, seed bed preparation and fertiliser and pesticide application, the potential for compaction is a serious problem (Arden-Clarke and Hodges, 1987). Such practices mostly have a negative effect on soil structure, with loss of macropore space in the top 30 cm being a primary symptom (Chen *et al*, 1998; Neves *et al*., 2003). Tillage increases the volume of macropores over a short duration whilst decreasing the continuity of macropores and micropores. However, over a longer duration, tillage generally increases bulk density, decreases water stable aggregation and reduces plant-available water capacity and infiltration (Shulka *et al.*, 2003). Regular tillage can also lead to a decline in SOC through oxidation and a minimum return of crop residue (Loveland and Webb, 2003). Any such decline in SOC may impact on soil structure, particularly macroporosity and bulk density due to a decline in aggregate stability and aggregate size. Indeed, minimum tillage practice and regular organic amendments that retain and increase SOC have been shown to have a positive effect on soil structure (Amezketa, 1999).

In grassland systems, poaching from sheep and cattle in particular can cause physical degradation of soil, even between seasons. This is more likely with high stocking rates and can result in decreased soil permeability due to reduced pore space and continuity (Drewry *et al.,* 2000). The impacts of poaching are compounded in wet soil conditions (Drewry and Paton, 2000). However, for soils in the UK, Scholefield and Hall (1986) found that the loss of soil structure due to poaching occurs as a progressive degradation and not immediately on treading of wet soil. Previous work by Scholefield and Hall (1985) found that poaching is also influenced by clay content, bulk density, sward strength, weather and management. Most structural damage and compaction from poaching occurs at 10 cm below the soil surface and poaching may even reduce bulk density in the top 10 cm (Scholefield *et al*., 1985).

Usefulness of bulk density as an indicator of soil quality

To be included in a MDS of soil quality indicators for the UK, bulk density will need to demonstrate sufficient variation between and within soil and land use groupings and be interpretable indicator of change. To this end, the derived measure of packing density may be more applicable for comparison of bulk density between different soils, as it allows for the effect of clay content. However, the usefulness of bulk density in its own right has been reported in regional and national scale studies. From a long-term study of cultivated soils in Illinois, USA, Wander and Bollero (1999) recommended that bulk density should be included in a refined set of soil quality parameters because of its sensitivity to soil type and management and also its environmental relevance. In New Zealand, Drewry *et al*. (2000) reported that bulk density under grazed pasture showed significant differences both between soil groups and for rapid seasonal changes within soil groups. Also in New Zealand, but for silt-loam soils only, Sparling *et al.* (2000) found bulk density between land uses to decrease in the order arable > pine forest > pasture > native forest.

Method to determining bulk density

The laboratory determination of soil bulk density is a simple and robust procedure (Hall *et al*., 1977). Intact soil cores are collected and the known volume of soil in the cores is dried at 105^{o}C, and weighed. Bulk density is the mass per unit volume, usually expressed as Mg/m^3. (Sparling *et al.*, 2003). Advantages of including bulk density in the MDS is that it is cheap and quick to measure and, if intact cores are retained, it can be conveniently combined with measurements of hydraulic conductivity, moisture release characteristics and total and macroporosity. A disadvantage of collecting intact cores is that great care is needed to avoid damage and loss of soil volume during sample extraction, transport and handling (Hernanz *et al*., 2000). However, there is no need to retain an intact core for the sole determination of bulk density. A notable disadvantage is that bulk density measurements may not be appropriate for stony soils. One possible technique is to use a replacement method, which is pouring a known volume of sand (or plastic beads) into an excavated hole. The bulk density of the fine soil can then be calculated from the total weight of the sample and the weight of the stones (ADAS, 1977; Hall *et al*., 1977).

Sampling protocol for bulk density is extremely important in terms of data interpretation. The chosen scale and level of statistical interpretation, whether national, regional or local, will also dictate the level of importance attached to the location and timing of sample collection. In terms of sampling location, one main difficulty in measuring physical parameters such as macroporosity is that they vary in position in the soil profile (Neves *et al*., 2003). For example, processes of compaction and disaggregation induced by machinery or livestock may be spatially heterogeneous. This heterogeneity is rarely considered but will result in considerable variation in measurements in time and space. Topographic location (e.g. position on hillslope) may also affect macroporosity due local variation in clay and organic matter content (Bissonnaise *et al*., 2002). Once a sampling site has been selected, consideration should be given to temporal resolution of sampling and the temporal variation that can occur in bulk density measurements.

The sampling depth is also an important consideration for bulk density in terms of identifying change. Sparling *et al.* (2000) noted that the top 10 cm of the soil that is most relevant for biological and some chemical characteristics, accounting for the highest nutrient concentrations and over 80% of SOM and microbial biomass and activity. Therefore, what depth should be sampled? In cultivated systems that undergo primary tillage, secondary seed bed preparation and further traffic from fertiliser/pesticide application and harvesting, soil compaction can affect all soil from 0-30 cm (Chen *et al.*, 1998; Hernanz *et al.*, 2000). Brejda *et al*. (2000a, 2000b) and Shulka *et al.* (2003, 2004) selected a less than 10 cm sampling depth for cultivated systems in the USA. Other researchers have selected 0-30 cm sampling depths for detailed investigation of cultivation-induced compaction. However, it is likely that the 0-10 cm horizon is sufficient to indicate bulk density change in conventional and minimum tilled systems. On arable land, cultivation and subsequent settling and slaking of the soil, along with the effects of various episodes of traffic during crop management, will increase bulk density in and sometimes below the tilled horizon (Harrod, *Pers. Comm.*). In grazed systems of New Zealand, researchers report that soil physical damage incurred by poaching is limited to the surface 10 cm (Drewry *et al*., 2000; 2001). However, for UK conditions, Scholefield *et al.* (1986) found that most compaction occurred at 10 cm depth and below.

It appears that no single depth can provide a definitive picture of physical soil change. The horizon most susceptible to physical change will vary between soils and management. In particular, the top 10 cm may be too ephemeral to be interpreted as a SQI. Confining what is said about soil bulk density to the near surface will make it hard to arrive at a meaningful interpretation of soil quality. For example, most plants root to about one metre depth and water from this depth may be vital for plant sustainability during dry periods. Expert advice advocates the importance of taking three depth measurements for bulk density: 0-10 cm, middle of the topsoil and top of the subsoil (M.J.Reeve and T. Harrod, *Pers. Comm.*).

What does change mean?

A change in soil bulk density over time represents a change in soil compaction or loosening and an associated decrease or increase in total porosity (particularly macroporosity), respectively. More indirectly, a change in bulk density might also imply a change in other physical parameters, such as macroporosity, aggregate stability, aggregate size, hydraulic conductivity, water release curve or chemical and biological parameters such as SOC and mineralizable N, respectively.

When considering broad-scale national or regional changes for different soil and land use combinations, bulk density is likely to provide a useful indicator of change. However, we must consider whether we intend to investigate local-scale as well as national/regional-scale changes in soil quality. If so, the national monitoring strategy will have to be designed accordingly.

What are the rates of change?

Topsoil bulk density can respond rapidly to perturbation by livestock and cultivation, most notably under wet soil conditions, and it can change seasonally due to natural amelioration processes that occur with time after perturbation, such as consolidation of loose aggregates, wetting and drying and freeze-thaw cycles, worm activity and root activity (Drewry *et al*., 2001; Shulka *et al*., 2004). These natural amelioration processes will vary depending on climate and soil properties.

Poaching is particularly degrading under high stocking densities and/or wet soils (Drewry *et al*., 2001; Drewry and Paton, 2000; Singleton *et al*., 2000). Drewry and

Paton (2000) reported that a reduction in grazing, or an omission of grazing, from months to a few years is sufficient to improve soil physical structure. Similar temporal variation is also reported for cropped systems where changes in topsoil physical properties (0-30 cm) can be rapid and highly variable (Shulka *et al*., 2003; 2004). Lodgson and Cambardella (2000) reported statistically significant changes in soil bulk density in the top 20 cm (random increases and decreases, ranging from 0.91 to 1.44 Mg/m^3), both between and within successive sampling years. However, Chen *et al*. (1998) noted that changes would obviously be less pronounced under minimum tillage. Under UK conditions, which you could argue are marginal for minimum tillage, it is not unknown for a compacted or smeared pan to be produced under the thin tillage horizon.

Therefore, measurements of bulk density will be strongly influenced by depth of topsoil sampled and timing in relation to livestock grazing, cultivation, soil moisture conditions, grazing/cultivation history and natural amelioration processes. Consideration of these factors is critical in terms of interpreting long-term soil quality trends against background noise. In terms of sample timing for the MDS, it is recommended that soil bulk density is measured in the spring when the soil is at its maximum field volume.

Is change measurable?

Researchers have found bulk density to be a statistically useful indicator of soil quality change at a regional (Brejda *et al*., 2000a, 2000b; Sparling *et al*., 2000) and national scales (Sparling and Schipper, 2002, 2004). However, in terms of local mitigation, an important question is how useful bulk density will be as an indicator of long-term change at a specific location. This will depend on the robustness and consistency of sampling protocol in terms of temporal and spatial factors.

Trigger values for soil bulk density

A healthy soil bulk density is equally important in terms of sustainable soil productivity and environmental well being. In an applied study of bulk density as an indicator of soil physical quality in New Zealand, Sparling *et al*. (2003) defined bulk density trigger values for each of four major soil groupings and a group of all other soils. There was no accounting for land use in the grouping of data. The trigger values are shown in Table 2.3.3.1. Acceptable bulk density was defined as the range between the values denoted in bold type data – that is, between 'no significant impact' and 'above target range'. The data suggests that, with the exception of organic soils, sustainable bulk density ranges are generally consistent across soil groups. These groupings are agreeable with but just a little low for UK soil.

Table 2.3.3.1: Provisional quality classes and target ranges for bulk density – applicable to all land uses (Sparling *et al*., 2003)

Soil group	Bulk density (Mg/m^3)					
	Very loose	Loose	Accept-able	Compact	Very compact	
Semi-arid, pallic and recent soils	0.3	**0.7**	0.9	1.3	**1.4**	1.6
Allophanic soils		**0.5**	0.6	0.9	**1.3**	
Organic soils		**0.2**	0.4	0.6	**1.0**	
Pumice and podzol		**0.6**	0.7	1.2	**1.4**	1.6
All other soils	0.3	**0.6**	0.8	1.2	**1.4**	1.6

Table 2.3.3.2: Bulk density data for major land use classes in New Zealand (Schipper and Sparling, 2000; Sparling and Schipper, 2004)

Land use description	Soil description	Bulk density (Mg/m^3) (mean ±S.E)	Bulk density (Mg/m^3) (range)
Natural forest	All soils	0.76 (0.04)	
Natural forest	Silt-loam		0.57–1.08
Pine forest	All soils	0.78 (0.03)	
Pine forest	Silt-loam		0.71–0.88
Dairy pasture	All soils	0.82 (0.02)	
Drystock pasture	All soils	0.91 (0.02)	
Tussock grassland	All soils	0.95 (0.03)	
Pasture	Silt-loam		0.72–0.82
Arable crop	All soils	1.07 (0.04)	
Mixed crop	All soils	1.22 (0.04)	
Arable	Silt-loam		0.89–1.21

More detailed analysis of the same dataset showed bulk density data in relation to land use for all soils (Sparling and Schipper, 2004) and for silt-loam soils only (Sparling *et al*., 2000). The data is shown in Table 2.3.3.2. The data ranges are particularly limited, even though it is largely only silt-loam soils that are considered. Nonetheless, the data suggests that land use will have a strong influence on soil bulk density. In particular, data for all soils shows bulk density decreases in the order arable > pasture > pine and natural forest. Data for silt loam suggest bulk density decreases in the order arable > grassland, pine and natural forest. The standard errors and ranges are small, suggesting little variation within land use groupings. All bulk density values also fall within the ranges defined as acceptable for most soil types in Table 2.3.3.1.

After assessment of available literature, Shukla *et al.* (2004) defined sustainability was based on the method developed by Lal (1993) for tropical soils. The index of Shulka *et al.* (2004) is shown in Table 2.3.3.3. Also, the sustainable bulk density of 1.5 Mg/m^3 in Table 2.3.3.3 would be defined as very compact and unsustainable using the New Zealand and UK conditions (Harrod, *Pers. Comm.*). Indeed, for UK conditions one would expect severe hydrological degradation of a soil with bulk density >1.5 Mg/m^3. For arable cropping on silt-loam topsoil in New Zealand, Sparling *et al.* (2000) found that descriptors (productivity and environmental factors) for silt-loam cropping systems in the USA. The sustainability index an increase in bulk density to 1.21 Mg/m^3 defined a point at which point macroporosity and hydraulic conductivity were reduced and root growth was starting to be inhibited.

Table 2.3.3.3: Bulk density sustainability index for tilled, silt-loam soils in the USA (Shulka *et al*., 2004)

Sustainability	Bulk density (Mg/m^3)
Highly sustainable	<1.5
Sustainable	1.5–1.55
Sustainable with high management input	1.55–1.6
Sustainable with another land sue	1.6–1.65
Unsustainable	>1.65

Bulk density data exists for all soils in England and Wales. This data was collected in the 1960-80s and resides in various databases. The NSRI holds a lot of this on the LandIS information system. Summary statistics of topsoil bulk density for major soil textural classes and land use categories are shown in Table 2.3.3.4. Regardless of soil textural group, arable soils consistently have a higher bulk density than the other land uses. Bulk density tends to decrease in the order sand > clay > loam soils (Table 2.3.3.4).

By consideration of the functions that are required for the soil to perform and further consideration of the connectivity between the indicator and those functions, it is possible to produce a table based on broad soil types. An example is given in Table 2.3.3.5 with potential trigger values derived from a range of sources based on review of the data in Tables 2.3.3.1 to 2.3.3.4.

Table 2.3.3.4: Summary statistics of bulk density for major soil textural groups and land use categories in England and Wales

Textural class	Statistic	Bulk density under different land use (Mg/m^3)		
		Arable	Grassland	Other
Clay	*n*	432	470	72
	range	0.43-1.75	0.43-1.74	0.54-1.42
	mean	1.24	1.22	1.09
	SE	0.01	0.01	0.02

Textural class	Statistic	Bulk density under different land use (Mg/m^3)		
		Arable	Grassland	Other
Clay-loam	*n*	115	118	112
	range	0.83-1.49	0.77-1.27	0.61-1.28
	mean	1.24	1.06	1.01
	SE	0.01	0.01	0.01
Silty-clay-loam	*n*	39	44	84
	range	0.83–1.42	0.58–1.15	0.71–1.4
	mean	1.21	0.96	1.09
	SE	0.02	0.02	0.02
Silty-clay	*n*	7	11	19
	range	0.78–1.23	0.67–1.23	0.25–1.26
	mean	0.99	0.9	0.92
	SE	0.07	0.05	0.06
Silt-loam	*n*	8	10	21
	range	1.16–1.47	0.91–1.14	0.76–1.34
	mean	1.29	1.02	1.04
	SE	0.04	0.02	0.03
Sandy-clay-loam	*n*	16	16	30
	range	0.9–1.42	0.78–1.35	0.86–1.47
	mean	1.23	1.11	1.22
	SE	0.04	0.04	0.03
Sandy-loam	*n*	82	101	181
	range	0.87–1.57	0.6–1.36	0.54–1.52
	mean	1.35	1.12	1.19
	SE	0.01	0.01	0.01
Sand	*n*	7	6	16
	range	1.11–1.57	0.88–1.22	0.77–1.42
	mean	1.37	1.1	1.12
	SE	0.06	0.05	0.05

Table 2.3.3.5: Proposed critical bulk density (Mg/m^3) thresholds for broad soil and habitat groupings in England and Wales

Function	Soil type		
	Mineral	Peaty	Calcareous
Habitat support table			
Calcareous grassland			>1.3
Mesotrophic grassland	>1.3	>1	
Acid grassland	>1.3	>1	
Dwarf shrub heath	>1.3	>1	
Biomass production			
Arable & horticultural	>1.3	>1	>1.3
Improved grassland	>1.3	>1.3	>1.3

Integration of bulk density within other national monitoring schemes

Bulk density is one of the foremost indicators of soil physical quality and is one of the principal parameters in existing soil datasets for England and Wales. Most bulk density data is determined using the simple dry weight bulk density method; thus, new data will be readily comparable with old. Therefore, bulk density could be easily integrated and cross-compared with existing and future datasets.

Cost and suitability for broad-scale monitoring

Bulk density is a simple and robust method that is cheap in terms of both collection and analytical determination. There is one generally accepted measure of bulk density - dry weight bulk density - so there is no scope for confusion or deliberation in selecting the best technique to take forward into a national monitoring program. The collection of intact cores and a measure of bulk density also complement the determination of other physical parameters such as hydraulic conductivity and macroporosity.

In terms of broad-scale monitoring, bulk density has been already been demonstrated as an effective indicator for soil quality (Brejda *et al*., 2000a, 2000b; Drewry *et al*., 2000; Shulka *et al.,* 2004; Sparling and Schipper, 2004).

Conclusion and recommendation

It is recommended that bulk density is included in the MDS. It fulfils all of the criteria required for broad-scale monitoring. The method is simple, cheap and robust; it has already been demonstrated as a useful and interpretable indicator of soil quality change; and it is a useful primary indicator, via packing density, for other physical soil properties such as macroporosity, aggregate stability, mean weight diameter and hydraulic conductivity. It is also a useful secondary indicator to infer possible changes in soil chemical and biological factors such as SOC and microbial biomass and activity.

However, as with all soil quality indicators, the sampling protocol must be carefully designed to answer questions that will be posed with the future dataset. The primary concern would be that bulk density is susceptible to high temporal and spatial variation. On a national or regional scale, collection of sufficient data will enable broad-scale changes in soil quality to be measurable. However, at the local scale, the sampling protocol must be sufficiently robust to minimise variation within and across sampling sites. In particular, selection of sampling units must be consistent and timing of sampling in relation to annual climate, plant growth and management factors must be considered.

References

ADAS (1977) *Techniques for measuring soil physical properties*. MAFF, London.

Amezketa, E. (1999) Soil aggregate stability: A review. *J. Sust. Agric.*, 14, 83-151.

Arden-Clarke, C and Hodges, R.D. (1987) The environmental effects of conventional and organic/biological farming systems. I Soil erosion, with special reference to Britain. Biolog. Agric. Hort., 4, 309-357.

Bissonnaise, Y.E, Cros-Cayot, S and Gascuel-Odoux, C. (2002) Topographic dependence on aggregate stability, overland flow and sediment transport. *Agromie*, 22, 489-501.

Brejda, J.J, Moorman, T.B, Douglas, L.K and Thanh, H.D. (2000a) Identification of regional soil quality factors and indicators: I central and southern High Plains. *Soil Science Society of America Journal*, 64, 2115-2124.

Brejda, T.B, Douglas, Smith, J.L and Allan, D.L. (2000b) Identification of regional soil quality factors and indicators: II northern Mississippi loess hills and palouse prairie. *Soil Science Society of America Journal*, 64, 2125-2135.

Chen, Y, Tessier, S and Rouffignat, J. (1998) Soil bulk density estimation for tillage systems and soil textures. Transactions of the ASAE, 41, 1601-1610.

Drewry, J.J and Paton, R.J. (2000) Effects of cattle treading and natural amelioration on soil physical properties and pasture under dairy farming in Southland, New Zealand. *New Zealand Journal of Agricultural Research*, 43, 377-386.

Drewry, J.J, Littlejohn, R.P and Paton, R.J. (2000) A survey of soil physical properties on sheep and dairy farms in southern New Zealand. *New Zealand Journal of Agricultural Research*, 43, 251-258.

Drewry, J.J, Cameron, K.C and Buchan, G.D. (2001) Effect of simulated dairy cow treading on soil physical properties and ryegrass pasture yield. *New Zealand Journal of Agricultural Research*, 44, 189-190.

Hall, D.G.M, Reeve, M.J, Thomasson, A.J and Wright, V.F. (1977) *Water Retention, Porosity and Density of Field Soils*. A.R.C Soil Survey Technical Monograph 9. Harpenden.

Hernanz, J.L, Peixoto, H, Cerisola, C and Sanchez-Giron, V (2000) An empirical model to predict soil bulk density profiles in field conditions using penetration resistance, moisture content and soil depth. *J Terramechanics*, 37, 167-184.

Lal, R. (1993) Tillage effects on soil degradation, soil resilience, soil quality, and sustainability – introduction. *Soil and Tillage Research*, 27, 1-8.

Logsdon, S.D and Cambardella, C.A. (2000) Temporal changes in small depth-incremental soil bulk density. *Soil Science Society of America Journal*, 64, 710-714.

Loveland, P and Webb, J. (2002) Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research*, 70, 1-18.

Neves, C.S.V.J, Feller, C, Guimaraes, M.F, Medina, C.C, Tavares, J. and Fortier, M. (2003) Soil bulk density and porosity of homogeneous morphological units identified by the Cropping Profile Method in clayey Oxisols in Brazil. *Soil and Tillage Research*, 71, 109-119.

Schipper, L.A and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses. *Soil Science Society of America Journal*, 64, 300-311.

Scholefield, D and Hall, D.M. (1985) A method to measure the susceptibility of pasture soils to poaching by cattle. *Soil Use and Management*, 1, 134-138.

Scholefield, D, Patto, P.M and Hall, D.M. (1985) Laboratory research on the compressability of four topsoils from grassland. *Soil and Tillage Research*, 6, 1-16.
Scholefield, D and Hall, D.M. (1986) A recording penetrometer to measure the strength of soil in relation to the stresses exterted by a walking cow. *Journal of Soil Science*, 37, 165-176.
Shukla, M.K, Lal, R and Ebinger, M. (2003) Tillage effects on physical and hydrological properties of a typic aggriquoll in central Ohio. *Soil Science*, 168, 802-811.
Shukla, M.K, Lal, R and Ebinger, M. (2004) Soil quality indicators for the north apalachian experimental watersheds in Coshocton, Ohio. *Soil Science*, 169, 195-205.
Singleton, P.L, Boyes, M and Addison, B. (2000) Effect of treading by dairy cattle on topsoil physical conditions for six contrasting soil types in Waikato and Northland, New Zealand, with implications for monitoring. *New Zealand Journal of Agricultural Research*, 43, 559-567.
Sollins, P, Homannn, P and Caldwell, B.A. (1996) Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74, 65-105.
Sparling, G.P, Shepherd, T.G and Schipper, L.A. (2000) Topsoil characteristics of three contrasting New Zealand soils under four long-term landuses. *New Zealand Journal of Agricultural Research*, 43, 569-583.
Sparling, G and Schipper, L. (2002) Soil quality at a national scale in New Zealand. *J. Environmental Quality*. 31, 1848-1857.
Sparling, G, Lilburne, L and Vojvodic-Vukovic, M. (2003) Provisional target for soil quality indicators in New Zealand. Landcare Research, Palmerston North, New Zealand.
Sparling, G.P., Schipper, L.A., Bettjeman, W. and Hill, R. (2004) Soil quality monitoring in New Zealand: practical lessons from a 6-year trial. *Agriculture, Ecosystems and Environment*, 104, 523 - 534.
Wander, M.M and Bollero, G.A (1999) Soil quality assessment of tillage impacts in illinois. *Soil Science Society of America Journal*, 63, 961-971.

2.3.4 Catchment hydrographs

Background

A hydrograph reflects the flow of water measured at a specific point on a stream or river in a catchment. Three components contribute to this flow - groundwater, through-flow and direct flow into watercourses.

Direct flow depends on the interaction of incident precipitation, vegetation, topography and hydraulic properties of the soil surface but has a much faster contribution to stream flow than either groundwater or through-flow. Direct flow not only includes overland flow, but also direct (soil-unattenuated) flow into under-drainage systems, for example through permeable fill above drainage pipes. It is the major component of pulses in downstream flow and flooding.

Through-flow occurs as a direct result of incident precipitation but is attenuated by a range of soil physical properties such as topsoil and subsoil permeability and moisture status. Most rapid through-flow will be through major soil voids such as shrinkage cracks. Flow through macropores (fissures between peds and worm/old root channels) will be somewhat slower. The slowest component of through-flow will be flow through the soil matrix and underlying substrates and rocks and this will be highly dependent on soil moisture status.

Table 2.3.4.1: Sub-processes affecting the hydrograph

Process	Influenced by:	
	Variables	Constants
Precipitation	Amount of precipitation	
	Spatial variation in precipitation	
	Intensity and duration of precipitation	
	Character of precipitation (rain or snow)	
Flow development	Vegetation and land use	Contributing surface areas
	Evaporation	
	Soil moisture conditions and presence or absence of a watertable	
	Soil physical conditions and substrate hydrogeology	
	Groundwater contribution	
Flow concentration	Channel morphology, deposition, vegetation	Topography and slope
		Length of river network

Groundwater is usually the least variable of the contributing sources and can be a major component of base flow. It is made up of water that is not diverted through overland flow and lateral through-flow routes, nor is stored in the soil by fine pores. However, in some catchments, the contribution of groundwater to surface watercourses is negligible. The various processes affecting the hydrograph are summarised in Table 2.3.4.1 (Breinlinger *et al.*, 1992).

Impact of soil type

It is clear from the above that there will be a correlation between downstream flow and the fundamental properties of soils and rocks in the catchment.

Studies into the link between catchment characteristics and the catchment hydrograph have tended to be stimulated by devastating flood events, initially following a series of disastrous Alpine Floods in the 1860s and 1870s (Robinson and Whitehead, 1992). Since then there have been a multitude of studies on the relationship between changes in land use and catchment flows, particularly upland afforestation and deforestation. Probably the most comprehensive study to date, of Coweeta in North Carolina, focussed on evaluating hydrological response to different types of forest management. Many of the detailed UK studies of the past 30 years (such as Plynlimon, Kirby *et al*., 1991; Coalburn, Robinson *et al*., 1998) have focussed on upland land use changes.

However, the extensive soil mapping programmes undertaken over the last half century in North America and Western Europe have provided better soils data. In the USA, soils were classified into four hydrologic soil groups by the United States Department of Agriculture’s Soil Conservation Service, to indicate the minimum rate of infiltration obtained for bare soil after prolonged wetting (US-SCS, 1964):

Group A soils (deep, well- to excessively-drained sands and gravels) have low run-off potential and high infiltration and water transmission rates even when thoroughly wetted.

Group B soils (moderately deep to deep, moderately well- to well-drained soils with moderately fine to moderately coarse textures) have moderate infiltration and water transmission rates.

Group C soils (moderately fine to fine texture and/or a layer that impedes downward movement of water) have low infiltration rates and water transmission.

Group D soils (clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material) have high run-off potential due very low infiltration and water transmission rates.

This classification has been used as part of commercial software (HydroCad) for modelling stormwater run-off and undertaking drainage calculations, and in web-based decision support systems to examine the effects of agriculture, urbanization, and land use management practices on the hydrology and water quality (Ehle *et al*., 1999).

In the UK, the first significant attempt to classify soils according to their hydrological response was the Winter Rain Acceptance Potential (WRAP) scheme (Farquharson *et al*., 1978). This combined soil hydrological properties (water regime class, depth to slowly permeable layer and permeability above it) with three slope categories to arrive at a five-category classification. Once again the primary aim was flood estimation. The WRAP classification was subsequently developed into the HOST (hydrology of soil types) classification (Boorman *et al.,* 1995). Eleven conceptual soil hydrological response models were subdivided according to substrate hydrogeology to arrive at a 29 class system. Groupings of the classification for low flow forecasting were found to be moderately well correlated with actual low flows and the system was also found to be fairly accurate for estimating standard percentage run-off (the percentage of rainfall that causes the short-term increase in flow seen at the catchment outlet). The validation of run-off and base flow indices by Boorman *et al.* (1995) used the 1:250,000 National Soil Map. Use of more detailed soil information, where available, might be expected to improve accuracy.

From the above it is clear that soils have an important influence on hydrological processes. Physical properties such as infiltration rate, permeability and moisture retention capacity govern the storage and transmission of water within and across the soil. These not only have effects at the small (within field) scale but can also be seen at the sub-catchment scale and in the integrated response of whole catchment systems.

Sensitivity of catchment hydrographs

Most of the previous research has been targeted at explaining an effect (such as flooding) by a hypothesised cause (such as major afforestation or other changes in land use). Is it possible then that the catchment hydrograph could be used as a surrogate indicator of changes in soil quality, changes that are likely to be rather subtler than major changes in land use? Furthermore, can the effects of the variables affecting the hydrograph (precipitation, land use and soils) be separately identified?

Separating climate effects from land use effects

Archer (2000) looked at separating climate effects from land use effects with regard to detecting the effects of land use change, which are thought to alter the ‘flashiness’ of rivers. Indeed, there is a widespread belief that Pennine and other upland rivers in Britain are now subject to spates of greater intensity and shorter duration than a few decades ago.

Archer decided that an effective method to define hydrological disturbance with respect to influences of land use change should have the following properties:

- It should focus on those attributes of flow which are said to have been influenced by land use change, that is, the number and frequency of spates and their duration
- The measurement interval considered should be sufficiently short to detect effects of land use on small to medium-sized catchments. Daily mean flows are likely to be an inadequate basis for analysis when catchment lag is less than one day. A continuous record or measurement at a sub-daily interval is required.
- There is a natural variability in the level of flow disturbance regime from year to year due to the sequence of weather and climate. There should be a means of

decoupling the effect of climate and weather of a particular period from the effects of land use.

- It should permit the detection of step changes and trends in disturbance characteristics at a site and their validation by statistical tests.
- It should provide a means of comparison between rivers and between different locations on the same river.
- The indices should also have a demonstrated link to measurable ecological properties such as living and total biomass and species richness and diversity.

A steep Pennine moorland catchment of 172 km^2 on the upper Wear was selected for analysis by Archer (2000), with flow monitored at Stanhope. The river has been persistently reported to become more flashy with spates of shorter duration than formerly and the change has been ascribed to moorland drainage or to changes in upland farming practices.

Definition and evaluation of the degree of flashiness was based on the frequency and duration of pulses above selected threshold flows. A pulse is an occurrence of a rise above a selected flow and pulse duration is the time from rising above the threshold to falling below the same threshold (Figure 2.3.4.1). For each year the total number of pulses was counted and the total duration above 17 thresholds (0.5M, M, 2M, 3M, 4M, 5M, 6M, 7M, 8M, 10M, 15M, 20M, 30M, 40M, 50M, 60M and 80M where M is the median flow) and the mean duration per pulse were computed.

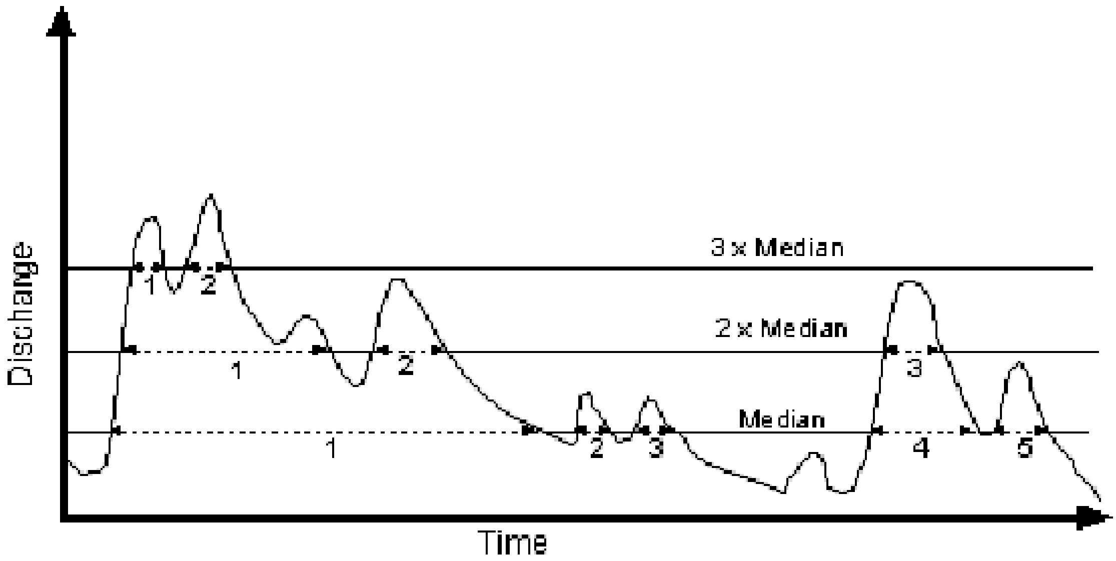
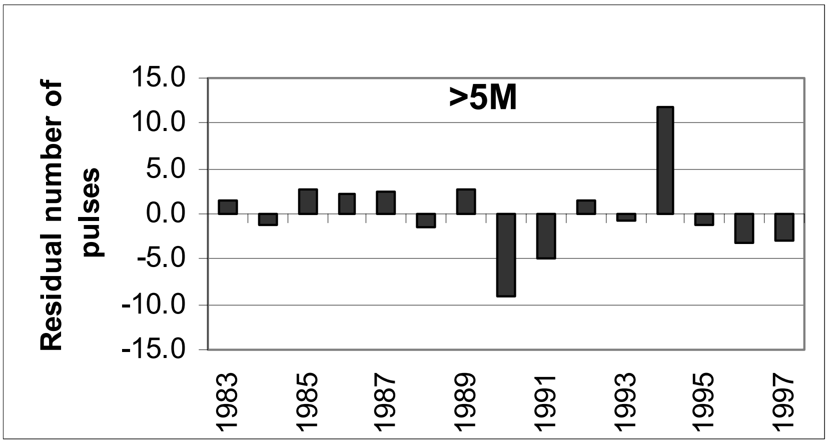


Figure 2.3.4.1: Definition diagram for assessing the number of pulses above selected thresholds and the duration (dashed) of pulses (Archer, 2000)

To assess the impact of climate, regression analysis was carried out between annual catchment rainfall for 1983 to 1997 as provided by the National Water Archive and the number and duration of pulses for each multiple of the median. Where a high level of correlation was achieved (for pulse number and total duration), the regression relationship was then used to compute an expected number of pulses for given rainfall and the residual from the observed was calculated (Figure 2.3.4.2).

This research not only provides evidence that there has been no change in the hydrological disturbance characteristics of the River Wear at Stanhope over the period from 1960 to 1998, but the procedure described provides an effective method for more general use in assessing the effects of different processes on the hydrological regime with particular emphasis on flow variability.



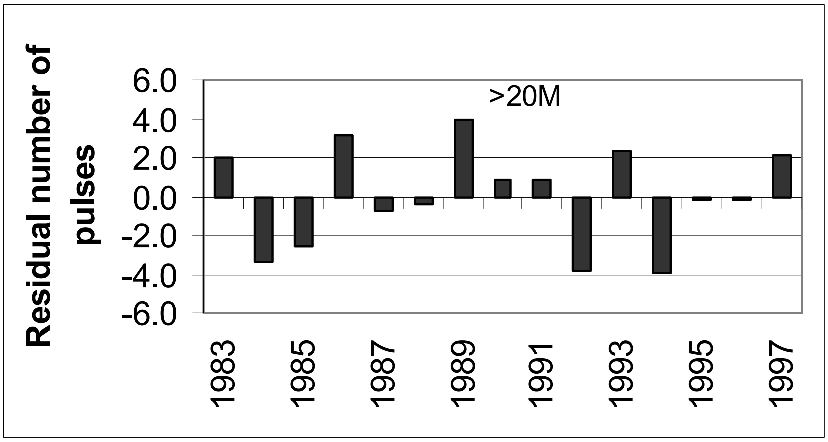


Figure 2.3.4.2: Time series of residual pulse numbers (after removal of climatic effect) for flows in excess of 5x and 20x the median flow for the River Wear at Stanhope (Archer, 2000)

Separating soil effects from land use effects

The research by Archer suggests that it is possible to separate out the effects of land quality and use from climate on the catchment hydrograph but it is uncertain whether soil quality *per se* can be separated from land use. In many ways the two are intimately linked.

Soil quality parameters and the hydrograph

The two principal properties likely to affect catchment hydrographs are the water-holding capacity of the soils and the rate of infiltration or drainage capacity.

Water-holding capacity

Water-holding capacity reflects the capability of the soils to absorb and retain rainfall before through-flow and run-off begin. It is positively correlated with soil organic matter but negatively correlated with bulk density, the content of coarse particles >100 μm (Hall *et al*., 1977) or any loss in topsoil thickness.

Organic matter can be lost as the result of long-term arable farming, erosion by wind and water, oxidation as the result of drainage of peat soils, commercial activities such as turf cutting or peat cutting or exportation of topsoil to other uses. Some of these causes can also go hand-in-hand with an increase in bulk density. A decrease in organic matter is likely to result in an increase in run-off and through-flow. Organic matter can be gained by addition of wastes to the soils. This can include sewage sludge, paper waste, farmyard manure and various other organic waste streams currently going to land. The effect of increasing water-holding capacity may be to decrease base flow at the catchment outfall, not necessarily desirable. The role of any changes in topsoil properties in this context will depend on the overall thickness of the soil profile. For some soils, for example the Rankers soil group, there is little more than the topsoil. Conversely, some members of the Brown Earths soil group, weathered in thick alluvium or loess, are more than a metre deep and have subsoils with water storage well in excess of that in the topsoil. Furthermore, the particle size distribution of soils is unlikely to be changed sufficiently to have effects at catchment or sub-catchment level, except perhaps as the result of disappearance of peat from organic soils.

Topsoil depth can change significantly by the physical removal of soil for construction or other purposes or by erosion. The effective depth may be reduced by compaction in the upper soil levels during cultivation or construction. Compacted layers retard water reaching the storage reserves in the subsoil.

Infiltration/drainage capacity

The drainage capacity of a soil is principally a function of soil type. Slow or restricted drainage may be the result of slow movement of incident water through slowly permeable layers, by high groundwater levels limiting freeboard, or both. Many clay soils are intrinsically poorly drained with slow infiltration rates, and severe rainfall events can institute high levels of overland flow. Most sandy soils are freely draining with infiltration rates up to centimetres per hour, and most excess rain flows through the soils. Ill-timed cultivation, soil moving or construction work may lead to the formation of compacted layers and surface caps within otherwise more freely draining soils, and this has the effect of increasing the amount of lateral overland flow and through-flow. The capping of light soils (the blocking of surface pores by particles detached by rain impact) drastically reduces surface infiltration rates (Harrod and Fraser, 1999). This explains the apparent contradiction of freely draining soils being highly erosion prone.

Installation of intensive drainage systems can be considered as a change in soil properties, as the process of installing drains adds voids to the soils both as pipes and permeable fill. Studies of flow records (Robinson, 1990) from individual catchments indicate that the combined effects of field drainage and related arterial works is to increase stream flow peaks (and dry weather flows). At the regional scale, artificial drainage was a statistically significant parameter shortening catchment response times.

Is change measurable?

A change in land use from open moorland to woodland will undoubtedly have an effect on soil quality but it might be difficult to define whether that is a positive or negative effect. Surface layers of the soil might become mixed by pre-planting cultivation and some compaction of soil might occur below the wheelings of cultivation equipment. But on wet soils the major effects on the hydrograph are likely to be from drainage grips accelerating water movement to streams and rivers and, as the forest becomes established, the increased transpiration of moisture and interception of rainfall by trees compared with moorland plant species. Robinson *et al*. (1998) demonstrated that at Coalburn the hydrological response was dominated by the drainage network for approximately 12 years after planting and that the run-off was more flashy than under moorland cover. The contribution of soil compaction to that effect was not considered. After the initial 12 years the forest assumed a greater role in run-off response, which becomes much less flashy.

In a lowland situation, different factors come into play. As an example one can consider a catchment on the South Downs (although it is normally a dry catchment with no active watercourse). This catchment, with an area of 2.6 km^2, is comprised of soils that are mainly permeable over chalk within a depth of 50 cm. Only on interfluves are soils somewhat deeper over chalk with clay layers in the subsoil. In its normal state, all of this land would be in HOST Class 1, representing permeable soils with relatively unimpeded vertical drainage of incident rainfall. A gradual change in land use over several decades has converted much of the former downland grass to arable use and particularly to autumn-sown wheat crops. In hand with this has been a gradual decline in soil organic matter levels, a likely decline in soil structural quality and a propensity for the topsoils to become over-compacted by autumn cultivations and crop treatments. This has the effect of turning a catchment dominated by soils of HOST class 1 to one of HOST class 13, albeit temporally. Water movement becomes concentrated on the surface instead of percolating downwards to rock, as illustrated in Figure 2.3.4.3.

Similar changes affect capped and degraded sandy and silty soils where rain as little as 2 mm per hour can cause overland flow. Such changes go a long way to explain the contradiction of freely draining soils being erosion prone, even on gentle slopes. (Fraser, 2000).

Despite this example, the HOST classification is not that applicable to a soil quality monitoring protocol that concentrates its efforts on topsoil characteristics. The identification and assignment of HOST classes depends on knowledge of the underlying geology as well as a thorough inspection of the whole soil profile, which in most instances will include subsoil characteristics as well as topsoil characteristics.

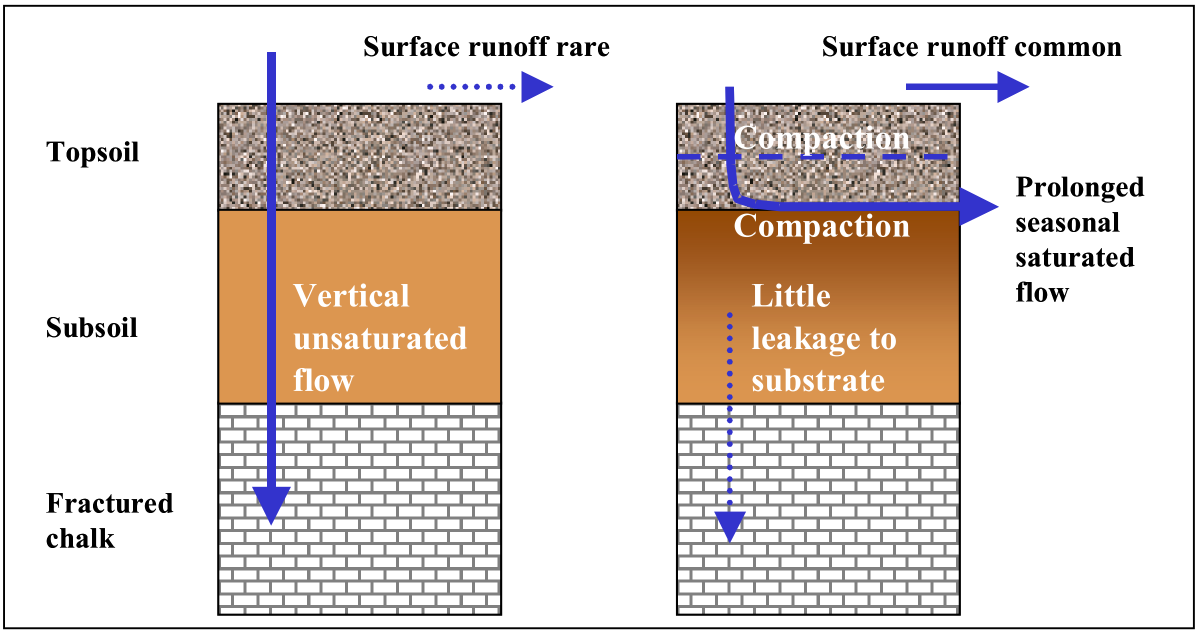


Figure 2.3.4.3: Effect of topsoil compaction and sealing on hydrological response of a permeable soil over chalk (Archer, 2000)

Scale issues are important. Land use changes occur as a patchwork over a catchment. A reduced rainfall run-off response time due to soil physical deterioration in lower or mid-catchment may have a counteracting effect on a reduced response time in the upper catchment. While soil quality or land use changes at the field or small catchment scale may have a closely defined interval of occurrence, such changes at the whole catchment scale may be spread over a period of years or decades, with land use and soil quality effects on run-off operating in different directions. Also, with increasing catchment size goes the likelihood of greater diversity of soils, and with that variety of hydrological response.

Monitoring in relation to current national catchment monitoring programmes

In view of the uncertainty of identifying soil quality changes from the catchment hydrograph, it is suggested that any monitoring for soil quality purposes should be based on the current network of gauging stations and research catchments.

The UK has a comprehensive network of flow gauging stations with 1,156 in England and 160 in Wales. These record the flows from sub-catchments smaller than 1 km^2 to major river system catchments (such as the Thames, Severn, Trent) of almost 10,000 km^2. Flow data is typically captured at 15 minute intervals but tends to be stored as daily mean flows on the National River Flow Archive (http://www.nwl.ac.uk/ih/nrfa). Daily data is useful in describing the flow regime of a catchment and durations of more than one day for describing low flows. For flood flows it is more usual to look at data that is at a finer interval than one day.

Other monitoring programmes include LOCAR (NERC Lowland Catchment Thematic Research Programme). This is a project studying key water resource issues in the lowlands of the English chalklands and Midland sandstones. It is based on intensive study of three catchments – Frome/Piddle and Pang/Lambourn on the Chalk and Tern on Triassic sandstones. The Environment Agency is a stakeholder in the programme and one of the questions it seeks to answer is whether information on the impacts of land use for one catchment can be used to predict impacts in other catchments, or whether local factors (such as river-road drainage networks, topography, soil type and farming practice) are so influential that experiences are not transferable. All data recorded by LOCAR will be managed by the Centre for Ecology and Hydrology as part of the National Water Archive.

The ECN (Environmental Change Network) records a large number of terrestrial and freshwater parameters, including hydrograph measurements, at 15 sites across England and Wales. Its aims are to:

- establish and maintain a selected network of sites within the UK from which to obtain comparable long-term datasets through the monitoring of a range of variables identified as being of major environmental importance;
- provide for the integration and analysis of this data, so as to identify natural and man-induced environmental changes and improve understanding of the causes of change;
- distinguish short-term fluctuations from long-term trends, and predict future changes;
- provide, for research purposes, a range of representative sites with good instrumentation and reliable environmental information.

CHASM (Catchment Hydrology and Sustainable Management) is a programme based on four catchments, with two in England and Wales - the Upper Severn (187 km^2) and the Eden (427 km^2). Outputs from the programme are, as yet, uncertain.

Conclusion and recommendation

It is uncertain whether the catchment hydrograph can be used as either a direct or an indirect indicator of changes in soil quality. A change in land use or land management practices could well affect the catchment hydrograph, as demonstrated by Archer (2000). In the long term, the change, if sustained, will influence soil properties such as organic matter, structure and density. This will be relatively long term, however, and it is doubtful whether a hydrograph is sufficiently sensitive to detect these rather subtle changes.

It seems likely that in large catchments there will so many confounding influences on the catchment hydrograph that the soil factor is unlikely to show through. In smaller catchments (say smaller than 10 km^2) there is a slight possibility that changes in soil quality might be detected in the hydrograph, particularly if land use changes were minimal and could be discounted over the period of monitoring.

There exists a good network of hydrograph measuring locations across England and Wales that are being monitored for other purposes. Some are backed up by intensive environmental study in the catchments they monitor. If, for example five small catchments can be identified on each of the main soilscapes of England and Wales (www.magic.gov.uk) from within the existing monitoring network, there might be some merit in using the catchment hydrograph as an early-warning indicator of possible soil quality changes. If this cannot be achieved using the existing network, the benefits of creating new monitoring locations are likely to be unattractive. It is likely that changes in soil quality will be detected earlier and more cost-effectively by other, more direct, monitoring techniques.

In conclusion, the catchment hydrograph is not a good indicator for soil quality *per se*. It does, however, provide an integrated indicator of land use, soil quality, climate change factors that affect diffuse pollution, flood defence and aquatic ecology. In this sense, and especially with regard to the EU Water Framework Directive, the catchment hydrograph is the best and most forward thinking way into effective and integrated catchment management.

References

Archer, A. (2000) Indices of flow variability and their use in identifying the impact of land use changes. *Proceedings of the Seventh National Symposium of the British Hydrological Society*, Newcastle 2000, 2.67–2.73.

Boorman, D B, Hollis, J M and Lilly A. (1995) *Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom*. Report No 126. Institute of Hydrology, Wallingford, UK.

Breinlinger, R, Duster, H and Weingartner, R. (1992) Methods of catchment characterisation by means of basin parameters. In: *Methods of Hydrological Basin Comparison*. Report No 120. Institute of Hydrology, Wallingford, UK.

Ehle, M, Kyoung Jae LimEngel, B A, Youngsug Kim, Budhendra Bhaduri, and Harbor J. WWW SYSTEM TO EVALUATE THE EFFECTS OF LAND USE CHANGE ON HYDROLOGY AND WATER QUALITY. 1999 ESRI User Conference Proceedings (http://gis.esri.com/library/userconf/proc99/proceed/papers/pap417/p417.htm)

Farquharson, F A K, Mackney, D, Newsom, M D and Thomasson, A J. (1978) *Estimation of run-off potential of river catchments from soil surveys*. Special Survey No. 11. Soil Survey of England and Wales.

Fraser, A I. (2000) *Towards a systematic framework for quantifying and extrapolating diffuse phosphorus transfer from agricultural fields to the national scale*. Unpublished Ph D thesis, Cranfield University.

Hall, D G M, Reeve, M J, Thomasson, A J and Wright, V F. (1977) *Water retention porosity and density of field soils.* Soil Survey Technical Monograph No 9. Soil Survey of England and Wales.

Harrod T.R. and Fraser A.I. (1999) *Phosphorus loss from agriculture*. Unpublished SSLRC report to MAFF, project NT1003.

Kirby, C, Newson, M D and Gilman, K eds. (1991) *Plynlimon research: the first two decades*. Report No 109. Institute of Hydrology, Wallingford, UK.

Robinson, M. (1990) *Impact of improved land drainage on river flows*. Report No 113. Institute of Hydrology, Wallingford, UK.

Robinson, M and Whitehead, P G. (1992) A review of experimental and representative basin studies. In: *Methods of Hydrological Basin Comparison*. Report No 120. Institute of Hydrology, Wallingford, UK.

Robinson, M, Moore, R E, Nisbet, T R and Blackie, J R. (1998) *From moorland to forest: the colaburn catchment experiment.* Report No 133. Institute of Hydrology, Wallingford, UK.

U.S. Soil Conservation Service (1964) Hydrology. Section 4, *SCS National Engineering Handbook*. Washington, D.C.

2.3.5 Mineralisable nitrogen

Mineralisable nitrogen has some international support as an indicator of soil quality, for example in the New Zealand and Dutch soil monitoring networks and by Doran and Safley (1997). Loveland and Thompson (2001) did not propose it as an indicator for further consideration. But the importance of nitrogen to soil-plant-microbial systems is widely known and an indicator relating to soil productivity is desirable. This review considers potentially mineralisable nitrogen (PMN) as a soil quality indicator for inclusion in the minimum dataset of a national monitoring scheme. Indicators are to be used broadly to assess relatively high level trends in soil quality and risks or threats to sustainable land management practices. Their use should provide a basis for evidence-based decision making and policy development for soils.

Questions and points to be addressed for this indicator include:

- How does the indicator relate to soil function?
- What is the rate of change and what does it mean? Is there an indication of the inherent variability?
- Is there a range or envelope of trigger values for different soil types?
- Will the indicator integrate with other monitoring schemes?
- What are the likely costs, ease of use and suitability to broad-scale monitoring?
- Recommendation.

How does the indicator relate to soil function?

Nitrogen (N) is a macronutrient essential to the growth of plants. Perturbations to its cycle will affect the flow of N through soil-plant-microbial systems. The availability of mineralisable N (that is, in forms readily usable by plants) has potential as an indicator of soil quality through its relationship with soil productivity and nitrogen supplying potential. The amount mineralised depends on temperature, moisture, soil matrix and the N source. Many studies suggest that the primary source of mineralisable N is the microbial biomass N. For example, 55-89% of N was released from microbial biomass in study by Bonde *et al.* (1988). As such, mineralisable N may be used to model soil processes and as a surrogate indicator of biological activity (biomass).

The mineralisation of soil organic N through to nitrate reflects the quality and quantity of soil organic N and links the substrate to a range of soil organisms. A large range of soil organisms are able to decompose organic N and release ammonium N, but the subsequent oxidation of ammonium to nitrite and nitrate is predominantly performed by small specialist groups of chemotrophic bacteria which are more fastidious in their growth requirements. The mineralisation of N therefore reflects previous biological activity in the soil, general decomposer activity (ammonification) and the activity of specialist groups (ammonium and nitrite oxidisers) and has the potential as an indicator of soil quality on several levels.

What does change mean?

The rapid accumulation of nitrate in soil indicates the presence of mineralisable organic N in addition to favourable soil conditions for the functioning of ammonifier organisms and nitrite and nitrate oxidising bacteria. Nitrogen mineralisation could therefore be a useful indicator of both organic matter quality and microbial processes in soil (Sparling, 1997).

Land users interested in plant production will favour high levels of nutrients and utilisation of soil organic reserves, that is, high N mineralisation rates. But this may conflict with environmental values in terms of an increased chance of nitrate leaching from soils and eutrophication of receiving waters. The latter also has economic implications for water companies who must meet drinking water standards for nitrate.

The following factors may affect mineralisable N and are discussed below:

- the N source such as litter quality, vegetation type and microbial biomass;
- temporal environmental factors such as temperature and soil moisture content;
- site matrix such as soil texture and pH;
- grazing organisms.

The N source

Plants obtain the majority of their nitrogen from the soil by absorption through roots, although the fraction of soil N available for uptake is rarely more than 1% (Gosz, 1981). The complex interplay of temperature, moisture and type and amount of organic matter can affect biological mineralization, so it is not surprising that great diversity exists within and between N cycles of different ecosystems.

Many studies have suggested that PMN is a measure of maximum N mineralised from soil organic N that may become available to plants. Dalal (1998) reviewed microbial biomass studies and found that some researchers reported close correlation between mineralisable N and microbial biomass N. But other studies have shown that the amount of PMN measured (6-10%) far exceeds the amount of microbial biomass N (2-6 % of total N) in soil. So, a substantial amount of non-biomass N, but labile N, is also mineralised (in this instance, during long periods of laboratory incubation under anaerobic conditions). Bonde *et al.* (1988) suggest that 67-78% of total microbial biomass consists of a dormant inactive fraction which could potentially be mineralised during the incubation period.

PMN may also be increased through the addition of N in fertiliser, plant material, farmyard manure or other organic wastes such as sewage sludge. These enter the labile fraction of soil organic N. Fertiliser may have a direct effect, such as ammonium nitrate or urea, but mainly an indirect effect by increasing plant growth and therefore litter production. Fertiliser application is likely to be the most important factor affecting PMN in many parts of the UK. That is, any given soil could have a range of net N-supplying power dependent upon applied N and level of intensity of management. This is due to immediate production (and attenuation on withholding fertiliser) of N-rich residues with narrower C:N ratios.

Vegetation types

In very general terms, vegetation types can be ranked in order of decreasing PMN; re-seeded grassland > undisturbed grassland > arable > hardwood or mixed forestry > coniferous forestry. Low N availability typifies forest soils. This condition (for example, ammonium-N < 2 ppm) is largely created by forests partly owing to the high C:N ratio of materials deposited as litter fall. But, forests have been shown to take up N at rates similar to intensively managed agricultural land when N availability is unlimited (Van Veen *et al.*, 1981).

Mineralisation rates affected by climate and site characteristics are discussed in other sections but litter quality, species differences and fire/clearance relating to vegetation type are summarised here.

Plant litter input into soil (Staaf and Berg, 1981)

In most ecosystems, plant litter is the major pathway for supplying energy and nitrogen to the soil community. Litter formation can be regarded as a flow where the chemical composition of litter regulates the directions and rates of N flows within and from the soil processes, which in turn give feedback to plants and influence losses from the ecosystem. The genetic control of plant production, phenology, life-span of organs and redistribution of N within plants are the primary regulating factors of N input to soil via plant litter. Environmental factors such as temperature, moisture, snow and wind also exhibit the same control, albeit often unpredictably. Depending upon which factor has the greatest influence, plants differ between how they circulate N, either within the plant or via the litter. Ecosystems with low N availability in the soil tend to hold plants with a more marked within-plant circulation strategy compared with N-rich ecosystems (Gosz, 1981). Species of early successional stages or species occupying niches with a locally higher N supply may differ by having a faster transfer of biomass N to soil – in other words, preferring a litter circulation strategy.

Species adapted to low N availability in soil can invade, for example *Calluna vulgaris* on heathlands, where species with high N concentrations or with decomposition-enhancing mechanisms are removed from an ecosystem. Conversely, invading or remaining plants offering rapid transfer of biomass N into litter will raise the general fertility of soil, e.g. grasses, herbs and deciduous trees in the early stages of secondary succession of boreal forests. High N concentrations in litter will facilitate N mineralisation in three ways:

- a high decomposition rate;
- a large amount of N released from each unit weight of litter undergoing decomposition;
- a lower percentage of the nitrogen immobilised in the litter.

The first point tends to be less important where litter has a high lignin content,such as coniferous litter.

Coniferous forests (Gosz, 1981)

Conifer ecosystems as a group have lower rates of mineralisation and uptake of N than deciduous forests; for example, N mineralisation in kg/ha/yr for conifers is 50-100 compared with 100-300 for hardwoods. Coniferous woods exhibit a large canopy biomass with long leaf persistence and, coupled with their low N uptake requirements, mean they increase their effectiveness of internally redistributing N. As a consequence litter fall is low in N content, resulting in low decomposition and mineralisation rates. Other environmental stresses can cause an increased production of organic acids in leaves which further slows decomposition and mineralisation rates. But leaching losses are very low from undisturbed coniferous ecosystems irrespective of the rate or amount of N cycling. Intense fire appears to be the only disturbance capable of causing large losses of N from these systems.

Coniferous species have adapted where nutrient supplies are different. For example, some species prefer to take up N as ammonium (low N-demanding species) and others as nitrate (high N-demanding species); others will utilise both forms. Better growth of species on ammonium than nitrate and those with an appreciable lack of nitrate reductase activity correlate with low rates of nitrification in soil. Clear felling of trees can increase N mineralisation. For example, this has been seen following the removal of *Pinus radiata* in New Zealand. This effect may be due to microbial activity recovering from the previously suppressing influence of mycorrhizal fungi. The fungi extensively colonise litter and physically exclude decomposers from microsites in the litter, preventing decomposition and mineralisation.

In summary, N cycling in coniferous forests is very slow but relatively non-leaky.

Deciduous forests (Melillo, 1981)

As in coniferous forests, leaching and erosion losses of N from deciduous forests is low. In general they are accumulators of nitrogen. The majority of N (around 90%) can be found on the forest floor or as organically bound N in soil complexes; most of the remaining N is located in the vegetation as organically bound N; and at any one time, usually <2 % is in the soil in inorganic form, mainly as ammonium. A mobile pool of N recycles annually within vegetation and is available for physiological processes while decreasing the dependence of plants on N cycled through soil pools.

Ellenberg (1971) reported that among central European deciduous forests, oak-birch forests have an ammonium economy; deciduous forests growing on a loamy soil have a mixed ammonium-nitrate economy; and deciduous forests growing on calcareous soils have a nitrate economy.

A study of mean annual net mineralisation rates in central European deciduous forests showed that the range is large. The lowest rates being in mountainside forest on rendzina soils at 0-24 kg/ha/yr and the highest being ravine forests on colluvial braunerde at 375-399 kg/ha/yr. The mean value was approximately 110 kg/ha/yr.

Grassland (Woodmansee et al., 1981)

Grassland ecosystems have similar pathways and components as other ecosystems, but their rates of processes differ vastly from those with woody components. That is, they are much more rapid. For example, PMN measurements taken from undisturbed grasslands at two sites in the UK ranged from 40 to 400 mg/kg (at 0-10 cm soil depth) in summer and autumn of one year (Bhogal *et al.*, 2000). This compares with 20-175 mg/kg for forestry (Table 2.3.5.1). Essentially all top growth of perennial grasses dies each year, returning N to the soil, although some tussock grass can persist for up to 3 years. Other short-lived perennials, annuals and shrubs also contribute to the mobilisation of N from their dying tissues by translocation to actively growing tissues (one-third to two-thirds total N in living tissue) whilst the remainder falls to the soil in litter. Freezing and herbivory kills grassland vegetation without the concomitant conservation of N, although grazers will return nitrogen in a relatively N-rich state (faeces) back to the soil.

During periods of rapid plant growth, vegetation and microbes take up inorganic N simultaneously. In unfertilised grassland this often exceeds the rate of N mineralisation potential, rendering all mineral N production immobilised in living organisms (mostly as ammonium). Microbes have the competitive advantage due to their intimate association with soil and as a result plant growth can be limited. The greatest source of mineral N in grassland appears to be a combination of recently dead plant and microbial debris.

N losses from grassland through leaching appear to be minimal. But wind and water erosion can lead to N losses where grassland cover is inadequate (e.g. due to intense grazing, drought or fire). Soil conditions required for denitrification are poor aeration, supply of a suitable substrate for microbial growth and the presence of nitrate and denitrifying organisms. These are thought to seldom occur on native grasslands, so N loss as NOx or N_2 is considered negligible; however, a better understanding of this area is recognised as a research need. Stable grassland ecosystems must maintain balanced N transfers within their actively cycling portion over long periods of time. If a prolonged imbalance occurs, then ecosystem succession may result.

Species differences impact on N mineralisation. Wedin and Tilman (1990) showed that rates of mineralisation of nutrients under a C4 grass was one-third that of rates under a C3 grass growing under similar nutrition and yielding similar amounts. Plants growing in a nutrient-poor soil are much more recalcitrant than those growing in a nutrient-rich (agricultural) soil, due to plant types and the need to employ herbivory defence mechanisms (Scholefield, 2003).

Bhogal *et al.* (2000) measured the distribution of N pools in the soil profile of undisturbed and re-seeded grasslands in the UK. Cultivation and re-seeding significantly increased the total soluble N concentration of the soil profile (0-90 cm), but by contrast the PMN behaved differently and was unaffected by cultivation and declined with increasing soil depth. Cultivation and re-seeding is well recognised as usually increasing PMN levels, so these results were surprising. The authors concluded that a substantial proportion of the PMN pool was immobilised into more stable organic forms and that soluble organic N should be measured where cultivated soils are being monitored.

Spatial variability of N mineralisation can occur due to the presence of areas of N-fixing organisms, poor drainage, season, drought, or patches of intense grazing. Seasonal N balances could vary considerably so timing of monitoring is important. Leaching of nitrate is likely to be influenced by cool wet winter and early spring conditions, whereas spring and early summer are periods of active plant growth. Also, year-to-year variability seems to be great. A 3-fold difference between inputs from fixation and deposition was seen between years in Canadian prairie (Vlassak *et al.,* 1973). This extreme dynamic may be lessened in perennial grasslands.

McCrea *et al.* (2004) reported less mineralisable N and organic matter in soils of artificially created urban meadows than semi-natural meadows. Although N is strongly associated with vigorous growth of agricultural grasses and rapid decline in diversity, the authors thought that high soil phosphorus concentrations were the main reason why relatively few species colonise or survive in grassland on many urban soils. It is well known that phosphorus is the enemy of wild flower meadows.

Relationship with soil organic carbon

In Figure 2.3.5.1, data from 60 arable soils shows a relationship between soil organic carbon and PMN (measured using the anaerobic incubation technique). This does not show a direct relationship between PMN levels and biological activity. But we can infer a *likely* relationship with biological activity, given that soil respiration rates and biomass carbon levels are related to soil organic carbon (matter) levels. PMN measurements are likely to be less affected by temporal variation than respiration rates and biomass carbon levels. This is the potential advantage in using PMN as an indicator of biological activity.

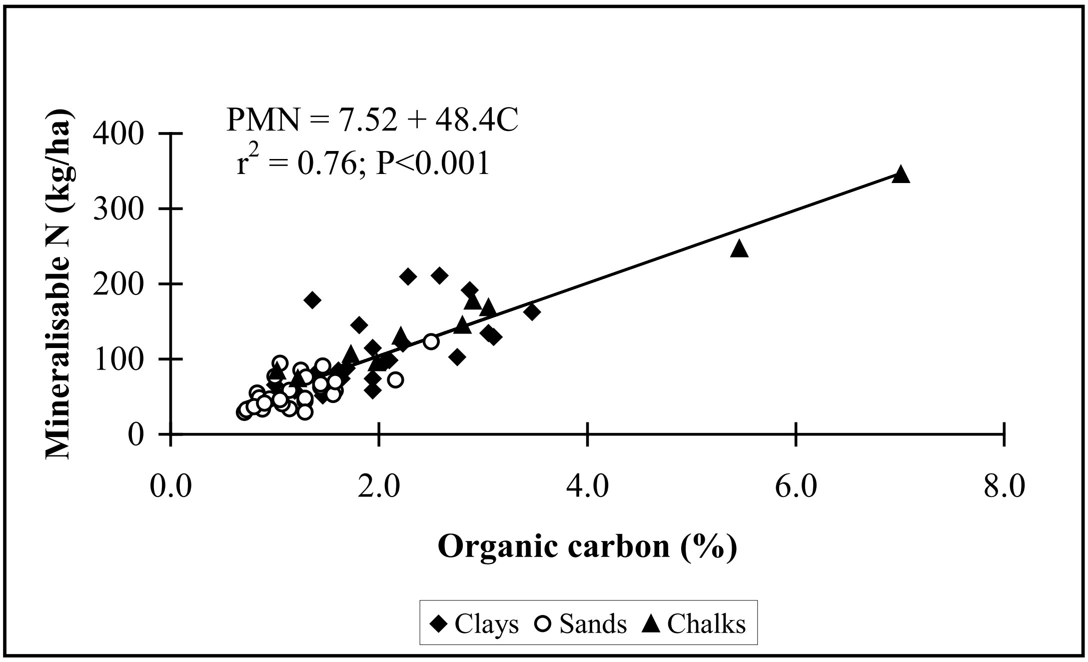


Figure 2.3.5.1: Relationship between topsoil organic carbon content and PMN

In summary, the N source is important to the amount of PMN available and the rate in which it is generated. Vegetation type and cover will affect the quality and quantity of leaf litter and rate of decomposition. If PMN is recommended as an indicator of soil quality, then vegetation cover or land use should be indicated also. It is probable that these additional parameters will be recorded for all indicators. This is how the New Zealand and Dutch authorities describe PMN in their soil quality monitoring networks.

Temporal environmental factors

The influence of climate on plant productivity and ecosystem development is well known and it follows that climatic factors will affect the availability and cycling of N. Measurements of plant productivity and N uptake suggest that mineralisation will be affected by climate. But measurements of total N are not thought to correlate well, particularly in coniferous forests that are undisturbed or with large old individuals where N retention in trees is efficient.

Temperature and moisture influence N turnover in ecosystems in particular. This also has serious implications for standardising the methods used to measure N mineralisation.

Temperature

The rate of N mineralisation has been shown to increase two- to three-fold for each 10 °C increase in soil temperature between 5 and 35 °C (in Power, 1981). Freeze–thaw cycles have little effect upon the mineralisation of soil N but they severely retard nitrification, resulting in ammonium accumulation (Power, 1981). This is supported by Adams and Jan (1999), who saw that decreases in soil temperature during autumn resulted in a modest decrease in net N mineralisation but a much more substantial decrease in the rate of nitrification, which has implications for nitrate leaching in cultivated systems. However, Addiscott (1983) didn't measure any temperature sensitivities in mineralisation or nitrification on arable soils until below 5 °C.

Moisture

The water content of soils is highly significant to the N mineralisation process. Campbell *et al.*, (1995), in their field trials on Scottish soils, showed that soil nitrate is strongly and positively correlated with soil moisture content. This is in agreement with general findings that nitrification rates are sensitive to moisture content (Miller and Johnson, 1964). Even small changes in soil water content may affect the subsequent potential for N mineralisation (West *et al.*, 1988). Mineral N tends to accumulate in low soil-water conditions. Plant uptake of N is greatly reduced and microbial growth is restricted but some microbes can remain active and capitalise on bright sunshine (warmth), dew or light rainfall to mineralise N (Woodmansee *et al.*, 1981).

Also, litter decomposition rates have been shown to significantly correlate with actual evapotranspiration (moisture and temperature interaction) values for a range of climates (Gosz, 1981).

Dry–wet cycles

The enhancement of N mineralisation following a dry–wet cycle has long been recognised as an important phenomenon in the process of N turnover (Birch, 1960), although underlying mechanisms are not well understood. Enhanced mineralisation is thought to result from the increased availability of organic substrates, due either to chemical changes induced by drying and re-wetting or to the death of microbial cells during drying. Marumoto *et al.* (1982) found that for a wide range of soil types, on average 77% of the mineral-N flush, after a dry–wet cycle, was derived from the biomass pool. Stockdale and Rees (1994) suggested that if microbial biomass is the source of much plant-available N, then reliable indices of N availability should depend on methods which preferentially extract N from the biomass pool.

Soil matrix

Soil texture has been frequently demonstrated to influence the soil microbial biomass, with high clay content soils promoting faster build up than corresponding coarse-textured soils. This has been attributed to a number of reasons including superior ability of clay to hamper turnover of organic products (enhancing microbial biomass retention), absorbing nutrients and reducing decomposition rates, buffering pH changes and protecting microbes from predation.

Decomposition, N immobilisation and mineralisation rates were shown to be slower in soils of heavy clay texture than in sandy soils (Van Veen *et al.*, 1985). West *et al.* (1988) also found an effect on N flush by reducing moisture content of soils with different textures (clay-loam, silt-loam and sandy soil). On further investigation, they reported that the proportional decline in N flush following the drying regime was greatest in clay-loam than sand. This is supported by Sparling *et al.* (1987) although they suggested that the reason may be due to microbes from droughty soils being pre-adapted to moisture stress and being better at resisting laboratory drying regimes, rather than the findings being related to soil texture *per se*. In a study of Australian *Pinus radiata* monoculture, with all other factors being equal, sandy soil was shown to induce the most stressful conditions. This caused low N content in leaf litter, extensive development of complex proteins in the leaves and consequently low rates of N mineralisation (Lamb, 1975).

Site characteristics such as soil water-holding capacity, aeration and slope tend to modify the effects of climate on N turnover. Site factors that improve moisture and temperature with respect to certain species will increase N cycling rates.

pH

A number of studies have shown increased mineralisation on limestone soils (although these may not necessarily be calcareous). These are supported by studies where liming conifer soils has increased decomposition, mineralisation and nitrification (Gosz, 1981). Also, in a review of microbial biomass by Wardle (1992), it was concluded that pH is an important factor in influencing soil microbial biomass in terms of the proportion of N that is immobilised there (but not the biomass itself). Where soil pH has been experimentally lowered (to simulate acidification), microbial biomass or its activity has not been severely affected until soils are pH < 2 or 3.

Grazers

Soil fauna have been shown to alter rates of substrate utilisation and nutrient release. They mechanically process litter and detritus, influencing organic matter distribution, temporal homogeneity of decomposing sites and ultimately the decomposing process. They excrete N as ammonium, urea or amino acids, which can further stimulate decomposition processes and primary production.

Van Veen *et al.* (1981), in their review of nitrogen cycle models, report that in the absence of grazers there is experimental evidence of bacteria mineralising little or no N or phosphorus (P). Experimental work also shows that the effect of grazers on mineralisation is different in coarse or fine-textured soils.

Grazers may be crucial where plants have comparatively short growing seasons and nutrient availability is crucial. In van Veen *et al.* (1981), models demonstrate the importance of amoebas and nematodes in soil microcosms, but we need a better understanding of the abundance and trophic relationships of all soil organisms in natural ecosystems to judge how they might affect mineralisation. The Dutch Soil Quality Network is looking at the main functional groups of the soil food web in an effort to relate the structure of the soil community to functions (Bloem *et al.*, 2003).

Above-ground grazers can also influence N input to soil by consumption of nutritive parts of plants and by excretion products. For example, during severe attacks by leaf-eating insects, the annual N input to soil in some European deciduous forests was found to rise 30-50 per cent (Staaf and Berg, 1981).

Inherent variability

The accumulation of N, its distribution amongst pools, and its exchange rates will vary over orders of magnitude in relation to ecological succession. But there is sufficient understanding to generalise and predict variation in biomass and detritus (and hence mineralisation) with disturbance and ecological succession. For example, from the higher proportions of N held in forest biomass, removal or death of biota will have much greater immediate impacts on the N cycle of forests than grassland.

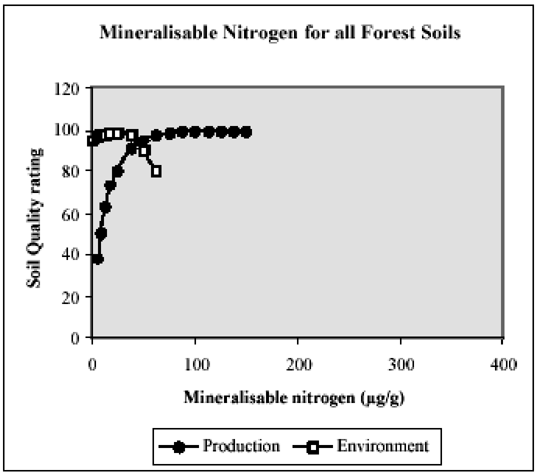
The New Zealand group concluded that there was no effect of soil order on variation in mineralisable N, but land use affected variability, with croplands having the lowest variability followed by pasture and plantation forests.

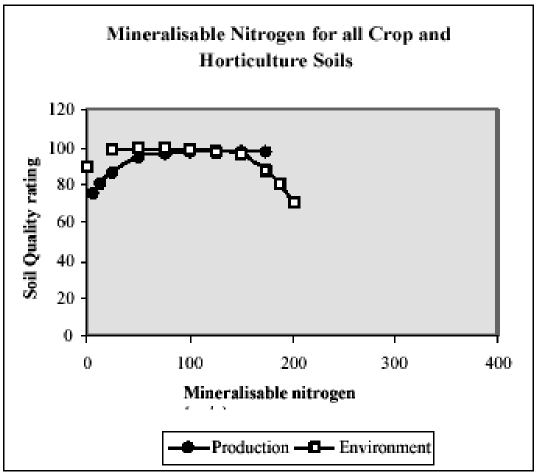
Transformations of soil N occur and are controlled at soil microsites. Inclusion of biomass in models directs attention to microsites and may provide a base from which various systems can be normalised and compared (Van Veen *et al.*, 1981).

In summary, strongly acidic soils, cold temperatures, very dry conditions and poorly aerated or waterlogged soils inhibit mineralisation. Factors such as the presence and abundance of N fertiliser applications, moderate or neutral soil pH, adequate moisture retention and residue incorporation (decomposition) favour accumulation of N in plants and organic matter. Soils can be ranked in terms of N-mineralisation, but it is difficult to define particular values of N mineralisation as representing good or poor soil quality, though this has been attempted in the next section.

Envelopes of normality

The New Zealand group has published its criteria for mineralisable N in forest, crop and horticulture and pasture soils (Figure 2.3.5.2 from Sparling *et al.*, 2003). For the purposes of this review we are only interested in the soil quality rating for the environment, but the divergence of the curves highlights the potential conflict between environmental regulators and some landowners.





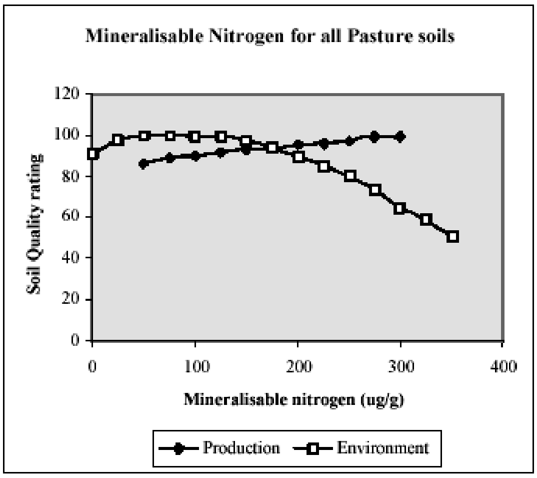


Figure 2.3.5.2
New Zealand soil quality response curves for anaerobic mineralisable N

The New Zealand group has defined baseline values for a range of soil types and land uses through the use of these response curves and expert judgement (Sparling *et al.*, 2003). The trigger values in Table 2.3.5.1 show provisional quality classes and target ranges for anaerobically mineralised N. The numbers in bold represent lower and upper limits (µg N/g). These data are applicable to all soil orders.

Pasture	25	**50**	100	200	200	**250**	300
Forestry	5	**20**	40	120	150	**175**	200
Cropping and Horticulture	5	**20**	100	150	150	**200**	225

Table 2.3.5.1

Very Low	Low	Adequate	Ample	High	Excessive

In Table 2.3.5.2, separate N mineralisation values for coniferous and deciduous forestry have been compiled. The typical or mean values compare well with those of the New Zealand group, but the extremes would trigger concern if they were measured there.

Table 2.3.5.2: Range of variation in rates of N mineralisation (Clark and Rosswall, 1981)

	Minimum value	Typical values	Maximum value
Coniferous forests[1]	9	30–50	125
Deciduous forests[2]	0–24	110 (mean)	375–399

* Mineralisation rates are in kg/ha/yr
[1] Gosz, 1981.
[2] Melillo, 1981.

When the New Zealand group tested their soil quality indicators, some showed little difference between each other across soil groups, whereas others, such as potentially mineralisable N, had a 15-fold range. Cropped soils generally had less potentially mineralisable N (microbial respiration and biomass C) than the equivalent soil under pasture, radiata pine, or indigenous forest. The overall variability of each indicator was obtained by calculating the coefficient of variation (CV) at each site and then averaging across all the sites. The CV for mineralisable N was 13.2% (Sparling *et al*., 2004).

Strong correlation between indicators means it may not be necessary to measure them all. There was a strong linear correlation ($R = 0.63$, $P < 0.001$) between microbial biomass C and potentially mineralisable N, which was also observed elsewhere (Hart *et al.*, 1986; Myrold, 1987). The correlation suggested that the more readily determined measure of potentially mineralisable N could serve as a satisfactory surrogate for microbial biomass (Schipper and Sparling, 2004). This is what has been recommended by the New Zealand and mineralisable N is currently their only biological indicator of soil quality.

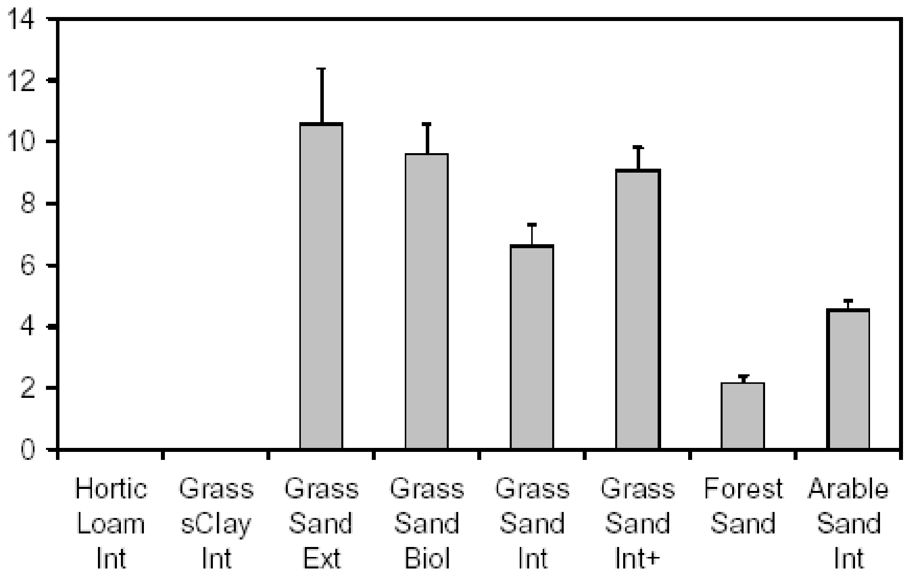


Figure 2.3.5.3: PMN (mg N/kg soil per week) from the Netherlands Soil Quality Monitoring Network

The Netherlands Soil Quality Network measures N mineralisation in 6-week laboratory incubations. The Dutch saw a similar trend in N mineralisation with land use as the New Zealand group. PMN was very low in forest, higher in arable land and highest in grassland (Figure 2.3.5.3). Further definition was seen between types of farming. At extensive and organic [Biol] grassland farms N mineralisation was about 50% higher than at intensive farms. But at highly intensive [Int+] farms mineralisation was not reduced, probably because more (pig) manure was applied. Note that Dutch soil texture classes do vary from those in the UK.

Integration with other frameworks

Most countries with soil monitoring frameworks have identified PMN as an indicator of soil quality. New Zealand uses PMN as their only biological indicator - specifically long-term fertility changes – in the minimum dataset. Hart *et al.* (1986) found a highly significant correlation between microbial biomass C and mineralisable N in a range of plant litters, peats and soils of moderate to low pH. Both measurements were sensitive to previous changes in management from indigenous forestry to exotic forestry and grassland development. This supported the findings of other New Zealand monitoring studies and the relationship strongly influenced New Zealand's decision to use mineralisable N as an indicator of soil fertility and microbial biomass.

The Netherlands Soil Quality Network measures N-mineralisation in 6-week laboratory incubations and has steadily been increasing the land uses and soil types monitored.

N mineralisation is recommended for use in the Environment Agency and Defra's draft ecological risk assessment framework for assessing harm to ecosystems from contaminated soils. However, the test may be limited to a screening assessment as the assay does not fully produce dose-response results (desirable when assessing chemical contamination of soil), nor is it as sensitive as lower organisational bioassays such as enzyme assays (Spurgeon *et al.*, 2002).

Costs and ease of use

Numerous chemical and biological methods have been developed for determining mineralisable N. Several reviews discuss the pros and cons of chemical methods (Drinkwater *et al.*, 1996). Although chemical methods tend to be more rapid and convenient than biological methods, they do not simulate the activities of soil microbes. Neither do they selectively release the fraction of soil organic N that is made available for plant growth by microbial activity (Drinkwater *et al.*, 1996), relying instead upon subsequent correlation with biological measurements of N availability such as crop yield.

Biological methods involve measuring the amount of mineral N released in the soil by microbial activity during incubation. These methods are time-consuming, but they are more ecologically relevant since they rely on normal biological mineralisation processes.

Stockdale and Rees (1994) compared the following N availability methods using Scottish soils: available mineral N (MIN-N); microbial biomass N (BIO-N); N released by anaerobic incubation (AN-N); N released by drying and reincubation (DRY-N); chemically extracted N (CHEM-N); and uptake in pot experiment with ryegrass (RYE-N). They found highly significant differences between soils for all measurements. CHEM-N showed a near perfect linear relationship with total soil N and had markedly different correlations with soil organic carbon, total soil N and soil C:N ratio from all the other techniques. CHEM-N extracted N from a different pool or pools more closely related to the total soil N than the biological techniques. Similar correlation coefficients were seen for BIO-N, AN-N and DRY-N, which supported the theory that they extracted N mainly from the same N pool, presumably microbial biomass. Myrold (1987) and Marumoto *et al.* (1982) both support this hypothesis but Stockdale and Rees were unable to confirm that the extracted N pool was from microbial biomass.

Drinkwater *et al.* (1996) recommend an aerobic and an anaerobic method for measuring mineralisable N. Aerobic incubations provide optimal temperature, moisture and aeration conditions for the microbial population responsible for mineralisation under most field conditions. The major disadvantage is maintaining optimal soil water content during incubation, although this can be alleviated by covering vessels with pierced parafilm (to allow gas exchange) and checking water content (60 $\pm$ 5%) at regular intervals. Drinkwater *et al.* (1996) consider there is less of a consensus for anaerobic methods as they may measure the potential for soil to deliver N, but not necessarily reflect microbial biomass N (such as composition or metabolic status of the microbial community) or the composition of the organic substrate being mineralised. In contrast, Hart *et al* (1986), Myrold (1987) and Williams and Sparling (1988) found a highly significant correlation between microbial biomass carbon (or N) and mineralisable N (using the anaerobic method) and Sparling recommends the anaerobic method for soil monitoring in New Zealand.

Recommended methods for measuring (aerobic) nitrogen mineralisation are described by Barraclough (1995) and O'Dowd *et al.* (1999). The N-mineralisation test should be reproducible, as it has successfully been ring-tested and standardised by OECD (2000) and ISO (1997). Sample handling presents the greatest challenge in terms of standardisation (Drinkwater *et al*, 1996). The following practices are recommended to promote standardisation:

- Timing - Samples collected shortly after management practices that stimulate microbial activity, such as residue additions or tillage, will have greater N mineralisation potentials. This may be difficult to distinguish from soil preparation unless land management practice is recorded at the time of sampling. Also, seasonal effects on microbial biomass will also impact on mineralisation potentials.

- Sieving - This increases N mineralisation potential, particularly in no-till systems, by exposing previously protected labile organic matter. But sieving increases homogeneity of soils. Drinkwater *et al.* (1996) recommends sieving soils when N-mineralisation is being used as an indicator of soil quality.

- Air-drying and re-wetting - Air-drying has been shown to increase the amount of mineral N produced compared with field-moist soil, and this increase is positively correlated with the length of time the air-dry sample is in storage prior to incubation (Birch, 1960). Conversely, Keeney and Bremner (1966) found that air-dried soils correlated better with crop response than field-moist soils. Dry–wet cycles in the environment will influence N mineralisation potential, so Drinkwater *et al.* (1996) recommend collecting samples at field capacity, such as after rainfall, and then air-drying to standardise soils prior to routine incubation. Soil samples should be kept at 4 °C prior to incubation.

- Temperature - Drinkwater *et al.* (1996) recommend soil temperature to be between 30 and 35 °C and that this temperature should remain constant throughout the duration of the test.

- Incubation conditions - Recommended optimal soil conditions for aerobic incubation are 60% water-holding capacity, soil temperature between 30 and 35 °C, and soil pH between 6.6 and 8.0.

This test requires a moderate level of technical equipment (scintillation counter (^{15}N); automated Flow Injection Analyser). The ISO guidelines recommend an exposure period of 28 days.

Opinion is divided over which method to use. We recommend that samples are air-dried and sieved (with standardised antecedent conditions) because of the difficulties in standardising sampling, storage and pre-treatment protocols with intact cores. We also recommend that biological methods are followed though there isn't a clear preference for aerobic or anaerobic incubations. The anaerobic method has been used in all recent Defra work to rank soils on their mineralisation capacity. But the Environment Agency and the ISO guidelines recommend aerobic incubations.

The costs associated with this test are dependent upon how many ammonium and nitrate analyses are required. For example, a recent price for running the N-mineralisation test on 150 soil samples (inclusive of replicates) was approximately £2,700 (£18 per sample). This included adjusting soils to a fixed moisture content, running the incubation at a fixed temperature, initial and final potassium chloride extractions and using an auto-analyser to measure ammonium and nitrate concentrations. Prices rise incrementally where more extractions and analyses are required, such as determining the rate of change (trends) in N-mineralisation over a number of weeks.

Another estimate from a government laboratory in the UK was £281.25 per sample for the same number of extractions. This is likely to be a top-end price.

£20 per sample is considered to be an average price.

In the Bhogal *et al.* (2000) study of grassland soils, potentially mineralisable N decreased down the soil profile to a depth of 90 cm. But the findings showed that 40-50% of the decline was observed in the top 10 cm.

We recommend that the top 15 cm is the biologically active and important layer to sample.

Giltrap and Hewitt (2003) analysed spatial variability for each NZ indicator. They concluded that mineralisable N required >30 m spacing between data points (the number of data points and analyses per monitoring site is unknown at present). They also concluded that there was no effect of soil order, but land use affected variability, with croplands having the lowest variability followed by pasture and plantation forests.

Recommendation

There are mixed views in the literature and expert opinion over the suitability of PMN as an indicator for use in the minimum dataset (MDS). The advantages of including PMN are its ability to indicate arable cropping fertility and potential leaching of nitrate from soils, and as a surrogate for biological activity. PMN has advantages over measuring biomass N (or carbon) in that it should be relatively insensitive to sample timing, whereas biomass N is very sensitive to temporal variation.

Conversely, PMN can be viewed as unsuitable as an indicator in the MDS because it is so variable. It is very sensitive to fertiliser and organic manure applications and it can be linked to changes in environmental interaction but interpretation is difficult.

It is the very virtues of PMN as an integrator of chemical, physical and biological aspects of soil health – that is, combining the accumulation of N through previous biological activity, the present organic matter status of the soil, and the current N mineralisation activity of the soil microbes - that makes PMN measurements so difficult to interpret. It would be challenging to compare PMN measurements across soil types and land uses (as demonstrated by the New Zealand group). But PMN may be easier to interpret in relatively undisturbed systems (such as forests, upland grassland).

Choosing a minimum dataset and ranking of indicators remains a subjective exercise. Looking at monitoring programmes established in other countries, it appears that microbial biomass, respiration and PMN are commonly regarded as part of a minimum dataset (Bloem *et al.*, 2003). If PMN were to be used in soil quality monitoring (beyond the MDS) then we would need to pay attention to the selection of appropriate triggers. We recommend that triggers for environmental interaction are considered separately from those for production. There would be a conflict between the two at high thresholds, where excessive values could indicate N inefficiencies and breakdown in retention function for environmental interaction.

Ultimately a direct and interpretable link between PMN and soil quality cannot be proven.

References

Adams, W. A. and Jan M T. (1999) Utilisation of nitrogen accumulated by a clover containing ley following cultivation. *Soil Use and Management*, 15, 247-253.

Addiscott, T.M. (1983) Kinetics and temperature relationships of mineralisation and nitrification in Rothamsted soils with differing histories. *Journal of Soil Science*, 34, 343-353.

Barraclough, D. (1995) N-15 isotope dilution techniques to study soil nitrogen transformations and plant uptake. *Fertilizer Research*, 42, 185–92.

Bhogal, A., Murphy, D.V., Fortune, S., Shepherd, M.A., Hatch, D.J., Jarvis, S.C., Gaunt, J.L. and Goulding K.W.T. (2000) Distribution of nitrogen pools in the soil profile of undisturbed and re-seeded grasslands. *Biology and Fertility of Soils*, 30, 356-362.

Birch, H.F. (1960) Nitrification in soils after different periods of dryness. *Plant and Soil* XII, No.1, 81-96.

Bloem, J., Schouten, T., Didden, W., Jagers, O.P., Akkerhuis, G., Keidel, H., Rutgers, M. and Breure, T. (2003) Measuring soil biodiversity: experiences, impediments and research needs. *OECD expert meeting on soil erosion and soil biodiversity indicators*, 25-28 March 2003, Rome, Italy.

Bonde, T.A., Schnurer, J. and Rosswall, T. (1988) Microbial biomass as a fraction of potentially mineralisable nitrogen in soils from long-term field experiments. *Soil Biology and Biocehmistry*, 20: 447-52.

Campbell, C.D., Chapman, S.J. and Urquhart, F. (1995) Effect of nitrogen fertiliser on temporal and spatial variation of mineral nitrogen and microbial biomass in a silvopastoral system. *Biology and Fertility of Soils*, 19, 177-184.

Dalal, R.C.. (1998) Soil microbial biomass – what do the numbers really mean? *Australian Journal of Experimental Agriculture*, 38: 649-65.

Doran, J.W. and Safley, M. (1997) Defining and assessing soil health and sustainable productivity. In *Biological Indicators of Soil Quality*, edited by CE Pankhurst, BM Doube and V S R Gupta, 1-28. Wallingford: CAB International.

Drinkwater, L.E., Cambardella, C.A., Reeder, J.D. and Rice, C.W. (1996) Potentially mineralisable nitrogen as an indicator of biologically active soil nitrogen. *Soil Science Society of America*, Special Publication, 49, 217-229.

Clark, F.E. and Rosswall, T. (editors) (1981) Terrestrial nitrogen cycles. *Ecological Bulletin,* 33.

Ellenberg, H. (1971) Nitrogen content, mineralisation and cycling – In: Duvigneaud P (Ed). *Productivity of Forest Ecosystems. Proceedings from a Brussels symposium*, October 1969, 509-514. Paris: UNESCO.

Gosz, J.R. (1981) Nitrogen cycling in coniferous ecosystems. *Ecological Bulletin,* 33, 405-426.

Hart, P.B.S., Sparling, G.P. and Kings, J.A. (1986) Relationship between mineralisable nitrogen and microbial biomass in a range of plant litters, peats and

soils of moderate to low pH. *New Zealand Journal of Agricultural Research*, 29, 681-686.
ISO (1997) *Soil Quality – Biological methods – Determination of nitrogen mineralization and nitrification in soils and the influence of chemicals on these processes.* International Standards Organisation, Geneva.
Keeney, D.R. and Bremner, J.M. (1966) Comparison and evaluation of laboratory methods of obtaining an index of soil nitrogen availability. *Agronomy Journal* 58, 498-503.
Lamb, D. (1975) Patterns of nitrogen mineralization in the forest floor of stands of *Pinus radiata* on different soils. *Journal of Ecology,* 63, 615-625.
Loveland, P. and Thompson, T. (2002) *Identification and development of a set of national indicators for soil quality.* Environment Agency R&D Technical report P5-053/2/TR
Marumoto, T., Anderson, J.P.E. and Domsch, K.H. (1982) Mineralisation of nutrients from soil microbial biomass. *Soil Biology and Biochemistry,* 14 (5): 469-475.
McRea, A.R., Trueman, I.C. and Fullen, M.A. (2004) Factors relating to soil fertility and species diversity in both semi-natural and created meadows in the West Midlands of England. *European journal of Soil Science*, 55, 335-348.
Melillo, J.M. (1981) Nitrogen cycling in deciduous forests. *Ecological Bulletin*, 33, 427-442.
Miller, H.G. and Johnson, D.D. (1964) The effect of soil moisture tension on carbon dioxide evolution, nitrification and nitrogen mineralisation. *Soil Science Society American Proceedings*, 28, 644-647.
Myrold, D.D. (1987) Relationship between microbial biomass nitrogen and nitrogen availability index. *Soil Science Society of America Journal*, 51, 1047-1049.
O'Dowd, R.W., Barraclough, D. and Hopkins, D.W. (1999) Nitrogen and carbon mineralization in soil amended with D- and L-leucine. *Soil Biology and Biochemistry*, 31, 1573–8.
OECD (2000) *Guidelines for the Testing of Chemicals:216. Soil Microorganisms, Nitrogen Transformation Test*. Organisation for Economic Co-operation and Development, Paris, France.
Scholefield, D. (2003) Some aspects of crop quality on environment and biodiversity. *Aspects of Applied Biology*, 70, 53-61.
Sparling, G., Lilburne L, Vojvodiæ-Vukoviæ M. (2003) *Provisional Targets for Soil Quality Indicators in New Zealand*. Landcare Research New Zealand Ltd.
Sparling G. 1997. Soil microbial biomass, activity and nutrient cycling as indicators of soil health. In *Biological Indicators of Soil Health*. Ed Pankhurst et al; pp 97-119.
Sparling G P, Milne J G, Vincent K W. 1987. Effect of moisture regime on microbial contribution to Olsen P values. *New Zealand Journal of Agricultural Research*: 30, 79-84.
Sparling, G.P., Schipper, L.A., Bettjeman, W. and Hill, R. (2004) Soil quality monitoring in New Zealand: practical lessons from a 6-year trial. *Agriculture, Ecosystems and Environment,* 104, 523-534.
Spurgeon, D., Svendsen, C., Hankard, P., Weeks, J., Kille, P. and Fishwick, S. (2002). *Review of sublethal ecotoxicological tests for measuring harm in terrestrial ecosystems.* Environment Agency R&D Technical Report P5-063/TR1
Staaf H and Berg B. 1981. Plant litter input to soil. *Ecological Bulletin*, 33, 147-162.
Stockdale E A and Rees R M. 1994. Relationships between biomass nitrogen and nitrogen extracted by other nitrogen availability methods. *Soil Biology and Biochemistry*, Vol 26, No 9, 1213-1220.

Van Veen, J.A., McGill, W.B, Hunt, H.W., Frissel, M.J. and Cole, C.V. (1981) Simulation models of the terrestrial nitrogen cycle. *Ecological Bulletin*, 33, 25-48.
Vlassak, K., Paul, E.A. and Harris, R. (1973) Assessment of biological fixation in grassland and associated sites. *Plant Soil*, 38, 637-649.
Wedin, D.A. and Tilman, D. (1990) Species effects on nitrogen cycling: a test with perennial ryegrass. *Oecologia*, 84, 433-441.
West, A.W., Sparling, G.P., Speir, T.Q. and Wood, J.M. (1988) Comparison of microbial C, N-flush and ATP, and certain enzyme activities of different textured soils subject to gradual drying. *Australian Journal of Soil Research*, 26, 217-229.
Williams, B.L. and Sparling, G.P. (1988) Microbial biomass carbon and readily mineralized nitrogen in peat and forest humus. *Soil Biology and Biochemistry*, 20, 579-581.
Woodmansee, R.G., Vallis, I. and Mott, J.J. (1981) Grassland nitrogen. *Ecological Bulletin*, 33, 443-463.

2.3.6 Soil macroporosity

Background

Macroporosity, also referred to as air-filled porosity, is a measure of the volume of large pores >60 µm in the soil. Although total pore space (micro- and macropores) typically comprises half the soil volume, it is the volume of macropore space that is most critical to soil sustainability, both in terms of productivity and environmental well being. Macropores are the primary conduits for air and water penetration into and through the soil, and they also provide the optimum environment for solute transport, effective root development and microbial activity. The 60 µm approximates to the pore diameter and is the pore size beyond which gravitational drainage occurs, so macroporosity also has a crucial role in soil hydrology. When a soil is subject to physical compaction or another form of structural degradation, macropore volume is the first to be lost. A decrease in macroporosity reduces aeration and water availability to plants and soil microbes and limits the physical environment for root development. It also restricts infiltration and drainage, thus increasing overland flow and soil erosion. Ultimately, all these factors impact on vegetation cover, biomass and crop yields. (Sparling *et al.*, 2003).

Links with other measurable soil parameters

Macropore space is reduced through direct compaction or through mechanical, chemical or biological breakdown of large soil aggregates into more densely packed smaller aggregates and particles. A direct measure of macroporosity is important because it provides understanding of pore size distribution and poor continuity. Indeed, changes in pore size distribution and continuity (such as through tillage) can vary greatly over time and it is not possible to gain this knowledge from bulk density alone. A measure of macroporosity also supports our understanding of aggregate stability, mean weight diameter, hydraulic conductivity, solute diffusion/transport and assessments of soil strength for crop production (Chen *et al*., 1998). Also, for all soil groups in England and Wales, Loveland and Webb (2003) found that increasing pore size and an increasing proportion of air-filled pore space were related to soil organic carbon (SOC) levels. In terms of biological indicators, such as mineralizable nitrogen, any decline in soil structure, particularly macropore volume, is also frequently associated with decreases in microbial biomass and activity (Neves *et al*., 2003; Sparling *et al*., 2000). For example, it was reported by Sollins *et al.* (1996) that 95% of the pore spaces in a degraded silt-loam were too small to be accessible to bacteria.

Pressures on soil macroporosity

For arable systems in the UK, where up to 90% of a field may be crossed by wheelings in the course of a single year due to ploughing, seed bed preparation and fertiliser and pesticide application, the potential for compaction is a serious problem (Arden-Clarke and Hodges, 1987). Such practices mostly have a negative effect on soil structure, with loss of macropore space in the top 30 cm being a primary symptom (Chen *et al*., 1998; Neves *et al*., 2003). Tillage increases the volume of macropores over a short duration whilst decreasing the continuity of macropores and micropores. However, over a longer duration, tillage generally increases bulk density, decreases water stable aggregation and reduces plant available water capacity and infiltration (Shulka *et al.,* 2003). Regular tillage can also lead to a decline in SOC through oxidation and also minimum return of crop residue (Loveland and Webb, 2003). Any such decline in SOC may impact on soil structure, particularly macroporosity and bulk density due to a decline in aggregate stability and aggregate size. Indeed, minimum tillage practice and regular organic amendments that retain and increase SOC have been shown to have a positive effect on soil structural (Amezketa, 1999).

In grassland systems, poaching from sheep and in particular cattle can cause physical degradation of soil between seasons. This is more likely with high stocking rates and can quickly result in decreased soil permeability due to reduced pore space and continuity (Drewry *et al.,* 2000). The impacts of poaching are compounded in wet soil conditions (Drewry and Paton, 2000). However, for soils in the UK, Scholefield and Hall (1986) found that the loss of soil structure due to poaching occurs as a progressive degradation and not immediately on treading of wet soil. Previous work by Scholefield and Hall (1985) found that poaching is also influenced by clay content, bulk density, sward strength, weather and management. Scholefield *et al*. (1985) also noted that most structural damage and compaction occurred at 10 cm below the soil surface and that poaching might even reduce bulk density in the top 10 cm.

Potential of macroporosity as an indicator of soil quality

Macroporosity is an excellent complementary measure to bulk density as it reveals important information on pore size distribution. To be included in a minimum dataset (MDS) of soil quality indicators for the UK, macroporosity will need to demonstrate sufficient variation between and within soil and land use groupings and be an interpretable indicator of change. Its usefulness has been reported in national-scale studies. In New Zealand, Drewry *et al*. (2000) reported that bulk density and macroporosity under grazed pasture showed significant differences both between soil groups and for rapid temporal changes within soil groups. Also in New Zealand, but for silt-loam soils only, Sparling *et al.* (2000) found there to be significant differences in macroporosity between land use classes in decreasing order of arable > pine forest > pasture > native forest.

Method to determine macroporosity

A series of measures based on moisture release characteristics, particle and bulk density are needed to calculate macroporosity. Moisture release characteristics are obtained from drainage on pressure plates at specific tensions (-5, -10, -100 and -1500 Kpa). The proportion of macropores is calculated from the total porosity and moisture retention data: $S_m = S_t - \theta$, where S_m is macroporosity, and θ is the volumetric water content at –10 Kpa (field capacity) tension (although some researchers claim that -5 kPa is more relevant). The total porosity is calculated from the formula: $S_t = 100[1-(p_b/p_p)]$ where S_t is the total porosity, p_p is the particle density and p_b is the dry bulk density. The dry bulk density is obtained from the mass of an intact soil core of known volume dried at 105°C. The weight of the oven-dry soil, expressed per unit volume, gives the bulk density. The particle density is measured by the pipette method and used to calculate total porosity as explained above.

Porosity is expressed as the proportion of pores per unit volume of soil (v/v percent). Relying on bulk density data, one disadvantage of the macroporosity method is that, without careful handling, the intact cores are susceptible to damage and loss of soil volume during sample extraction, transport and handling (Hernanz *et al*., 2000). A second disadvantage is that macroposity can be greatly modified (usually increased) by dislodging stones during core sampling. Indeed, soil stoniness is a far more pertinent factor for macroporisty than bulk density.

Similar to bulk density, there are issues of protocol for the selection of macroporosity sampling site and sampling time. The issues described for bulk density in Section 2.3.3 are detailed again here for completeness.

The chosen scale and level of statistical interpretation, whether national, regional or local, will also dictate the level of importance attached to the location and timing of sample collection. In terms of sampling location, one main difficulty in measuring physical parameters such as macroporosity is that they vary in position in the soil profile (Neves *et al*., 2003). For example, processes of compaction and disaggregation induced by machinery or livestock may be spatially heterogenous. This heterogeneity is rarely considered but will result in considerable variation in measurements in time and space. Topographic location (such as position on hillslope) may also affect macroporosity due local to variation in clay and organic matter content (Bissonnaise *et al*., 2002). Once a sampling site has been selected, consideration should be given to temporal resolution of sampling and the temporal variation that can occur in macroporosity measurements. These temporal issues are discussed in the next section of this report.

The sampling depth is also an important consideration for macroporosity in terms of identifying change. Sparling *et al.* (2000) noted that the top 10 cm of the soil is most relevant for biological and some chemical characteristics, accounting for the highest nutrient concentrations and over 80% of soil organic matter and microbial biomass and activity. Therefore, what depth should we sample? In cultivated systems that undergo primary tillage, secondary seed bed preparation and further traffic due to fertiliser/pesticide application and harvesting, soil compaction can affect all soil from 0-30 cm (Chen *et al.,* 1998; Hernanz *et al.,* 2000). On arable land, cultivation and subsequent settling and slaking of the soil, along with the effects of various episodes of traffic during crop management, will reduce macroporosity in and sometimes below the tilled horizon (Harrod, *Pers. Comm.*). However, in grazed systems, soil physical damage incurred by poaching is limited to the surface 10 cm (Drewry *et al*., 2000; 2001).

For all land use and soil combinations in New Zealand, Sparling and Schipper (2002, 2004) determined macroporosity for the top 10 cm, although the percentage of land under cropping is small. Nonetheless, Brejda *et al*. (2000a, 2000b) and Shulka *et al*. (2003, 2004) selected a <10 cm sampling depth for cultivated systems in the USA. Other researchers have selected 0-30 cm sampling depths for detailed investigation of cultivation-induced compaction; however, it is likely that the 0-10 cm strata is sufficient to indicate bulk density change in conventional and minimum tilled systems.

No single depth can provide a definitive picture of physical soil change. The horizon most susceptible to physical change will vary between soils and management. In particular, the top 10 cm may be too ephemeral to be interpreted as a SQI. Confining what is said about soil macroporosity to the near surface will make it hard to arrive at a meaningful interpretation of soil qualities. For example, most plants root to about one metre depth and water from this depth may be vital for plant sustainability during dry periods. Expert advice advocates the importance of taking three depth measurements for bulk density and macroporosity: 0-10 cm, middle of the topsoil and top of the subsoil (M.J.Reeve and T. Harrod*, Pers. Comm*.).

What does change mean?

A change in soil macroporosity over time demonstrates a change in pore size distribution that may result from compaction or loosening or breakdown of aggregates and soil structure. The pore size distribution also relates to other physical parameters such as hydraulic conductivity and the water release curve or chemical and biological parameters such as SOC and mineralizable N, respectively.

When considering broad-scale national or regional changes for different soil and land use combinations, macroporosity is likely to provide a useful indicator of change. However, we must consider whether we intend to investigate local-scale as well as national/regional-scale changes in soil quality. If so, the national monitoring strategy will have to be designed accordingly.

What are the rates of change?

Macroporosity responds rapidly to perturbation by livestock and cultivation. It will also change between seasons due to natural amelioration processes that occur with time after perturbation (such as consolidation of loose aggregates, wetting and drying and freeze–thaw cycles, worm activity and root activity) (Drewry *et al*., 2001; Shulka *et al*., 2004).

Topsoil physical properties (0-10 cm) can change rapidly in response to poaching, especially in conditions of high stocking rates and high soil moisture (Drewry *et al*., 2001; Drewry and Paton, 2000; Singleton *et al*., 2000). Drewry and Paton (2000) also reported that a reduction in grazing, or an omission of grazing, from months to a few years is sufficient to improve soil physical structure. Similar temporal variation is also reported for cropped systems, where changes in topsoil physical properties (0-30 cm) can be rapid and highly variable (Shulka *et al*., 2003, 2004). Lodgson and Cambardella (2000) reported statistically significant changes in soil bulk density in the top 20 cm (random increases and decreases, ranging from 0.91 to 1.44 Mg/m^3), both between and within successive sampling years. However, Chen *et al*. (1998) noted that changes will obviously be less pronounced under minimum tillage.

Measurements of soil physical parameters, such as bulk density and macroporosity, will be strongly influenced by timing in relation to livestock grazing, cultivation, soil moisture conditions, grazing/cultivation history and depth of topsoil sampled. Consideration of these factors is critical in terms of interpreting long-term soil quality trends against background noise. Timing of sample collection is therefore a critical consideration for assessing changes in soil physical quality. The best time to assess field soil physical attributes is in the spring when the soil is near to field capacity. This timing is particularly important for clayey soils, which will be at their maximum field volume (lowest bulk density) in spring (Hall *et al*., 1977).

For all measured SQIs, measured data must be interpreted in terms of the potentials and constraints of specific land use, management systems and soil types. Each combination should be considered for any combination-specific relationship that exists between sustainable land use and measured SQI (Shulka *et al*., 2003).

Is change measurable?

In terms of demonstrating differences between soil and land use, macroporosity has been suggested as a measurable and useful indicator at a national scale (Sparling and Schipper, 2002, 2004). However, the New Zealand soil quality indicator study showed that macroporosity, by itself, may not show sufficient variation to be a useful indicator. However, macroporosity data for England and Wales may be more informative. Similar to bulk density, an important question is how useful macroporosity will be as an indicator of long-term change at a specific location? This will depend on the robustness and consistency of sampling protocol in terms of temporal and spatial factors.

Trigger values for macroporosity

In an applied study of macroporosity as an indicator of soil physical quality in New Zealand, Sparling *et al*. (2003) defined macroporosity trigger values for two land use groupings with no discrimination of soil type. The trigger values are shown in Table 2.3.6.1. Acceptable macroporosity was defined as the range between the values denoted in bold type data, that is, between ‘no significant impact’ and ‘above target range’. The data suggest that, with the exception of organic soils, sustainable macroporosity ranges are generally consistent between land use groups.

Table 2.3.6.1: Provisional quality classes and target ranges for macroporosity – applicable to two land use groupings for all New Zealand soils (Sparling *et al*., 2003)

Landuse group	Macroporosity (%v/v)				
	Very low	Low	Adequate		High
Pasture, cropping and horticulture	0	**6**	8	**30**	40
Forestry	0	**8**	10	**30**	40

Hall *et al.* (1977) used < 5% as an indicator of a very low macroporosity likely to be associated with soil quality deficiencies but later research and experience suggests that 7-8% might be a more relevant threshold. This accords reasonably well with Sparling’s New Zealand work. However, the textural range of soils in New Zealand is limited in comparison to England and Wales. (T. Harrod and M.J. Reeve, *Pers. Comms*.).

More detailed analysis of the same dataset showed macroporosity data in relation to land use for all soils (Sparling and Schipper, 2004) and for silt-loam soils only

(Sparling *et al.,* 2000). The data are shown in Table 2.3.6.2. The data would suggest that land use does not have a strong influence on macroporosity. In particular, the data for all soils show macroporosities are similar for arable, pasture and natural forest and only noticeably higher for pine forest. Data for silt loam suggests the same. The small standard errors suggest little variation within land use groupings. Only arable soils have macroporosity values that are below the levels defined as acceptable for most soil types in Table 2.3.6.1.

In the New Zealand study, Sparling and Schipper (2004) noted that any high macroporosity found under arable was most likely attributable to recent cultivation – cultivation aerates the soil and requires periods of several months for the topsoil to be re-consolidated. They also noted that low macroporosity was found on a high proportion of pasture soils with high stocking densities. Macroporosity <10% has been shown to affect pasture production adversely (Drewry *et al.*, 2000; Drewry and Paton, 2000; Singleton *et al.*, 2000; Drewry *et al.*, 2001). However, the New Zealander group found there to be no data relating to macroporosity trigger values for any other land use.

Table 2.3.6.2: Topsoil macroporosity data for major land use classes in New Zealand (Schipper and Sparling, 2000 and Sparling and Schipper, 2004)

Landuse description	Soil description	Macroporosity (%v/v)	
		(mean ±S.E)	(range)
Natural forest	All soils	9.3 (1.3)	
Natural forest	Silt-loam		9–17
Pine forest	All soils	25.6 (1.2)	
Pine forest	Silt-loam		9–28
Dairy pasture	All soils	10.1 (0.5)	
Drystock pasture	All soils	13.3 (0.6)	
Tussock grassland	All soils	15.6 (1.3)	
Pasture	Silt-loam		6–9
Arable crop	All soils	14.7 (1.3)	
Mixed crop	All soils	9.3 (1.3)	
Arable	Silt-loam		2–12

Table 2.3.6.3: Summary statistics of macroporosity for major soil textural groups and land use categories in England and Wales

Textural class	Statistic	Topsoil macroporosity under different land use (%v/v)		
		Arable	Grassland	Other
Clay	*n*	39	38	73
	range	7.3–13.9	8.6–14.4	7–14.8
	mean	11.1	12.3	11.5
	SE	0.2	0.2	0.2
Clay-loam	*n*	116	119	232
	range	9.6–17.2	11.4–19	9.9–23.2
	mean	12.6	14.3	13.8
	SE	0.1	0.1	0.1
Silty-clay-loam	*n*	40	45	85
	range	7.4–15.5	9.7–23.1	8.1–16.8
	mean	11.1	13.2	12.3
	SE	0.2	0.3	0.2
Silty-clay	*n*	8	12	20
	range	7.9–14.3	9.1–20	8.2–17.2
	mean	10.9	12.9	12.1
	SE	0.4	0.8	0.6
Silt-loam	*n*	9	11	22
	range	10.9–14.6	14.9–22.5	11.8–17.9
	mean	14.6	17.4	14.2
	SE	0.4	0.6	0.3
Sandy-clay-loam	*n*	17	17	31
	range	13.8–21.6	14.9–22.5	13.2–20.4
	mean	15.9	17.4	16.1
	SE	0.5	0.6	0.4
Sandy-loam	*n*	83	102	182
	range	13.5–30.7	14.5–36.5	13.5–36.9
	mean	18.1	21.3	20.5
	SE	0.4	0.4	0.3
Sand	*n*	8	7	17
	range	11.9–38.4	15.7–44.3	15.2–46.5
	mean	25.8	31.1	32
	SE	2.6	3.3	2.2

Macroporosity data for all soils in England and Wales has been calculated from the total porosity and volume of occupied pore space at 5Kpa suction data on the National Soil Resouce Insitute (NSRI) database. This data was collected in the 1970s and 80s. Summary statistics of topsoil macroporosity from the database are shown in Table 2.3.6.3 for major soil textural classes and land use categories. For each of the soil textural classes, land use appears to have little effect on macroporosity. Although differences in means are small between land use classes for each soil texture, the standard errors suggest the differences are statistically significant in most cases. There is considerably more variation between soil textural groupings. All mean macroporosities fall within the sustainability range defined by Sparling *et al*. (2003) and corroborated by Reeve (*Pers. Comm.*). The data show that macroporosity increases in the order clay < loam < sand.

Integration of macroporosity within other national monitoring schemes

Macroporosity is calculated from bulk density, water release curve and particle density. These parameters exist in available soil datasets for England and Wales. Macroporosity could therefore be easily integrated and cross-compared with existing and future datasets.

Cost and suitability for broad-scale monitoring

Macroporosity is derived from parameters that are generally recorded for standard soil assessments. The methodology is simple and robust method but it is not efficient in terms of time and cost of analytical determination. For robust measurements, a soil has to first be sampled very carefully in moist field conditions. In the lab, the cores need full wetting and then putting on a sand suction table (one can also use Bruchner funnels as in the Netherlands) to equilibrate at -50 or -100 kPa. They will need to be left there for two weeks before removing and weighing. This method is very hungry in terms of both laboratory space and time. The only alternative is to try and sample close to field capacity and assume that all non water-filled pores are macropores. This short-cut method has been tried by Soil Survey but it is not recommended as being sufficiently accurate for inclusion in a soil monitoring programme (Reeve, *Pers. Comm.*). It is strongly advised to use the one accepted measure of macroporosity so there is no scope for confusion. Sample collection is the same as for bulk density. Sampling should be carried out in spring. Ideally samples would be taken from 1-10 cm, the middle of the topsoil and the top of the subsoil.

In terms of broad-scale monitoring, macroporosity is less indicative than bulk density (Brejda *et al*., 2000a, 2000b; Drewry *et al*., 2000; Shulka *et al.,* 2004; Sparling and Schipper, 2004). However, a measure of macroporosity complements bulk density and, used together, both parameters provide considerable value to our understanding of soil quality in terms of cost-benefit.

Conclusion and recommendation

It is not recommended that macroporosity is included in the MDS. Macroporosity can be estimated by bulk density, clay content and the use of laboratory-derived statistics (Harrod and Reeve, *Pers. Comms*.). Also, macroporosity determination does not fulfil the criteria of a simple and cost effective method. However, when coupled with bulk density data, a reliable measure of macroporosity is a powerful aid in soil quality assessment and should definitely be considered as part of a Tier 2 assessment. If used in a Tier 2 assessment, macroporosity would prove a useful secondary indicator for other physical soil properties such as aggregate stability, mean weight diameter and hydraulic conductivity as well as changes in soil chemical and biological factors such as SOC and microbial biomass and activity.

References

Amezketa, E. (1999) Soil aggregate stability: A review. *Journal of Sustainable Agriculture*, 14, 83-151.

Arden-Clarke, C. and Hodges, R.D. (1987) The environmental effects of conventional and organic/biological farming systems. I Soil erosion, with special reference to Britain. *Biological Agriculture and Horticulture*, 4, 309-357.

Bissonnaise, Y.E., Cros-Cayot, S. and Gascuel-Odoux, C. (2002) Topographic dependence on aggregate stability, overland flow and sediment transport. *Agronomie*, 22, 489-501.

Brejda, J.J., Moorman, T.B., Douglas, L.K. and Thanh, H.D. (2000a) Identification of regional soil quality factors and indicators: I central and southern High Plains. *Soil Science Society Of America Journal*, 2115-2124.

Brejda, T.B., Douglas, L.K., Smith, J.L and Allan, D.L. (2000b) Identification of regional soil quality factors and indicators: II northern Mississippi loess hills and palouse prairie. *Soil Science Society Of America Journal*, 64, 2125-2135.

Chen, Y., Tessier, S. and Rouffignat, J. (1998) Soil bulk density estimation for tillage systems and soil textures. Transactions Of The ASAE, 41, 1601-1610.

Drewry, J.J. and Paton, R.J. (2000) Effects of cattle treading and natural amelioration on soil physical properties and pasture under dairy farming in Southland, New Zealand. *New Zealand Journal Of Agricultural Research*, 43, 377-386.

Drewry, J.J., Littlejohn, R.P. and Paton, R.J. (2000) A survey of soil physical properties on sheep and dairy farms in southern New Zealand. *New Zealand Journal Of Agricultural Research*, 43, 251-258.

Drewry, J.J., Cameron, K.C. and Buchan, G.D. (2001) Effect of simulated dairy cow treading on soil physical properties and ryegrass pasture yield. *New Zealand Journal Of Agricultural Research*, 44, 189-190.

Hall, D.G.M., Reeve, M.J., Thomasson, A.J. and Wright, V.F. (1977) *Water Retention, Porosity and Density of Field Soils*. A.R.C Soil Survey Technical Monograph 9. Harpenden.

Hernanz, J.L., Peixoto, H., Cerisola, C. and Sanchez-Giron, V. (2000) An empirical model to predict soil bulk density profiles in field conditions using penetration resistance, moisture content and soil depth. *Journal Of Terramechanics*, 37, 167-184.

Logsdon, S.D. and Cambardella, C.A. (2000) Temporal changes in small depth-incremental soil bulk density. *Soil Science Society Of America Journal*, 64, 710-714.

Loveland, P. and Webb, J. (2002) Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research*, 70, 1-18.

Neves, C.S.V.J., Feller, C., Guimaraes, M.F., Medina, C.C., Tavares, J. and Fortier, M. (2003) Soil bulk density and porosity of homogeneous morphological units identified by the cropping profile method in clayey oxisols in Brazil. *Soil and Tillage Research*, 71, 109-119.

Schipper, L.A. and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses. *Soil Science Society Of America Journal*, 64, 300-311.

Scholefield, D and Hall, D.M (1985). A method to measure the susceptibility of pasture soils to poaching by cattle. *Soil Use and Management*. Vol 1, No 4, 134-138.

Scholefield, D., Patto, P.M. and Hall, D.M. (1985) Laboratory research on the compressability of four topsoils from grassland. *Soil and Tillage Research*, 6, 1-16.

Scholefield, D. and Hall, D.M. (1986) A recording penetrometer to measure the strength of soil in relation to the stresses exterted by a walking cow. *Journal of Soil Science*, 37, 165-176

Shukla, M.K., Lal, R. and Ebinger, M. (2003) Tillage effects on physical and hydrological properties of a typic aggriquoll in central Ohio. *Soil Science*, 168, 802-811.

Shukla, M.K., Lal, R. and Ebinger, M. (2004) Soil quality indicators for the north apalachian experimental watersheds in Coshocton, Ohio. *Soil Science*, 169, 195-205.

Singleton, P.L., Boyes, M. and Addison, B. (2000) Effect of treading by dairy cattle on topsoil physical conditions for six contrasting soil types in Waikato and Northland, New Zealand, with implications for monitoring. *New Zealand Journal Of Agricultural Research*, 43, 559-567.

Sollins, P., Homannn, P. and Caldwell, B.A. (1996) Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74, 65-105.

Sparling, G.P., Shepherd, T.G. and Schipper, L.A. (2000) Topsoil characteristics of three contrasting New Zealand soils under four long-term landuses. *New Zealand Journal Of Agricultural Research*, 43, 569-583.

Sparling, G. and Schipper, L. (2002) Soil quality at a national scale in New Zealand. *Journal Of Environmental Quality*, 31, 1848-1857.

Sparling, G., Lilburne, L. and Vojvodic-Vukovic, M. (2003) *Provisional target for soil quality indicators in New Zealand.* Landcare Research, Palmerston North, New Zealand.

Sparling, G. and Schipper, L. (2004) Soil Quality monitoring in New Zealand: trends and issues arising from a broad-scale survey. *Agriculture, Ecosystems and Environment*, 104, 545-552.

2.3.7 Olsen P

Phosphorus (P) is an essential element and is a macronutrient. A measure of soil P as a soil quality indicator in a minimum dataset, particularly for environmental interaction, is necessary. It is recognised that relatively low concentrations, in an agricultural context, of P in surface water (< 20 μg/l) can cause detrimental effects through nutrient enrichment and subsequent eutrophication. Further, recent evidence suggests that significantly more P may leave soils and enter into adjacent environmental compartments than previously thought, via eroded soil material and overland flow, and also through land drains and soil through-flow (Johnes and Hodgkinson, 1998). Finally, excessive soil P is understood to alter plant successional dynamics and community composition and for semi-natural, nutrient-poor habitats this can reduce diversity (Roem *et al*., 2002).

Olsen P or phosphorus concentrations as extracted from soil by a solution of sodium bicarbonate (0.5M $NaHCO_3$) have been or are likely to be used in a range of soil quality indicator programmes (De Clerk *et al*., 2003; Sparling *et al*., 2003). Its possible inclusion here is due to the need for a more responsive measure of P behaviour in soils than total P. As the total amount of P held in soil solution at any time is very small relative to the total P content of the soils, measuring total soil P content gives little indication of the labile amount of P in soils and likely environmental behaviour of P. This is because of the strong sorption of P to soil matrices and the very low solubility of P compounds, leading both to very long P residence times in the soil (> 1000 years) and to soil solution P (the most labile form) concentrations in the order of < 0.002% (Tisdale *et al*., 1993). Therefore, the use of a dilute chemical soil extractant, with its well known drawbacks (Barber, 1995), does give a broad indication of an operationally defined measure of P availability and potential environmental mobility (McDowell *et al*., 2001). There are, however, many methods for measuring P availability (around 26) and the actual forms of P within this available fraction are numerous and temporally and environmentally highly variable. Nevertheless, links have been made between Olsen P concentrations and surface water quality (Heckrath *et al*., 1995).

The key concern for this paper is the use of Olsen P as a soil quality indicator for the function ‘environmental interaction’. More specifically, the aims are to challenge the use of Olsen P as a soil quality indicator. What does change in soil Olsen P mean? What triggers in Olsen P exist? Is it possible to integrate Olsen P with other indicators? How much does Olsen P measurement cost and what is the sensitivity and likely variability of the method used? Again, as with pH, methods for Olsen P may be standardised but its interpretation would not be, depending on major soil group and land use. This is an acknowledgement that different soil conditions are desirable for different land uses; that is, the soil condition is fit for that particular land use (Schipper and Sparling, 2000).

What does change mean?

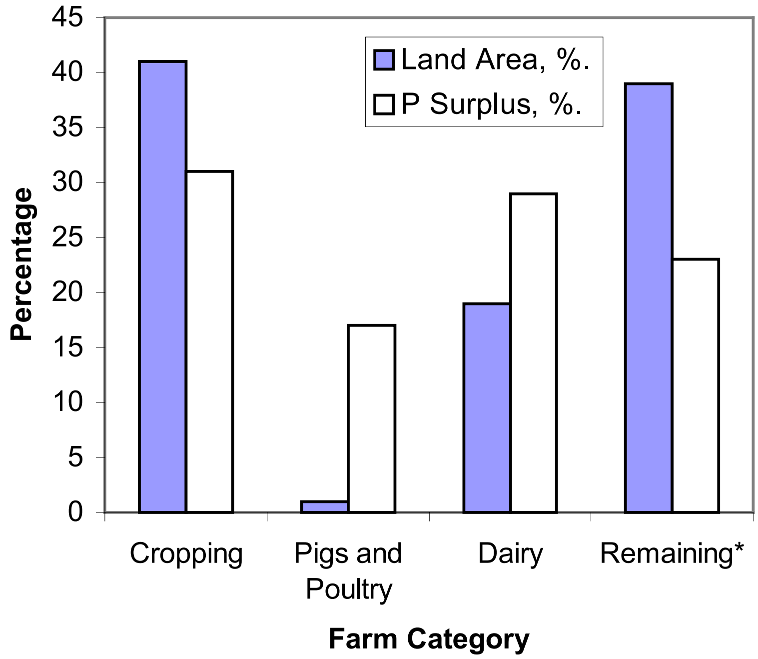
Phosphorus, as an essential macronutrient, is required by all living organisms. It is involved in energy transfer reactions and plays a role in plant photosynthesis, respiration, cell enlargement and division – indeed, almost every metabolic reaction of any significance proceeds via a phosphate derivative. Phosphorus also promotes early root formation and growth, and as plants mature most P moves into seeds and/or fruiting bodies; hence the quality of grain, fruit and vegetable crops is greatly improved with an adequate supply of P (Tisdale *et al*., 1993). Phosphorus, along with water and nitrogen, tends to be the dominant yield-limiting factor for agricultural crop growth.

In agricultural systems, too little available soil P results in stunted crops and reduced yield, and in grazing livestock may cause reduced milk yield, poor live-weight gains, reduced food intake and fertility. Therefore, it is broadly accepted that in intensive agricultural systems there is a significant requirement, for most soil types, to apply appreciable amounts of P in fertilizer or manurial form.

Too much soil phosphorus

The need for appreciable P inputs to soils from mineral and organic fertilizers in UK agriculture over the last 45-50 years has led to a present day soil P surplus compared to the agronomic requirement of use. Repeated, inefficient applications of P to soils or through feed to livestock and the behaviour of P on entering the soil has led to this accumulation. For example, between 1979 and 1985, an increase in topsoil (0 – 15 cm) P of over 200 mg/kg has occurred under both arable and grassland systems, representing an increase of almost 20% (MAFF, 2000).

This shift in the balance between P input and crop offtake in the UK is mirrored across Europe, with Belgium and the Netherlands having the greatest annual surpluses of 40 kg P/ha (*cf.* UK 16 kg P/ha) (Brouwer *et al*., 1995). Cropping systems are regarded as having the greatest P surplus, with 269 kg/ha, while losses to drainage channels depend not only on surplus size but also on land management and farming system (Figure 2.3.7.1) (Edwards and Withers, 1998). One of the results of this imbalance between inputs and outputs is a build up in soil P to levels that are of environmental rather than agronomic concern (Daniel *et al*., 1998).



* Remaining includes mixed farms and horticulture.

Figure 2.3.7.1: The contribution of farming types to the UK annual P surplus, as a percentage of the total agricultural land (Edwards and Withers, 1998)

The transfer of P from agricultural systems to other environmental compartments is complex, but is dependent upon three factors: the source of P; the mechanism of release from soil to water; and the hydrological pathways by which mobilised P moves from the land. The source of P, for example manure or inorganic fertiliser, determines the P form within the soil, that is, whether it is in an organic or inorganic form. The predominant mechanisms of release from soil to water are solubilisation (release to solution), detachment (primarily the erosion of soil particles) and incidental transfers (loss of P before incorporation into the soil). Detachment is the most important mechanism of release, accounting for approximately 60% of total P loss, whereas solubilisation and incidental losses account for 20% each (Defra, 2002).

The magnitude, form and extent of P transport along the pathways shown in Figure 2.3.7.2 will vary considerably with land use, catchment topography, soil physical and chemical properties (including P content), rainfall duration and intensity (and preceding hydrological conditions) and proximity to stream corridors (Johnes and Hodgkinson, 1998). Losses of P are always greater for cultivated soils than uncultivated and semi-natural habitats, and losses via subsurface pathways greatest on poorly drained, high organic matter and heavily manured soils. Leaching and subsurface loss of P from agricultural systems would suggest that P is being transferred primarily in the dissolved form. This is not always the case; finer soil fractions may also be transferred, with P sorbed to soil colloids, along subsurface pathways and overland flow. The proportion of P transferred in particulate form can range from 17-60% of the P lost via drainage water (Heckrath *et al.*, 1995; Hooda *et al.*, 1996).

It is estimated that 43% of the phosphorus entering UK waterways is from agriculture, with the rest coming from point source discharges, such as sewage treatment works (Morse *et al*., 1993). For Olsen P to be a useful soil quality indicator in a minimum dataset and have relevance to the soil function of environmental interaction, there must be an explicit link between measured values in the soil and likely impacts on broader environmental quality (*cf*. McDowell, 2001). However, it is important to stress that soil P surplus or Olsen P values are not the only factors that determine P loss from soils, although links between Olsen P content of soils and deleterious effects upon water quality have been made with varying degrees of success by a range of workers (Pote *et al*., 1996; Djodjic *et al*., 2004).

The 'change point' is the soil P concentration at which the solubility of P to soil solution markedly increases. In a study examining the existence and behaviour of a change point in soil P release from soils under a range of management systems from the UK, New Zealand and the USA, McDowell *et al*. (2001) examined the relationship between quantity and intensity of P supply. The plots of intensity, as measured by calcium chloride extraction, against the quantity, as measured by Olsen P, showed change points ranging from 20 – 112 mg Olsen P kg^{-1}. The change points are values above which release of P into soil solution occurs at a greater rate per unit increase in soil P concentration. Hooda *et al*. (2001) demonstrated the link between Olsen P and potential environmental implications of P in soils under intensive fertilizer and manuring practice. Saturation of the soil with P to 25% of its estimated total capacity was suggested to result in excessive P concentrations in run-off. In order to reduce this risk, an Olsen P trigger, again matching agronomic optimum values (25 < 45 mg Olsen P kg^{-1}) was suggested, in particular on soil receiving inorganic P addition.

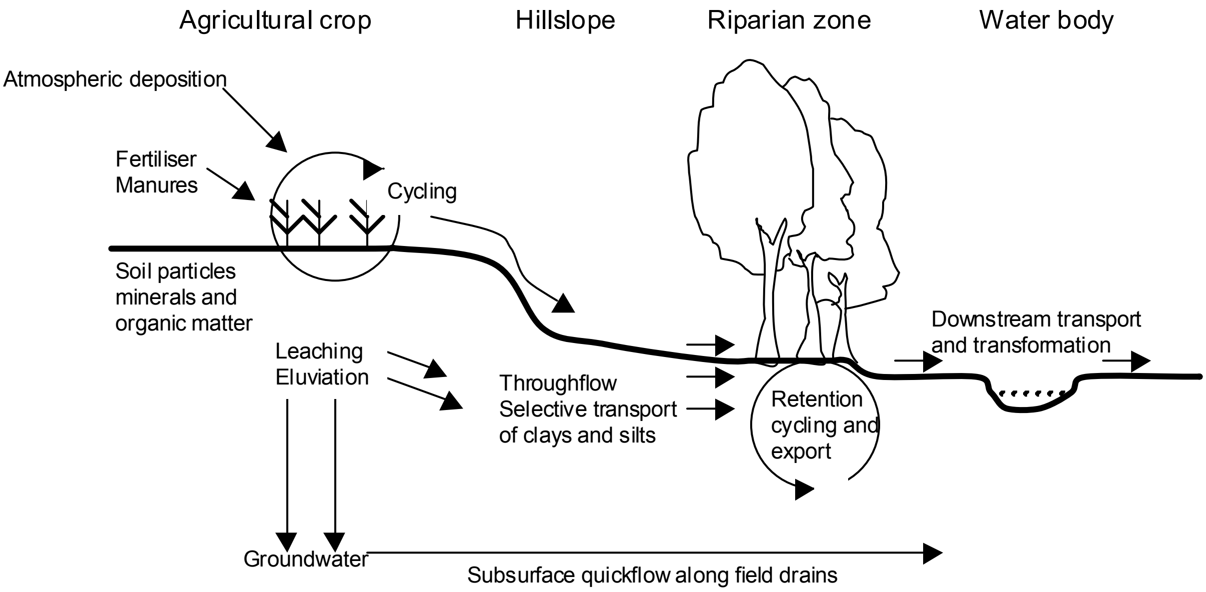


Figure 2.3.7.2: P inputs from non-point sources and main hydrological flow pathways (Johnes and Hodgkinson, 1998)

It is important to note that the change point theory only relates to the process of solubilisation (soluble P release from soils to run-off); as mentioned previously, the predominant mobilisation process for P is detachment. Currently, there is no simple routine test to predict particulate P loss, however, the DESPRAL project (www.despral.org.uk) developed a water dispersion test to predict soil vulnerability to particle detachment and initial P mobilisation in fields. Further work is still required to examine changes in soil P dispersibility across different scales and to combine the test with a hydrological component for use in risk assessment.

At a larger scale, for example the catchment, a greater degree of complexity is introduced through consideration of hydrological, geographical and land use factors, making interpretation of Olsen P data for soils significantly more onerous with regard to water quality (*cf*. Edwards and Withers, 1998). Yet it should be the role of an indicator in the minimum dataset to highlight likely environmental risk, and it is clear that Olsen P values have been used to establish the potential for adverse effects on the broader environment (Jordan *et al*., 2000). Change in soil Olsen P values indicate an increase in the available P content of the soil and likely risk of nutrient enrichment to aquatic systems. For example, He *et al*. (2003) observed an increase in Olsen P in a range of soils under intensive irrigated agriculture in response to fertilizer application and a positive correlation with loss of P via overland flow. Further, organic carbon was noted as being the soil factor that explained 50% of the variability in total P load in surface run-off, followed in importance by Olsen P and then fertilizer rate.

Elevated soil P concentrations, as measured through Olsen P, may also have a detrimental effect upon ecosystems adapted to low P conditions. Such effects may include reductions in species richness (Reom *et al*, 2002), symbiotic plant associations (Smith and Read, 1997) and organism diversity. The enrichment of soil P levels, as reflected in large or excessive values of Olsen P compared to controls, can occur at sensitive sites adjacent to farmland having received unintended fertilizer inputs (Stevenson, 2004), road sides (Brewer and Cralle, 2003) and reclaimed farm or horticultural land (Gough and Marrs, 1990).

Gough and Marrs (1990) noted that the re-establishment of semi-natural vegetation on previous agricultural land was limited by the excessive soil P concentrations, expressed as Olsen P. Stevenson (2004) suggested that increased values of Olsen P (<10 times compared to soils at control sites) reduced the long term stability of indigenous forest fragments located adjacent to permanent pasture. A reason for this was thought to be the limiting nature of P availability on microbial activity in the indigenous forests control sites, but not in the fragments of forest near pasture.

Phosphorus toxicity, while not common, may occur at excessive P concentrations appearing as interveinal chlorosis and necrosis in younger leaves and the shedding of older leaves (Reuter and Robinson, 1997). Long-term immobilisation (and hence system inefficiency) is likely to be more of a concern in the majority of cases when thinking about upper thresholds.

Selection of triggers for Olsen P

Total P concentrations in topsoils are generally in the range of 100-3000 mg/kg. Yet, as has already been outlined, only a very small fraction of this total is available for plant uptake or movement through soils in solution. The total concentration of P in surface waters, which is considered critical in terms of biological production and potential eutrophication, is about 0.01- 0.1 mg/l. This is some 10 to 100 times lower than that required for nitrogen and 10 times lower than the concentration commonly thought to be required for crop growth. Therefore, what may be considered as a trivial solution P concentration in an agronomic context may be crucial with regard to water quality.

Agronomic requirements for P, developed through calibration against plant tissue analysis using Olsen P extraction, are widely available and indicate optimum P levels of fertilizer application to maintain P levels in soils at certain grain yields (MAFF, 2000). There is also strong positive correlation between inorganic and organic P fertilizer applications to agricultural soils and values of Olsen P (Rowell, 1994). Differences in values of Olsen P between conventional farm managed systems and biodynamic farming systems on the same soils can be up to 20 mg/kg and should be relatively easy to discern (66 against 46 mg P/kg, respectively) (Reganold *et al*., 1993).

A linear relationship between Olsen P concentrations in soils of a plough layer and the concentrations of dissolved reactive P in underlying tile drains were observed by Heckrath *et al*. (1995) at Olsen P values above 60 mg/kg. Below this concentration – termed the change point – concentrations in the tile drains were below 0.1 mg/l. From their work on change points, McDowell *et al*. (2001) suggested that most critical Olsen P concentrations, with regard to adverse effects upon water quality, were 40% of the optimum values for plant growth on a range of soil types and land uses. Therefore, remaining within the agricultural optimum should be protective and ultimately more economical than over-fertilizing. The source of fertilizer P, either organic or inorganic, has also been observed to have only a limited effect upon P loss from plots subject to simulated rainfall events, where change points were calculated for a range of soil types and were again close to the agronomic optimum for those soils (McDowell *et al*., 2003).

In an attempt to establish the sustainable soil Olsen P level that would minimise environmental effects on water quality but sustain optima crop growth, Jordan *et al*. (2002) estimated the change point to be 22 mg Olsen P/l (a P index of two, MAFF, 2000). This was calculated from estimating catchment loads of P for 56 rivers across Northern Ireland and measured values of Olsen P from 5615 soil samples from corresponding sites in the catchments. Importantly, a P index of three (around 26-45 mg P/l) was noted as being excessive environmentally and for crop production. From the Represetative Soil Sampling Scheme (RSSS) data, which solely considers agricultural land, 41 per cent (n = 396) of the sites sampled in 1969 showed Olsen P values at or above index three and in 2002, 43 per cent (n = 219) were in index three. Smith *et al*. (1998) suggested that in order to reduce leaching losses of P it was necessary to restrict topsoil Olsen P levels to below 70 mg/l. In practice it will not be possible to set absolute values that apply across the whole of England, hence the usefulness of ranges that are soil type and land use specific, as suggested below in Table 2.3.7.1.

Table 2.3.7.1: Target limits for Olsen P (μg/ml) for five broad land uses and three major soil types in New Zealand for both agricultural production and environmental interaction (Sparling *et al*., 2003)

Pasture on sedimentary and allophanic soils	0	15	20	50	100	200
Pasture on pumice and organic soils	0	15	35	60	100	200
Cropping and horticulture on sedimentary and allophanic soils	0	20	50	100	100	200
Cropping and horticulture on pumice and organic soils	0	25	60	100	100	200
Forestry on all soils	0	5	10	100	100	200
	Very Low	**Low**		**Adequate**	**Ample**	**High**

Figures in shade represent upper and lower limits.

The relevance of the soils in Table 2.3.7.1 and the assigned values is limited in terms of a UK context, but what this table does demonstrate is the ability to produce such a table as a screen for use of Olsen P in a minimum dataset. It could be anticipated that in the UK, if such a table does not already exist, it may be relatively straightforward to derive one.

The development of trigger values for Olsen P for use in low P habitats are not so readily available. Nevertheless, Walker *et al*. (2001) noted, in a study looking at semi-natural lowland grasslands, that the mean value of Olsen P at these sites was 2.2 mg/kg. It is thought that values of Olsen P greater than 5 mg/kg could indicate species-poor, nutrient-enriched pastures (Goodwin *et al*., 1998).

Sparling *et al.* (2003a) suggested that there were three possible methodologies by which trigger values for soil indicators could be derived. The first of these methods was to use the national soil baseline data – calculating median and lower quartile values for a parameter for each soil order. This approach can be followed with RSSS datasets from 1969 and 2002 to give an indication of the range and variability of Olsen P data across all land types sampled (Table 2.3.7.2). However, the justification for the use of the lower quartile is not entirely clear.

Table 2.3.7.2: Olsen P data from the RSSS data base from 1969 and 2002

	Olsen P, mg/l 1969 (n = 397)	Olsen P, mg/l 2002 (n = 218)
Mean	28	27
Standard deviation	20	16
95th Percentile	69	60
50th Percentile	22	23
25th Percentile	15	16
5th Percentile	7	10

Harrod and Fraser (1999) have also undertaken some work in the UK from National Soil Inventory (NSI) datasets on the comparison of total and Olsen P values in topsoils under a range of land uses, including semi-natural (Tables 2.3.7.3 and 4). The Olsen P median values from Tables 2.3.7.3 and 4 indicate contents by land use classes as follows: Horticulture > orchard > arable > ley> permanent grass > forestry and semi-natural.

Table 2.7.3.3: Total and [Olsen] P concentrations (mg/kg) in NSI topsoils under various landuses in England and Wales (Harrod and Fraser, 1999)

	Arable	Horti-culture	Ley	Perman't grass	Orchard	Conifers	Decid-uous
Mean	830 [32]	1023 [56]	922 [27]	939 [24]	882 [50]	527 [14]	608 [19]
Median	734 [26]	854 [45]	822 [20]	847 [17]	851 [44]	448 [11]	528 [12]
Standard error	9.5 [0.5]	81.8 [6.3]	17.3 [1.0]	11.0 [0.6]	66.6 [5.8]	24.4 [0.9]	23.0 [1.5]
Range	246-4189 [0-205]	280-2312 [4-160]	170-4535 [1-274]	188-4529 [0-337]	177-1925 [1-123]	41-1685 [0-94]	108-2636 [1-210]
Number	1888 [1868]	41 [39]	675 [668]	1560 [1538]	35 [34]	207 [203]	273 [267]

Table 2.3.7.4: Total and [Olsen] P concentrations (mg/kg) in NSI agricultural and semi-natural topsoils in England and Wales (Harrod and Fraser, 1999)

	Arable, ley and permanent grass	Upland heath and grass	Deciduous and coniferous woodland
Mean	887 [28]	762 [16]	572 [17]
Median	794 [21]	696 [12]	500 [11]
Standard error	6.7 [0.4]	20.3 [0.8]	16.9 [0.9]
Range	170-4535 [1-337]	142-2214 [1-157]	41-2634 [1-210]
Number	4111 [4111]	344 [344]	479 [479]

The final two methodologies suggested by Sparling *et al*. (2003a), by which trigger values for soil indicators could be derived, were modelling to set a limit values and using expert judgement to derive response curves for changes in the Olsen P level (on the x-axis) against soil quality (0-100%).

The link between soil quality indicator and the soil function of 'environmental interaction' is imperative. By consideration of the functions that are required for the soil to perform and the connectivity between the indicator and those functions, it is possible to produce a table based on broad soil types. An example is given in Table 2.3.7.5 with potential trigger values derived from a range of sources (MAFF, 2000; Smith *et al*., 1998). The habitats selected are those from the Biodiversity Action Plan (UK BSG 1995) that are likely to be affected by excessive values of Olsen P.

Table 2.3.7.5: Potential trigger values for Olsen P (mg/l) for 'environmental interaction'

Function	Soil type		
	Mineral	Peaty	Calcareous
Metal retention	[redacted]	[redacted]	[redacted]
Microbial function/ biofiltering	[redacted]	[redacted]	[redacted]
Gaseous emissions	[redacted]	[redacted]	[redacted]
Soluble phosphorus leaching	>60	>60	>60
Hydrological?	[redacted]	[redacted]	[redacted]
Habitat support table			
Calcareous grassland	[redacted]	[redacted]	>16
Mesotrophic grassland	>10	>10	[redacted]
Acid grassland	>10	>10	[redacted]
Dwarf Shrub Heath	>10	>10	[redacted]
Biomass production			
Arable and horticultural	16–45	16–45	16–45
Improved grassland	16–25	16–25	16–25

Integration with other national monitoring schemes

The NSI data and RSSS contain data collected on Olsen P. Importantly, the reported data in RSSS is based on w/v criteria (mg/l) and not w/w (mg/kg). To ensure the greatest continuity with these relatively large datasets it is recommended that measurements of Olsen P are undertaken on w/v and w/w.

Cost and suitability for broad-scale monitoring

The Olsen P methodology is thought to be most suitable for predicting fertility and crop recommendations in soils of pH > 7, while acid extracts (sometimes including fluoride to complex Al^{3+} and Fe^{3+}) frequently give better correlation with acid soils.

Costs of determination for Olsen P can be as much as £15 (National Laboratory Service, Environment Agency, *Pers. Comm.*), but is routinely less than £5 a sample.

Olsen P is commonly used in some, but not all, EU countries for the determination of P status in soils and fertilizer recommendations. Importantly, there is significant variability in methodologies and results obtained, with commensurate differences in interpretation (Sibbesen and Sharpley, 1997; Neyroud and Lischer, 2003).

Relative to soil pH determination, Olsen P is thought to be a more volatile or variable soil quality indicator. In a study across 29 sites on a range of land uses showed a coefficient of variation for Olsen P of 15.6% (*cf.* with the coefficient of variation for pH of 2.3%) (Schipper and Sparling, 2000). In a study of spatial variability of soil quality indicators in New Zealand across four land uses and one major soil order, Gilltrap and Hewitt (2004) noted that variability in Olsen P was related to land use, with native vegetation showing the greatest variability (Table 2.3.7.6).

Table 2.3.7.6: The coefficients of variation in Olsen P from soils under various land uses in New Zealand (Gilltrap and Hewitt, 2004)

	Cropping (cv%)	Native (cv %)	Pasture (cv %)	Pine (cv %)	All land uses (cv %)
Olsen P	14.1	29.7	16.4	21.4	17.1

Conclusion and recommendation

There is need to balance the P inputs to soils for optimum crop production and the P transferred from the leaky agroecosystem resulting in adverse environmental effects. Trigger values set for Olsen P will reflect the need to balance these two issues. Significant information is available for the former, and increasingly for the latter. Setting trigger values for Olsen P in soils in low P habitats may be potentially difficult, as less information is available to establish what may be critical levels.

Olsen P has been shown to be an indicator that responds directly to increased input of P to soils; further links have been made, and continue to be made, between soil Olsen P values and concentrations of P in surface waters. However, results are mixed and interpretation is not as straightforward as some have suggested (Sibbesen and Sharpley, 1997). Nevertheless, by remaining within the optimum values for crop growth, environmental impacts on water quality are limited. The estimation of change points from quantity and intensity relationships seems to be an effective way of deriving triggers.

For low P ecosystems, while limited data does exist as to possible effects of increased P, it is clear that it can have an adverse effect. The triggers of Olsen P at which that effect may occur will probably be defined by the organisms (and especially the plants) within that system (Stevenson, 2004). For very low P systems, it is possible that Olsen P may not provide a sensitive measure of changes in P status. This is a potential area for greater research.

The use of Olsen P as a measure of soluble P, the most readily available P form, seems to be an effective soil quality indicator. Although the particulate P fraction is more important, in terms of P losses in run-off, 10-90% can become available (Bostrom, 1988) and therefore this becomes limiting in terms of interpretation and understanding.

Soil Olsen P is linked with biomass production, water quality and biodiversity. Its inclusion in a minimum dataset as an indicator of soil quality for the function of ‘environmental interaction’ is recommended.

References

Barber, S.A. (1995) *Soil nutrient bioavailability; a mechanistic approach*. (2nd Edition) Wiley, New York.

Bostrom, B., Persson, G. and Broberg, B. (1988) Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiologia*, 170, 133-155.

Brewer, J.S. and Cralle, S.P. (2003) Phosphorus addition reduces invasion of longleaf pine savanna (Southeastern USA) by a non-indigenous grass (Imperata cylindrica). *Plant Ecology*, 167, 237-245.

Brouwer, F.M., Godeschalk, F.E., Hellegers, P.J.G.J. and Kelholt, H.J. (1995) *Mineral balances at farm level in the European Union.* Agricultural Economics Research Institute (LEI-DLO), The Hague.

Daniel, T.C., Sharpley, A.N. and Lemunyon, J.L. (1998) Agriculture phosphorus and eutrophication: A symposium overview. *J. Environ. Qual*., 27, 251-257.

De Clerk, F., Singer, M.J. and Lindert, P. (2003) A 60 year history of California soil quality using paired samples. *Geoderma*, 114, 215-230.

Defra (2002) *The effectiveness of changing farmer practice on the reduction of diffuse pollution of nitrogen and phosphorus*. http://www.defra.gov.uk/environment/water/quality/diffuse/agri/reports/pdf/dwpa06.pdf

Edwards, A.C. and Withers, P.J.A. (1998) Soil phosphorus management and water quality: a UK perspective. *Soil Use and Man.*, 14, 124-130.

Djodjic, F., Borling, K. and Bergstrom, L. (2004) Phosphorus leaching in relation to soil type and soil phosphorus content. *Journal of Environmental Quality*, 33, 678-684.

Giltrap, D.J. and Hewitt, A.E. (2004) Spatial variability of soil quality indicators in New Zealand soils and land uses. *New Zealand Journal of Agricultural Research*, 47, 167-177.

Goodwin, M.J., Parkinson, R.J., Williams, E.N.D. and Tallowin, J.R.B. (1988) Soil phosphorus extractability and uptake in a Cirsio-molinietum fen meadow and an adjacent Holcus lanatus pasture on the Culm Measure, North Devon, UK. *Agriculture, Ecosystems and Environment*, 70, 169-179.

Gough, M.W. and Marrs, R.H. (1990) A comparison of soil fertility between semi-natural and agricultural plant communities: Implications for the creation of species rich grassland on abandoned agricultural land. *Biological Conservation*, 51, 83-96.

Harrod, T, R. and Fraser, A.I. (1999) *Phosphorus loss from agriculture*. Unpublished SSLRC report to MAFF, Project NT1003.

He, Z.L., Zhang, M.K., Calvert, D.V., Stoffella, P.J., and Li, Y.C. (2003) Loading of phosphorus in surface runoff in relation to management practices and soil properties. *Soil Crop Sci. Florida Proc*, 62, 12-20.

Heckrath, G., Brookes, P.C., Poulton, P.R. and Goulding, K.W.T. (1995) Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment.J. Environ. Qual., 24, 904-910.

Hooda, P.S., Moynagh, M. and Svobada, I.F. (1996) A comparison of phosphate losses in drainage water from two different grassland systems. *Soil Use and Man*., 12, 224.

Hooda, P.S., Truesdale, V.W., Edwards, A.C., Withers, P.J.A., Aitken, M.N., Miller, A., and Rendell, A.R. (2001) Manuring and fertilization effects on phosphorus accumulation in soils and potential environmental implications. *Advances in Environmental Research*, 5. 13-21.

Johnes, P.J. and Hodgkinson, R.A. (1998) Phosphorus loss from agricultural catchments: pathways and implications for management. *Soil Use and Man.*, 14, 175-185.
Jordan, C., McGuckin, S.O. and Smith, R.V. (2000) Increased predicted losses of phosphorus to surface waters from soils with high Olsen-P concentrations. Soil Use and Man., 16, 27-35.
Neyroud, J.A. and Lischer, P. (2003) Do different methods used to estimate soil phosphorus availability across Europe give comparable results. *J. Plant Nutr. Soil Sci.*, 166, 422-431.
McDowell, R., Sharpley, A., Brookes, P. and Poulton, P. (2001) Relationship between soil test phosphorus and phosphorus release to solution. *Soil Science*, 66, 137-149.
McDowell, R.W., Monaghan, R.M. and Morton, J. (2003) Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland*. New Zealand Journal of Agricultural Research*, 46, 239-253.
MAFF (2000) *Fertiliser Recommendations for Agricultural and Horticultural Crops* (RB 209). 7th Edition. HMSO, London.
Morse, G.K., Lester, J.N. and Perry, R. (1993) The economic and environmental impact of phosphorus removal from wastewater in the European Community. Selper Publications, London.
Pote, D.H., Daniel, T.C., Sharpley, A.N., Moore Jr. P.A., Edwards, D.R. and Nichols, D.J. (1996) Relating extrcatable soil phosphorus to phosphous losses in runoff. Soil Sci. Soc. Am. J., 60, 855-859.
Reganold, J.P., Palmer, A.S., Lockhardt, J.C. and Macgregor, A.N. (1993) Soil quality and financial performance of biodynamic and conventional farms in New Zealand. *Science*, 260, 344-349.
Reuter, D.J. and Robinson, J.B. (1997) *Plant analysis, an interpretation manual*. 2nd Edition. CSIRO Publishing, Collingwood, Australia.
Roem, W.J., Klees, H. and Berendse, F. (2002) Effects of nutrient addition and acidification on plant species diversity and seed germination in heathland. *J. App. Ecol*, 39, 937-948.
Rowell, D.L. (1994) *Soil Science, methods and applications*. Longman Scientific and Technical. Harlow.
Schipper, L.A. and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses. Soil Sci. Soc. Am. J., 64, 300-311.
Sibbesen, E. and Sharpley, A.N. (1997) Setting and justifying upper critical limits for phosphorus in soils. In: *Phosphorus loss from soil to water*. Tunney, H., Carton, O.T., Brookes, P.C. and Johnston, A.E. (Eds). CAB International, Wallingford. pp 151-176.
Smith, K.A., Chalmers, A.G., Chambers, B.J. and Christie, P. (1998) Organic manure phosphorus accumulation, mobility and management. *Soil Use and Management*, 14, 154-159.
Smith, S.E. and Read, D.J. (1997) *Mycorrhizal Symbiosis*, Edition 2. Academic Press, London.
Sparling, G., Lilburne, L. and Vojvodic-Vukovic, M. (2003). *Provisional Targets for Soil Quality Indicators in New Zealand*. Landcare Research New Zealand Ltd, Palmerston North.
Sparling, G., Parfitt, R.L., Hewitt, A.E., and Schipper, L.A. (2003a) Three approaches to define desired soil organic matter contents. *Journal of Environmental Quality*, 32, 760-766.

Stevenson, B.A. (2004) Changes in phosphorus availability and nutrient status of indigenous forest fragments in pastoral New Zealand Hill country. *Plant and Soil*, 262, 317-325.

Tisdale, S.L, Nelson, W.L., Beaton, J.D. and Halvin, J.L (1993) *Soil fertility and fertilizers*. 5th Edition. Prentice Hall, New Jersey.

Walker et al. (2001) *Methodologies for restoring and re-creating semi-natural lowland grasslands: a review and quantitative model*. Countryside Council for Wales Contract No. FC 73-01-273A.

2.3.8 Soil quality indicators: metals and organic micropollutants

Soil needs to be protected as a limited resource for production (food and agricultural products), as a habitat for plants and animals, and for the numerous other functions it performs, such as neutralising contaminants and acting as a base for parks and buildings, sinks for carbon dioxide and reservoirs for water (MAFF, 2000). The aim of this report is to present changes in concentrations of metals and organic contaminants as potential indicators of soil quality. The presence and fluctuation of metals and contaminants in soil may indicate stress, particularly for soil microbial communities; however, this may be difficult to relate to the soil functions of environmental interaction.

Metals and organic micropollutants (OMPs) may enter the soil through natural processes, such as weathering of parent materials for metals or burning of vegetation for polycyclic aromatic hydrocarbons (PAHs), and through human activities, such as atmospheric deposition from industrial and vehicle emission or direct application of waste (sludge/slurry, landfills), pesticide and fertiliser (Thornton, 1995; MAFF, 2000). Levels of metals and organic contaminants must therefore be viewed in relation to soil types, land uses and other external influences.

In the MAFF report, indicators for sustainable agriculture included accumulation of heavy metals in agricultural topsoils. However, this report also stated that developing reliable indicators for pesticides is difficult given the high number of authorised products (>300; MAFF, 2000). The same may be said for organic contaminants more generally. Although checking for pesticide levels may be useful to indicate the effectiveness of current usage policies (minimal usage to control pests without environmental impact), it was agreed that this is not part of the SQI remit. Representative sampling would also be difficult, where overland flow and soil erosion might result in hotspots of residues on footslopes, for example (Harrod, 1994). Moreover, whilst there are numerous datasets for metals in UK soils, it is difficult to hindcast and model background concentrations of organic micropollutants due to the limited number of temporally extensive studies (*cf*. Alcock *et al*., 1993).

In summary, can changes in concentrations of metals (and some metalloids) and organic micropollutants be used to assess soil quality? Interpretation of the indicators should consider soil type, land uses and data (measured or modelled) of other inputs/outputs.

Questions to be addressed:

- What does change mean?
- What triggers could be used to initiate further investigation?
- Can the indicators be integrated with national monitoring schemes?
- Are the indicators suitable for broad-scale monitoring in cost, ease of analysis and suitability?
- Can the indicators be recommended as SQIs?

METALS

What does change mean?

Data from SSLRC (Soil Survey and Land Research Centre, data 15 years apart) showed that national trends for total metal concentrations in agricultural soils were difficult to identify due to the heterogeneous nature of soil, underlying geology and spatially variable metal sources. In general, total topsoil zinc had decreased, possibly following deeper ploughing; copper had increased in some light soils, but decreased elsewhere; cobalt, cadmium and nickel remained mostly stable, while chromium increased in organic soils. Targets for indicators of sustainable agriculture could not be established on the basis of this data, due to the imperfect state of knowledge of soils and soil processes and therefore of metal availability to plants and animals (MAFF, 2000).

Metals and metalloids such as copper, zinc and molybdenum are essential trace elements, and deficiencies may lead to reduction in crop yield for agricultural systems, or deleterious effects on development and reproduction in animals. However, high concentrations of copper and zinc can damage soil fertility, whilst accumulation of cadmium and lead in the food chain can affect human and animal health (MAFF, 2000). Moreover metals such as lead, cadmium and mercury have no known biological functions. It is recognised that metals can have deleterious effects, hence safe limits (based on total concentrations) are set for various regulated activities, such as sludge and compost application, in order to prevent accumulation of metals in soil. More generally, critical limits and a critical load methodology are being developed for toxic metals within the United Nations Economic Commission for Europe (UN/ECE) Convention on Trans-boundary Air Pollution (CLRTAP). The critical limit defines an acceptable maximum concentration of metal below which no significant harmful effects to ecosystems should occur. Recent research has focused on the potential for using available metal concentration as the basis for calculating critical loads, rather than total concentrations (Ashmore *et al*., 2000). Changes in soil chemistry will affect the availability and subsequently bioavailability and thus toxicity of metals in soil, as illustrated in Figures 2.3.8.1 and 2.3.8.2.

Changes in the land use/location/soil type will likely mean changes in total concentrations of metals, due to fluctuations in inputs/outputs (atmospheric deposition, direct application, soil processes, land use), or changes in the available fraction of the metals resulting from chemical, physical and microbiological processes and changes in environmental conditions. In order to be meaningful SQIs, concentrations of metals (and OMPs) must therefore be understood from an effects perspective, to allow interpretation of changes in metal concentration. For example, what concentrations may have deleterious effects on the soil? What factors control the concentrations and behaviour/form of the metals over time?

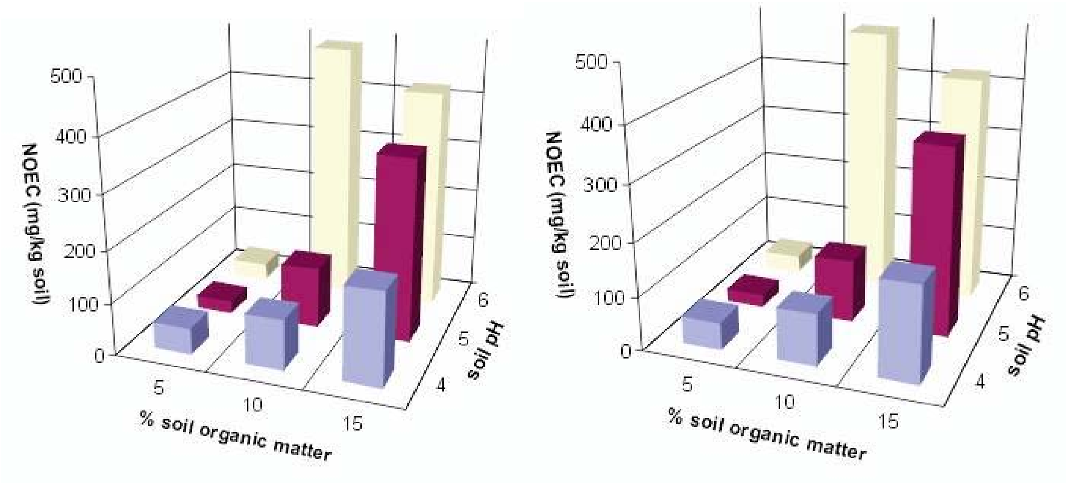


Figure 2.3.8.1: The variation in the NOEC (no observed effect concentration) of zinc for effects on the reproduction of the earthworm *Eisenia fetida* in soils of varying pH and organic matter content (Lofts *et al*., 2004)

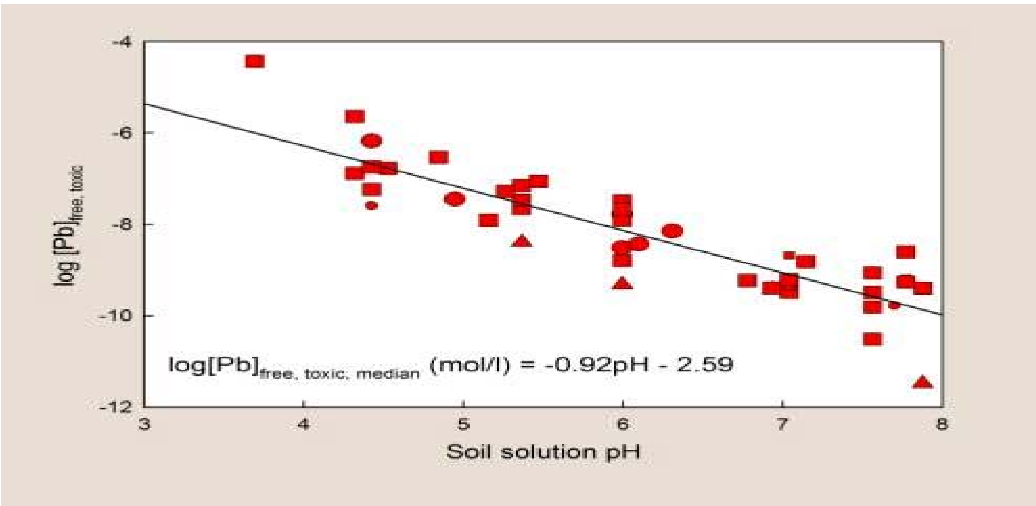


Figure 2.3.8.2: Plot of $\log[Pb]_{free,\ toxic,\ median}$ against soil solution pH for chronic effects on plants (triangles), invertebrates (circles) and microbial processes (squares). The median regression line is shown (Lofts *et al*., 2004)

The significant effect of soil physico-chemical properties upon metal availability and potentially on ecotoxicology makes the use of total soil metal concentrations a very blunt instrument with which to assess soil quality with regard to the function of environmental interaction. The drawbacks of using total metal concentrations in a regulatory context have been widely acknowledged across the scientific and regulatory community (Drexler *et al*., 2003; Impellitteri *et al*., 2003; The Netherlands Risk Assessment Report: Zinc metal, 2004; Basta *et al*., 2005; Cu RAR, 2005; Meers *et al*., 2005). These drawbacks include interpretation of total metals data in the context of environmental behaviour and toxicological effect, lack of ecological relevance and consideration of the huge natural variation in metal background concentrations in soils.

Triggers

A number of trigger values for total soil metals exist nationally and internationally, including soil guideline values (SGV), maximum permissible soil concentrations following sewage sludge application (Table 2.3.8.1) and Dutch target and intervention values (Table 2.3.8.2). There are, of course, many more potential trigger values for metals in soils; however, few are derived with the protection goal of soil function for environmental interaction. For example, SGVs are for human health, the sewage sludge values have the key protection targets of soil fertility and crop production and the Dutch target and intervention values are for contaminated land and often used in site clean-up values. Therefore, the relevance of the values for use in a SQI context, from the perspective of environmental risk is not clear.

The use of total metal soil limit values (as in Tables 2.3.8.1 and 2), taking into account pH and organic matter/clay contents of the soils, have generic application and at least acknowledge and in part take account of soil factors influencing metal speciation. Nevertheless, the relevance and interpretation of total metal concentrations to the soil function of environmental interaction and broader environmental risk is still open to question (Meers *et al*., 2005).

Table 2.3.8.1: Maximum concentrations (total) of metals in UK soils following sewage sludge applications (MAFF, 1993)

Metals	Max permissible concentrations in soil (mg/kg) in UK
Arsenic	50
Cadmium	3
Chromium	400
Copper	80 (pH 5-5.5); 100 (pH 5.5-6); 135 (pH 6-7); 200 (pH > 7)
Lead	300
Mercury	1
Molybdenum	4
Nickel	50 (pH 5-5.5); 60 (pH 5.5-6); 75 (pH 6-7); 110 (pH > 7)
Selenium	3
Zinc	200 (pH 5-5.5); 250 (pH 5.5-6); 300 (pH 6-7); 450 (pH > 7)

Table 2.3.8.2: Dutch target and intervention values for soil remediation (Sanaterre Environmental, 2004)

Metals	Target value (mg/kg) [1]	Intervention value (mg/kg) [2]
Antimony [3]	3	15
Arsenic	29	55
Barium	160	625
Cadmium	0.8	12
Chromium	100	380
Cobalt	9	240
Copper	36	190
Lead	85	530

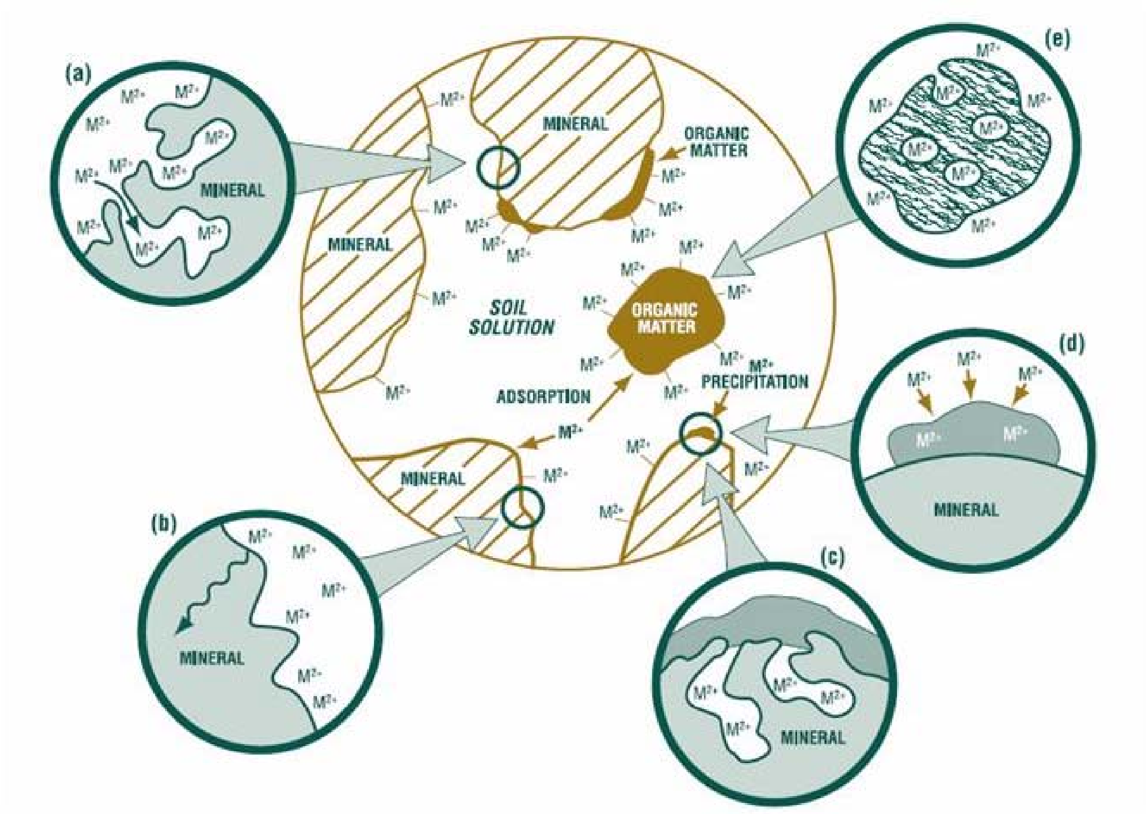
Mercury	0.3	10
Molybdenum	3	200
Nickel	35	210
Selenium	0.7	100
Zinc	140	720

1. Baseline concentration value below which elements are known or assumed not to affect the natural properties of the soil.
2. Maximum tolerable concentration above which remediation is required. This occurs if one or more elements in concentrations equal to or higher than the intervention value is found in more than 25 m^3 of soil.
3. Values for metals and arsenic, with the exception of antimony, molybdenum and selenium, depend on the clay content and/or organic matter content. Standard values are converted to values applying the actual soil concerned, on the basis of measured organic matter and clay content. Values can then be converted into measured concentrations in the soil.

At any point in time, it is the available fraction of the total metal concentration that may interact with and affect the soil biota - that is, the amount of metal in the soil that is in equilibrium with the metal in soil solution. However, this fraction is not easily measured and interpreted. The meaning and measurements of bioavailability are still debated (Drexler *et al*., 2003). Its determination is often operationally defined, that is, it is dependent on the extraction method or organism used, with no consensus on how the data obtained can be interpreted or compared (Alloway, 1995; EC, 2004). Some workers have found excellent correlation between metals extracted from soils with dilute solutions of salts (such as ammonium nitrate) or complexing agents (such as DTPA) and biological uptake of metals grown on corresponding soils (De Vries, 1983; Jing and Logan, 1992). However, others have found the predicting power of such extracts poor in relation to biological uptake, especially for Cu and Pb (Hooda and Alloway, 1993; Miner *et al*., 1997).

The significance of the bioavailable fraction relative to the available fraction and the total concentration, and how it may change with time in response to external factors, is impossible to quantify without direct reference to an organism. Soil properties such as pH, organic matter, iron oxide and clay contents, and Eh affect solubility and the species of metals and subsequently metal availability (*cf*. Figure 2.3.8.3).
These factors <u>and</u> the biological receptor - its specific pathophysiological characteristics (such as route of entry, duration and frequency of exposure) - will govern bioavailability.

Figure 2.3.8.3: Adsorption, precipitation and ageing processes in soil



Adsorption and precipitation moves metals form the solution to the solid phase (soil surfaces). Ageing moves metal from soil surface to deeper/stronger in the solid phase through (a) surface pore diffusion, (b) solid state diffusion, (c) occlusion of metals through precipitation of other phases, (d) precipitation of new metal solid phases, and (e) occlusion in organic matter (McLaughlin, 2001).

Attempts to link effects measured on biological receptors, for example soil microbial communities, invertebrates and plants and soil properties, often find that differences in experimental approaches between studies render this extremely difficult. Giller *et al*. (1998) concluded that although the factors affecting metal availability and toxicity are known, quantifying their relative effects is not yet possible. Nevertheless, progress on the biotic ligand model (BLM) approach for soils is being made (Allen *et al*., 2004). The model is based on the assumption that bioavailability is related to the interaction of free metal ions with receptor sites of the organisms – the biotic ligands. The model takes account of inorganic and organic ligand competition with the biotic ligands for the available metal. The metal becomes toxic when the fraction of sites at the biotic ligand occupied by the metal exceed a certain level – which is organism specific. However, this work is still under development, is available for only a few metals and has not been trialled within a regulatory or trigger setting context. But, if the terrestrial BLM follows the progress of the aquatic BLM, it could eventually be used to set trigger values (US EPA, 2003).

Building on the links with ecotoxicological data, the work on metal risk assessment driven through the Existing Substances Regulation (793/93/EEC) has seen significant developments, namely the incorporation of soil factors that modify availability into the derivation and understanding of the environmental effects of concentrations of metals in soils. In particular, factors that account for ageing of metals and their removal from soil solution (Figure 2.3.8.3), background concentrations (usually ambient) and availability and in a few situations bioavailability have been included. For data-rich metals such as Cu and Zn, these modifications have been applied to environmental concentrations for comparison with laboratory toxicological data in risk assessment. The potential exists to set local trigger values for metals and the Environment Agency is currently undertaking work to establish the practicalities of doing this within a broader soil sampling regulatory context, with the focus upon ecological risk assessment. The project will enable soil total metal concentrations to be modified in order to compare them with ecotoxicological data generated in laboratories (Merrington *et al*., 2006). The project will deliver at the end of June 2006 and will be of relevance with regard to trigger values for the soil function of environmental interaction.

The trigger values, or values which give an indication of when the soil is outside a manageable range, will obviously depend upon the metal. With regard to which metals would potentially appear in a minimum dataset for the soil function of environmental interaction, it is recommended that it should be metals of national importance, those with known detrimental effects in relation to environmental interaction, ecologically relevant and potentially multiple-source metals. Through the Existing Substance Regulations (Ni, Zn) and Cu Voluntary Risk Assessment, legislation and guidance notes (Water Framework Directive, IPPC; MAFF, 1993; MERAG, 2004) and scientific literature sources (Stevens *et al*., 2003), the metals initially selected are Cu, Ni and Zn. Within the tiered approach to soil sampling it is possible to analyse for other metals through the identification of increased risk at particular sites or locations (Tiers 2 or 3).

Integration with other national monitoring schemes

Data from the National Soil Inventory (NSRI 1983 and 1995), MASQ (Monitoring and Assessing Soil Quality in Great Britain) (MASQ, 2000) and soil/herbage surveys (reports to be published by the Environment Agency in 2006) can be used to map metal loading in the UK relative to land use and soil properties. In particular, the NSI data could be of great value in attempting to derive local or regional background metal concentrations. Further, the recent work carried out under the FOREGS programme (Forum of the European Geological Surveys Directors - http://www.gsf.fi/foregs/geochem/) could provide baseline data for ambient metal concentrations.

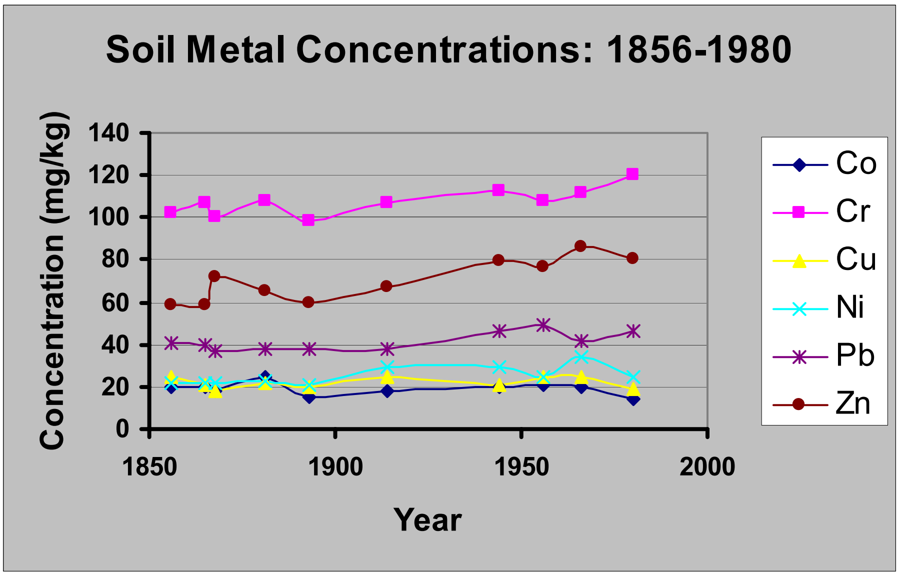


Figure 2.3.8.4: Total soil metal concentrations from archived soil samples from Rothamsted (Jones *et al*., 1987)

Data from long-term trials and analysis of archived soils may also be pertinent with regard to trends and relevant sources. For example, data from the Rothamsted archived soils indicates long-term increases (or not) in key metal concentrations (Figure 2.3.8.4) (Jones *et al*., 1987). This data is from untreated Broadbalk plots and so reflects mainly aerial deposition and not other potential agricultural inputs of metals.

Data below (Figure 2.3.8.5), for the plots receiving N, P, potassium (K), sodium (Na) and magnesium (Mg) fertilizers, illustrates any additional effect of the fertilizer additions, and suggests that for these metals, inputs from fertilizers are modest and that aerial deposition is the most significant source.

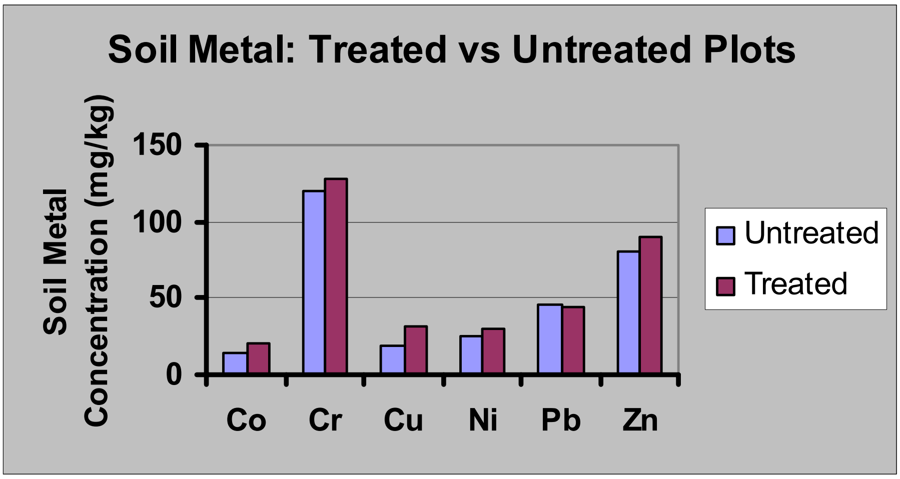


Figure 2.3.8.5: Total soil metal concentrations from archived soil samples from Rothamsted showing control and soil treated with fertilizers (Jones *et al*., 1987)

Cost, suitability and ease of analysis

The Environment Agency and the National Laboratory Services (NLS) have indicated that a general metal scan using an Inductively Coupled Plasma Mass Spectrometer (ICPMS) for soil sample analysis, following digestion using aqua regia, would cost around £30 per sample. The methods are standardised and should be widely available. If a more specific analysis is required, a separate quote is needed specifying the actual elements of interest.

ORGANIC MICROPOLLUTANTS

What does change mean?

One difficulty in using organic contaminants as SQIs is the large number of possible contaminants. The European Comission (EC) working group on parameters and indicators suggested that halogenated compounds (HCH, DDT/E), PAHs, polychlorinated biphenyls (PCBs) and di-benzofurans/dioxins were likely to be of greatest concern. However, their monitoring would be restricted to specific sites (EC, 2004).

The fate and effects of organic contaminants in soil depend on numerous factors. Prevalent environmental conditions, soil characteristics including soil properties and biota, and the properties of the compounds themselves – polarity, solubility, volatility – will determine the fate of chemicals in soil (O'Neill, 1993; Moorman, 1996; MacLeod *et al*., 2001). The amount of compound present in soil declines over time via loss processes, including physical transport, chemical reactions and biological activity (Northcott and Jones, 2000; MacLeod *et al*., 2001). However, a significant amount of the compound may remain in soil. As the compound ages, it may become less available and bioavailable, as determined by a decrease in the labile fraction of the compound, coupled with an increase in bound residues (Alexander, 2000; Jones *et al*., 2000; Northcott and Jones, 2000; Gevao *et al*., 2003). Although these processes are significant and are widely studied, there is at present no consensus on how bioavailability should be measured, and how data obtained via the various methods available (pore water concentrations, extractants of increasing 'harshness', bioassays) can be interpreted robustly and consistently, in relation to potential impact.

Nevertheless, changes in the context of SQI can be interpreted as changes in the identity of the contaminants present or an increase or decrease in contamination level. If a change in environmental conditions (such as soil properties with land use or perturbation) occurs, changes in the potential impact of a contaminant loading as the bioavailable fraction[1] increases or decreases (the total fraction may remain the same or also be affected).

If we assume that organic contaminants will only be assessed on a site-specific basis, temporal changes will be measured relative to the first analysis following concern, and to available background data, taking into account a specific set of circumstances. A discussion of the significance of changes for this SQI may therefore not be relevant at this stage. If organic contaminants are part of the MDS, priority substances would need to be chosen, and monitoring would give an indication of temporal changes in levels of these substances of concern in UK soils.

[1] If we assume that effects are due to interaction between biota and the bioavailable fraction, given that the latter is entirely dependent on the organism of interest and therefore not an absolute value. As such, it is a much contested term.

Triggers

If organic contaminants are chosen at a higher tier assessment, the choice of analytes will depend on the geographical location of the site, its use and expert judgement and/or data available on potential inputs (empirical or modelled). General analytical scans can be performed, but resolution will be low and may not detect some of the substances potentially present on site. Triggers for concern may include concentrations above the background concentration for the area (see below), or may be derived from ecotoxicological data. For the latter, it should be noted that the potential impact of long-term exposure of low-level organic contaminants is not well documented. Moreover, sites will most likely contain a complex mixture of organic and inorganic contaminants. However, a watching brief approach could be taken to inform SQI, particularly to address the lack of data on how to translate low-level long-term exposure into actual effects on organisms and ecosystems.

Integration with other national monitoring schemes

A database containing data from previous surveys, such as soil and herbage surveys (reports to be published) and MASQ (Monitoring and Assessing Soil Quality in Great Britain; MASQ, 2001), and from peer-reviewed publications, including soil properties and geographical information system data if available, would be a useful though complex and time-consuming tool, for example to help model soil as source and sink. It would provide information on existing concentrations of organic contaminants in UK soils and could be used to model potential temporal changes and areas at risk.

Cost, suitability and ease of analysis

Cost is difficult to assess. If a large number of compounds need to be quantified accurately in soil samples, the cost can escalate rapidly. Moreover, analysis of some compounds is likely to be restricted to a small number of laboratories, due to the equipment and expertise required to adequately analyse the samples – that is, appropriate clean-up methods and good quantification with low limits of detection.

Example of cost from NLS:

A basic mass scan would cost £55.00 per sample. Depending upon what was found the costs are as follows:

OCP/PCBs = £270.00 per sample.

PAHs = £55.00 per sample.

Methods would involve harsh solvent extractions in order to quantify maximum potential risk and actual levels in soil. If a more subtle approach is required, sequential extractions with different solvents and bioassays may be used to quantify the fraction likely to be affecting soil function. However, this adds to the level of complexity and could only be performed for specific sites, as it is in effect the approach taken for ERA (Ecological Risk Assessment) of contaminated land.

Conclusions and recommendations

For both metals and organic micropollutants, caution needs to be taken when using the data, as publicly detrimental views of contaminated land, i.e. containing heavy metals or organic pollutants - may have serious financial and development implications.

The inclusion of metals within a minimum dataset for measuring the soil quality for environmental interaction is not without a range of complex methodological issues and important caveats. Total soil metal concentrations have only marginal ecological and environmental relevance and provide no indication of ecotoxicology effects. Partial or dilute soil extractants as measures of metal availability have been used extensively, particularly in relation to soil fertility and estimating plant uptake. While accounting, in part, for some of the soil's physico-chemical characteristics affecting metal speciation, correlations with biological effects remain mixed, with some metals presenting useful and consistent relationships (often Zn) and others being particularly poor (Cu and Pb). Bioavailable metal concentrations, as outlined above, are relevant only to the specific organism under consideration, and not all organisms in the soil.

Therefore, as a first screen within a tiered risk assessment, it is suggested that modification of total soil metal concentrations be undertaken in accordance with work under the ESR and Voluntary Risk Assessment to enable rapid comparison with terrestrial toxicological data. This relatively precautionary approach is currently being assessed by the Environment Agency for its value within ecological risk assessment. On this basis, it is recommended to measure metal concentrations in a minimum dataset, with the inclusion of Cu, Ni and Zn.

The inclusion of organic micropollutants for measuring soil quality may be politically attractive; there is no doubt that many organic contaminants are toxic, such as dioxins and PCBs, and widespread in the environment. However, relating the presence of a contaminant or groups of contaminants to, for instance, effect on soil communities in the field is nigh impossible. Moreover, the effects of long-term low-level exposure on soil (and water) systems are at present unknown or at least uncertain. Organic micropollutants thus reflect potential pressure (pollution) rather than function. Further, *routine* monitoring of many organics which have half lives of < 10-20 days in soil is pointless. If monitoring is to be undertaken it should more closely reflect the PBT (persistence, bioaccumulation and toxicity) hazard criteria.

Although the presence of organic micropollutants could be viewed in the context of soil quality for environmental interactions (for example, potential water contamination or soil as a source/sink for aerial deposition), overall, organic micropollutants pose major problems of selection (both in terms of the choice of substances and, as for metals, the fraction of relevance – total, available, bioavailable) and analysis. Consequently, any monitoring is likely to be carried out at particular sites suspected of having problems and not routinely countrywide.

Therefore, it is recommended that organic micropollutants be considered as higher tier/site-specific indicators. On this basis, samples should be archived to allow for retrospective analysis and re-calibration of analytical methods if re-sampling.

References

Alcock, R.E., Johnston, A.E., McGrath, S.P., Berrow, M.L. and Jones, K.C. (1993) *Environ. Sci. and Tech*, 27, 1918-

Allen, H., Ponizovsky, A.A., Di Toro, D.M., Shi, Z., Thakali, S. and Metzler, D.M. (2004) Development of a terrestrial BLM for trace metals. Paper presented at SETAC Europe 14th Annual Meeting, Prague, 18-22 April. SETAC Europe, Brussels.

Alloway, B.J. (1995) *Heavy metals in soils*. 2nd edition. Blackie Academic and Professional, London.

Ashmore, M. *et al*. (2000) *Development of a critical load methodology for toxic metals in soil and surface waters: stage II*. EPG 1/3/144, Final contract report, Part I. http://www.bradford.ac.uk/acad/envsci/Research/PIG/metalscritloads/criticalloads(stage3)/EPG1_3_144partI.pdf

Alexander, M. (2000) Aging, bioavailability, and overestimation of risk from environmental pollutants. *Environmental Science and Technology,* 34, 4259-4265.

Basta, N.T., Ryan, J.A., and Chaney, R.L. (2005) Trace element chemistry in residual-treated soil: key concepts and metal bioavailability. *Journal of Environmental Quality*, 34, 49-63.

Cu-RAR (2005) *European Union Risk Assessment Report. Copper, copper(ii) sulphate pentahydrate, copper(i) oxide, copper(ii) oxide, dicopper chloride trihydroxide*. Voluntary risk assessment, Chapter 3, draft May 2005. European Copper Institute.

De Vries, M.P.C. (1983) Investigations on twenty Australian sewage sludges – their evaluation by means of chemical analysis. *Fertilizer Research*, 4, 75-87.

Drexler, J., Fisher, N., Henningsen, G., Lanno, R., McGreer, J., and Sappington, K.G. (2003). *Issue paper on the bioavailability and bioaccumulation of metals*. US EPA Risk Assessment Forum, Washington.

EC (2004) Task groups on parameters, indicators and harmonisation. Group leader: Loveland, P. *Soil Thematic Strategy*, EC, Bruxelles

Gevao, B., Jones, K.C., Semple, K.T., Craven, A., and Burauel, P. (2003) Nonextractable pesticide residues in soil. How can we resolve conflicts in definition, regulatory requirements, and significance? *Environmental Science and Technology*, 37, 139-144A.

Giller, K.E., Witter, E., and McGrath, S.P. (1998) Toxicity of heavy metals to microorganisms and microbial processes in agricultural soils: a review. *Soil Biology and Biochemistry*, 30, 1389-1414.

Hooda, P.S. and Alloway, B.J. (1993) The plant availability and DTPA extractability of trace metals in sludge-amended soils. *The Science of the Total Environment*,149, 39-51.

Impellitteri, C.A., Saxe, K.J., Cochran, M., Janssen, G.M.C.M., and Allen, H.E. (2003) Predicting the bioavailability of Cu and Zn in soils: modelling the partitioning of potentially bioavailable Cu and Zn from soil solid to solution ratio. *Environmental Toxicology and Chemistry*, 22, 1380-1386.

Jing, J. and Logan, T.J. (1992) Effects of sewage sludge cadmium concentration on chemical extractability and plant uptake. *Journal of Environmental Quality*, 21, 73-81.

Jones, K. C., Symon, B. D. and Johnston, A. E. Retrospective Analysis of an Archived Soil Collection I. Metals. (1987) *The Science of the Total Environment*, 61, 131-144

Jones, K.C., Gevao, B., Mordaunt, C., Northcott, G.L., and Semple, K.T. (2000) Special issue: Non-extractable residues in soils and sediments: characterisation and environmental significance. *Environmental Pollution*, 108, 1-119

Lofts, S., Spurgeon, D., Svendsen, D., and Tipping, E. (2004) Derivation of critical limits for ecotoxicological effects of Cu, Zn, Cd and Pb in soils. http://www.bradford.ac.uk/acad/envsci/Research/PIG/metalscritloads/criticalloads(stage3)/critlimitsstage3.htm

MacLeod, C.J.A., Morriss, A.W.J., and Semple, K.T. (2001) The role of microorganisms in ecological risk assessment of hydrophobic organic contaminants in soils. *Advances in Applied Microbiology*, 48, 171-211

MAFF (1993) *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Potentially Toxic Elements*. Ministry of Agriculture, Fisheries and Food, London.

MAFF (2000) *Towards sustainable agriculture: a pilot set of indicators*. MAFF Publications, London.

MASQ (2001) *Countryside Survey Module 6: Soils and Pollution*. CEH R&D Technical Report from Countryside Survey 2000, project P5/053/01. Environment Agency, Bristol.

McLaughlin MJ, 2001. *Ageing of metals in soils changes bioavailability*. Fact Sheet 4 on Environmental Risk Assessment, ICME (International Council on Metals and the Environment), 6p.

Meers, E., Unamumo, V., Vandegehuchte, M., Vanbroekhoven, K., Geebelen, W., Samson, R., Vangronsveld, J., Diels, L., Ruttens, A., Laing, G.D., and Tack, F. (2005) Soil-solution speciation of Cd as affected by soil characteristics in unpolluted and polluted soils. *Environmental Toxicology and Chemistry*, 24, 499-509.

MERAG (Metals Risk Assessment Guidance) (2004) Background Document, Draft Report 18th December 2004. EURAS, Commissioned by ICMM and Eurometaux.

Merrington, G., Fishwick, S. and Brooke, D. (2006) The derivation and use of soil screening values for metals for the ecological risk assessment of contaminated land: a regulatory perspective. *Land Contamination and Reclamation* (In Press).

Miner, G.S., Gutierrez, R. and King, L.D. (1997) Soil factors affecting plant concentrations of Cd, Cu and Zn on sludge-amended soil. *Journal of Environmental Quality*, 26, 989-994.

Moorman, T.B. (1996) Assessing organic chemical contaminants in soil. In *Methods for assessing soil quality* (Doran, J.W. and Jones, A.J., eds), pp311-326, SSSA Inc, Madison.

Northcott, G.L., and Jones, K.C. (2000) Experimental approaches and analytical techniques for determining organic compound bound residues in soil and sediment. *Environmental Pollution*, 108, 19-43.

O'Neill, P. (1993) *Environmental Chemistry*, Second Edition. Chapman and Hall, London

Stevens, D.P., McLaughlin, M.J., and Heinrich, T. (2003) Determining toxicity of Pb and Zn run-off in soils: salinity effects on metal partitioning and on phytotoxicity. *Environmental Toxicology and Chemistry*, 22, 3017-3024.

Sanaterre Environmental (2004) The Dutch target and intervention values. www.sanaterre.com/guidelines

The Netherlands (2004) Risk Assessment Report: Zinc metal. Final draft of December 2004 (R073_0412_env).

Thornton, I. (1995) *Metals in the global environment: facts and misconceptions*. ICME, Ottawa

US EPA (2003) *Draft update of ambient water quality criteria for copper*. USEPA 822-R-03-026. November 2003.

2.3.9 Total nitrogen

Nitrogen is an essential constituent of all nucleic acids, amino acids and proteins, and therefore fundamental to the reproduction and growth of all organisms. The ultimate source of the N in soils used by plants is N_2 gas from the atmosphere. The conversion of N gas into a form that can be utilised by plants is dependent on transformations including microbial N fixation in the soil, fixation as oxides of N by lightning and through fixation as ammonia and nitrate by the manufacture of synthetic N fertilizers.

The widespread use of nitrogenous fertilizers in agriculture over the last 50-70 years has had a dramatic effect upon crop and livestock production. Yet, like phosphorus, it is the loss of N in various forms from the agroecosystem that has heightened environmental concern. Losses of N from soils may occur naturally via drainage water and leaching (mostly as NO_3^-), in run-off and as gases from denitrification and volatilization (NH_3, N_2 and N_2O). The increased inputs to and accelerated loss of N from the agroecosystem is the reason behind the concern (Figure 2.3.9.1).

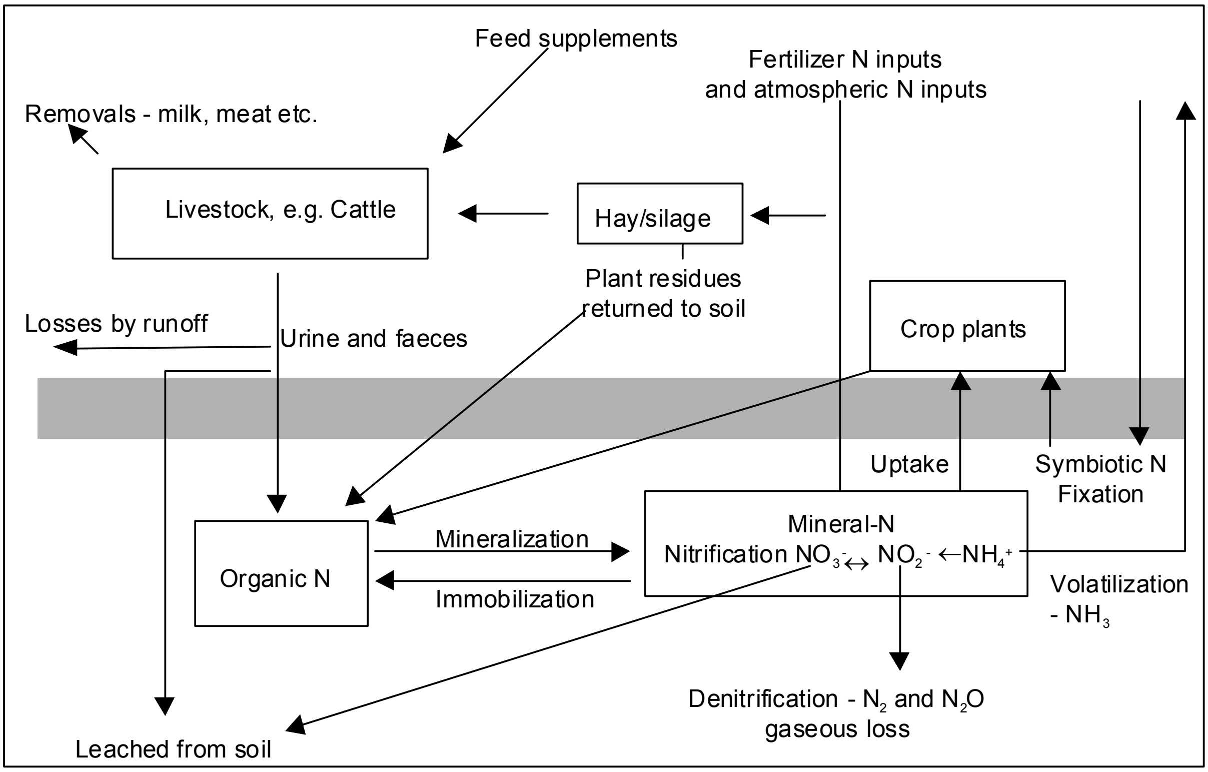


Figure 2.3.9.1: The agricultural N cycle (adapted from Rowell, 1994)

Significant inputs of N to soil may also occur from animal manures - where 70% of the N consumed by cattle and 80% by sheep is excreted as dung and urine (Haygarth *et al.*, 1998) - organic wastes and aerial deposition (estimated at 30-50 kg N per ha per year deposited on agricultural land from the atmosphere in southern and eastern England). Again, like P, soil N surpluses above optimum economic values for production are now common in the UK, although declining (70 kg/ha in the 80s, 25 kg/ha in the late 90s, FMA, 1998).

Measuring total soil nitrogen (N) gives an indication of the total amount of all forms of N in the soil and also the size of organic N reserves and organic matter quality (Schipper and Sparling, 2000). In most topsoils, organic matter N would comprise greater than 90% of the total N. For N to be useful as a soil quality indicator it must provide relevant, interpretable and sensitive information on changes in soils in relation to the function of environmental interaction. The key concern for this paper is to challenge the use of total soil N (TSN) in terms of the following: what does a change in TSN mean? What triggers in TSN exist? Is it possible to integrate TSN with other indicators? How much does TSN measurement cost and what is the sensitivity and likely variability of the method used?

What does change mean?

Increasing levels of N used in agriculture are thought to be responsible for increases of N in drainage waters, with the subsequent detrimental effects upon human and ecological water quality (Royal Society, 1983; DoE, 1986; House of Lords, 1990; MAFF, 1993; Isherwood, 2000). These effects result from the contamination of groundwaters and surface waters, to an extent where pre-treatment for nitrate removal is required before human consumption, and of the formation of algal blooms and eutrophication.

Gaseous losses of N from soils can occur as nitrous oxide and nitric oxide from nitrification and denitrification and also through volatilisation of ammonia. These losses tend not be as great in the UK as quantities of nitrate, but can be crudely linked to total soil N content (Fowler *et al*., 1996).

Increased N in soils often results in increased plant growth and vigour manifested by greener foliage. However, too much available N can adversely effect crop plants; for example, barley grown for malting needs grain with as much starch and as little protein content as possible (ideally not in excess of 1.6% N). Too much N may also reduce the sugar content of sugar beet (Isherwood, 2000). Excessive amount of N produces vegetative growth with large, succulent, thin-walled cells which may mean leaves and stems are more readily attacked by insect pests and fungal diseases and that stems are less mechanically strong and crops prone to 'lodging'.

Crop plants tend to take up N as nitrate, and occasionally ammonium, relatively rapidly during growth. However, the nitrate left in the soil following harvesting at the end of the cropping season, and the quantity that is mineralised when crop growth is insignificant in terms of uptake, is readily susceptible to leaching and subsequent loss. Soils are thought to be prone to leak substantial quantities of NO_3^- in drainage water when TSN values > 40 kg N/ha in many UK soils (Lord and Anthony, 2000).

Total soil nitrogen is closely related to soil organic matter. It can be combined with soil carbon values for an indication of soil N mineralisation potential, that is, the potential of a soil to convert organic N compounds to inorganic forms. As the C:N ratio increases so N mineralisation decreases, giving an indication of the potential activity of soil microbial populations (White, 1987). The ratio of C:N is relatively constant in temperate agricultural soils and falls between 10-12:1. The C:N ratio is particularly useful when looking at organic materials applied to soils; as values increase above 30, the soil biomass becomes limited by the quantity of N and will not be able to utilize the C. This reduction in available N is termed immobilization and can last a significant time (a growing season) depending on a range of factors, but particularly the C:N of the added residue (Rowell, 1994). Therefore, through monitoring changes in total N an indication of potential N behaviour in soil is determined.

The adverse effects of elevated soil N values in N-poor semi-natural ecosystems are well documented and referred to through the ‘critical loads’ programme (CEH, 2004). These N inputs are only indirectly related to agriculture, and are more likely due the burning of fossil fuels. There is a calculated 67% exceedance of the critical loads for nutrient nitrogen in all UK habitats, The greatest accumulated exceedance is for managed confierous woodland apart from in England where again, like P, managed broadleaved woodland is at risk. Adverse effects of increased soil N in low N habitats are, like P excess, manifested through diversity loss and increased dominance of non-indigenous or exotic species.

Critically, speciation of N compounds greatly affects behaviour, fate and transfer of N in soils (cf. Tisdale *et al*., 1993). As mentioned above, it is available N or mostly nitrate that is taken up by plants and also leached from soils. Monitoring changes in TSN does not always provide the sensitivity required to determine the likely environmental risks (Ashrad and Coen, 1992). Indeed, TSN is often regarded as being of limited relevance to soil processes and functions, with greater importance being placed on measures of mineral N, which take into account many of the factors that influence soil N dynamics and subsequent behaviour (Goulding, 2000; Wilson *et al*., 2001). This insensitivity of TSN as an indicator for environmental interaction is well illustrated by Adair *et al*. (2004) who noted similar TSN values in two stretches of alluvial soils on the Green and Yampa Rivers in Colorado. Yet N turnover and mineralization rates differed by up to 4 times, indicating significantly different N behaviour and environmental risk. Further, old, permanent grassland, heathland, and upland organic soils in the UK may have accumulated >10 tonnes of organic N in the topsoil, but could be quite infertile (due to low pH and/or temperature for example) and have a low potential for N losses unless applied with fertiliser. For example, the ‘zero N grazed ’ treatment plots of the Rowden Moor drainage experiment at North Wyke, Devon, has produced little over 5 tonnes of herbage dry matter on average over the last 20 years and has caused nitrate leaching to about 2 kg N per ha per yr on average over the same period, despite having a soil total N value of about 10 t per ha. With 400 kg N per ha per yr inorganic fertiliser applied, the same soil yields >11 t per ha and causes N leaching within the range 50-150 kg per ha per yr (David Scholefield *Pers. comm*.).

Selection of triggers for total soil nitrogen

Depending upon the cropping history of the soil, its mineral composition and prevailing environmental conditions, agricultural soils contain 2,000-6,000 kg N per ha, almost all of which is in organic form and therefore unavailable for crop uptake (Powlson, 1993). Values for TSN are often combined with a measure of SOC to give C:N ratios, which are often used as triggers for soils in an agricultural context (*cf*. Tisdale *et al*., 1993).

Nevertheless, the New Zealand group have defined baseline values for a range of soil types and land uses and established triggers, through the use of response curves and expert judgement (Sparling *et al*., 2003). The trigger values in Table 2.3.9.1. are applicable to all soil orders.

Table 2.3.9.1: Total soil nitrogen trigger values developed by Sparling *et al*. (2003)

Pasture	0	0.25	0.35	0.65	0.7	1.0
Forestry	0	0.10	0.20	0.60	0.7	1.0
Cropping and horticulture*						
	Very depleted	**Depleted**	**Adequate**	**Ample**	**High**	

*Exclusion due to dependency on crop
Values in shade are upper and lower limits

It has been suggested by Sparling et al. (2003) that there may be three possible methods by which trigger values for soil indicators could be derived. These methods are:

- using the national soil baseline data – calculating median and lower quartile values for a parameter for each soil order;
- modelling to set a limit set value that still permitted recovery to a value within a 25 year period; is this possible for TSN?
- expert judgement - used in a workshop format to derive response curves for changes in the TSN level (on the x-axis) against soil quality (0-100 %).

However, data is limited on TSN in national monitoring programmes or extensive sampling studies, which greatly hampers attempts at interpretation and evidence-based expert judgements.

Integration with other national monitoring schemes

Few monitoring or soil monitoring studies have used TSN. In the UK, TSN has not been measured in NSI, RSSS or CS2000, which greatly limits the opportunity to integrate measures or compare data.

The incorporation of TSN with a measure of soil organic matter or soil organic carbon is a distinct possibility. Indeed, the close relationship between TSN and organic matter is such that estimates of soil N content can be readily made from organic matter (Tisdale *et al*., 1993; Motta *et al*., 2002).

Cost and suitability for broad-scale monitoring

Total soil nitrogen can be measured as gaseous N following high temperature combustion, a relatively rapid method allowing significant throughput of samples and automation. Yet some laboratories may still use the Kjeldahl diesgtion method, which is somewhat slower and certainly less suitable for automation. Cost of TSN determination can be as much as £15 per sample (National Laboratory Service, Environment Agency*, Pers. Comm.*)

Total soil nitrogen is often viewed as a less variable or volatile soil quality indicator (Gilltrap and Hewitt, 2004). Schipper and Sparling *et al*. (2000) measured a range of soil quality indicators across 29 sites in New Zealand and showed the coefficient of variation for TSN to be 12%. The use of N as a stable benchmark in some studies does illustrate its possible unsuitability as a soil quality indicator compared to other soil nitrogen indicies in that it is insensitive to change (Beauchamp *et al*., 2003).

Topsoil TSN may give only a limited indication of the capacity of soil N budget, due to exclusion of a relatively large portion of N in the coarse soil fraction. Analysis of soils is traditionally undertaken on the < 2 mm or fine earth fraction, but Whitney and Zabowski (2004) showed that for 0.3-37 % of the combined fractions, TSN was present in the coarse fraction (> 2 mm). This fraction is probably not terribly dynamic, but somewhat devalues TSN as a measure of total N pool and potential N behaviour.

Many groups measuring soil quality have chosen TSN in their minimum dataset (Sparling *et al*., 2002; De Clerk *et al*., 2003). Other groups undertaking soil monitoring have not, citing the inability of TSN to provide a discriminatory measure of the threats and risks posed by N species to the broader environment (Stevenson, 2004). In particular for non-agricultural soils or N-sensitive habitats, TSN may not provide the required detail during interpretation.

Conclusion and recommendation

Total N provides an indication of the total N pool in soils; also, when linked with SOC, it may provide an indication of the likely mineralizable N supply. It is relatively straight-forward to measure and provides a broad indication of N behaviour in soils. The close association with SOC suggests that measuring both would not be necessary and that as SOC is a relative certainty in the MDS, total N could be dropped.

Further, in regard to environmental interaction, TSN does not provide sensitive, interpretable information to gauge risks and threats to surface and groundwater quality.

The use of total soil nitrogen in a minimum dataset is not recommended.

References

Ashrad, M.A. and Coen, G.M. (1992) Characterization of soil quality: physical and chemical criteria. *Am. J. Alternative Ag*., 7, 25-31.

Adair, E.C., Binkley, D. and Andersen, D.C. (2004) Patterns nitrogen accumulation and cycling in riparian floodplain ecosystems along the Green and Yampa rivers. *Ecologia*, 139, 108-116.

Beauchamp, E.G., Parararajasingham, R. and Kay, B.D. (2003) Relationships of total soil nitrogen to several soil nitrogen indicies. *Communications in Soil Science and Plant Analysis*, 34, 505-518.

CEH (2004) Addendum: The status of UK critical load exceedances. April. CEH, Monks Wood, Huntingdon.

De Clerk, F., Singer, M.J. and Lindert, P. (2003) A 60 year history of California soil quality using paired samples. *Geoderma*, 114, 215-230.

DoE (1986) *Nitrate in Water: a report by the Nitrate Co-ordination Group*. Department of the Environment Pollution, HMSO, London. Paper No. 26.
FMA (2000) *The fertilizer review 2000*. Fertilizer Manufacturers Association, Peterbourough.
Fowler, D., Sutton, M.A., Skiba, U. and Hargreaves, K.J. (1996) Agricultural nitrogen and emissions to the atmosphere. In: *Issues in Environmental Science and Technology 5: Agricultural Chemicals and the Environment* (Eds. Hester, R.E. and Harrison, R.M.) The Royal Society of Chemistry, Letchworth. pp 57-84.
Giltrap, D.J. and Hewitt, A.E. (2004) Spatial variability of soil quality indicators in New Zealand soils and land uses. *New Zealand Journal of Agricultural Research*, 47, 167-177.
Goulding, K. (2000) Nitrate leaching from arable and horticultural land. *Soil Use and Man*., 16, 145-151.
Haygarth, P.M., Chapman, P.J., Jarvis, S.C. and Smith, R.V. (1998) Phosphorus budget for two contrasting grassland farming systems in the UK. *Soil Use and Man*., 14, 160-167.
House of Lords (1990) *Nitrate in Water: 16th Report of the Select Committee on the European Communities* (HL Paper 73). HMSO, London.
Isherwood, K.F. (2000) *Fertilizer use and the environment*. (Revised Edition) International Fertilizer Industry Association. Paris.
Lord, E.I. and Anthony, S.G. (2000) MAGPIE: A modeliing framework for evaluating nitrate losses at national and catchment scales. *Soil Use and Man*., 16, 167-174.
MAFF (1993) *Solving the Nitrate Problem: progress in research and development*. Publication No. 1092, Ministry of Agriculture, Fisheries and Food Publications, London.
Motta, A.C.V., Reeves, D.W., and Touchton, J.T. (2002) Tillage intensity effects on chemical indicators of soil quality in two coastal plain soils. *Commun. Soil Sci. Plant Anal*., 33, 913-932
Powlson, D.S. (1993) Understanding the soil nitrogen cycle, *Soil Use and Man.*, 9, 86-94.
Royal Society (1983). *The Nitrogen Cycle of the United Kingdom: a Study Group Report.* Royal Society, London.
Rowell, D.L. (1994) *Soil Science, methods and applications*. Longman Scientific and Technical. Harlow.
Schipper, L.A. and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses. Soil Sci. Soc. Am. J., 64, 300-311.
Sparling, G., Lilburne, L. and Vojvodic-Vukovic, M. (2003). *Provisional Targets for Soil Quality Indicators in New Zealand.* Landcare Research New Zealand Ltd, Palmerston North.
Stevenson, B.A. (2004) Changes in phosphorus availability and nutrient status of indigenous forest fragments in pastoral New Zealand Hill country. *Plant and Soil*, 262, 317-325.
Tisdale, S.L, Nelson, W.L., Beaton, J.D. and Halvin, J.L (1993) *Soil fertility and fertilizers.* 5th Edition. Prentice Hall, New Jersey.
White, R.E. (1997*) Principles and Practice of Soil Science, the Soil as a Natural Resource.* (3rd Edition). Blackwell Science Ltd, Oxford.
Whitney, N. and Zabowski, D. (2004) Total soil nitrogen in the coarse fraction and at depth. Soil Sci. Soc. Am. J., 68, 612-619.

Wilson, S.M., Pyatt, D.G., Malcolm, D.C. and Connolly, T. (2001) The use of ground vegetation and humus type as indicators of soil nutrient regime for an ecological site classification of British forests. *Forest Ecology and Management*, 140, 101-116.

2.3.10 Soil organic carbon

Soil organic carbon (SOC) is likely to be one of the minimum dataset of soil indicators for reasons unconnected with its value as an indicator of soil quality. Nevertheless, it is appropriate to challenge it using the same approach as that employed for the other potential indicators.

Thus, the key questions are:

- What does SOC indicate about the key soil function of environmental interaction?
- Is it measurable – that is, are there accepted analytical techniques? This is not discussed in this report.
- What is the feasibility of detecting temporal changes in SOC against the noise of spatial and temporal variability? This is discussed in the accompanying report by Smith and Fang for the Environment Agency, presented in Section 2.3.11.
- Is it possible to identify either breakpoints or trigger values which, if breached, would indicate significant environmental risk?

What does SOC indicate about the key soil function of environmental interaction?

Numerous studies show a relationship between SOC and aggregate stability, which in turn influences surface sealing and hence flood initiation (Kemper and Koch, 1966). But, as with the data shown here in Figure 2.3.10.1, there is no clear indication of a breakpoint or threshold below which aggregate stability declines sharply.

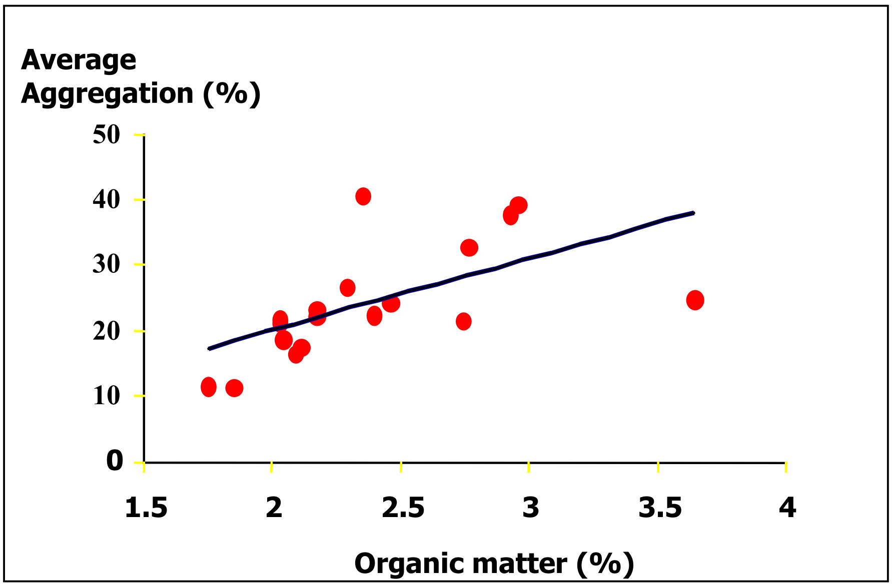


Figure 2.3.10.1: Aggregate stability vs soil organic matter

Analysis of the 6,500 soil profile data in the Landis database of British soils shows a relation between SOC and water stored at 5 kPa (the nominal field capacity) (Hollis *et al*., 1977). This effect is related to the aggregate stability noted above. There is a suggestion of a breakpoint at 4% SOC above which increasing SOC has little effect; below 4%, water storage declines linearly (Figure 2.3.10.2).

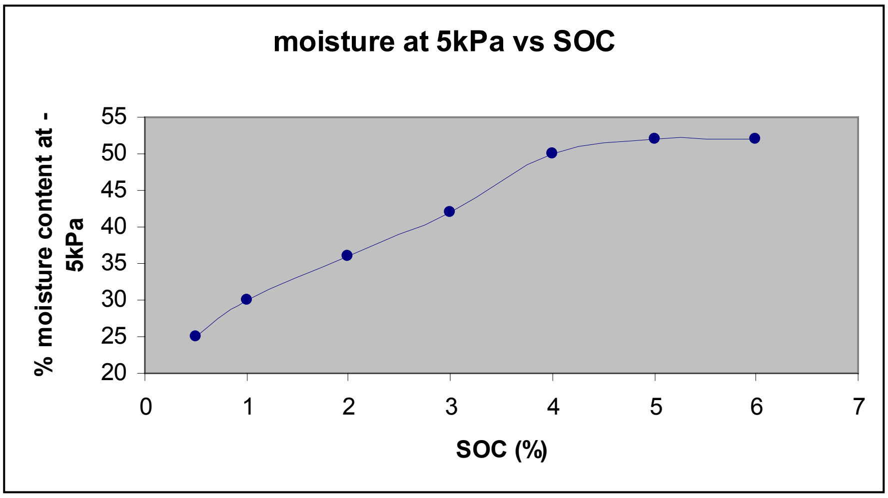


Figure 2.3.10.2: The relationship between soil moisture content and soil organic carbon.

SOM/SOC influences pesticide leaching through the retention of material in the organic matrix and the resultant smearing out of the pesticide pulse as it moves down the soil profile. Pesticide adsorption is described by:

$$K_d = C_{sorbed}/C_{solution}$$

$$\text{and } K_d = f_{oc}.\ K_{oc}$$

Where K_d = soil–water partition coefficient
f_{oc} = fraction of organic carbon in soil
K_{oc} = coefficient for organic carbon sorption

Note: organic carbon is related to organic matter by the approximate relation:

SOC = 0.58*SOM

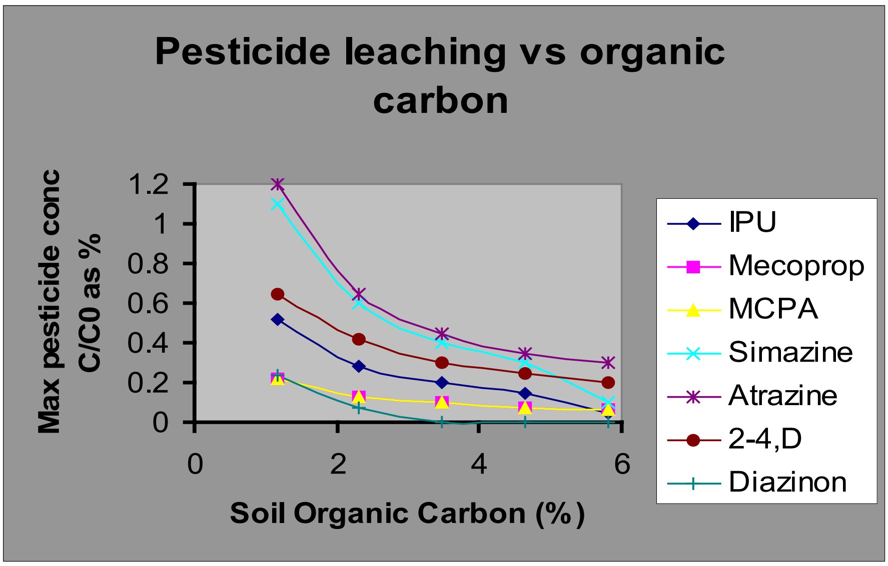


Figure 2.3.10.3:

Simulation models of pesticide leaching which account for both adsorption and degradation can show the effect of changes in SOC on pesticide leaching. The example in Figure 2.3.10.3 shows the effect of SOC on the peak concentration of a number of pesticides which appear most frequently in concentrations exceeding 0.1 μg/l in English surface waters (Pesticides in the Environment Working Group, 2000). The simulations assume 30 cm of topsoil and 290 mm of winter drainage (typical of much of central England). There is a clear relation between declining SOC and increasing pesticide maximum concentrations, but evidence of a breakpoint or threshold is ambiguous.

However, shallower soils show a more pronounced dependence on SOC. Where SOC is mainly in the upper 15 cm, pesticide leaching does increase at SOC levels below 3-4%.

SOM is an indicator of environmentally relevant and important functions which determine water acceptance (aggregate stability), water storage (retention at 5 kPa) and bio-filtration (pesticide leaching). Of these functions, only water storage shows a clear threshold or breakpoint at around 4% SOC. Where soil depth is 30 cm or greater, there is little evidence of a trigger or breakpoint; at depths of 15 cm or less, pesticide leaching increase at SOC levels less than 3-4%.

Identifying breakpoints or envelopes of normality

The discussion above suggests the following:

Aggregate Stability	No readily identifiable breakpoint. Confirmed by a number of studies.
Water Storage	Increases in SOC above 5 % SOC (i.e. 8.6% SOM) show no increase in water storage; below 8.6% SOM, water storage declines linearly. Relation appears robust (n = 77 and 74% of variance explained).
Filtration	Simulation of pesticide leaching suggests peak pesticide concentration increases with SOC. At soil depth of 30 cm or greater, no ambiguous threshold; at depths of 15 cm or less, pesticide leaching increases when SOC below 3-4%.

In a recent review, Loveland et al. (2001) concluded there was little consistent evidence that there were critical thresholds of SOC above or below which *soil physical properties* (and hence some of the environmental interactions above) changed significantly.

Identifying trigger values for SOC is further complicated by temporal changes in SOC driven by changes in land use and cropping, which alter the inputs and turnover/loss of soil carbon. The simplest model of SOC depicts a change from, for example, long-term grassland to arable as a transition between two equilibrium states.

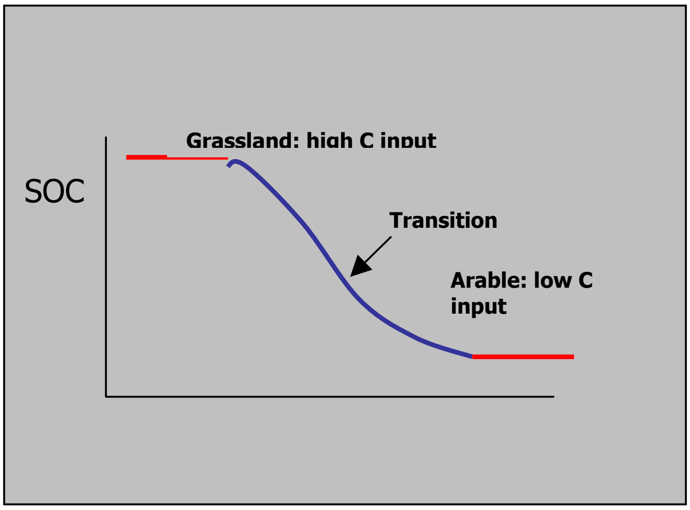


Figure 2.3.10.4: The potential change in SOC with land management.

In reality, this transition (Figure 2.3.10.4) is subject to noise because land use rarely changes from grass to arable; there are often intermediate cropping periods when grass leys are re-instated. And even if mono-cropping is established, annual yields and therefore returns of C to the soil vary with climate, fertiliser use and periods of vegetable growing.

However, the broad model is supported by data collected over thirty years at 14 sites, on SOM changes following the ploughing out of the rich fen silt soils in the East of England (Campbell and Watson, 2001). A long-term grassland equilibrium SOM of around 15% declines to an equilibrium value of 2-3% under arable (Figure 2.3.10.5).

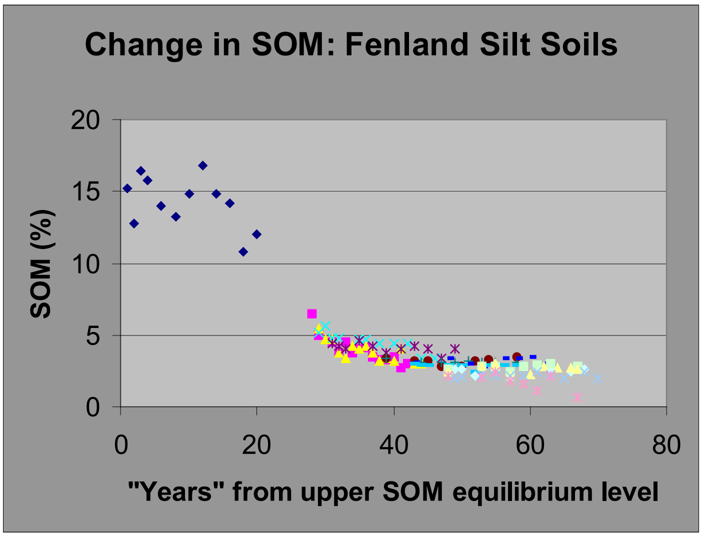


Figure 2.3.10.5: Decline in SOM in Fenland silt soils when land use change from grassland to arable.

SOC is also strongly influenced by soil texture. There is considerable evidence that finer textured soils such as clays protect SOM and, as a result, have higher SOM levels than coarser textured soils (see below). The influence of cropping and soil texture can be seen in Table 2.3.10.1, which uses data from the NSRI Landis dataset to derive envelopes of normality based on the 5 and 95 percentile distributions.

Table 2.3.10.1: Envelopes of normality for SOC with land use

			SOC (%)	
Land use	Soil texture	5 percentile	50 percentile	95 percentile
Arable	Clay	2	3.2	7.6
Arable	Sandy loam	1	1.8	5.6
Permanent Grass	Clay	2.6	5.4	9.2
Permanent Grass	Clay	1.6	3.2	11.2

Identifying envelopes of normality for semi-natural soils (such as woodlands) is problematic since the majority of the available data relates to agricultural soils.
The Countryside Survey 2000 (CS2000) (Black *et al*., 2002) gives the following ranges for soils supporting a number of semi-natural habitats (Table 2.3.10.2).

Table 2.3.10.2: Range of SOC in semi-natural habitats in the UK

	SOC (%)	
Habitat	Minimum	Maximum
Calcareous grassland	13.1	21.2
Neutral grassland	1.16	28.7
Broadleaf woodland	2.16	56.0
Coniferous woodland	1.74	56.6
Improved grassland	1.85	53.6
Acid grassland	3.7	56.4
Bog	6.6	56.7
Dwarf Shrub Heath	4.2	56.8
Bracken	4	55.3

These are not true envelopes of normality but extreme ranges, and the data may well include sites which are not functioning well in ecological terms. However, at present the dataset is limited.

Conclusions

- While there are clear relationships between SOC and parameters for environmental interaction, evidence for unambiguous breakpoints or trigger values is less clear.
- Water storage decreases at less than 4% SOC; and where soil depths are 15cm or less, pesticide leaching increases more sharply at SOC levels below 3-4%.
- Identifying trigger values is further complicated by the dependence of SOC on land use and cropping and by the SOC protection observed in finer textured soils.
- The functioning of soil in environmental interactions (water acceptance, water storage, pollutant attenuation) increases with SOC – so more is certainly better.
- It is possible to identify envelopes of normality as defined by the 5 and 95 percentiles for some land use/soil texture combinations.

- For many semi-natural soils and habitats, only extreme ranges are available.

Recommendations

- From the perspective of environmental interaction, a provisional SOC minimum level of 3% should be adopted for all soils receiving pesticide additions. Coincidentally, this contrasts with the general agronomy view that soils with SOM of <1.7% (equivalent to SOC of around 1%) are in a pre-desertification stage.
- From the perspective of environmental interaction, grassland soils not receiving pesticides should have a provisional SOC level between of 2 and 9%.
- From the perspective of environmental interaction, the extreme ranges of SOC observed in CS2000 for many semi-natural habitats should be regarded as provisional: more work is needed to identify reliable envelopes of normality.

References

Black, H. I. J. et. al. 2002. *MASQ: Monitoring and Assessing Soil Quality in Great Britain. Countryside Survey Module 6: Soils and Pollution* (The Environment Agency, Bristol.

Kemper, W. D. & Koch, E. J. 1966. *Aggregate stability of soils from western United States and Canada.* USDA Technical Report 1355, Washington DC.

Hollis, J. M., Jones, R. J. A. & Palmer, R. C. 1977. The effects of organic matter and particle size on the water retention properties of some soils in the West Midlands of England. *Geoderma* **17**, 225-238.

Loveland, P. J., Webb, J. & Bellamy, P. 2001. In *Sustainable Management of Soil Organic Matter* (eds. Rees, R. M., Ball, B. C., Campbell, C. D. & Watson, C. A.) 9-23, CABI, Wallingford.

Pesticides in the Environment Working Group. Monitoring of pesticides in the environment *(The Environment Agency, Bristol, 2000).*

2.3.11 UK SOIL QUALITY INDICATORS: IDENTIFICATION OF MINIMUM DATASET – SOIL ORGANIC CARBON

Changming Fang & Pete Smith
School of Biological Sciences, University of Aberdeen, Aberdeen AB24 3UU, UK

Introduction

Measuring changes in soil organic carbon (SOC) is necessary in many circumstances, including for verification of soil sinks under the Kyoto Protocol (Smith, 2004a), for assessing soil fertility (Smith and Powlson, 2003) and for assessing potential changes in soil quality (Sparling *et al*., 2003). If SOC is to be used as an indicator of soil quality, accurate measurements are critical, both for assessing spatial and temporal variations in SOC and for assessing possible management induced changes in SOC stocks.

For the purposes of identifying a SOC stock change, soils can be regularly sampled over a period of time. Changes in soil carbon stock can be determined and the change SOC stocks, ΔSOC, can be calculated as:

$$SOC = 10^{-2} \times BD \times C\% \times D, \quad (1)$$

$$\Delta SOC = (SOC_{t_2} - SOC_{t_1})/(t_2 - t_1), \quad (2)$$

where *BD* is soil bulk density (g/cm^3), *C*% is organic carbon content of soil dry mass and *D* is soil sampling depth (cm). The unit of SOC is ton C/ha. SOC_{t1} and SOC_{t2} are carbon stock at time t_1 and t_2, respectively.

The 1996 IPCC guidelines for C accounting recommend that changes in soil C stocks be calculated over a period of 20 years to a depth of 30 cm following a shift in management. Agricultural management is assumed to have the greatest influence over this time frame and portion of the profile (IPCC, 1997), although this may limit its usefulness if there are pervasive impacts at greater depths (Guo and Gifford, 2002). A significant change in SOC can occur in deeper soil layers after 13 years of an alternative system of tillage (Sisti *et al*., 2004). Soil bulk density often changes with land use and the soil C per unit ground area to a fixed depth will also change even without any change in the mass fraction of C in dry soil. This problem will generally arise when soil C accounting is taken to a fixed depth. For accuracy in determining the land use change effects on soil C, soil sampling should be referred to a fixed dry soil mass per unit ground area (Gifford and Roderick, 2003). For determining SOC stock change at benchmark sites, sampling soil to a depth greater than 30 cm may be the simplest way to account for the influence of changing bulk density on SOC. However, if SOC concentration/density in surface soil (0-30 cm or down to a greater depth) is being monitored for the purpose of using SOC as an indicator of soil quality, measuring the change of C stock per unit ground area is not necessary. Temporal changes in bulk density and soil depth can be excluded from the monitoring scheme. Many variables drive SOC change over time. The most important natural factors are climate (temperature, precipitation, etc.) and the net primary production (NPP) which controls the input of C to the soil. These factors change under natural climatic variability, and under climate change. SOC is also greatly influenced by changes in land use and agricultural management (Guo and Gifford, 2002; Smith *et al*., 1997, 2000). Such practices include fertilization (Smith *et al*., 2002), adding organic residues to soils (Jenkinson and Johnston, 1977), or changing vegetation type (Garten and Wullschleger, 1999), and shifting from the conventional to no-till farming (Smith *et al*., 1998; Sisti *et al*., 2004). Attributing SOC changes to natural causes of human-induced change such as land use and agricultural management is still a challenge (Smith, 2005). A common technique is using paired sites of undisturbed vegetation and agricultural use. Based on this method, IPCC suggested an equation to estimate SOC change under different agricultural land use and management practices (IPCC, 1997):

$$SOC_{\text{managend}} = SOC_{\text{native}} \times F_{\text{base}} \times F_{\text{tillage}} \times F_{\text{input}} \qquad (3)$$

The base factor (F_{base}) represents changes in soil organic matter associated with conversion of the native vegetation to agricultural use. Tillage ($F_{tillage}$) and input (F_{input}) factors account for effects of various management practices of lands in agricultural use. The latter two factors can be used to capture changes in management trends that have occurred over the counting period. Due to the high spatial variability in SOC, and the fact that most agricultural soils are cultivated, choosing a suitable reference site to assess land use and management activity on SOC stocks and soil quality is difficult (Ogle *et al*., 2003). The use of mathematical models of SOC under different scenarios will be useful for helping to assess the relative contributions of natural and land use/management to SOC change (Smith, 2005).

Our analyses and discussions aim to address the questions: 1) whether, and how, a management-induced change in SOC can be detected with acceptable precision against a background of natural variation; 2) whether appropriate criteria can be defined for using SOC content as an indicator in a minimum dataset (MDS) to assess soil quality.

The spatial variability of SOC and the ability of soil sampling to detect change

Large spatial variabilities have been reported, both in terms of SOC stocks and in terms of C fluxes, for many studies on various soils (Conen *et al*., 2003; 2004). Figure 2.3.11.1 shows SOC data in a middle-aged Sitka spruce plantation in Scotland, where SOC is highly variable due to site preparation in the late 1970s when the site was converted from farmland to plantation (F. Conen, *Pers. Comm.*). A 100×100 m plot was divided into 1×1 m grid and 100 samples were randomly collected from the 1×1 m subplot. The range of SOC content was 1,487-29,725 g C per m^2, with an area average of 10,021 g C per m^2.

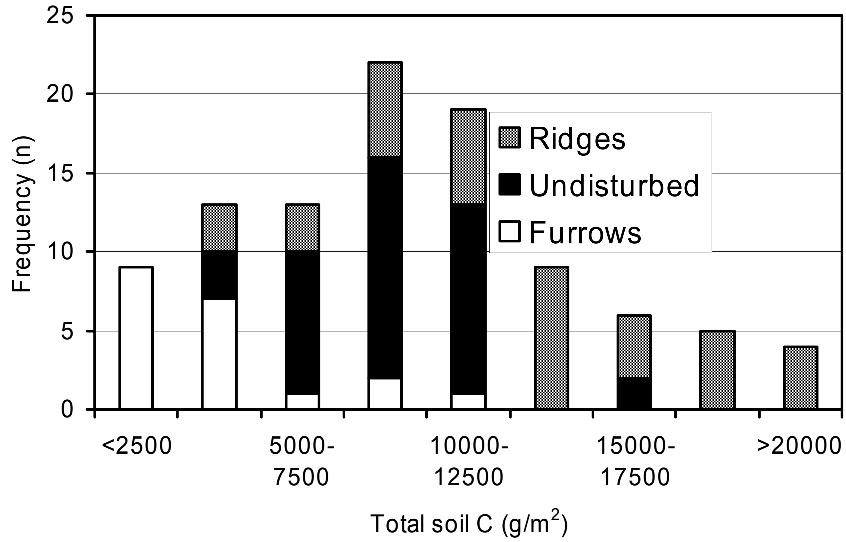


Figure 2.3.11.1: Distribution of total SOC in a Sitka spruce plantation, Griffin, Scotland

To obtain reliable estimates of total SOC, a large number of samples are required. In this case, 20 samples over the 100 ×100 m plot will allow the standard deviation of sampling to be within 15% of background carbon level. Fifty samples will secure a standard deviation below 10%. To achieve a sampling accuracy of 5% of background carbon level, 170 samples are needed (Figure 2.3.11.2). Sampling positions from furrow, ridge and undisturbed area will lead to different results for SOC content. However, if a large number of samples are taken, stratification seems have little influence on the relationship between standard deviation and sample size.

Conen *et al*. (2003) collected soil samples from a desertified steppe, and a dry steppe site where SOC was less spatially variable (200×200 m plot, 2 samples taken from each of 40×40 m subplot). In the desertified steppe site, the standard deviation (stratified) as a fraction of the mean was slightly smaller in the top 10 cm (3.4%) than at the 10–20 cm depth (3.8%). For the combined 0–20 cm depth interval, it was 2.9%. In the dry steppe site, the standard deviation (stratified) as a fraction of the mean decreased with increasing depth from 2.5% in the top 10 cm to 1.8% at 20–30cm depth. For combined samples (0–30 cm depth), the standard deviation was 1.7% of the mean. The differences in soil carbon concentrations between both plots were highly significant in the 0–10 and 10–20cm layers.

In an arable soil, Garten and Wullschleger (1999) showed that the smallest difference that could be detected was about 1 ton C per ha (2-3% of the background carbon level), and adequate statistical power (90% confidence) could only be achieved when using a very large sample size (>100). The minimum difference that could be detected with a reasonable sample size (16) and a good statistical power (90% confidence) was 5 ton C per ha (10-15% of background carbon level).

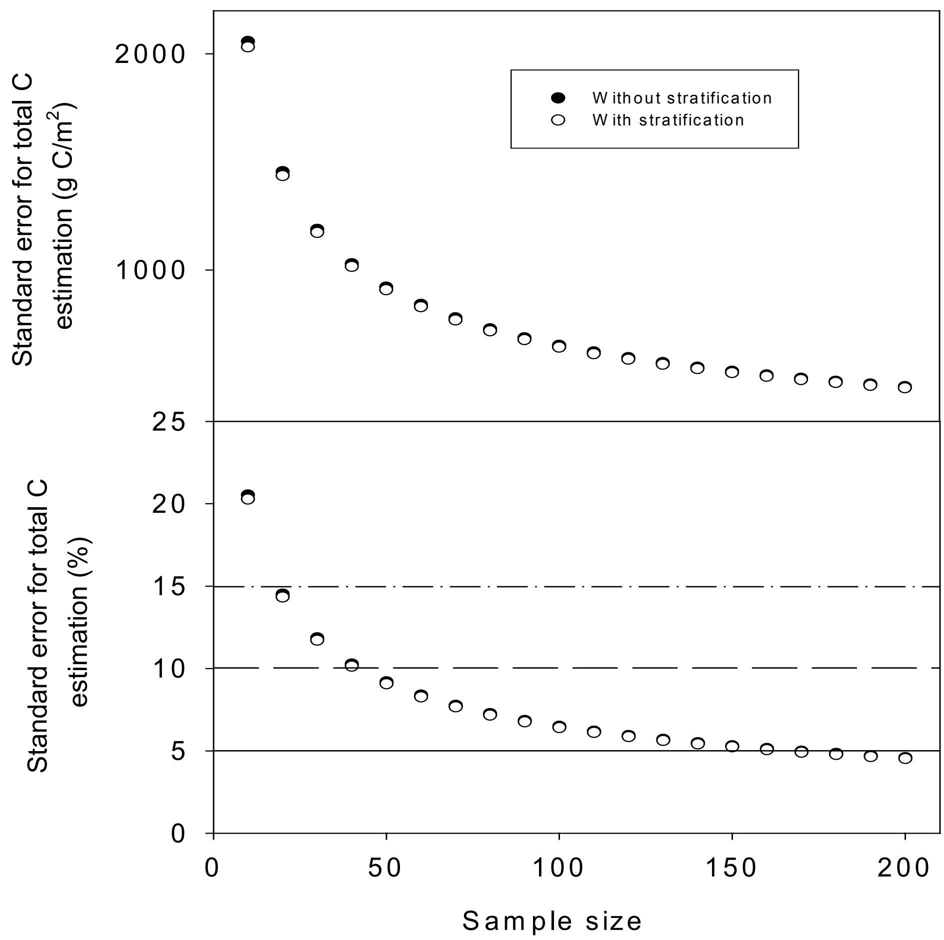


Figure 2.3.11.2: Accuracy of soil C sampling at Griffin depending on sample size

From the above studies, the ability to detect change in SOC by sampling at the plot scale is probably about 2% of background level at best, and not less than 7% of background total C at worst, even with a sample size over 100. If using the concentration/density of SOC, rather than the dry mass per unit ground area, to account for soil carbon, sampling precision may be significantly improved. However, there is no published data available to allow us to estimate how much the sampling precision will be improved. Detecting SOC change at scales of farm or land use type is of more importance for preparing C inventories and sustainable management than sampling at the plot scale. At a larger spatial scale, the spatial variability of SOC could be much greater than that suggested in the above studies, though much of the variability may occur at scales smaller than the field (K. Paustian, *Pers. Comm.*). There is insufficient data available to assess the ability of sampling techniques to detect the minimum change in SOC at these scales.

The minimum years required to detect changes in SOC

The number of years over which a change in SOC can be detected depends on the spatial variability of the soil, sampling method (mostly the number of samples) and the rate of systematic change in SOC. In Conen *et al*. (2003), the hypothesis that a 10 % reduction of current SOC will occur over the next 50 years due to climate change could be tested after about 43 years in desertified steppe site (with type I error probability $\alpha = 0.05$ and statistical power = 0.90). The same calculation for the more homogenous site of dry steppe (0–30cm depth interval) yields a time interval of 26 years. The time interval could be halved to 22 and 13 years, if sample sizes were increased to 200 at each plot. Attempting to detect significant changes in even shorter time would not be possible.

An model simulation showed that in FACE (and other) experiments where C inputs increase by a maximum of about 20–25%, SOC change could be detected with 90% confidence after about 6–10 years if a sampling regime allowing 3% change in background SOC level (probably requiring a very large number of samples) were used, but could not be detected at all if a sampling regime were used that allowed only a 15% change in background SOC to be detected (Smith, 2004a). If increases in C inputs are much below 15%, it might not be possible to detect a change in soil C without an enormous number of samples. Relationships between the change in C inputs and the time taken to measure a change in SOC were found to be robust over a range of soil types and land uses (Smith, 2004a).

Most agricultural practices will not cause soil carbon accrual rates as high as 1 ton C per ha (2-3% of the background carbon level, Garten and Wullschleger, 1999) during a 5-year commitment period (Smith *et al.*, 1997). This means that detecting a change in SOC over a short period of time will be extremely difficult, even if taking a very large (>100) number of samples (Smith, 2004b).

If measuring precision is 2% of the background C level (with a sample size > 100) and if SOC is changing at a rate of 0.5% per annum, the minimum number of years to detect the change in SOC (with a confidence of 90%) would be about 10 years. If the precision is 10% of background level, the minimum years would be 40-50 years. Falloon and Smith (2003) showed that sampling and analysis regimes for existing long-term experiments and monitoring sites need to be improved if they are to be used either to test models, or to demonstrate changes in soil carbon over short periods. The above estimates may be optimistic since they assume a change in SOC in one direction. If users are monitoring changes occurring due to less direct pressures, changes in SOC would require even more samples and a longer period to detect a change.

The natural variation in SOC

SOC can be driven by many natural variables, such as climate, NPP and changes in biodiversity. At present, only the climate forcing and change in NPP can be generalized and simulated by models. Figure 2.3.11.3 shows a plot derived from the simulation of SOC for European cropland soils with measured climate data over last 100 years (1900-2000). Climate-induced change (direct impacts in this case) over that period comprise two components: an overall loss in SOC and fluctuations in that loss between years, depending on the climate. Over the 20th Century, European cropland soils lost just over 5% SOC (Smith *et al.*, in prep.), while the standard error was less than 0.5% (with a variation range less than ±1.0%) against the trend. In some regions in Europe, climate alone can induce an increase in SOC by about 23% over that period (due to soil drying which slows decomposition), with a standard error of slightly more than 2% (variation range -3-6%) against the trend.

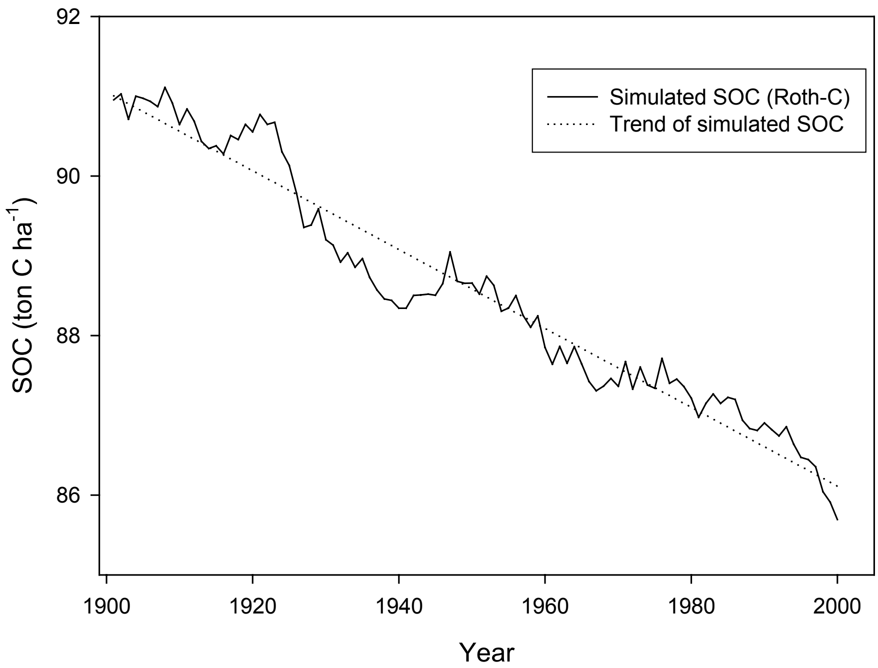


Figure 2.3.11.3: Simulated SOC of European arable soils driven by climate alone over the period of 1900-2000

SOC stocks are an average of all grid cells (16,807 cells at 10′ × 10′ resolution), simulated by the Roth-C model. The trend is a linear regression of simulated SOC (R^2=0.95)

Over short periods (10 years), climate-induced SOC change in the European cropland varied 0.2-0.8% (with a confidence at 90%). Over 20 and 50 years, climate-induced change was about 0.7-1.4% and 2.2-3.1%, respectively. For a selected grid cells, these figures could be as high as 0.3-4%, 2.0-7.4% and 9-15.4%, respectively.

The temporal variability of measured SOC in long-term experiments

The temporal variability in measured SOC is also important when attempting to detect an SOC change. This has received less attention in the literature than spatial variability in SOC. Measured SOC in existing long-term experiments shows a large temporal variation, against the trend of SOC (Figure 2.3.11.4). In the Muencheberg long-term fertilizer experiment site, for example, the control soil lost about 26% of its original C over the

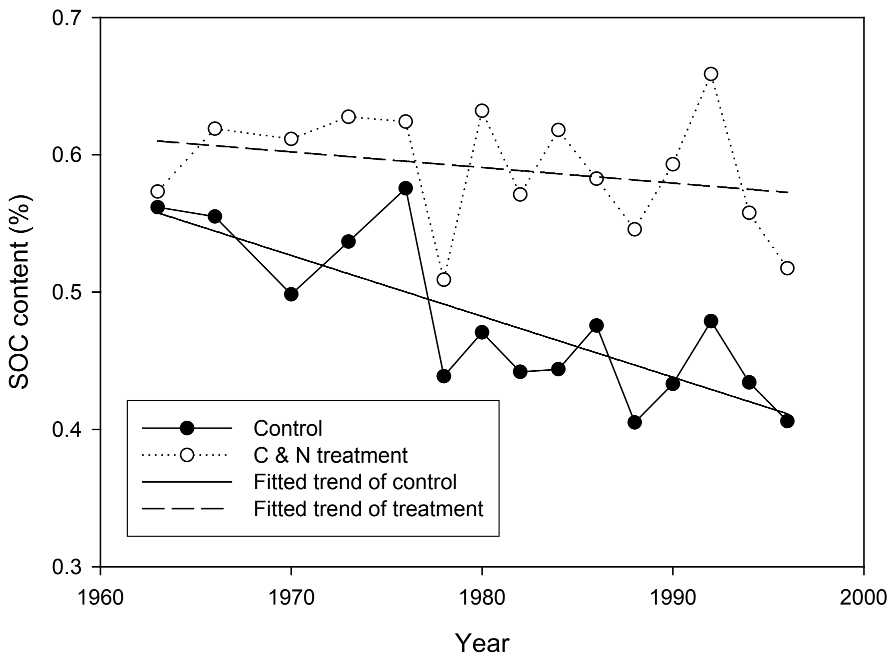


Figure 2.3.11.4: Measured SOC content (0-25 cm depth) in a long-term fertilizer experiment site in Muencheberg, Germany. Plotted values were averages of 10-15 samples (data from SOMNET database, Smith *et al.*, 2002)

period of experiment (33 years), caused by both natural factors (such as climate change) and cultivation. The standard error against the fitted trend in SOC change is 7% of the background C level. For the fertilizer treatment, soil lost only around 6% of its original C, indicating that treatment has increased SOC by 20% of the original SOC. The standard error was at the same level as control (7% of background C). At a confidence of 90%, the lost of SOC was significant in control, but far from significant in treatment.

There is no doubt that a part of the temporal variation in measured SOC could have been caused by the methodology used (for example, inadequate soil sampling or a change in the method of SOC analysis). In most existing long-term experiments, SOC is determined from less than 20 samples (Smith, 2004a; 10-15 samples in the above Muencheberg site). This apparent temporal variation may be significantly reduced by a better sampling protocol in future. The studies described above by Conen *et al.* (2003; 2004) and Garten and Wullschleger (1999) indicated that measured standard deviation can be reduced from 15% with 20 samples to about 7% with 100 samples in variable sites, or from 15% with 16 samples to 2-3% with 100 samples in less variable sites. For a rough estimate, the above temporal variation in measured SOC could be reduced to about 3% to 1% of the background C level if using 100 soil samples. The temporal variability of SOC needs to be taken into account when attempting to detect a change in SOC.

Distinguishing management-induced SOC change from natural variation

Management-induced SOC change depends on the intensity of the management. Most agricultural practices may not cause soil carbon accrual rates as high as 0.5% of initial soil C per annum (Smith *et al*. 1997). As simulated in Figure 2.3.11.5, an increase of 0.5% per annum in SOC can be achieved by an increase of about 40% in C input. Such changes are feasible but do not occur commonly or without incentives to do so (Smith et al., 2004). We assume a change in SOC of 0.5% per annum induced by management to estimate how many years is required to detect this change.

Based on the above studies, we assume that the best achievable sampling precision with respect to the spatial variability of SOC is 3% of the background C level (≥100 samples), where noise due to natural variation is 5% of the background C level (single site over 50 years, climate forcing change only and climate-driven trend removed by modelling), while the temporal noise in measured SOC is 2% of background level (increasing sample size to 100, a part of climate-induced noise may be included in measured SOC). The error in detecting a management-induced SOC change depends on all three errors discussed above. Estimating the combined precision involves a complex statistical calculation but as a rough estimate, the combined precision should be worse than the largest single error but better than the sum of all errors, that is, the combined precision is between 5 and 10%.

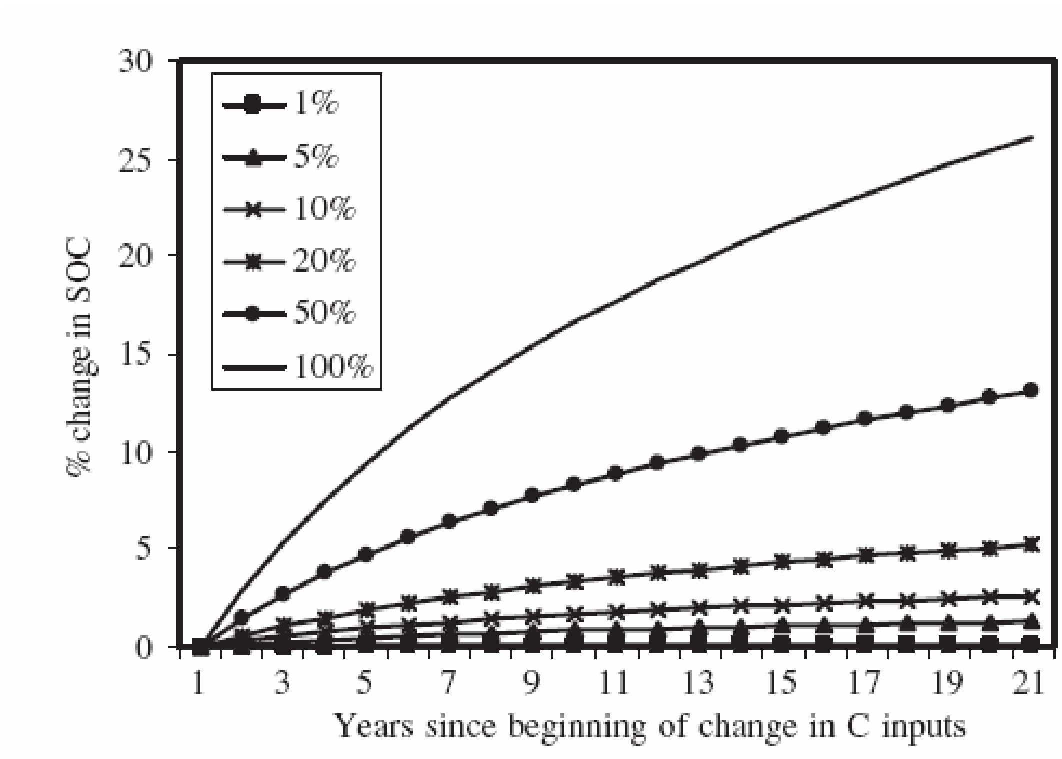


Figure 2.3.11.5: The change in soil organic carbon (SOC) over a 20-year period under different changes in carbon input ranging from a 1% to 100% increase for the reference scenario (23.4% clay, equilibrium SOC content of 50 t C per ha and a spring barley crop) (Smith, 2004a)

For a confidence level of 90%, the minimum years required for detecting a change of 0.5% per annum induced by agricultural management is between 20 and 40 years. In other words, if we check SOC over a period of 20 years, the management-induced rate of change in SOC would need to be 0.5-1.0% per annum to be detectable. Over a period of 5 or 10 years, management-induced change in SOC is unlikely to be distinguishable from natural variation with adequate confidence.

These rough estimates could be improved by more complex analyses, but serve to give an idea of the magnitude of difficulty in measuring SOC change, even under favourable management. At best, 20 years will be required to detect a management-induced trend in SOC against the natural variation, which is much longer than that estimated based purely on the spatial variability of SOC. The minimum detectable change in SOC will be affected by soil type and land use. The relationship between minimum detectable SOC change and clay content is not yet clear and requires further work. If the SOC change is determined by the percentage of background C level, clay content should not significantly affect the minimum detectable change in SOC, unless clay content influences sampling precision.

Improving precision when attempting to detect a change in SOC

If the purpose of monitoring SOC is only to detect the temporal change rather than the accurate SOC stocks, the number of samples required may be significantly smaller than 100 samples suggested above. Within a plot, SOC content is spatially variable, so a large number of samples have to be collected to provide an acceptable precision. However, the temporal variation is relatively small compared with the spatial variation and the relationship between sampling points is relatively consistent over time (i.e. a sampling point with a high content of SOC at time 1 is likely to have a high content at time 2). With a clear understanding of relationships between the spatial variation, temporal variation and samples size, the number of samples for re-sampling may be significantly reduced. At present, experiments taking a very large number of samples (such as 100 samples or more per plot) only started few years ago and the second-time sampling should be scheduled for several years after the beginning of the experiment.

Multiple measurements over a short period (such as five years) will provide much better determination of SOC than a two-point measurement over a longer period (such as 20, 25 or even 50 years). Despite the fact that a minimum of 20-40 years are required to detect a change in SOC induced by agricultural management, regular re-sampling during a short period will help to improve the measurement precision, and the measured trend of SOC will provide information as an early indicator. With a two-point measurement over 20 years or longer period, it may be possible to detect whether management-induced SOC changes are significant or not, but it is difficult to quantify these changes. Regular sampling over a shorter period will narrow the range of errors and provide more quantitative information. Measured trends in SOC can be combined with model simulations to estimate whether, or after how long, the soil will breach a critical value of SOC.

Models and experiments can be combined to estimate natural variation and management-induced SOC change. For European soil, when simulated with projected climate change, SOC is predicted, on average, to remain unchanged in the first 10 years of the 21st century. In the next 10 years, climate-induced SOC loss will be around 1%, and will be up to 4% by the year 2050. For a specific site or soil, climate-induced SOC change should be much bigger. If comparing soils across sites at a large scale, climate-driven change in SOC may be in opposite directions. After a model has been calibrated with site-specific information, we should be able to simulate climate-induced change in the site or other natural forcing with future development in modelling. Using long-term datasets and modelling will help us to understand SOC change in the past and to estimate the possible trend in future. Running models under different scenarios will help to estimate relative contributions from different causes and explain underlying mechanisms. This will help us to assess management-induced change in soil C stocks, and will be useful for setting the envelope of normality for different soil groups and considering possible management options to mitigate unfavourable changes in soil quality. Further work is needed but the approach clearly has potential (Smith, 2005).

Defining criteria for identification of a minimum dataset of SOC

Sparling *et al*. (2003) tested three approaches to define maximum and lowest desirable soil C contents for four New Zealand soil orders. Approach 1 used the New Zealand National Soils Database (NSD). The maximum C content was defined as the median value of long-term pastures, and the lower quartile was used to define the lowest desirable soil C content. Approach 2 used the CENTURY model to predict maximum C contents of long-term pasture and targeted C contents were set at 80 % of maximum C contents. Lowest desirable content was defined by the level that still allowed recovery to the targeted C content over 25 years after the land was converted from cropland to long-term pasture. Approach three used an expert panel to define desirable C contents based on production and environmental criteria. Upper and lower limits of C content by approaches 1 and 2 were similar. Expert opinion was that C contents could be depleted below these values with tolerable effects on production, but less so for the environment.

Defining the lowest desirable C content as the level that allows recovery to 80% of maximum C content over 25 years may be a useful criterion for New Zealand soils, but may be difficult to use in practice in UK soils or in other countries. In New Zealand, long-term pasture, which was used as a reference in Sparling *et al*. (2003), is a major type of land use. In other countries, most agricultural soils may have been cultivated for shorter periods and often have achieved equilibrium under arable conditions. Uncultivated soils may have different environmental conditions from cultivated soils. Natural soils may not be a good reference in these countries. Secondly, there are several agricultural management practices to allow soil C recovery; the recovery speed depends on which practice is used and the intensity with which it is implemented. As simulated above (Smith, 2004a), increasing C input by 50% will allow SOC to be increased by more than 12% over 20 years, but increasing C input by 20% will increase SOC by only about 5%. If defining the lower limit of SOC as the level that can be recovered to 80% over 25 years, the lower limit will vary with chosen management. Thirdly, since all arable soils cannot be converted to long-term pasture (or forest in other cases), most arable soils will still be used for growing food or other products in future. The hypothetical land use change that might allow SOC to recover should be regarded as optimistic because although soils could recover, the management practice/land use to enable them to recover may not be feasible. A criterion defining the lower limit of SOC, based on the presumption of converting to long-term pasture (or other natural soils), will not provide a practical guide for soil management since it may not be possible to implement. The threshold might be set higher to account for this.

The so-called envelope of normality is determined by the lower limit of soil C content since the upper limits of SOC do not present a problem. It is difficult to decide criteria defining the lower limit of soil C content (see above – we do not regard the limits of Sparling *et al.*, 2003, as satisfactory). Further substantive research work is needed, beyond the scope of this report. Before new methods or limits are defined, further consultation of experts will be required on the following points:

- Using the national soil database to determine the distribution of soil C content for each soil group or land use/management category. The sample size of each soil group in the UK national soil database (as reported in Bellamy and Kirk, 2005) should be sufficient to draw a distribution curve.
- Based on the distribution curve of SOC for each group/category, the lower limit of SOC can be decided by experts by considering ecology, techniques and economics. For example, the lower limit of SOC can be set as the 5% (or 10%) of a soil group/category which has the lowest SOC content, is under threat and is in need of changing management in order to improve soil quality. The critical value of SOC (trigger value) which should not be breached can be determined from the distribution curve.
- Criteria should be regularly reviewed (for example, every 10 or 20 years) by experts to take into account advances in ecology, technology and economic development.

We believe that this method would be more appropriate than those suggested by Sparling *et al*. (2003). The lower limits of SOC content determined here are based on a more comprehensive set of considerations, not just soil quality. Criteria used in approach 1 and 2 in Sparling *et al*. (2003) can be linked with those defined in this method. Approach 1 of Sparling *et al*. (2003) left 12.5-25% of soils below the lower limit and in need of management change. The wide range of estimates of soils below the lower limit (12.5-25%) across soils/categories will also lead to large uncertainties when attempting to use this criterion (the percentage of soils under threat varies with soil group/category and is significantly affected by SOC datasets). After model calibration, the criteria outlined in approach 2 of Sparling *et al*. (2003) can be marked in the distribution curves of SOC for each soil group. The expert approach in Sparling *et al*. (2003) is somewhat qualitative, rather than quantitative, and might be difficult to operate.

'Plus or minus 100% around the median may not be a good criterion. If the SOC content of a group of soils scatters over a wide range, ±10% around the median will leave a large part of soils breaching the lower limit, though they may not be under threat. We suggest that a detailed study be undertaken. Different criteria (particularly approach 2 of Sparling *et al*. (2003) which can be run with various scenarios) can be marked on the distribution curve of SOC for soil groups and landuse/management categories for deciding a suitable criterion.

At present, several national databases are available for considering the criterion of desirable SOC content with the method outlined as above for MDS. For example, the

National Soil Inventory (NSI) database includes soil information from around 6,000 sites across UK, where soils were sampled in both 1978-1983 and 1994-2003. The Representative Soil Survey Scheme (RSSS) currently surveys about sixty farms at 5 year intervals since 1969 (1969-2002). Detailed information about soil properties from RSSS will be useful to help set criteria of desirable SOC content and to link SOC content to other soil properties, such as N concentration, which are recognized to be important components of soil quality. UK national wide simulation of SOC in the 20^{th} and 21^{st} century by Smith et al. (2004, 2005) provides long-term SOC changes of UK soils in the past and the future in the context of climate and environment change. By integrating information from these resources together, it should be feasible for us to set up provisional SOC criteria for UK soils, though further work is required. After a wide discussion by soil scientists or experts in relevant areas, SOC criteria for MDS can be achieved for UK soils.

In summary, soil C may form a useful part of a minimum dataset to detect changes in soil quality, but because of small changes relative to large background stocks, and high spatial heterogeneity, changes may not be detectable for many (>20) years and even then only with very large sampling campaigns. Simulation modelling may help to define trends in SOC change, and could be use to attribute various causes for SOC change (such as management vs. climate), but models must be soundly tested against benchmark data (from intensively sampled, tier three benchmark sites) and should be well documented and archived. The suggested criteria for use of SOC as a soil quality indicator will likely be feasible for UK soils but requires further detailed study.

References

Bellamy, P. and Kirk, G. (2005) *Changes in organic carbon in the soil across England and Wales from 1978-2003.* NSRI Internal Report.

Conen, F., Yakutin, M. V., and Sambuu, A. D. (2003) Potential for detecting changes in soil organic carbon concentrations resulting from climate change. *Global Change Biology*, 9, 1515-1520.

Conen F, Zerva A, Arrouays D *et al.* (2004) The carbon balance of forest soils: detectability of changes in soil carbon stocks in temperate and boreal forests. In: *The Carbon Balance of Forest Biomes*. eds Griffith H, Jarvis PG, Bios Scientific Press, London (in press).

Falloon, P. and Smith, P. (2003) Accounting for changes in soil carbon under the Kyoto Protocol: the need for improved long-term datasets to reduce uncertainty in model projections. *Soil Use and Management*, 19, 265-269.

Garten, C.T. and Wullschleger, S.D. (1999) Soil carbon inventories under a bioenergy crop (Switchgrass): measurement limitations. *Journal of Environmental Quality*, 28, 1359-1365.

Guo, L.B. and Gifford, R.M. (2002) Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, 8, 345-360.

Gifford, R.M. & Roderick, M. L., 2003, Soil carbon stocks and bulk density: spatial or cumulative mass coordinates as a basis of expression? *Global Change Biology*, 9, 1507-1514.

IPCC (1997) Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual (Vol. 3). Eds by Houghton, J.T. et al., *Intergovernmental Panel on Climate Change*. Hadley Centre, Bracknell, UK.

Jenkinson, D.S. and Johnston, A.E. (1977) *Soil organic matter in the Hoosfield continuous barley experiment.* Rothamsted Experimental Station, Report for 1976, Part 2, pp. 87–101.

Ogle, S. M., Breidt, F. J., Eve, M. D. and Paustian, K. (2003) Uncertainty in estimating land use and management impacts on soil organic carbon storage for US agricultural lands between 1982 and 1997. *Global Change Biology*, 9, 1521–1542.

Sisti, C.P.J., dos Santos, H. P., Kohhann, R., Alves, B.J.R., Urquiaga S. and Boddey, R.M. (2004) Change in carbon and nitrogen stocks in soil under 13 years of conventional or zero tillage in southern Brazil. *Soil and Tillage Research*, 76 , 39–58.

Smith, P., Powlson, D.S., Glendining, M.J. and Smith, J.U. (1997) Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. *Global Change Biology*, 3, 67-79.

Smith, P., Powlson, D.S., Glendining, M.J. and Smith, J.U. (1998) Preliminary estimates of the potential for carbon mitigation in European soils through no-till farming. *Global Change Biology*, 4, 679-685.

Smith, P., Powlson, D.S., Smith, J.U., Falloon, P.D. and Coleman, K. (2000) Meeting Europe's Climate Change Commitments: Quantitative Estimates of the Potential for Carbon Mitigation by agriculture. *Global Change Biology*, 6, 525-539.

Smith, P., Falloon, P.D., Korschens, M. et al., (2002) EuroSOMNET – a European database of long-term experiments on soil organic matter: the WWW meta-database. *Journal of Agricultural Science*, Cambridge, 138, 123–134.

Smith, P. and Powlson, D.S. (2003) Sustainability of soil management practices – a global perspective. In: *Soil Biological Fertility - A Key To Sustainable Land Use In Agriculture.* ed. L.K. Abbott and D.V. Murphy, Kluwer Academic Publishers, Dordrecht, Netherlands: 241-254.

Smith, P. (2004a) How long before a change in soil organic carbon can be detected? *Global Change Biology*, 10, 1878-1883.

Smith, P. (2004b) Monitoring and verification of soil carbon changes under Article 3.4 of the Kyoto Protocol. *Soil Use and Management*, 20, 264-270.

Smith et al. (2004) *CarboEurope GHG: Greenhouse gas emissions from European croplands.* CarboEurope GHG, University of Tuscia, Viterbo, Italy, 68pp.

Smith, P. 2005. An overview of the permanence of soil organic carbon stocks: influence of direct human-induced, indirect and natural effects. *European Journal of Soil Science* (in press).

Smith, P., Smith, J.U., Zaehle, S., Wattenbach, M., Fang, C., Heiderer, R., Jones, R.A., Montanarella, L., Rounsvell, M., Reginster, I. and Ewert, F. (in prep.) Annual soil carbon fluxes in European croplands during the 20th Century. To be submitted to *Global Biogeochemical Cycles*, Spring 2005.

Sparling, G., Parfitt, R. L., Hewitt, A. E. and Schipper, L. A. (2003) Three approaches to define desired soil organic matter contents. *Journal of Environment Quality*, 32, 760-766.

2.4 Peer review

The reports shown in the previous section were sent to our selected expert group, several weeks prior to a technical peer review meeting on the 2-3rd of February in Bath. Accompanying the reports was a delegate pack that outlined the processes undertaken up to that date, the form of the challenge processes and methodology of SQI uses (i.e. the Tiered Approach). Further, a list of working definitions was provided to the experts to give context and confine debate to meet the aims of the meeting. Those aims were to:

- review each potential indicator, agree recommendations for use and justify the decision (based on evidence);
- agree proposals for trigger values for each indicator;
- agree proposals for a suitable network for soil monitoring in the UK;
- inform the experts of the Environment Agency's negotiating position on this project with regard to the consortium and Defra

Experts' comments were collated before the meeting and formed the basis for discussion of each of the potential indicators. The experts and report authors were then asked the following questions regarding each indicator:

1. Can we interpret change in this indicator?
2. What does it mean if change occurs and how might this influence other soil properties?
3. What is the indicator's relevance to the soil function of environmental Interaction?
4. Can we integrate it with another indicator?
5. Other factors to be considered – measurability, cost, ease of sampling, predictability, signal-to-noise ratio?

The experts were asked to provide evidence to corroborate their views. During the meeting the group referred to Figure 2.4 below, particularly with regard to question three. If, for example, a potential indicator seemed appropriate for interpretability but was not relevant to any of the components in Figure 2.4, then it was deemed unlikely to be useful.
A brief summary of the meeting is given in Appendix I. The reports were then amended, as given in sections 2.3.1 to 2.3.11, and passed to consortium members for comment. Following the incorporation of their views, the first section of the challenge process was completed.

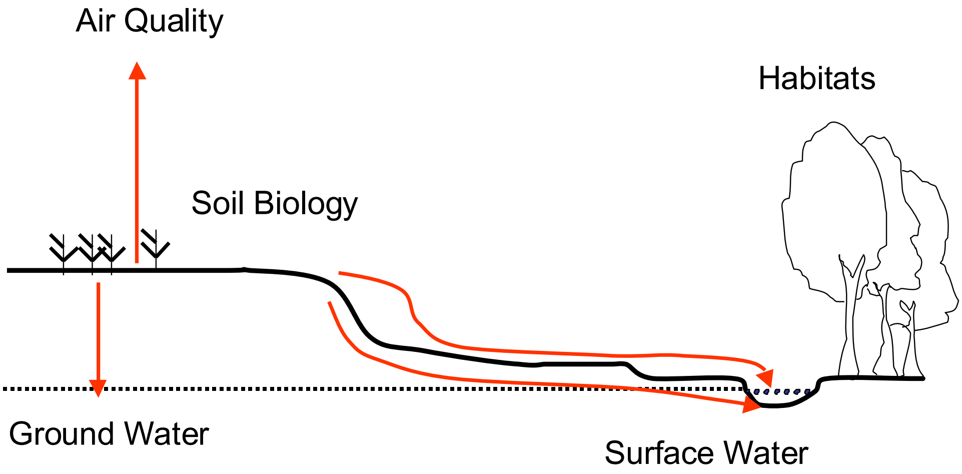


Figure 2.4: Key components of the soil's interaction with the environment

In Figure 2.5, the indicators selected from the peer review meeting are listed under each of the soil functions to which they are most relevant.

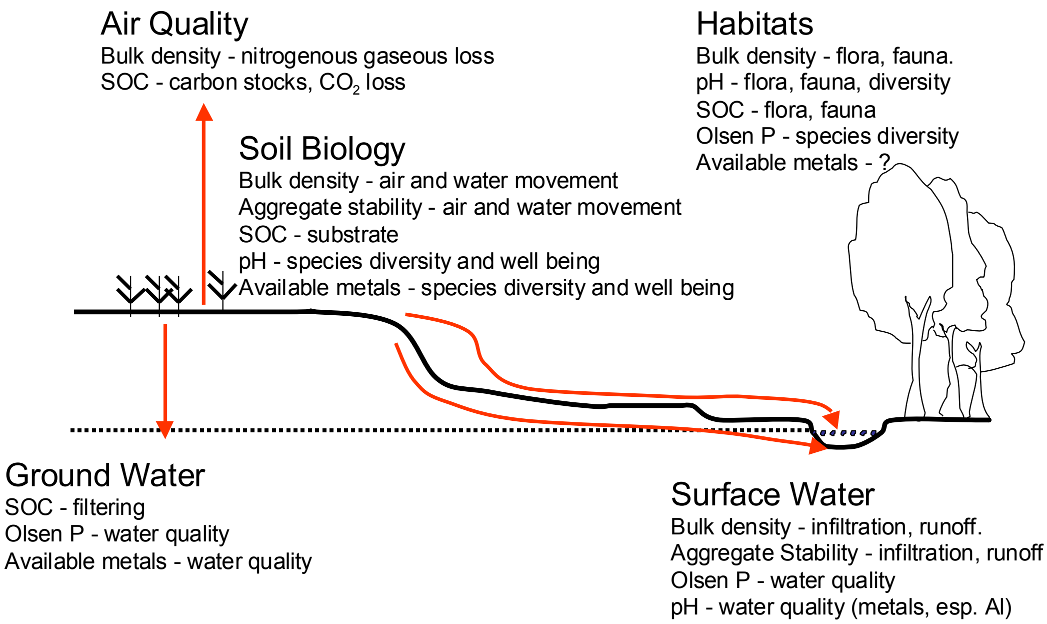


Figure 2.5: Key components of the soil's interaction with the environment and related potential indicators

2.5 Technical workshop

The final stage in the challenge process was a technical workshop held on the 28-29 June 2005. The aim of the workshop was to reach a consensus on appropriate SQIs for the soil function of environmental interaction, which could then be presented to Defra and the Scottish Executive. The goals of the workshop were to:

- reach an agreement that the overall process followed was appropriate;
- confirm that the correct indicators had been selected
- confirm that the correct minimum dataset had been selected;
- identify appropriate workable ranges for the indicators for the road test phase of the work (expected to start in late 2006).

Breakout groups and a professional facilitator were used in this workshop (more information is given in Appendix II). Each breakout group was chaired by a staff member of the Environment Agency, and each group was charged with tackling the following questions:

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures, such as atmospheric deposition, climate change or agricultural or industrial practice?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Is the parameter you are discussing a relevant environmental indicator, and can you justify your conclusion?

A key difference between the peer review meeting and the technical workshop was the use of extrapolated datasets, in particular to answer question 4. Indicators were expected to be interpretable over 50 to 100 years based on data generated for this time period. The data was extrapolated with the use of existing temporal datasets, models (such as Roth C for soil organic carbon) and literature sources. For each indicator, data for at least two scenarios of change were generated.

For example, for soil organic carbon, Pete Smith and Michael Haft of the University of Aberdeen used four scenarios based on the long-term experiments at Rothamsted, against which the model has been tuned. These scenarios were:

1. No land use or climate change based on the long-term dataset from Park Grass.
2. No land use change (Park Grass) but with a climate change driver based on the Med-High scenario of the UK Climate Impact Panel, UKCIP (Appendix III).
3. Land use change: arable to woodland regeneration, no climate change, based on the Geescroft Wilderness data.

4. Land use change with climate change: arable to woodland regeneration, climate change included based on the Med-High scenario of UKCIP.

For the purposes of the workshop, the model outputs were taken to represent the actual changes in SOC that would result from changes in either land use or climate. All of the potential indicators selected from the peer review meeting had accompanying long-term datasets; the delegate note and data are given in Appendix III. Each note also outlines critical model and parameter assumptions, likely spatial variability, data sources and scenarios, but does not specifically link the data to the scenarios. The breakout groups were tasked with interpreting these datasets and correctly identifying the scenarios. A critical test of whether the indicator was robust for use in a UK soil monitoring network was whether the assembled experts were able to interpret the data on the basis on the soil function of environmental interaction.

Prior to the meeting the invitees were sent a delegate pack, including the reports (Sections 2.3.1-2.3.10), an explanation of the process and a set of ground rules. A list of the invitees, an agenda and complete delegate pack is provided in Appendix II. The invitees were selected on the basis of experience in soil science and UK soil conditions as well as understanding of monitoring and SQI programmes. Invitees were randomly assigned to breakout groups and remained in these groups over the period of the workshop. Each group was chaired by a member of the Environment Agency and had a member of the UK Soil Indicators Consortium as Reporter.

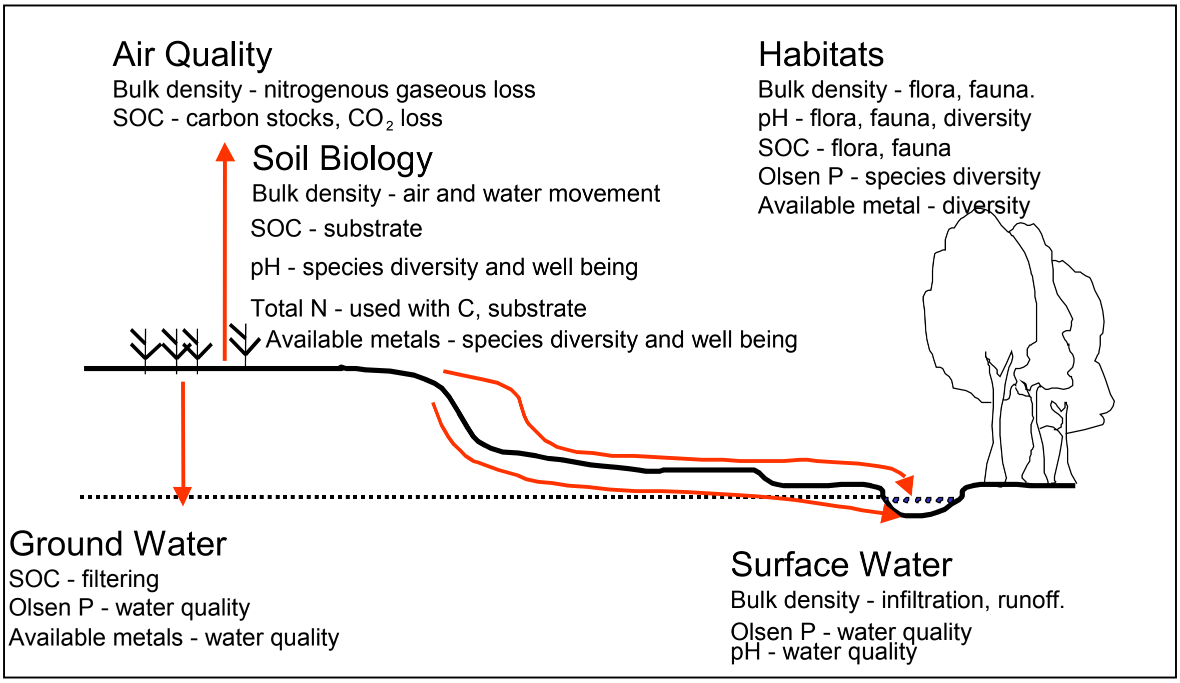


Figure 2.6: Key components of the soil function of environmental interaction and selected potential indicators - modified

Summaries of the deliberations and decisions of each of the breakout groups are also given in Appendix II. Included in these are recommendations of indicators and needs for further work. The recommended indicators are shown in the key component diagram (Figure 2.6).

3 Discussion

3.1 Indicators

The soil quality indicators selected for tier one, as a minimum dataset, for the soil function of environmental interaction are:

- Bulk density
- Total nitrogen
- Total and adjusted copper (Cu), nickel (Ni) and zinc (Zn)
- Olsen P
- Soil organic carbon content
- Soil pH in water

The measures are to be performed on topsoils, apart from bulk density which is likely to be determined at a range of depths depending on the contextual information known about the location being sampled. The contextual information gathered about the sampling location is crucial with regard to interpretation and understanding of the indicator; it would include such information as soil type, land use, land use history and management, climatic conditions and known potential risks with regard to the function of environmental interaction. The other indicators within the minimum dataset would also add to the weight of evidence with regard to single indicator interpretation.

3.2 Trigger values for the indicators

From the peer review by selected experts and workshop summaries from the technical workshop, trigger values have been derived for some of the indicators in the minimum dataset. It is important to stress that such trigger values are for use within the tiered assessment and are therefore relatively conservative. Further, the values of the triggers are to be viewed in the context of interpretative information also collected at the site on soil type, land use and so on. The triggers are to be used to build up ‘weight of evidence’ to determine if there is a potential issue with the quality of the soil in relation to the function of environmental interaction.

Bulk Density

The trigger values for bulk density were initially to be developed on the basis of Table 3.1, as it was thought this would cover key land uses and depths. However, from a review of available data from the Soil Survey and Land Research soil physics database of the mid-1980s by Malcolm Reeve, it was clear that it was not possible to cover all these land uses. Further, as discussed at the workshop, there was a strong view that the triggers should be developed for broad textural classes, taking account of sand, silt and clay content. However, following analysis of this data ($n \approx 5{,}000$) and an examination of the strength of the relationships, it was established that many of the limits overlapped textural classes and that the driving factor explaining the variance in bulk density is SOC (Figure 3.1)

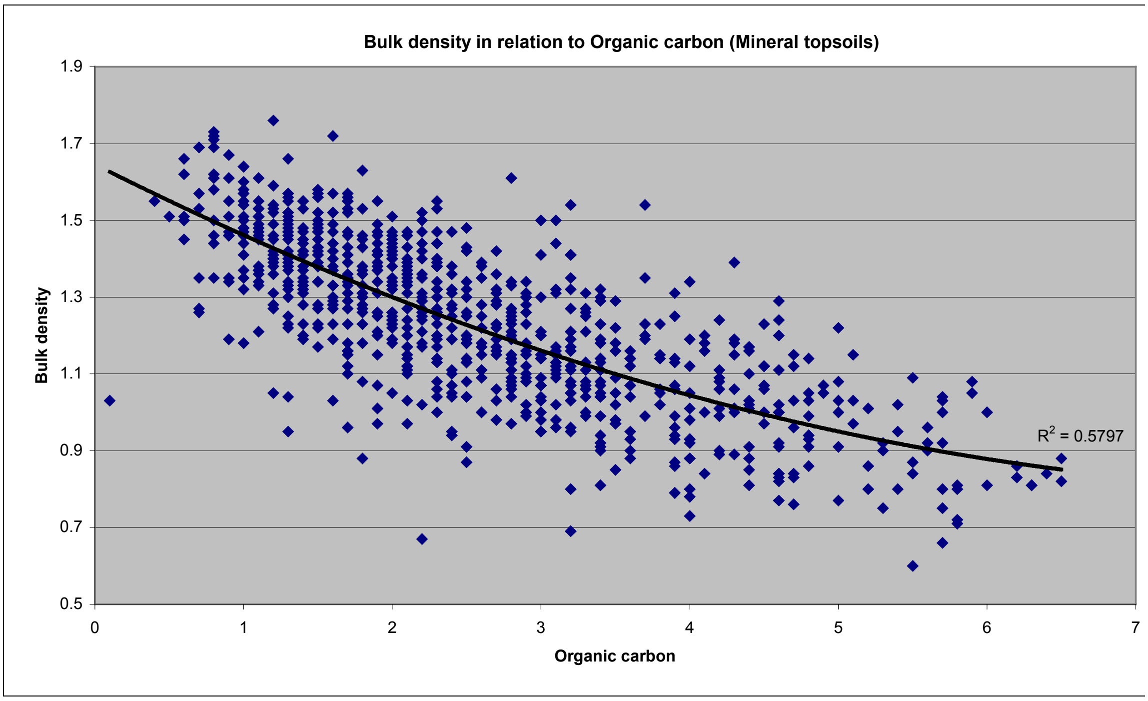


Figure 3.1: Relationship between SOC (%) and bulk density (g cm^{-3}) for mineral topsoils in the UK (Malcolm Reeve, *pers. comm.*).

Table 3.1: The potential categories for which bulk density triggers could be developed.

Land use	Depth of measurement
Arable	Below the cultivated layer (upper subsoil)
	Topsoil
Grassland	Topsoil
Woodland	Deep subsoil (relating to harvesting)
	Topsoil
Urban	?

Therefore, Malcolm Reeve has derived trigger valuesfor both tilled and untilled top and subsoils related to soil organic matter (Tables 3.2 and 3.3). Tilled top soils are largely arable and ley, whereas untilled was predominantely permanent pasture and rough grazing. These values have been calculated from the Soil Survey data. Further investigation would be triggered when measured values exceed these levels. Further, it was strongly recommended that sampling of bulk density, as with many other of the indicators was to be carried out in spring.

Table 3.2: Bulk density (Mg/m^3) trigger values for topsoils with different organic matter content for the UK.

Organic matter content (%)*		
	Tilled land	*Untilled land*
Mineral soils		
<2	>1.60	>1.50
2-3	>1.50	>1.40
3-4	>1.40	>1.35
4-5	>1.30	>1.25
5-6	>1.25	>1.20
6-8	>1.20	>1.15
Organic mineral soils	>1.00	

* if measured as organic carbon, multiply by 1.72 to obtain organic matter content

Table 3.3: Bulk density *(*Mg/m^3*)* trigger values for subsoils with different organic matter content for the UK.

Parameter	
Clay >50%	>1.35
Clay <50%	>1.50*
Peats	>0.50

Total N

For total N, as was highlighted in the summary sheet from the workshop, the merit of its measurement is in the combination of C with N in a C/N ratio. This would be particularly important in soils for which the natural C/N falls outside the usual ratio of 9-12. Table 3.4 gives C/N ratios based on broad habitats from the Joint Nature Conservation Committee; the values are calculated from 1,200 soil samples taken from 0-15 cm depths from CS2000 and produced with kind permission of Helaina Black. These values can be considered as 'normal' ranges, further investigation would be undertaken when the calculated values of C/N ratio from the field fall out of these ranges for those specific habitats.

Table 3.4: Trigger values for the C/N ratio for a range of habitats

Habitat	$25\text{-}75^{th}$ percentile range
Calcareous grassland	11-14
Neutral grassland	10-14
Broadleaf woodland	12-17
Coniferous woodland	16-26
Improved grassland	10-12
Acid grassland	14-21
Arable and horticultural	9-13
Bog	20-31
Dwarf shrub heath	19-29
Bracken	13-18

Soil concentrations of Cu, Ni and Zn

Trigger values for soil concentrations of Cu and Zn have been determined using information from the Existing Substances Regulations (Directive 98/8/EC) programme. While it was acknowledged in the workshop summary for metals that the trigger values could be informed by the current sewage sludge regulations (MAFF, 1993), but questions with regard to the protective nature of the sludge values for sensitive habitats and/or biological systems remain (Dahlin *et al*., 1997; Jarausch-Wehrheim *et al*., 1999; McBride *et al*., 1999; Bhogal *et al*., 2003). Therefore, the triggers being recommended are predicted no-effect concentrations (PNECs) based on soil ecotoxicological data and are relevant for the soil function of environmental interaction. Further information on the methods used can be obtained from the Technical Guidance Document found at ECB (2003). In particular, the risk assessment programmes on Zn (The Netherlands Risk Assessment Report: Zinc metal, 2004) and Cu (Cu- Risk Assessment Report, 2005) have provided significant advances in understanding in metal behaviour in soils and importantly, how this information can be used to set trigger values in regulation. The methodology and example presented below is from Merrington *et al*. (2006) and illustrates the processes for incorporating site-specific soil factors (such as pH or background metal concentrations). This provides an ecologically relevant assessment of potential risks posed by concentrations of Cu and Zn in soils. However, the method has its detractors, but has largely been adopted by the Environment Agency as a technical approach, in-line with broader UK agreements from other EU regulatory fora, and arguably represents some of the best of the current thinking on metal behaviour in soils.

This methodology also allows for a single trigger value to be set across the UK, as the manipulations identified in the example are to the <u>sampled site soil</u>, not to the trigger value. This provides consistency of application, promotes understanding across interested groups and makes use of site-specific soil properties.

Step 1 in this process is a comparison of a measured soil field concentration with the trigger value ($PNEC_{soil}$). Step 2 is the adjustment for the background metal concentration. For this step a tiered approach for the use of background soil metal concentration is being recommended. The first tier, as suggested in the example below, is to use a 10th percentile value of soil background concentration as taken from the UK National Soil Inventory and in line with EU Technical Guidance Document generic refinement (ECB, 2003). This is to avoid excluding sites that have both a low ambient concentration and higher local contamination levels (in other words, false negatives). The second tier is currently under development and will attempt to use relationships developed between soil structural components (such as Fe and Mn) and trace metal concentrations as outlined by Hamon and McLaughlin (2004). The relationship between soil Fe and background trace metal copncentrations can be especially marked. For example, the correlation between soil Fe and Zn from the UK National Soil Inventory is 0.61 and Ni and Fe 0.80 (n >5,000), without the removal of outliers (Steve McGrath *Pers. Comm*.). This offers the potential to differentiate background metal concentrations from anthropogenically enriched concentrations on the basis of their relationship with soil structural components, such as Fe. The final tier would be to use a site-specific background concentration that has been measured at or near the site and established through the use of local knowledge of the soil, site conditions and land use. Such a background should represent relatively pristine ambient conditions.

Step 3 adjusts for the difference in metal toxicity between the laboratory and field situations. In many cases no toxicity is observed in the field at very high metal concentrations - the levels are often 1-2 orders of magnitude higher than the PNEC in the laboratory.

The final step is a refinement that effectively enables a comparison of the tigger value with the result that would have been obtained if toxicity tests were carried out in the soil at a monitoring site

The Ni trigger value will be based on the Ni Risk Assessment Report (Directive 98/8/EC) at the end of 2006.

The examples given in the box below are for Cu and Zn and are hypothetical, in that they use a fictitious soil sample, but utilise data and technical information given in the risk assessment reports (The Netherlands Risk Assessment Report: Zinc metal, 2004; Cu- Risk Assessment Report, 2005) (Merrington *et al*., 2006).

	Zinc	Copper
Trigger value §	The trigger value ($PNEC_{soil}$) for zinc is 26 mg/kg dry weight. The value was derived using species sensitivity distribution method.	The trigger value ($PNEC_{soil}$) for Cu is 63 mg/kg$^{\infty}$ dry weight. The value was derived using species sensitivity distribution method.
Site soil level	The hypothetical site has been sampled and soil analysed for total Zn using *aqua regia*. The concentration of Zn in the soil is 200 mg/kg and represents the C_{total}.	The hypothetical site has been sampled and soil analysed for total Cu using *aqua regia*. The concentration of Cu in the soil is 150 mg/kg and represents the C_{total}.
Step 1 Compare to trigger value	When the measured concentration of zinc at the site is compared with the zinc trigger value there is a possible risk to ecosystems at the site. C_{total}/ trigger value > 1? For zinc: 200/ 26 is >1 Where: C_{total} = 200 mg/kg Trigger value = 26 mg/kg	When the measured concentration of Cu at the site is compared with the Cu trigger value there is a possible risk to ecosystems at the site. C_{total}/ trigger value > 1? For Cu: 150/ 63 is >1 Where: C_{total} = 150 mg/kg Trigger value = 63 mg/kg
Step 2 Adjusting for background metal	A mean concentration of zinc in a range of UK soils is 85 mg/kg (McGrath and Loveland, 1992). The 10^{th} percentile of this range is	A mean concentration of Cu in a range of UK soils is 23 mg/kg (McGrath and Loveland, 1992). The 10^{th} percentile of this range is taken to be ≈ 9 mg/kg and is

	taken to be 40 mg/kg and is given as $C_{b(UK)}$. But this only reduces the measured concentration (C_{add}) to 160 mg/kg (200 – 40). The soil at the hypothetical site is a sandy loam. An average background concentration for zinc on sandy loam soils is considered to be 60 mg/kg ($C_{b(soil\_type)}$). The C_{add} is then reduced to 140 mg/kg but still exceeds the trigger value. Finally an adjustment is made to the measured concentration by considering the <u>local</u> background concentration (e.g. 120 mg/kg). The measured concentration (C_{add}) is still above the trigger value following the adjustment of background concentrations of zinc. So, the assessment continues to the next step. $C_{add} = C_{total} - C_{b(local)}$ For zinc: C_{add} = 200 – 120 C_{add} = 80 mg/kg	given as $C_{b(UK)}$. But, this only reduces the measured concentration (C_{add}) to 141 mg/kg (150 – 9). The soil at the hypothetical site could be from a textural class with an average background Cu concentration of 23 mg/kg ($C_{b(soil\_type)})^{\alpha}$. The C_{add} is then reduced to 127 mg/kg but still exceeds the trigger value. Finally an adjustment is made to the measured concentration by considering the <u>local</u> background concentration (e.g. 30 mg/kg). The measured concentration (C_{add}) is still above the trigger value following the adjustment of background concentrations of Cu. So, the assessment continues to the next step. $C_{add} = C_{total} - C_{b(local)}$ For Cu; C_{add} = 150 – 30 C_{add} = 120 mg/kg
Step 3 Adjusting for the difference in toxicity between the lab and the field.	The zinc risk assessment discusses the different results obtained in laboratory and field studies and derives a refinement factor R_{L-F} of 3. This is applied to the concentration for the site adjusted for local background level of zinc. This value is just above the trigger value*. C_{add} / R_{L-F} for zinc is 80/ 3 = 27 mg/kg	The copper risk assessment discusses the different results obtained in laboratory and field studies and derives a refinement factor R_{L-F} of 2. This is applied to the concentration for the site adjusted background level of Cu. C_{add} / R_{L-F} for Cu is 120/ 2 = 60 mg/kg
Step 4 Accounting for soil factors effecting	The zinc risk assessment includes relationships between soil properties and observed toxicity of zinc to plants, invertebrates and	The Cu risk assessment did not consider soil factors affecting availability. Therefore,

$$\frac{C_{add} / (R_{L-F})}{\text{trigger value}} > 1?$$

metal 'availability'	microbes. These are based on eight different soil types of which the closest to our site is arable sandy soil that had a factor of 1.1 ($BioF_{site}$). This is applied to the result from step three giving a final adjusted measured concentration for zinc of 24 mg/kg. $$\frac{C_{add} / (R_{L-F} \bullet BioF_{site})}{\text{trigger value}} > 1?$$ For zinc; $$\frac{80/(3 \times 1.1)}{26} = 0.9$$	For Cu; $$\frac{120/(2)}{63} = 0.95$$
Decision	The adjusted soil Zn concentration is below the trigger value.	The adjusted soil Cu concentration is below the trigger value.

In practice, the difference between 26 mg/kg and 27 mg/kg is within measurement error, however the additional step is given for illustrative purposes of how the processes would operate.

$^{\infty}$ From the Cu RAR (2005) the PNEC represents a 'worst case scenario' for Cu behaviour in soils – for example the toxicity data used has been normalised to soil conditions which would represent the greatest Cu availability (i.e. low SOC, low CEC, etc.).

$^{\alpha}$From Cu RAR (2005).

Olsen P

From the technical workshop it was recommended that trigger values be set as 'no increase values', especially when soils are at the top of Index 2 (25 mg/l) or within Index 0 (0-9 mg/l). However, such triggers need to be contextualised with information on catchment hydrology, sensitivity of receiving waters, erosion risk and so on. It was thought that with this information the potential risks associated with Olsen P values could be interpreted appropriately. As a risk-based tool, inherent caution in the trigger values is not a problem, as long as the values are set to ensure an ecologically relevant screen. Table 3.5 gives the Olsen P potential trigger values on the basis of functions/habitats and broad soil types. The soil types would be delineated according to MAFF (1988), where mineral soils are those with <6% organic matter, organic soils have 6-20% organic matter, peaty soils >20% organic matter and finally calcareous soils are those containing 'free calcium carbonate'.

Table 3.5: Potential trigger values for Olsen P (mg/l) for the soil function of environmental interaction

Function	Soil Type		
	Mineral and organic	Peaty	Calcareous
Metal retention			
Microbial function/ biofiltering			
Gaseous emissions			
Soluble phosphorus leaching*	>60	>60	>60
Hydrological			
Habitat support table			
Calcareous grassland			>16
Mesotrophic grassland	>10	>10	
Acid grassland	>10	>10	
Dwarf Shrub Heath	>10	>10	
Biomass production*			
Arable and forage	<15	<15	<15
Vegetables	<26	<26	<26
Grassland	<15	<15	<15

*Need to collect contextual/activity information.

Soil pH

From the soil pH report and the technical workshop, the potential triggers values as measured in water are given in Table 3.6. These values are relatively conservative for each of the soil types, habitats and functions. The soil type conditions are the same as those described for Olsen P above (MAFF, 1988).

Table 3.6: Potential trigger values for topsoil pH in water for the soil function of environmental interaction

Function	Soil type		
	Mineral and organic	Peaty	Calcareous
Metal retention	<6	<5.5	
Microbial function/ biofiltering	<5	<4.5	
Gaseous emissions			
Habitat support table			

Calcareous grassland			<7
Mesotrophic grassland	<5>7	<5>7	
Acid grassland	>5	>5	
Dwarf Shrub Heath	>4.5	>5	
Biomass production			
Arable and Forage	<6.5	<5.8	
Vegetables	<6.5	<5.8	
Grassland	<6	<5.3	

Soil organic carbon

From the technical workshop on soil organic carbon there was strong feeling that SOC monitoring at tier 1 should be across all monitoring sites, but that tier 2 should include reference sites. Further, it was thought that there was a need to assess the implications of different sampling strategies in terms of minimum detectable differences.

As outlined in the paper at the end of the SOC report, by Fang and Smith (Section 2.3.11), sampling and analysis regimes at existing reference sites would need to be improved if they are to be used either to test models or to demonstrate changes in soil carbon over short periods. The authors of the report also suggested that 20 years of sampling would be required to detect a management-induced trend in SOC against the natural variation, which is much longer than the time estimated based purely on the spatial variability of SOC. Although methods may exist to understand and interpret changes in SOC, benchmark data is lacking and there is a recommendation of further work to assess the form and characteristics of this data.

3.3 Road-testing the indicators and triggers

The soil quality indicators and respective trigger values are to be tested in a desktop pilot trial and field validation exercise. The latter is likely to be undertaken at well-established soil monitoring sites and sites for which significant historical and soil characterisation information is known. The testing will be carried out using all the selected indicators from all of the functions identified by Loveland and et al (2002) which have been tested by members of the UK Soil Indicators Consortium.

4 Conclusions and recommendations

The outcomes of this report are:

- the derivation and use of a method to challenge soil quality indicators for their suitability to assess environmental interaction for a national soil monitoring network;
- a tiered, risk-based approach to indicator use catering for the broad soil interest of different agencies and stakeholders
- agreement on a minimum dataset of soil quality indicators for the soil function of environmental interaction: that is, soil organic carbon, total nitrogen, Olsen P, available and total Cu, Ni and Zn, bulk density and pH;
- trigger values for indicators to flag potential 'headline' issues of concern with soils;
- a rigorous approach and structured use of expert judgement, which was considered to greatly reduce uncertainty and provide a sound methodology to the selection of SQIs;
- agreement from representatives of the UK soil science community and UK Soil Indicators Consortium on approaches and findings.

The recommendations from this work to take these findings forward in part are reliant on the work of other members of the UK Soil Indicators Consortium but include:

- bringing together the indicators for environmental interaction with those from the other soil functions to assess the minimum dataset;
- undertaking a road test of the methodology (all three tiers), indicators and triggers in a desktop pilot with a range of soil data, scenarios and locations;
- developing a monitoring strategy for the UK, through which the indicators can be further tested;
- using well-characterised soils at sites with documented histories to field test the method, indicators and monitoring strategy in a range of locations across the UK.

5 Glossary of terms

Environmental Interaction - soils form a crucial link between the atmosphere, underlying geology, water resources and land use. They filter substances from water and intercept particles from the atmosphere; they emit and adsorb atmospheric gases (including the suite known as greenhouse gases); they can store carbon and regulate the flow of water from rainfall to aquifers and surface water sources, vegetation and back to the atmosphere. These are the functions that the soil performs.

Soil Quality - The capacity of a specific soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health habitation.

Indicator - property or situation. Responds promptly and accurately to perturbation and a change in its value or status can be interpreted with regard to a desirable condition for that soil and/or land use.

Triggers - values or ranges of values above or below which a level of change is understood to be critical in terms of the soil's fitness for a specific use.

Minimum dataset - minimum number of key soil quality indicators to be used for high level policy issues. The indicators that would be used nationally across the monitoring network.

6 References

Arshad, M.A. and Martin, S. (2002) Identifying critical limits for soil quality indicators in agro-ecosystems. *Agricultue Ecosystems and Environment*, 88, 153-160.

Bhogal, A., Nicholson, F.A., Chambers, B.J. and Shepherd, M.A. (2003) Effects of past sewage sludge additions on heavy metal availability in light textured soils: implications for crop yields and metal uptakes. *Environmental Pollution*, 121, 413-423.

Cu-RAR (2005) *European Union Risk Assessment Report. Copper, copper(ii)sulphate pentahydrate, copper(i)oxide, copper(ii)oxide, dicopper chloride trihydroxide.* Voluntary risk assessment, Chapter 3, draft May 2005. European Copper Institute.

Dahlin, S., Witter, E., Mårtensson, A., Turner, A. and Bååth, E. (1997) Where's the limit? Changes in the microbiological properties of agricultural soils at low levels of metal contamination. *Soil Biology and Biochemistry*, 29, 1405-1415

De Clerk, F., Singer, M.J., and Lindert, P. (2003) A 60 year history of California soil quality using paired samples. *Geoderma*, 114, 215-230.

Department for the Environment, Transport and the Regions (DETR), Circular 02/2000, Contaminated Land: Implementation of Part IIA of the Environmental Protection Act 1990. DETR.

Doran, J.W. and Jones, A.J. (1996) *Methods for assessing soil quality*. SSSA Spec. Publ. 49. SSSA, Madison.

ECB (2003) Technical Guidance Document on Risk Assessment in support of Commission Directive 93/67/EEC on Risk Assessment for new notified substances, Commission Regulation (EC) No 1488/94 on Risk Assessment of existing substances and Directive 98/8/EC of the European Parliament and of the Council concerning the placing of biocidal products on the market. European Commission, Joint Research Centre, EUR 20418 EN, 2003.

Goodwin, M.J., Parkinson, R.J., Williams, E.N.D. and Tallowin, J.R.B. (1988) Soil phosphorus extractability and uptake in a Cirsio-molinietum fen meadow and an adjacent *Holcus lanatus* pasture on the Culm Measure, North Devon, UK. *Agriculture, Ecosystems and Environment*, 70, 169-179.

Gufstafson, D.L., Shukla, R.K., Delbecq, A. and Walster, G.W. (1973) A comparative study of differences in subjective likelihood estimates made by individuals, interacting groups, Delphi groups and nominal groups. *Organisational Behaviour and Human Performance*, 9, 200-291.

Hamon, R.E. and McLaughlin, M.J. (2004) *Moving away from a single number background concentration for metals in soils.* Paper presented at SETAC Europe, 14th Annual Meeting, Prague, Czech Republic, 18-22 April. p.104.

Jarausch-Wehrheim, B., Mocquot, B. and Mench, M. (1999) Absorption and translocation of sludge-borne zinc in field grown maize (*Zea mays* L.). *European Journal of Agronomy*, 11, 23-33.

Kenney, E.A., Hall, J.W., and Wang, C. (2002) Temporal trends in soil properties at a soil quality indicator benchmark site in the Lower Fraser Valley, British Columbia. *Can. J. Soil Sci.*, 82, 499-509.

Lilburne, L., Sparling, G. and Schipper, L. (2004) Soil quality monitoring in New Zealand: development of an interpretative framework. *Ag. Ecosys. Env*. 104, 535-544.

Loveland, P.J., Thompson, T.R.E., Webb, J., Chambers, B., Jordan, C., Stevens, J., Kennedy, F., Moffat, A., Goulding, K.W.T., McGrath, S.P., Paterson, E., Black, H. and Hornung, M. (2002) *Identification and development of a set of national indicators for soil quality.* R&D Technical report P5-053/2/TR, Environment Agency, Swindon.

Loveland, P. and Webb, J. (2003) Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research*, 70, 1-18.

MAFF (1988) *Fertiliser Recommendations*. Reference Book 209. HMSO, London.

MAFF (1993) *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Potentially Toxic Elements*. Ministry of Agriculture, Fisheries and Food, London.

McBride, M.B., Richards, B.K., Steenhuis, T. and Spiers, G. (1999) Long-term leaching of trace elements in a heavily sludge-amended silty clay loam soil. *Soil Science,* 164, 613-623.

Merrington, G., Fishwick, S. and Brooke, D. (2006) The derivation and use of soil screening values for metals for the ecological risk assessment of contaminated land: a regulatory perspective. *Land Contamination and Reclamation* (In press).

Moffat, A.J. (2003) Indicators of soil quality for UK forestry. *Forestry*, 76, 547-568.

Motta, A.C.V., Reeves, D.W., and Touchton, J.T. (2002) Tillage intensity effects on chemical indicators of soil quality in two coastal plain soils. *Commun. Soil Sci. Plant Anal*., 33, 913-932.

Page-Dumroese, D., Jurgensen, M., Elliot, W., Rice, T., Nesser, J., Collins, T. and Meurisse, R. (2000) Soil quality standards and guidelines for forest sustainability in northwestern North America. For. Ecol. Man., 138, 445-462.

Pankhurst, C.E., Doube, B.M., Gupta, V.V.S.R. and Grace, P.R. (1994) (ed.) *Soil biota management in sustainable farming systems.* CSIRO, Melbourne, Australia.

RCEP (1998) Royal Commission on Environmental Pollution. *Setting Environmental Standards*, HMSO, London, UK.

Schipper, L.A. and Sparling, G.P. (2000) Performance of soil condition indicators across taxonomic groups and land uses.Soil Sci. Soc. Am. J., 64, 300-311.

Seybold, C.A., Grossman, R.B., and Pierce, F.J. (2003) On-site assessment of use-dependent soil properties in Michigan. Commun. Soil Sci. Plant Anal., 34, 765-780.

Smith, D. (1990) A better water quality indexing system for rivers and streams. *Water Res*. 24, 1237-1244.

Sojka, R.E. and Upchurch, D.R. (1999) Reservations regarding the soil quality concept. Soil Sci. Soc. Am. J., 65, 1039-1054.

Sparling, G.P. and Tarbotton, I.S. (2000) *Workshop to develop soil quality indicators*. Landcare Research Contract Report LC9900/078.

Sparling, G., Lilburne, L. and Vojvodic-Vukovic, M. (2003) *Provisional targets for soil quality indicators in New Zealand.* Landcare Research New Zealand Ltd, Lincoln, New Zealand.

Sparling, G., Parfitt, R.L., Hewitt, A.E., and Schipper, L.A. (2003a) Three approaches to define desired soil organic matter contents. *Journal of Environmental Quality*, 32, 760-766.

Sparling, G.P., Schipper, L.A., Bettjeman, W. and Hill, R. (2004) Soil quality monitoring in New Zealand: practical lessons from a 6-year trial. *Agriculture, Ecosystems and Environment*, 104, pps 523 - 534.

The Netherlands (2004) Risk Assessment Report: Zinc metal. Final draft of December 2004 (R073_0412_env).

Walker *et al*. (2001) *Methodologies for restoring and re-creating semi-natural lowland grasslands: a review and quantitative model*. Countryside Council for Wales Contract No. FC 73-01-273A

Appendix i

Summary from peer review by selected experts

Brief Summary of Meeting
Soil Quality Indicators – Bath Meeting 2-3 February 2005

Which indicators?

Other potential indicators were considered briefly. In particular the group acknowledged that significant gaps existed with regard to physical SQI. This was primarily due to underlying concerns with regard to measurability and interpretability of information form the limited data to hand.

Further, the lack of a suitable biological SQI (beyond SOC) is acknowledged, an action on attendees on PMN by the 15th of February.

The indicators will NOT be clustered as physical/chemical and biological when more broadly discussed, as it was thought the delineation was in some respects arbitrary.

Table showing the selection of potential indicators and additional comments for action

Potential indicator	In/out?	Comments
pH (H_2O – W/V)	IN	Look at consistency across soil types with regard to $CaCl_2$. Concerns with regard to sensitive sites.
Total N	OUT	Not sensitive to function. Lack of relevance, Brian to send data showing linkages to SOC.
Olsen P (W/W and W/V)	IN	Olsen P cf. Resin. Reasonably good links with water quality. Sensitive sites – not well covered by this. Rob to dig something up.
Organics	OUT	Maybe in at a later date. Politically could be in at a later date.
Metals (Cu, Ni, Zn) – total & available	IN	Total and available. Must have both to be interpretable and relevant to function – values may be set using ESR and VRA work.
SOC (concentration, not stock?)	IN	Measured as SOC, reported as SOM. Active/light as an additional measure on archived soil? Dave to provide data from Tyson, Brian from Fen Silts.
Potentially mineralisable N	OUT	Everyone by the 15th of February
Bulk density	IN	Just extreme ranges, measures in soil and base of cultivation layer.

Potential indicator	In/out?	Comments
Macroporosity	OUT	Tricky, although would be great if it could be measured simply and cheaply.
Aggregate stability	IN	Which method? Data from Brian and Dave
Catchment hydrograph	OUT	Great, if we could interpret it.

Monitoring Options:

Option 1. Grid
Option 2. Soil series
Option 3. Groups, sub groups, groups.
Option 4. Catchment-based, river basin district
Option 5. Point linked to area – 'intelligent sampling' (Chambers *Pers. Comm.*)

Why do we want to monitor?

Soil interactions with water storage, retention, movement.
Soil interaction with chemicals nutrients.
Soil interaction with air, aerial deposition.
Soil and resilience.
Soil to support a land use/ habitat.

Some questions to be tackled by the Environment Agency, including:

What are the implications of a different sized grid?
Is there a grid size that is optimum?
If you used a grid what questions could you answer?
What does 50 km x 50 km tell you?
Can a point represent an area?
What happens if we bulk less and sample more?

Also to draft success criteria for sampling programme: 'What do we want it to do?'

For example:
1. Demonstrate change over time at national level
2. Answer policy questions
3. Can the same set of measurements be used to fuel a model which would discriminate environmental Interaction?
4. Answer future policy questions.

Triggers

Could be considered as 'wake up and have a look' criteria.
If a trigger is exceeded then the first action will be to look at the other indicators, how do they interact?
As time progresses, and this is why soil type is included in a crude form, that not only magnitude but rate and direction of change is recorded.

See reports.

Timing and depth of sampling

Potential indicator	Depth	Timing
pH (H_2O – W/V)	0-7.5 cm, 0-15 cm	N/a
Olsen P (W/W and W/V)	As above	N/a
Metals (Cu, Ni, Zn) – total & available	0-15 cm	N/a
SOC (concentration, not stock?)	0-15 cm (stocks – 0-deeper?)	N/a
Bulk density	Top of topsoil, mid of topsoil, upper B	Spring
Aggregate stability	0-5 cm	Spring

Appendix ii

Delegate pack for technical workshop

Attendees at Bath Technical Workshop

Roger J Unwin Policy Adviser Soil Protection and Organic Farming, Defra	Tim Harrod Soil Survey and Land Research Centre (Retired)
Patricia Bruneau Soil Science Advisor/Soils Lead Coordination Network, Scottish Natural Heritage	Judith Stuart Policy Lead - Soils in the Built Environment, Soils Team, Defra
Sal Burgess Soils Research Manager, Soils Team, Defra	Sharon Ellis Head of Soil Team, Defra
Mark Aitken Policy Development Advisor (Land), Scottish Environmental Protection Agency	David Rimmer Senior Lecturer Civil Engineering & Geosciences , University of Newcastle Tyne
Karl Ritz Chair in Soil Biology, National Soil Resources Institute, Cranfield University	Peter Loveland Editor-in-Chief, European Journal of Soil Science, Rothamsted Research
Malcolm Reeve Land Research Associates Ltd, Lockington Hall	Stephen Nortcliff Head of Department, Professor of Soil Science Soil Science; School of Human and Environmental Sciences, University of Reading
Helaina Black Head of Soil Ecology Group, ECOPRO, Centre for Ecology and Hydrology, Lancaster	Mike Hornung Centre for Ecology and Hydrology, Lancaster
Rob Parkinson School of Biological Sciences, University of Plymouth	Ian Rugg Technical Services Division, Department for Environment, Planning and Countryside, Welsh Assembly
Dylan Williams Soil Science Policy Officer, Countryside Council for Wales	Jim Stevens Department of Agriculture and Rural Development, Belfast
Pete Smith Reader in Soils & Global Change, School of Biological Sciences, University of Aberdeen	Vanessa Kind Research Manager, Scotland and Northern Ireland Forum for Environmental Research (SNIFFER)
Michael Haft School of Biological Sciences, University of Aberdeen	Declan Barraclough Science Manager, Ecology and Soils, Environment Agency
Bob Jones	Jane Morris

NSRI Cranfield University	Policy Manager, Soil, Environment Agency
Brian J Chambers Principal Research Scientist, ADAS Gleadthorpe Research Centre	Graham Merrington Principal Scientist, Chemicals Team, Environment Agency
Andy Moffat Head of Environmental and Human Sciences Division, Forest Research, Alice Holt	Mark Crane Watts, Fawell & Crane Environment Ltd., Farringdon
Colin Campbell The Macaulay Institute, Craigiebuckler, Aberdeen	David Scholefield Principal Research Scientist, SEES, IGER, North Wyke

Ground Rules for Participation in Meeting

Be prepared for the meeting.

1. Come to the main meeting and breakout groups on time.
2. Start and end breakout and plenary sessions on time.
3. Speak in turn, as invited by plenary and breakout chairs, with only one person speaking at a time.
4. Participate fully.
5. Maintain positive group dynamics.
6. Question assumptions by asking clarifying questions, but assume good faith on the part of others.

7. Aim for consensus, but not 'consensus in error.' If there is a disagreement:
    a. Clarify exactly what the disagreement is about.
    b. Determine whether the disagreement is about facts or personal values.
    c. If the disagreement is about facts then determine what facts need to be obtained to resolve it (such as collation of existing information, or specific R&D), and ensure that this conclusion is recorded.
    d. If a disagreement is about personal values then ensure that this conclusion is recorded, and move on.
    e. If you have evidence to support or counter views expressed in the reports or datasets, please bring it with you.

MEETING AGENDA for the 28th and 29th of June 2005

Time	Activity	Person responsible
Day 1		
12.30	Arrive & Register	Ella Turner
13.00	Lunch	
14.00	Introduction (logistics & housekeeping)	Mark Crane
14.10	Background & context: why do we need soil quality indicators?	Sharon Ellis (Defra)
14.30	Process to develop soil quality indicators & outputs to date	Graham Merrington
14.50	Plenary: Delegates comment on strengths and weaknesses of process to date	Mark Crane

15.10	Introduction to breakouts	Mark Crane
15.15	Breakout 1: Consideration of selected indicators* ▪ Group 1 – Bulk density ▪ Group 2 – pH ▪ Group 3 – Metals (Cu, Zn, Ni) Tea & coffee available throughout	 Jane Morris Declan Barraclough Graham Merrington
17.15	Plenary feedback (15 minutes per group + 5 minutes questions per group)	Breakout group reporters
18.15	Break (or additional discussion time if required)	
19.00	Dinner	
Day 2		
8.30	Plenary: Discussion of what did and didn't work yesterday and amendment of schedule, if necessary	Mark Crane
9.00	Breakout 2: Consideration of selected indicators* ▪ Group 1 – Aggregate stability ▪ Group 2 – Soil organic carbon ▪ Group 3 – Olsen P Tea & coffee available throughout	 Jane Morris Declan Barraclough Graham Merrington
11.00	Plenary feedback (15 minutes per group + 5 minutes questions per group)	Breakout group reporters
12.00	Plenary discussion: Are these six soil quality indicators sufficient?	Mark Crane
12.30	Lunch	
Time	**Activity**	**Person responsible**
13.30	Breakout 3: Consideration of rejected indicators* ▪ Group 1 – Macroporosity & Catchment hydrograph ▪ Group 2 – Total N & Potentially mineralisable Nwiwigg ▪ Group 3 – Organics Tea & coffee available throughout	 Jane Morris Declan Barraclough Graham Merrington
15.30	Plenary feedback (15 minutes per group + 5 minutes questions per group)	Breakout group reporters
16.30	Summary, Conclusions & Next Steps	Mark Crane
17.00	End	

* Breakout groups will consider the following questions for each indicator in the light of a 100-year dataset based on real data:
- Is it relevant and, if it is, why?
- What does an overall change in this indicator mean?
- What rates of change in this indicator are important?
- What should trigger a closer analysis of this indicator?
- Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

- Are there any other factors that should influence the selection and use of this indicator?

People Responsible

Ella Turner - Environment Agency, Event Organiser
Mark Crane – Independent Facilitator, *WFC Environment*
Graham Merrington – Environment Agency, Programme Manager
Declan Barraclough – Environment Agency, Programme Executive
Jane Morris – Environment Agency, Programme Client

Introduction to processes undertaken (also see slides)

Selection of soil quality indicators

The selection of soil quality indicators to use in a UK monitoring network is potentially an onerous and contentious task. Indicators need to be meaningful and interpretable, with regard to the state of soils under a significant range of potential pressures, and linked to guidance on a management response. Furthermore, an important criterion of an indicator is that it responds promptly and accurately to perturbation and that this change can be interpreted with regard to a desirable condition for that soil and/or land use. Other criteria may include consideration of cost, the need to utilise existing data and practicality and simplicity.

The previous Environment Agency-led project (P5-053/2) identified 67 potential soil quality indicators and recommended a minimum set of nine. However, to ensure that this selection of nine is both appropriate and robust we are going to challenge the list of 67 by applying them, or the concepts behind them, to a wide range of soil and land use systems. The Environment Agency has responsibility for indicators with an ‘environmental interaction’ and these have been challenged using the methodology outlined in slide 11 of the introductory notes.

Focus for the internal reports:

The internal reports have focussed on the ‘Level 1’ type of indicators as used potentially in a minimum dataset (MDS), that is, a relatively few, headline indicators that are sampled extensively across the monitoring network. Potential MDS indicators have been selected for review by reference to other international soil monitoring networks, the previous phase of this work (P5-053/2), expert soil scientists’ views and Environment Agency experience. The selection is by no means exhaustive or definitive, indicators selected here may not be on the MDS list and similarly, indicators (following the review process or the workshop) not selected may be placed on the list.

The purpose of the internal reports is many-fold, but primarily to provide a thought piece for our internal experts and Steering Group members to comment. The reports <u>are not expected to be definitive guides</u> to a particular soil property. However, it is imperative that they avail <u>us</u> to the most important points for deliberation. They have been written in a style that is understandable to an informed lay audience.

Environment Agency

Section 2

Introductory Notes - pre reading
Francis Hotel, Bath
28 & 29 June 2005

Graham Merrington

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Environment Agency

Papers to bring to meeting

- These notes
- The potential minimum data set indicator reports (n = 10)
- Meeting Ground Rules
- Agenda
- Data sets if you wish

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Environment Agency

Why are we (Agency and Consortia) doing this project? Aims

- Answer Soil Policy questions
- Development of a National Soil Monitoring Network for the UK
- Enable improved ability to report on the state of UK soils
- Develop a common approach to development of indicators

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What is the role of the Environment Agency

Environment Agency

- Member of consortia (with Defra, SEERAD, CCW, DOENI, EH, NAWAD, EN, SEPA,)
- The Agency is responsible for Soil Quality Indicators in the context of the soil function 'Environmental Interaction'

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Some 'working' definitions as guidance (no more!)

Environment Agency

- **'Soil Quality'** - The capacity of a specific soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health habitation
- **'Indicator'** - property or situation. Responds promptly and accurately to perturbation and that this change can be interpreted in regard to a desirable condition for that soil and/or land use.
- **'Environmental Interaction'**- soils form a crucial link between the atmosphere, underlying geology, water resources and land use; they filter substances from water and intercept particles from the atmosphere; soils emit and adsorb atmospheric gases (including the suite known as greenhouse gases); they can store carbon; they regulate the flow of water from rainfall to aquifers and surface water sources, vegetation and back to the atmosphere. The function that the soil performs.

Continued.............

Environment Agency

- **Workable Range/ Envelope of normality -** values or ranges of values, above or below which a level of change is understood to be 'critical' in terms of the soils fitness for a specific use (we formally called these triggers in the reports).
- **'Minimum data set'** - minimum number of key soil quality indicators used to answer 'high level' Policy questions. The indicators that would be used nationally across the monitoring network. See slide on 'Methodology'.

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Overall aims of meeting

Environment Agency

- Agreement that the overall process followed to date has been appropriate
- Confirmation that the correct indicators have been selected
- Confirmation that the correct minimum dataset has been selected; and
- Identification of appropriate workable range for the indicators for ‘Road Test’

Why challenge the indicators?

Environment Agency

- We started with 67+Indicators (from Phase 1 if the project) - too many, are the 11 chosen from the first project appropriate? Are the ones selected by us appropriate for a minimum dataset?
- The aim of the external challenge is to confirm whether our thinking in regard to potential indicators, workable ranges and a minimum dataset are reasonable.
- Are the Indicators appropriate, relevant, practical and usable?
- Will the indicators fit into a soil monitoring network?
- The indicators must be supported by a scientific and interpretative framework which can extract long term trends from short term observations (see datasets).

What is ‘challenge’?

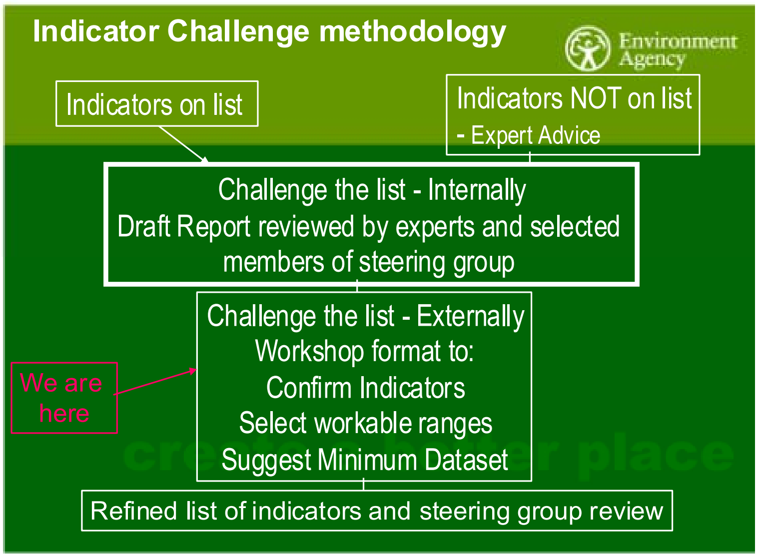
Environment Agency

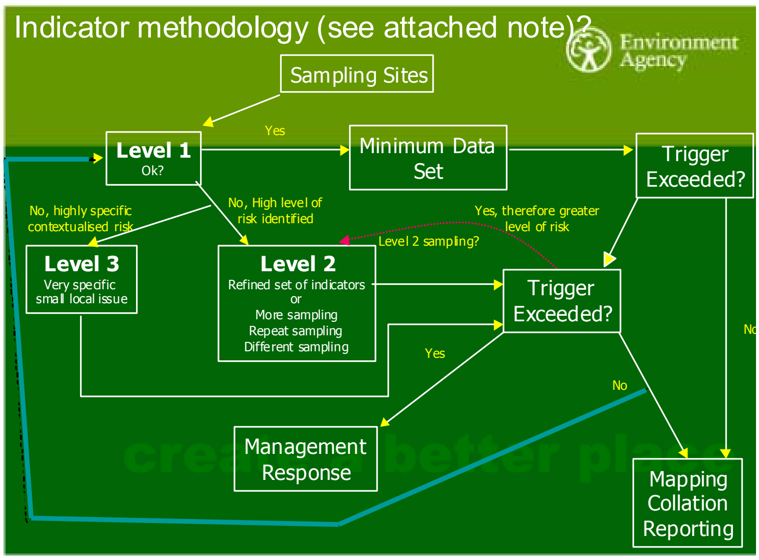
- Do the indicators fulfil the requirements of the ‘Indicator’?
- Do the Indicators stand scrutiny within the context of the project aims i.e. do they answer the policy questions?
- Could we justify the choice of an Indicator on the basis of being appropriate, relevant, practical and usable for a National Soil Monitoring Network?
- Scientifically acceptable?

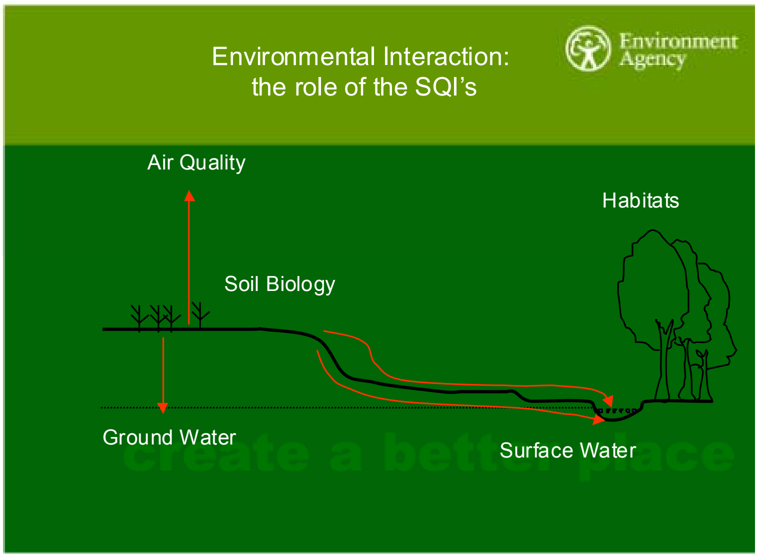
Questions to be considered by YOU

Potential indicators of soil quality should be assessed against relevance, sensitivity and discrimination.............

- Can we interpret change (see datasets - to be discussed in Breakouts)?
- What does it mean if change occurs (influence on other soil properties?) i.e. what questions can they answer?
- Relevance to the key soil function of Environmental Interaction?
- Can we integrate with another indicator?
- Measurability - Cost? Ease? Predictability? Signal to Noise ratio?

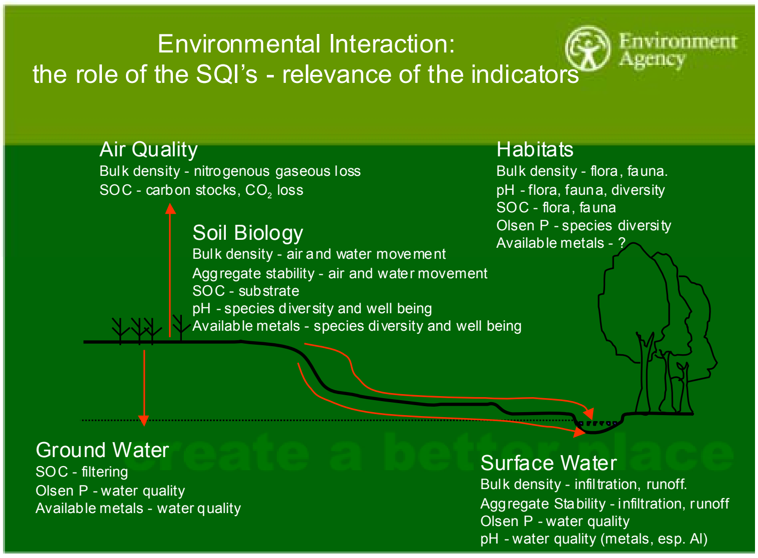






The eleven indicators challenged (see Section 3 Reports)

Bulk density
Macroporosity
Aggregate Stability
Catchment hydrograph
pH
Soil Organic Carbon
Olsen P
Potentially mineralisable nitrogen
Total N
Heavy metals/POPs



Workable ranges (formally triggers)

- Could be considered as 'wake up and have a look' criteria
- If value is exceeded then the first action - look at the other indicators, how do they interact?

Summaries from technical workshop

Breakout Group Summary – Aggregate Stability

Summary
- Aggregate stability was not considered to be an appropriate measure for use as a soil quality indicator at Level 1.
- The group recommended the use of surrogates at Level 1 (SOM and Bulk Density) which could then be used in conjunction with other factors to trigger measurement of aggregate stability at Level 2.
- Aggregate stability is important for providing information on soil biology, soil organic matter protection, infiltration and aquifer recharge – all within the environmental interaction soil function.
- The information from aggregate stability measurement must be interpreted in conjunction with information on soil type, changes in land use and land management and climate.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

An overall change in the indicator has an influence on:

- Infiltration and aquifer recharge;
- Soil organic matter protection;
- Soil biology;
- (Possibly) habitat support.

The main environmental pressures which affect aggregate stability and can cause changes are:

- Land use and land management;
- Climate change – this is indirect through a feedback loop from climate change driving changes in land use and management.

There may be more data on aggregate stability from the work on the protection of SOM by Cursent (published in EJSS).

Questions 2 and 3

Measurements of aggregate stability are informative and we can discern and interpret changes and trends over time but:

- We don't know what rates of change are important but we might be able to infer some information from it;
- We don't have a good method to measure it;
- Mean weight ratio method – reproducible but not automated plus too resource intensive;
- Dispersion method – high potential for operator error, quick but not reproducible/interpretable;
- We don't know what trigger values/rate of decline should trigger further investigation.

However, we did agree that if we were able to measure and reproduce the method then aggregate stability would be very informative.

In reality the group thought that we could use surrogates for the measurement of aggregate stability, namely SOC and bulk density to trigger a move to Level 2 monitoring where we could look at aggregate stability. Additionally, we thought it might be a good indicator for use at risky sites, for example where the soil had less than 2.5% SOC and greater than 12% clay and was not calcareous.

We therefore concluded that there was a need for further work on ranges and triggers for this indicator. Specific further work required:

- Malcolm Reeve offered to supply some information from a research student on progression of aggregate size of reclaimed land;
- We need a value-for-money method which is reproducible;
- We need to identify a normal range and triggers of change by texture (this is likely to require further data collection);
- In the urban environment for unmanaged green space bulk density may be more relevant but we need to investigate this further.

However we decide to measure aggregate stability in the future, the collection of significant quantities of land use history information is also required between sampling events to allow interpretation.

Question 4

Long term trends in aggregate stability can be discerned and interpreted over time but we would suggest the use of surrogates at Level 1 (bulk density and SOM).

These could then be used to trigger the need for aggregate stability measurement at Level 2.

Questions 5 and 6

Aggregate stability is not fit for use as an indicator and we would not recommend it for use in Level 1 monitoring.

Breakout Group Summary – Bulk Density

Summary

- Bulk density was considered to be an appropriate measure for use as a soil quality indicator.
- It gives information on soil biology, water quality, aquifer recharge and habitats – all within the environmental interaction soil function.
- The information from bulk density measurement must be interpreted in conjunction with information on soil type, changes in land use and land management and climate.
- Trigger values for the bulk density need to be produced for broad textural classes taking clay content into account.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

The group agreed that a change in bulk density is an important indicator. Land management and use were identified as the predominant environmental pressure to which changes in bulk density was related. However, bulk density is also affected by climate change, although this is predominantly through other factors such as changes in agricultural practices.

Where bulk density is measured within the soil profile under different land uses is critical to its interpretation. We therefore agreed the following:

Land use	Depth of measurement
Arable	Below the cultivated layer (upper subsoil)
	Topsoil
Grassland	Topsoil
Woodland	Deep subsoil (relating to harvesting)
	Topsoil
Urban	?

Question 2

Rates of change for bulk density are difficult to interpret and at the moment there is little temporal data available. Issues of particular note with respect to interpretation of rates of change are:

- information on soil type, changes in land use and land management and climate is required;
- some natural systems with indurated layers may have an increase in bulk density which is not a problem - this is different to other systems;
- free-draining soil could recover when the pressure is removed.

The group felt that further research was needed and in particular, data mining of the following sources was necessary:

- CEH scoping study regarding priorities in Wales;
- landslides in Scotland – PB to provide;
- MAFF-funded work on porosity and hydrological conductivity;
- Pont Bren – links between biology and bulk density;
- NSRI – trafficking and soil surface properties data;
- reclamation data to be supplied by Malcolm Reeve;
- Brimstone data looking at before and after cultivation on wet and dry soils (work done by Mike Goss and John Catt);
- Farringdon data;
- LANDIS.

This information should then be used to review and revise the information contained within the draft Environment Agency report on bulk density.

Question 3

The group agreed that it was possible to identify triggers that would signify further investigation, however trends in the data are very noisy. If triggers were set the group agreed that these should be established for broad textural classes. These need to relate to clay content and an increased number of divisions for mineral soils is required over and above those already identified in the report (Table 2.3.3.5, Section 2.3.3). If triggers were set for different textural classes the following data would be required for interpretation:

- soil type and texture;
- habitat/land use;
- climate;
- temporal.

Concerns were discussed over the ability to measure trends in stony soils.

Peter Loveland provided a box plot analysis with inter-quartile ranges from NSRI data which could be used as a reality check and an estimate of a range of normality.

Question 4

Long-term trends in bulk density data can be discerned and interpreted as long as the data is reviewed in the context of other environmental data such as those items identified under question 3 plus climate change.

Questions 5 and 6

Bulk density is fit for use as an indicator in terms of costs, and is essential for the interpretation of a number of other soil parameters. The timescales for repeat analysis of bulk density data is dependent on land use change.

Particular issues relating to the measurement of bulk density in forestry were raised. It was agreed that under forestry the measurement of bulk density would only be necessary when something had changed, for example after harvest or planting.

Other issues discussed

The predominant impact of land management and land use on bulk density means that we must have information on these factors to allow interpretation of bulk density data.

The misuse of data was of great concern. For bulk density it was agreed that it is difficult to prove that data is auto-correlated and therefore the mapping of bulk density would not be meaningful and could be misleading.

Breakout Group Summary – Catchment Hydrograph

Summary

- The catchment hydrograph was not considered to be an appropriate measure for use as a soil quality indicator.
- It gives information on infiltration, surface water and aquifer recharge, riparian habitat support and the influence of the built environment - all within the environmental interaction soil function.
- Research is needed to ascertain how far the impacts of soil quality changes can be interpreted from other factors which affect the catchment hydrograph.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

An overall change in the indicator has an influence on:

- infiltration;
- surface water and aquifer recharge;
- habitat support (riparian);
- built environment impacts.

The main environmental pressures which affect catchment hydrograph and can cause changes are:

- land use and land management;
- climate change;
- built environment.

Questions 2 and 3

The rates of change are important; however, they are land-related rather than a specific soil quality issue. The hydrograph cannot be interpreted in terms of changes in soil quality as a large number of factors may lead to changes in the hydrograph.

Further research is needed to ascertain exactly what the catchment hydrograph can tell us about changes in soil quality and how these changes can be identified from the other factors affecting the hydrograph. As part of this, it would be helpful to consider the hydrograph in combination with water quality data.

The group was unable to suggest any triggers or long-term trends which were important, due to the lack of understanding of how changes in soil quality can be identified within the hydrograph in an interpretable way. Further work on this is needed in conjunction with hydrologists.

Question 4

We cannot interpret changes in the hydrograph in terms of soil quality.

Questions 5 and 6

Catchment hydrograph is not fit for use as an indicator of soil quality at this stage. We would not recommend it for use in Level 1 monitoring.

Breakout Group Summary – Macroporosity

Summary

- Macroporosity was not considered to be an appropriate measure for use as a soil quality indicator for Level 1 monitoring.
- The group recommended the use of surrogates at Level 1 (SOM and bulk density) which could then be used in conjunction with other factors to trigger measurement of macroporosity at Level 2.
- Macroporosity is important for providing information on soil biology, water quality, aquifer recharge and habitats – all within the environmental interaction soil function.

Questions Considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

An overall change in the indicator has an influence on:

- infiltration and aquifer recharge;
- run-off;
- aeration;
- soil biology;
- habitat support;
- 'rootability'.

The main environmental pressures which affect macroporosity and can cause changes are:

- land use and land management;
- climate change – this is indirect through a feedback loop from climate change driving changes in land use and management.

Questions 2 and 3

Rates of change for macroporosity are difficult as it isn't pore size alone which is important in a soil, but the connectivity of pores. Having said this, the group suggested the following would be significant rate of change:

'A decline of 20% of the original coarse porosity over 5 years'.

The group agreed that triggers for different soil types could be set and that macroporosity was important as an indicator of soils as a transmitter of water (i.e. that porosity relates to hydrologic connectivity).

Question 4

Long-term trends in macroporosity can be discerned and interpreted with respect to water transmission over time if measured in soils in a saturated state, but not if they were in unsaturated or dry state. Additionally, we felt that it could be interpreted in terms of air transmission through soils but the link to the impacts on soil biology were less clear.

Maroporosity alone was not felt to provide sufficient information to assess the adequacy of changes in policy.

Questions 5 and 6

Macroporosity is not fit for use as an indicator and we would not recommend it for use in Level 1 monitoring. Bulk density can act as a surrogate trigger if we define thresholds in terms of broad textual groups (see aggregate stability paper).

The method to measure macroporosity was not agreed and it is resource-intensive and expensive to measure.

We would recommend the use of surrogates at Level 1 (namely SOM and bulk density) with macroporosity at Level 2 where risks are identified. This is largely down to the lack of a standard cost effective method and the difficulty of interpretation of results in terms of soil quality.

Other issues discussed

The profile description as each sampling site must include porosity when a site is baseline. This will help to identify high risks sites for Level 2 monitoring in conjunction with other factors.

Breakout Group Summary - Metals

Summary

- The metals copper (Cu) and zinc (Zn) where considered to be appropriate for use as soil quality indicators.
- These two metals give information on soil biology, water quality and habitats – all within the environmental interaction soil function.
- Soil metal concentrations are measured as pseudo (aqua regia) totals and from these, refinements can be made through the use of measured physico-chemical factors, such as pH, to estimate metal 'availability'. Both need to be undertaken to assess <u>rate</u> and <u>direction</u> of change.
- Trigger values for the total may be informed through the use of the sewage sludge values, although for habitat protection these may not be suitably precautionary.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Before consideration of the questions above, the group was asked if they believed that in copper (Cu), nickel (Ni) and zinc (Zn), the correct metals had been chosen for inclusion in a minimum dataset.

The group was in agreement that these metals were appropriate, but that mercury (Hg) and lead (Pb) were also potentially important, with significant diffuse inputs to soils. However, with Pb aerial deposition dropping significantly in all European capital cities and Hg use likely to be greatly reduced (as a Priority Hazardous Substance under Annex X, Article 16 of the Water Framework Directive) their inclusion at Levels 2 or 3 was deemed most appropriate. Further, the inclusion of cadmium (Cd) was also considered, but it was believed that this was more closely related to the soil function of food and fibre production, in relation to crop grain quality.

Question 1

This issue of change in metal levels in soils was discussed, with particular emphasis on the need for contextual information on the soil and land use practice. This should be provided on sample collection, in particular information on manure/sludge practice. The need for a proxy measure of soil metals, such as aerial deposition data (possibly at reference sites), to flag up potentially rapid changes in soil metal levels was thought to be a useful inclusion.

Total metal concentrations would be extracted using aqua regia, in the light of the significant amount work on soil metals previously carried out using this extractant. However, it was broadly agreed that a refinement to provide an indication of metal availability and subsequent likely ecological risk also needed to be undertaken. It was agreed that total soil metal concentrations were relatively insensitive but methodologies for refining this to indicate likely risk, such as those used in the European Existing Substance Regulations (ESR) for Zn, should also be used. These refinements are based on physico-chemical factors that affect metal availability, such as pH, SOC, cation-exchange capacity and background soil concentration, through relationships established by international research programmes.

It was understood that Cu and Zn both provided information for the soil function of environmental interaction on soil biology, water quality and habitats. It was not clear whether Ni provided similar detailed information, although ongoing research in this area (PDF files available for those interested) show compelling evidence.

Question 2

The rates of change thought to be significant could be estimated from the datasets (for example, for Cu around 8 mg/kg/yr). It was agreed that the datasets represented worst case conditions, but critical rates could be predicted for Cu and Zn with some additional work by the Environment Agency.

Question 3

The use of the sludge values (MAFF, 1993) as trigger values for total soil metal concentrations in an agricultural context were thought to be suitable, particularly as a guide to assess the significance of rates of change (in combination with other proxy measures, such as aerial deposition). However, for sensitive habitats more emphasis needs to be placed on the refined values than the totals. A report by the Environment Agency sets the likely procedure and methodology for the values in the context of ecological risk assessment.

The group expressed the view that it was appropriate to discern rates of change as an imperative over triggers, primarily because once metals are in the soil, it is technically challenging and prohibitively expensive to remove them on a broad scale.

Questions 4, 5 and 6

The metals data can be interpreted with regard to long-term trends. The modelled datasets were supplemented with metals data from the Broadbalk ‘untreated’ site to test the ease of interpretation.

Total soil metals were thought to be fit for use in a cost-effective, broad-scale monitoring scheme. In particular, with the use of ICP OES analytical facilities, multiple sweeps of elements are possible from a single extract at little if any extra cost. The methodologies and analysis are routine for most environmental laboratories.

Finally, the metals Cu and Zn were thought to be relevant soil quality indicators in a minimum dataset, to assess the soil function of environmental interaction in a national soil monitoring scheme. Further, it seems likely that Ni may also be included in the light of contemporary evidence from European soil research programmes.

References

Campbell, C.D., Paton, G.I., Towers, W., Paterson, E., Dawson, J.J.C., Cameron, C.M., Coull, M.C. and Christie, P (2001) *A biological classification scheme to assess the sensitivity of Scottish and Northern Ireland soils to heavy metals*. SNIFFER Technical Report SR(00) 09.

MAFF (1993) *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Potentially Toxic Elements*. Ministry of Agriculture, Fisheries and Food, London.

Merrington, G., Fishwick, S. and Brooke, D. (2005) *Use of soil screening values for metals for ecological assessment in England and Wales*. Paper presented at SETAC Europe, 15th Annual Meeting, Lille, France, 22-26 May. 83

The Netherlands (2004). *Risk Assessment Report: Zinc metal*. Final draft of December 2004 (R073_0412_env).

Breakout Group Summary – Olsen P

Summary

- Olsen P was considered to be a useful measure of available or reactive P in soils, and potentially sensitive to anthropogenic pressures.
  Olsen P can give information on habitat condition that is within the environmental interaction soil function. Further, secondary effects upon water quality may also be gained from the use of Olsen P (supported by contextual information at the site of sampling).
- Olsen P is 'less than perfect' as an SQI, not being a very sensitive measure in low P systems and having a broad range of acceptable values according to agricultural land use.
- Trigger values could be set as 'no increase values', especially when soils are at the top of Index 2 (25 mg l^{-1}) or within Index 0 (0-9 mg/l). However, such triggers need to be contextualised with information on catchment hydrology, sensitivity of receiving waters, erosion risk, and so on.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

It was agreed that Olsen P was a useful soil quality indicator that was sensitive to anthropogenic pressures. In particular, pressures in low P or P sensitive habitats, where elevated levels of available P may cause biodiversity loss. Further, that Olsen P was useful as a secondary indicator of water quality, and work both in UK and the US has demonstrated clear links with this measure of water quality.

There is also significant evidence (possible 30 years of data) from agriculture, where the measure is fairly widely accepted, that Olsen P is a valuable indicator of potential losses of reactive P.

However, in low P systems where kg/ha increases in P are not likely to be seen, Olsen P is too insensitive to pick up small but significant increases (within Index 0). Indeed, potentially irreversible damage may occur before significant differences in Olsen P are discernible from natural and measurement noise. Such sites may be more suitably placed at Level 3 where the P balance (inputs/outputs) can be monitored in detail.

Questions 2 and 3

Rates of change in Olsen P where thought to be important – but it was agreed that limits or triggers of 'no increase' in Olsen P would be appropriate for the soil function of environmental interaction. This trigger would need to be informed by activity at the site and in the catchment.

The table below, from the report, has the value of 60 mg/l for soluble leaching and was considered by the group to be too high. Discussion led to suggested values of > 25 mg/l to trigger further investigation. However, it was noted that the median Olsen P value for arable, ley and permanent grassland topsoils from the NSI was 21 mg/kg (Harrod and Fraser, 1999), suggesting limited sensitivity of the trigger. Therefore, values could only be interpreted with activity information collected at the time of sampling, including sensitivity of the receiving water, catchment hydrology and soil management (related to erosion risk, stocking density and manure management, etc.). This information will be enable the changes in Olsen P values to be put into the context of the soil function of environmental interaction.

Where excessive increases are noted (some livestock production systems), it was thought this demonstrated that such practices are unsustainable with regard to soil quality and the function of environmental interaction.

The use of reference sites was also discussed, possibly in line with limitations brought about under the Water Framework Directive. It may be possible to have well-studied catchments at which changes in Olsen P values may be readily interpreted and subsequently extrapolated to other catchments of similar typologies.

For habitats, the trigger table from the Olsen P report below was thought to be reasonable in the light of the lack of sensitivity of Olsen P at the lower ranges.

Potential trigger values for Olsen P (mg/kg and mg/l) for environmental interaction

Function	Soil type		
	Mineral	Peaty	Calcareous
Metal retention			
Microbial function/ biofiltering			
Gaseous emissions			
Soluble phosphorus leaching*	<60	<60	<60
Hydrological			
Habitat support table			
Calcareous grassland			<16
Mesotrophic grassland	<9	<9	
Acid grassland	<9	<9	
Dwarf shrub heath	<9	<9	
Biomass production*			
Arable and horticultural	16–45	16–45	16–45
Improved grassland	16–25	16–25	16–25

*Need to collect contextual/activity information.

Question 4

The interpretation of Olsen P data, as outlined above, can only really be undertaken when aligned with contextual data for site/catchment. For water quality issues this is reasonable, but for habitats and semi-natural sites this is unlikely to be straightforward.

Questions 5 and 6

Olsen P is fit for use as an indicator in terms of costs, and though not used UK-wide, was thought to be applicable broad-scale. Its limitations as a measure are understood and the clear lack of any alternative on a broad scale means that it is preferred.

The measurements can be undertaken on a wt/wt basis and a wt/vol basis.

Finally, Olsen P was thought to be a relevant soil quality indicator in a minimum dataset, to assess the soil function of environmental interaction in a national soil monitoring scheme. However, its use in a minimum dataset in order to answer high level policy questions is only going to be possible with the collection of additional contextual site information and the possible use of well-characterised reference sites.

References

Harrod, T, R. and Fraser, A.I. (1999) *Phosphorus loss from agriculture*. Unpublished SSLRC report to MAFF, Project NT1003.

Breakout Group Summary – Organic micropollutants

Summary

- Organic micropollutants (OMPs) were not considered to represent a potentially useful soil quality indicator for a minimum dataset.
- With variable degradation rates in soils and likely controls on emissions from international legislation, it was thought that OMPs are likely to represent Level 2 or 3 indicators.
- The links with the soil function of environmental interaction are relatively few, but with clear secondary poisoning and human health relevance.
- The use of reference sites to monitor persistent OMPs ensures that likely risks are identified.

Questions considered

1. What does an overall change in the indicator mean? Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
2. What rates of change in the indicator are important?
3. What should trigger a closer analysis of the indicator? Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?
4. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy?
5. Is the indicator fit for use in a cost-effective, broad-scale monitoring scheme?
6. Whether the parameter you are discussing is a relevant environmental indicator, with justification for your conclusion.

Question 1

It was thought that OMPs were sensitive to changes in environmental pressure, but that the meaning of change was not clear. Soil ecotoxicological data on many OMPs is very limited, hampering interpretation. This was particularly relevant for the datasets presented (PBDE) which had relatively short half-lives in soils, but are ubiquitous in their distribution.

Questions 2 and 3

Rates of change were distinguishable from the modelled datasets and from data obtained from the Broadbalk plots. It was noted that aerial deposition of many OMPs is decreasing with time as industrial controls of point and fugitive emissions driven by legislation at European and International levels take effect.

It was not possible to identify triggers, although it was acknowledged that through the use of reference sites, levels of OMPs from diffuse sources could be monitored. Further the use of a proxy measure of OMPs, such as OMPs in aerial deposition, may ensure that potential risks to soils are identified more rapidly than if measured, for example, five yearly in soils.

Questions 4, 5 and 6

It was thought that interpretation of long-term trends to inform policy would be difficult, primarily due the lack of understanding. The use of OMPs in a minimum dataset for a national monitoring network was also understood to be particularly expensive currently, but may significantly decrease as analytical techniques improve.

Although it was acknowledged by the group that OMPs responded to environmental pressure, their role as an indicator of the soil function of environmental interaction was not clear.

Breakout Group Summary - Soil organic carbon (SOC)

Question 1: Is it a relevant indicator of environmental interaction?

Individual initial comments

Mike Hornung (MH): Yes. Sensitive to land use change, habitat support, water-holding capacity and structure.
Pete Smith (PS): Yes. Baseline indicator of fertility, soil structure and climate change (C sequestration).
Heleina Black (HB): Yes. Important in habitat maintenance but major sampling issues and must be done with total N and bulk density.
Roger Unwin (RU): Yes.
David Scholefield (DS): Yes: C sequestration, gas emission, soil hydrophobicity on slopes.
Karl Ritz (KR): Yes; fundamental energy resource for soil.
Jim Stevens (JS): Yes; key indicator, biofilter for metals and organics.

Points in general discussion

- general feeling was that SOC was a fundamental indicator reflecting soil fertility, soil structure, retention of organics and pesticides, gas emission and nutrient cycling;
- main indicator integrating air and water.

Question 2: What does change mean?

Individual initial comments

Sharon Ellis (SE): sensitive to land use.
MH: we know enough to disentangle main drivers forcing C change.
DS: strong contextual component; more not necessarily good; old grasslands high in C and N leak nutrients.
KR: fundamental changes in energy flows and status.

Points in general discussion

- confusion of land use effects - may be difficult to disentangle main drivers. Need for reference sites with constant land use;
- need to consider implications of sample stratification on statistical power;
- an estimate was that there needed to be a minimum of 25 data points in each sub set;
- good system models will be needed to disentangle main drivers for SOC change.

Questions 3 and 4: What rate of change is important and what would trigger further investigation?

Note: these two questions were considered together.

Individual initial comments

SE: uncomfortable with trigger values as lines in the sand; happier with direction of change.
KR: agree; more useful to consider probability distributions of values and percentiles.
HB: also agree but emphasise the need to ensure statistical power in sub-groups derived from nested analysis.
JS: some useful info on sampling strategies in SP0124 (on Defra web site).
PS: *any* decline in upland SOC significant and cause for concern.

Points in general discussion

- strong feeling that SOC justified a stratified Level 1 and that Level 2 should include reference sites;
- climate change scenario predictions could be used as a diagnostic tool – for example, if observed changes in SOC agree with predictions from climate change models, that could be strong evidence that climate change was occurring;
- need for detailed power analysis to tease out the implications of different sampling strategies in terms of minimum detectable difference and s. on, Large datasets already available for UK which should be used in such exercises.

Question 5: Can we discern/interpret long-term trends?

Individual initial comments

None

Points in general discussion

- feeling was that changes in national C levels detectable over medium to long term;
- changes in C stocks more difficult. Need for soil datum level such as soot layer derived from Mount Helen;
- distinction drawn between changes in national distribution of soil C and localised, site-specific C. Underlined the need for reference sites at Level 2.

Question 6: Could SOC be a cost-effective part of the monitoring?

Points in general discussion

- SOC should be determined using CN analyser; thus, N also determined at little extra cost.

Conclusion

Soil SOC is almost a given. It is such a fundamental soil property, determining soil fertility, soil structural stability, retention of organics, sequestration of C and nutrient cycling. Modern analytical methods are cheap and accurate. Given the major effect of land use on equilibrium SOC levels, real thought is needed on the extent of stratification at Level 1 (to ensure statistical power for main soil types, main land uses, main habitat classes); and the use of reference sites in Level 2 to factor out the effects of changing land use and allow any climate change signal to be distinguished.

Breakout Group Summary - Soil pH

Question 1: Is it a relevant indicator of environmental interaction?

Individual initial comments

Mike Hornung (MH): Yes. Indicator of nutrient availability and Al^{3+} toxicity.
Pete Smith (PS): Yes. Trigger values likely to vary widely depending on context.
Heleina Black (HB): Yes. Valuable indicator of change giving information on environmental drivers. Integrator for air and water but do need other information to interpret.
Roger Unwin (RU): Yes.

Points in general discussion

- general habitat indicator/underlies vegetation change;
- proxy for risk of metal leaching.

Question 2: What does change mean?

Individual initial comments

HB: plant ranges are a surrogate for pH change.
RU: only interpretable with triggers.

Points in general discussion

- need other information to interpret changes in pH;
- need to stratify sampling to ensure areas of interest are adequately covered;
- significant spatial context - location of sensitive/important soils/habitats;
- the scale of change is important - national means and distribution vs local change not necessarily bad.

Question 3: What rate of change is important?

Individual initial comments

JS: significance of a given rate depends on how near a threshold/trigger the value was.
DS: 0.5 pH unit between pHs of 4.5-6 over 5 years would be significant.
HB: local change might trigger more extensive national/regional sampling.

Points in general discussion

- need to remember the scale is log;
- the significance of a given rate of change would depend on proximity to threshold and rate of change;
- soil solution measurements are more sensitive than bulk soil;
- not considered useful to report a national mean pH - more sensible to report distributions;

- some habitats will be sensitive to very low rates of change (such as already acid habitats, Al^{3+} leaching).

Question 4: What should trigger further analysis?

Individual initial comments

MH: need to identify trigger values for all the broad habitats.
HB: issue of whole community vs most sensitive species – likelihood is that tolerance narrows to most sensitive species.

Points in general discussion

- management is likely to mask many changes (such as lime use);
- real friction between setting triggers at existing levels or at ecologically derived thresholds.
- need to distinguish carefully between trigger values (which prompt further investigation) and target values (remediation/restoration objectives);
- Tier 1 triggers should always be the most sensitive.

Question 5: Can we discern/interpret long-term trends?

Individual initial comments

RU: Yes, already do. Need measure every 5 years, possible more frequently at Level 2.
JS: Yes we can discern trends, we already do.
DS: Yes. Measure every 2-3 years, possibly five.
MH: Tier 1 every five years; Tier 2 sensitive sites every year.
HB: 8-10 yerars for national monitoring; 2-4 years for Tier 2 sensitive sites; monthly for Tier 3?
PS: Relatively straightforward to do statistical power analysis using existing data to work out ability to discern changes over time.

Points in general discussion

- could do soil solution on a selection of sites.

Question 6: Could pH be a cost-effective part of the monitoring?

Points in general discussion

- for a very minor additional cost, it would be sensible to do pH in both water and $CaCl_2$.

Conclusion

Soil pH is a relevant indicator of environmental interaction. It could be included as a cost-effective part of a national soil monitoring scheme. It gives direct information about metal leaching and bioavailabilty (particularly Al^{3+}), habitat and vegetation support, and indirect information about aerial deposition and management effects. Existing long-term datasets indicate that it is possible to discern long-term trends. Recommendations are that it is measured every 5 years at Tier 1 sites; more frequently at Tiers 2 and 3. Trigger values prompting movement between tiers could be derived but these would have to be in the form of a matrix: habitat vs soil function.

Critical rates of change could also be identified; these would depend on proximity to trigger values.

Breakout Group Summary - Total nitrogen and potentially mineralisable N (PMN)

Note: These were indicators that had not made the initial selection and so the discussion was structured differently.

Question 1: Is it a relevant indicator of environmental interaction?

Individual initial comments

Mike Hornung (MH): Total N alone is of little use; C/N ratio needs to be done, particularly on litter and fresh inputs.
Heleina Black (HB): N must be done with C to yield C/N ratios. Not sure about usefulness of PMN.
Roger Unwin (RU): Sceptical about total N; happier with PMN.
Brian Chambers (BC): Total N yes mainly because of usefulness of C/N ratios; supports PMN but in Level 2.
Jim Stevens (JS): Supports N because of C/N.
David Scholefield (DS): Supports total N; likes PMN but has serious concerns over methodologies, lack of agreed protocols.

Points in general discussion

- point was made that the assumption that C/N ratios in topsoils fall between 9 and 12 very southern lowland biased - many upland soils show wider variation;
- C/N ratio was useful as indicator of acidification and hence metal leaching risk in upland soils;
- position on PMN mixed: some felt it was a useful integrative biological indicator, others had concerns about methodology. Lack of coherent historic data was raised as a drawback.

Question 3: What rate of change is important?

Points in general discussion

- more work needed on existing datasets to identify acceptable and unacceptable rates of change for Total N;
- concern that the lack of coherent data for PMN made it impossible to specify rates of change.

Question 4: What should trigger further analysis?

Points in general discussion

- identifying trigger values not useful;
- Direction of change more useful, particularly when considering C/N ratios. This C/N increasing may indicate a system “bleeding N”.

Question 6: Could Total N and PMN be a cost-effective part of the monitoring?

Points in general discussion

- consensus was that total N was cheap to do, given that C was already being determined on a C/N analyser and that the value of C/N ratios justified its inclusion;
- PMN elicited no such consensus. Concern over methodologies and costs meant it would be more appropriate to Level 2 but still more work need to investigate its biological relevance to environmental interaction.

Conclusion

Total N should be part of the Level 1 monitoring. PMN could be included in Level 2 but a lot more work is needed on methodology and interpretation before it can be included.

Appendix iii

Extrapolated datasets used at the technical workshop

Soil Bulk Density: Soil Quality Indicator Evaluation

Introduction

Soil bulk density was one of the supporting indicators identified in phase I of a project to identify potential indicators of soil quality[1]. Indicators were classified according to the key soil function they pointed to. Bulk density from 0-50 cm was highlighted as a supporting indicator of **environmental interaction/providing raw materials (water).**

The criteria employed are:

1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator?
5. What rate of change in the indicator is important?
6. What should trigger a closer analysis of this indicator?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on soil bulk density for use in the meeting on 28/29th June. If you have other relevant soil bulk density data, please bring it on 28/29th.

Challenging proposed indicators

The bulk density of a soil layer is a parameter that increases and decreases in response to land use activities and environmental conditions. These include:

- agricultural tillage;
- trafficking by vehicles, animals or humans;
- land levelling, installation of underdrainage and so on;
- soil organic matter changes;
- physical processes such as consolidation and shrink-swell;
- weather factors, such as frost and rainfall.

Some of the induced changes are ephemeral, some are seasonal but others may result in long-term changes to soil bulk density.

Information sources

There are few, if any, datasets on long-term changes in soil bulk density so the information in this paper is based on a variety of published and unpublished sources from the author's own research over a 30 year period:

- Soil Survey of England and Wales soil physics laboratory data from 1970s and 1980s (Hall *et al*., 1977)
- Land Research Associates soil physical measurements on restored and undisturbed land in the 1990s (Land Research Associates 1998 and Reeve *et al*., 1999)

The dataset (Bulk density spreadsheet.xls) has been created by using measurements derived from multiple locations on plots on a trial site in Hertfordshire which had undergone different histories and compactive forces. The basic data has been randomly extrapolated over a 50-year period but with the introduction of certain trends based on real situations.

Assumptions

1. Bulk density of the fine earth fraction is the key parameter – bulk density of the whole soil (if stones are present) merely introduces another variable.
2. Bulk density as a SQI will need to be measured within the topsoil, say at 10 cm or 15 cm depth, <u>and</u> at the top of the subsoil where long-term changes are likely to be reflected.
3. Bulk density would have to be measured when soils are moist, to allow reproducible sampling and avoid the efforts of shrinkage.
4. Minimum bulk density thresholds are less likely to be defined than maximum bulk density thresholds.
5. A standard error of measurement of 0.02 g/cm^3 has been assumed for the purpose of illustration.

Results

Bulk density spreadsheet.xls includes the bulk density dataset created for this exercise and two charts (Figures 1 and 2 below), one of the topsoil bulk densities over a 50 year period, one of the bulk density at the top of a subsoil over a 50-year period and one chart showing both trends. The bulk density data is created as measured in April – this would be well after autumn cultivations but only shortly after any spring cultivations that might be made.

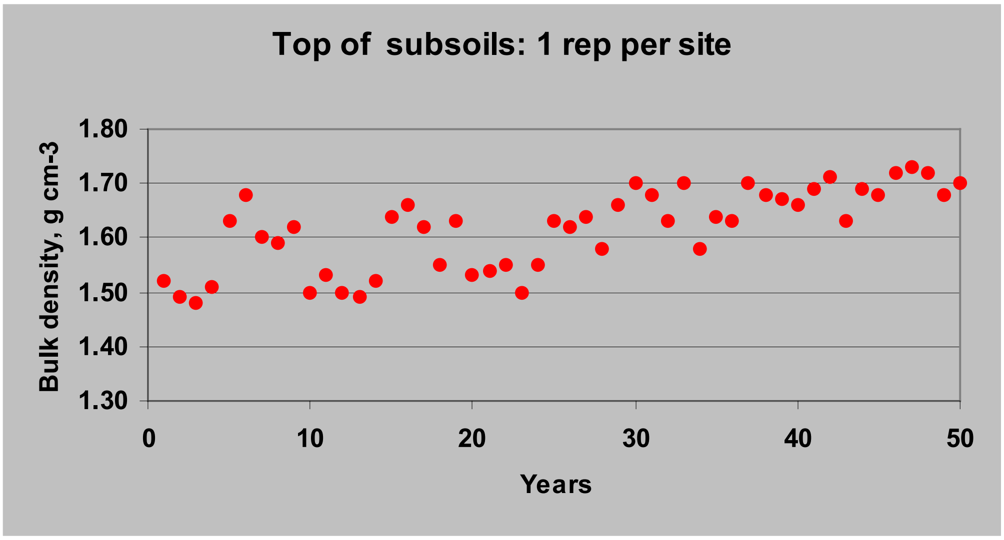


Figure 1: One replicate per site taken annually

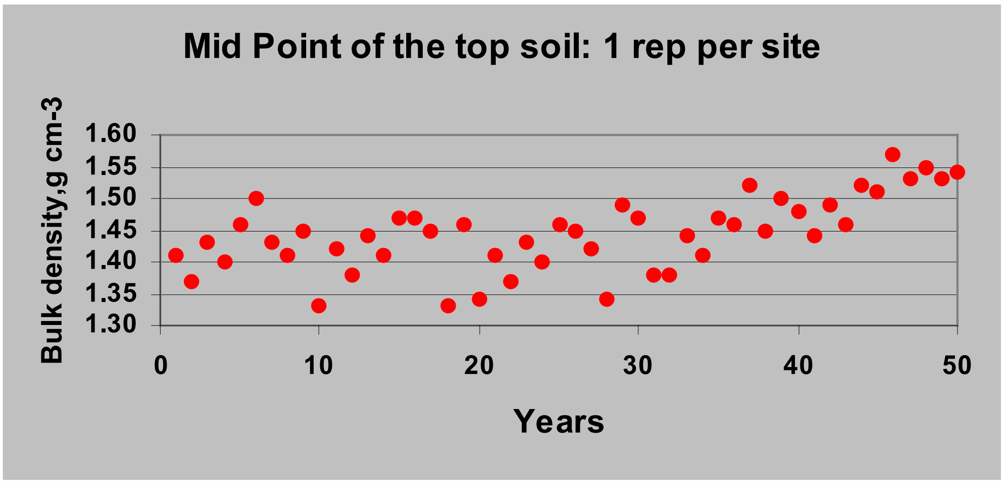


Figure 2: One replicate per site taken annually

Questions

Using these data (graphs and the Excel spreadsheet) and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is bulk density a relevant environmental indicator?
- Is it sufficiently sensitive to changes in environmental pressures (such as climate change)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce the effects of climate change?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Malcolm Reeve of Land Research Associates Ltd for the data generation and scenarios.

References

Hall, D.G.M, Reeve, M.J, Thomasson, A.J and Wright, V.F (1977). *Water Retention, Porosity and Density of Field Soils*. Soil Survey Technical Monograph 9. Harpenden.

Reeve, M.J., Heaven, F.W. and Duncan, N.A. The Bush Farm Experiment 15 years on: recovery in soil and agricultural quality. In *Land Reclamation: Achieving sustainable benefits*, eds. H.R.Fox, H.M.Moore & A.D.McInstosh. Balkema, Rotterdam.

Reeve, M.J., Heaven, F.W. and Duncan, N.A. (2000). *Evaluation of Mineral Sites Restored to Agriculture*. Defra (www.defra.gov.uk/environ/landuse/restore).

Soil Organic Carbon: Soil Quality Indicator Evaluation

Introduction

Soil organic carbon (SOC) was one of the headline indicators identified in phase I of a project to identify potential indicators of soil quality[1]. Indicators were classified according to the key soil function they pointed to. SOC was highlighted as a key indictor of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:

1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator?
5. What rate of change in the indicator is important?
6. What should trigger a closer analysis of this indicator?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on SOC for use in the meeting on 28/29th June. If you have other relevant SOC data, please bring it on 28/29th.

Challenging proposed indicators

The approach has been to use the Roth-C model to mimic a situation where 100 years of monitoring data is available and is being examined for signs of significant environmental change. Pete Smith and Michael Haft of the University of Aberdeen have used four scenarios based on the long-term experiments at Rothamsted, against which the model has been tuned:

- No land use or climate change based on the long-term dataset from Park Grass
- No land use change (Park Grass) but with a climate change driver based on the Med-High scenario of the UK Climate Impact Panel, UKCIP (see appendix for details)[2].
- Land use change: arable to woodland regeneration, no climate change, based on the Geescroft Wilderness data.
- Land use change with climate change: arable to woodland regeneration, climate change included based on the Med-High scenario of UKCIP.

For the purposes of this exercise, the model outputs are taken to represent the actual changes in SOC which would result from changes in either land use or climate.

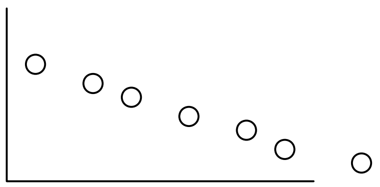
Adding spatial variability to the modelled curves

The model curves present an idealised picture of trends. In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by ‘fuzzying’ the modelled curves using estimates of the spatial variability in SOC. In this exercise we are using an assumed standard deviation for SOC equivalent to 15% of the mean (P.Smith *Pers. Comm.*)

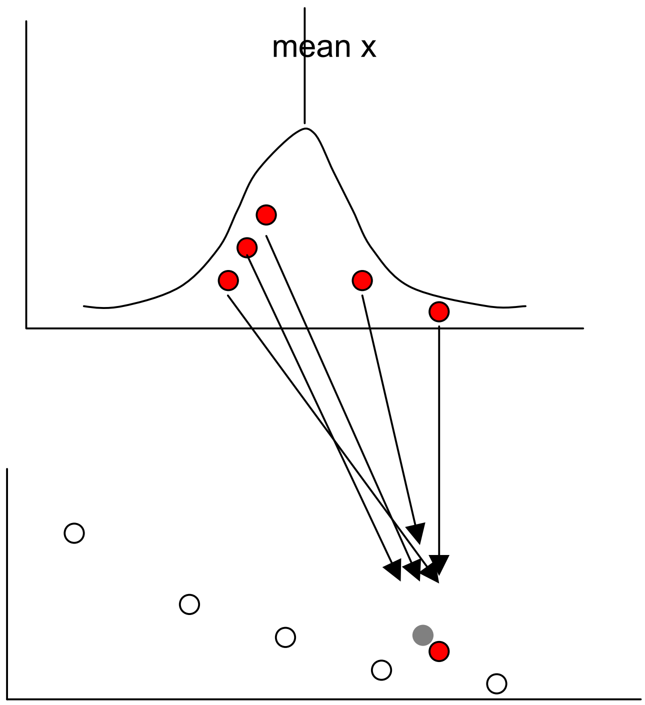
Thus, we assume that each point on the modelled “ideal” curve represents the mean value of a population which has a standard deviation equivalent to 15% of that mean value. We then regenerate that population using a probability package such as @Risk and sample it either once or five times to mimic the taking of one or five replicates at each sampling site.

The procedure is shown diagrammatically below.

1. Generate ideal model curve using ROTH-C



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y



Sample that population either once, five or fifteen times to regenerate measured mean value.

Hence, reconstitute a ‘roughed-up’ curve.

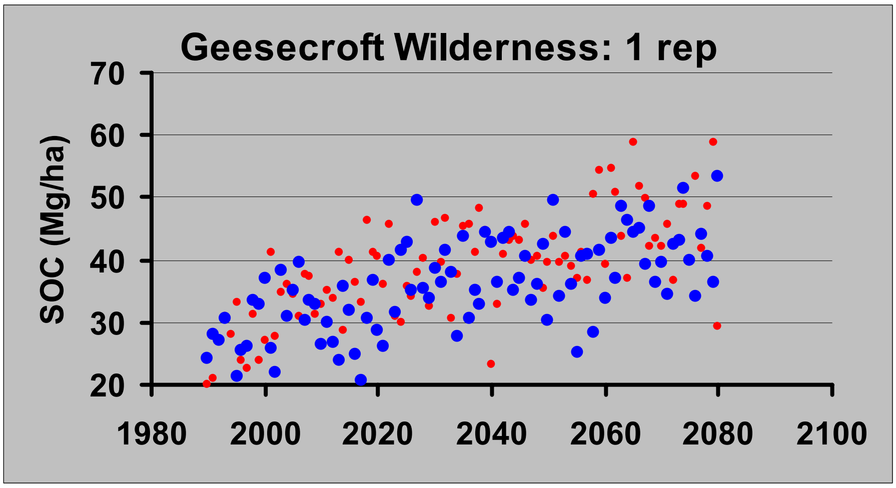
The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the mean value). We are also presenting you with Excel spreadsheets with the tabulated data on which you can perform more detailed statistics. That sheet also contains data for 10 replicates.

Geesecroft Wilderness

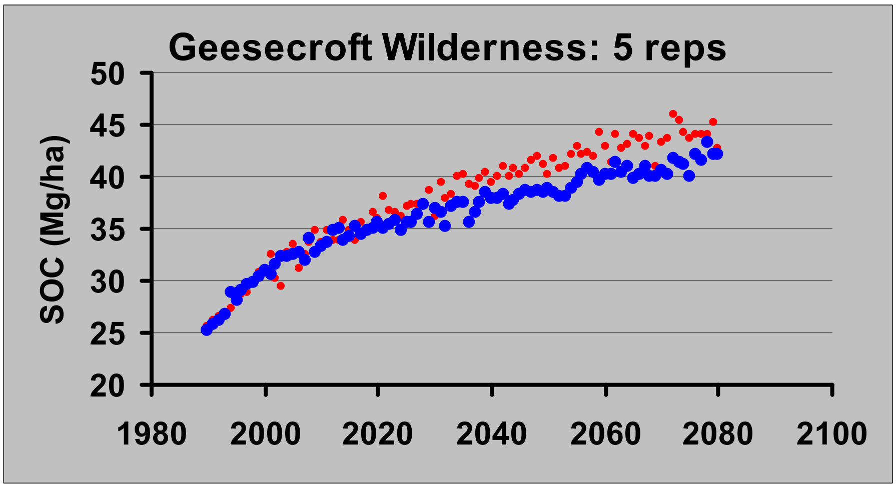
Land Use change: arable to woodland

Climate change scenario: none (red points); or med-high scenario (blue points).

One replicate per site taken annually



Five replicates per site taken annually

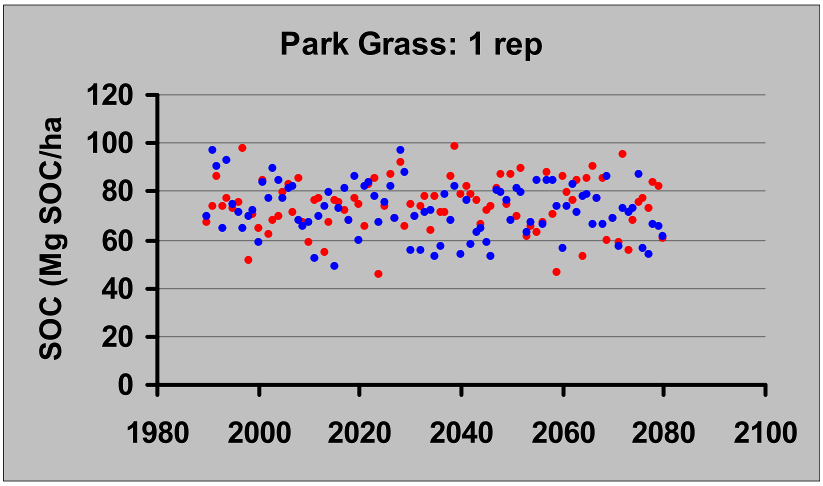


Park Grass

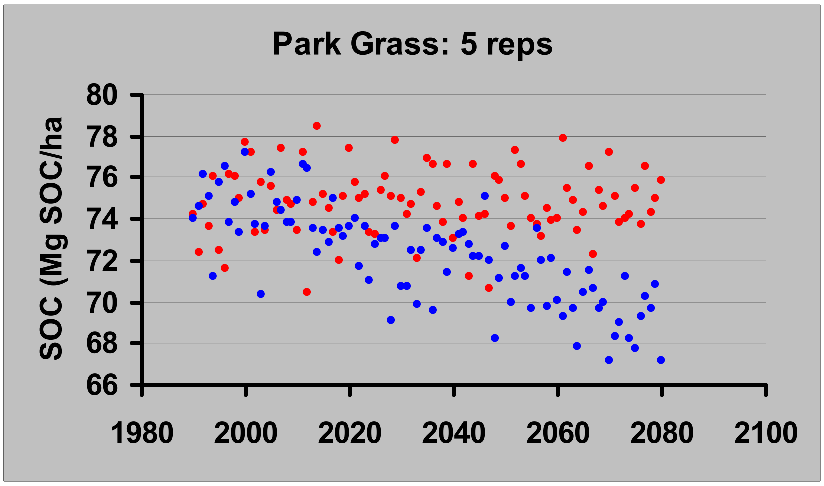
Land use change: none, permanent grass

Climate change scenario: none (red points); or med-high scenario (blue points)

One replicate per site taken annually



Five replicates per site taken annually



Questions

Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is SOC a relevant environmental indicator?
- Is it sufficiently sensitive to changes in environmental pressures (such as climate change)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce the effects of climate change?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Pete Smith and Michael Haft of the University of Aberdeen for generating the ROTH-C simulations.

References

1. Loveland, P. J. & Thompson, T. R. E. 41 (Environment Agency, Bristol, 2002).
2. Hulme, M. & Jenkins, G. J. 61 (Climate Reserach Unit, Norwich, 1998).

Appendix

The med-high climate change scenario is set out in detail in Hulme and Jenkins. One way of summarising its predictions is to set out the percentage of years for future thirty-year periods which would experience climate extremes. For temperature and rainfall, these are:

	1961-1990	2020s	2050s	2080s
Mean temperature				
hot august (+3.4 °C)	*2*	*15*	*32*	*40*
warm august (+1.06 °C)	*6*	*59*	*85*	*99*
Rainfall				
<50% average summer rainfall	*1*	*7*	*12*	*10*
2 year total <90% of average	*12*	*11*	*14*	*6*

Note: differences are with respect to 1961-1990 actual, not modelled climate.

Soil Total Nitrogen: Soil Quality Indicator Evaluation

Introduction

Soil total nitrogen was **not** one of the headline indicators identified in phase I of a project to identify potential indicators of soil quality[1]. We are including it for consideration as one of the indicators we are not proposing to include and the challenge, therefore, is to champion its inclusion.

The criteria employed are:
1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator?
5. What rate of change in the indicator is important?
6. What should trigger a closer analysis of this indicator?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on total N for use in the meeting on 28/29th June. If you have other relevant total N data, please bring it on 28/29th.

Challenging proposed indicators

Total nitrogen values have been derived from the soil organic carbon values simulated using Roth-C model (see the delegate pack on organic carbon for details). Carbon: nitrogen ratios for most topsoils fall between 9 and 12, with an average of 10 to 11. To derive total N from soil carbon, we have assumed a distributed C/N ratio with a mean of 11 and a standard error of one.

The scenarios are:
- No land use or climate change based on the long-term dataset from Park Grass
- No land use change (Park Grass) but with a climate change driver based on the med-high scenario of the UK Climate Impact Panel, UKCIP (see appendix for details)[2].
- Land use change: arable to woodland regeneration, no climate change, based on the Geescroft Wilderness data.
- Land use change with climate change: arable to woodland regeneration, climate change included based on the med-high scenario of UKCIP.

For the purposes of this exercise, the model outputs are taken to represent the actual changes in total N which would result from changes in either land use or climate.

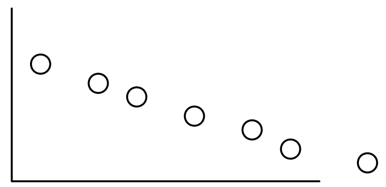
Adding spatial variability to the modelled curves

The model curves present an idealised picture of trends. In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by ‘fuzzying’ the modelled curves using estimates of the spatial variability in total N. In this exercise we are using an assumed standard deviation for total N equivalent to 15% of the mean, as for soil organic carbon.

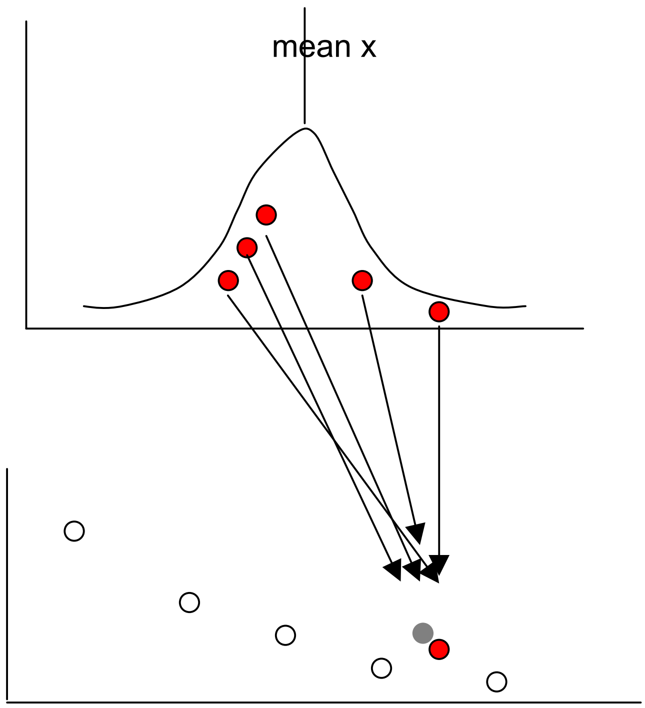
Thus, we assume that each point on the modelled ideal curve represents the mean value of a population which has a standard deviation equivalent to 15% of that mean value. We then regenerate that population using a probability package such as @Risk and sample it either once or five times to mimic the taking of one or five replicates at each sampling site.

The procedure is shown diagrammatically below.

1. Generate ideal model curve using ROTH-C



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y



Sample that population either once, five or fifteen times to regenerate measured mean value.

Hence, reconstitute a ‘roughed-up’ curve.

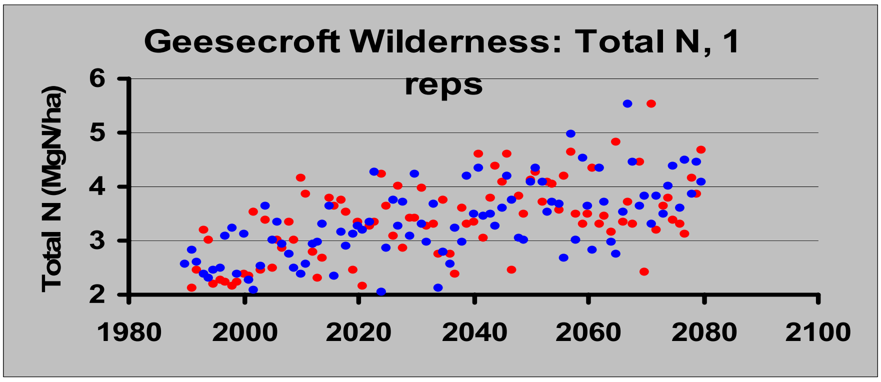
The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the mean value). We are also presenting you with Excel spreadsheets with the tabulated data on which you can perform more detailed statistics. That sheet also contains data for 10 replicates.

Geesecroft Wilderness

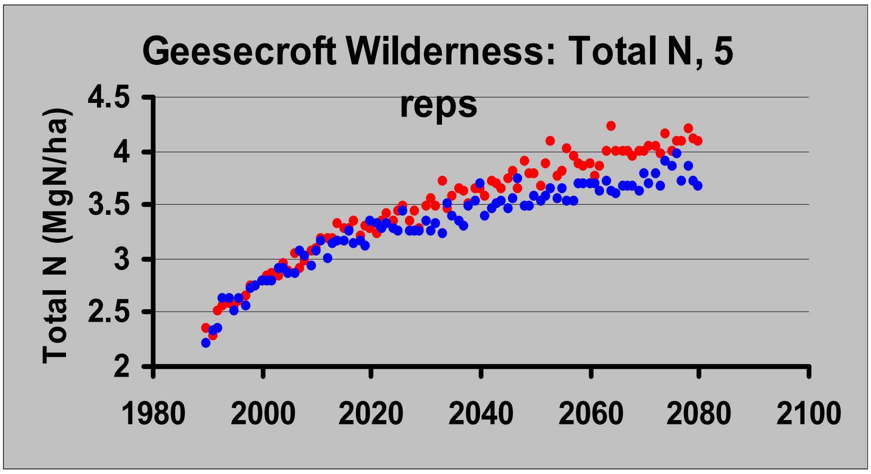
Land Use change: arable to woodland

Climate change scenario: none (red points); or med-high scenario (blue points).

One replicate per site taken annually



Five replicates per site taken annually

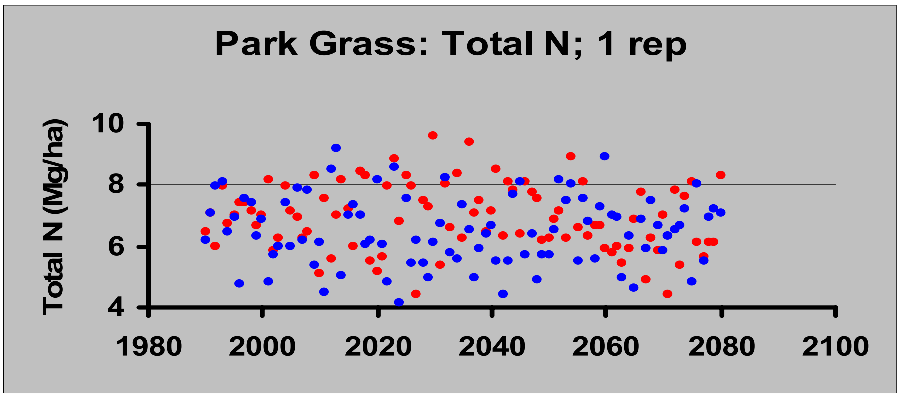


Park Grass

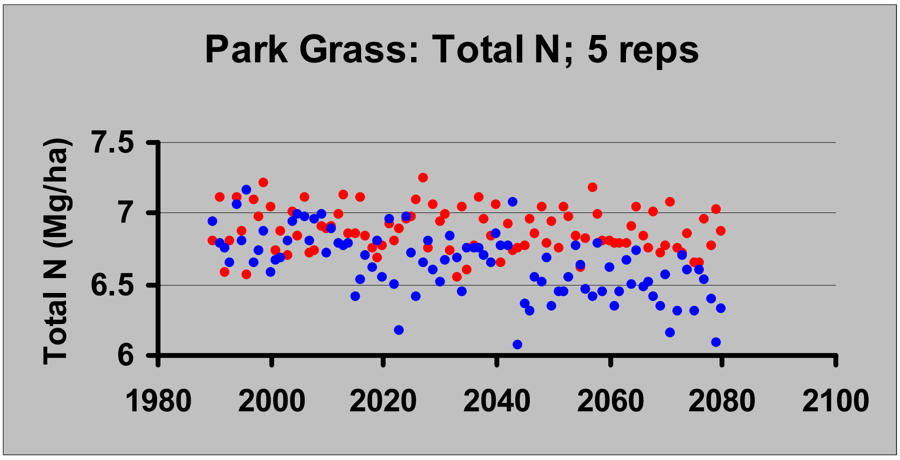
Land use change: none, permanent grass

Climate change scenario: none (red points); or med-high scenario (blue points)

One replicate per site taken annually



Five replicates per site taken annually



Questions

Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is total N a relevant environmental indicator? Does it tell us any more than SOC?
- Is it sufficiently sensitive to changes in environmental pressures (such as climate change)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce the effects of climate change?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Pete Smith and Michael Haft of the University of Aberdeen for generating the ROTH-C simulations.

References

1. Loveland, P. J. & Thompson, T. R. E. 41 (Environment Agency, Bristol, 2002).
2. Hulme, M. & Jenkins, G. J. 61 (Climate Research Unit, Norwich, 1998).

Appendix

The med-high climate change scenario is set out in detail in Hulme and Jenkins. One way of summarising its predictions is to set out the percentage of years for future thirty year periods which would experience climate extremes. For temperature and rainfall, these are:

	1961-1990	2020s	2050s	2080s
Mean temperature				
hot august (+3.4 °C)	*2*	*15*	*32*	*40*
warm august (+1.06 °C)	*6*	*59*	*85*	*99*
Rainfall				
<50% average summer rainfall	*1*	*7*	*12*	*10*
2 year total<90% of average	*12*	*11*	*14*	*6*

Note: differences are with respect to 1961-1990 actual, not modelled climate.

Soil Aggregate Stability: Soil Quality Indicator Evaluation

Introduction

Topsoil aggregate stability was one of the indicators identified in phase I of a project to identify potential indicators of soil quality [1]. Indicators were classified according to the key soil function they pointed to. Aggregate stability was highlighted as a key indictor of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:

1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator?
5. What rate of change in the indicator is important?
6. What should trigger a closer analysis of this indicator?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on aggregate stability for use in the meeting on 28/29th June. If you have other relevant data, please bring it on 28/29th.

Challenging proposed indicators

We could identify no long-term datasets on topsoil aggregate stability and have chosen to estimate it from soil organic matter using the relation derived by Kemper and Koch[2]:

$$\text{Aggregate stability} = 40.8+17.6*\log(\text{SOM}) +0.73*\log(\%\text{clay})$$

which on arable soils explained 44% of the observed variance.

As with soil total N, we have used the modelled estimates of soil organic carbon (SOC), corrected to SOM, derived from the Roth-C model. Details are given in the delegate pack on SOC.

We have derived aggregate stability for the following scenarios:

- No land use or climate change based on the long-term dataset from Park Grass
- No land use change (Park Grass) but with a climate change driver based on the Med-High scenario of the UK Climate Impact Panel, UKCIP (see appendix for details)[3].
- Land use change: arable to woodland regeneration, no climate change, based on the Geescroft Wilderness data.
- Land use change with climate change: arable to woodland regeneration, climate change included based on the Med-High senario of UKCIP.

For the purposes of this exercise, the model outputs are taken to represent the actual changes in aggregate stability which would result from changes in either land use or climate.

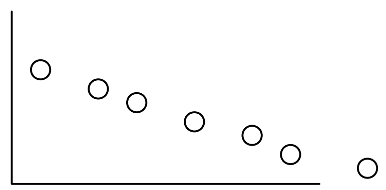
Adding spatial variability to the modelled curves

The model curves present an idealised picture of trends. In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by 'fuzzying' the modelled curves using estimates of the spatial variability in aggregate stability. In this exercise we are using statistical data on aggregate stability from Clement and Williams[4] who measured aggregate stability at 29 points in a single experimental block and obtained a standard deviation equivalent to 18% of the mean. We will assume a standard deviation equivalent to 20% of the mean.

Thus, we assume that each point on the modelled ideal curve represents the mean value of a population which has a standard deviation equivalent to 20% of that mean value. We then regenerate that population using a probability package such as @Risk and sample it either once or five times to mimic the taking of one or five replicates at each sampling site (note the accompanying spreadsheet also has data for 10 replicates).

The procedure is shown diagrammatically below.

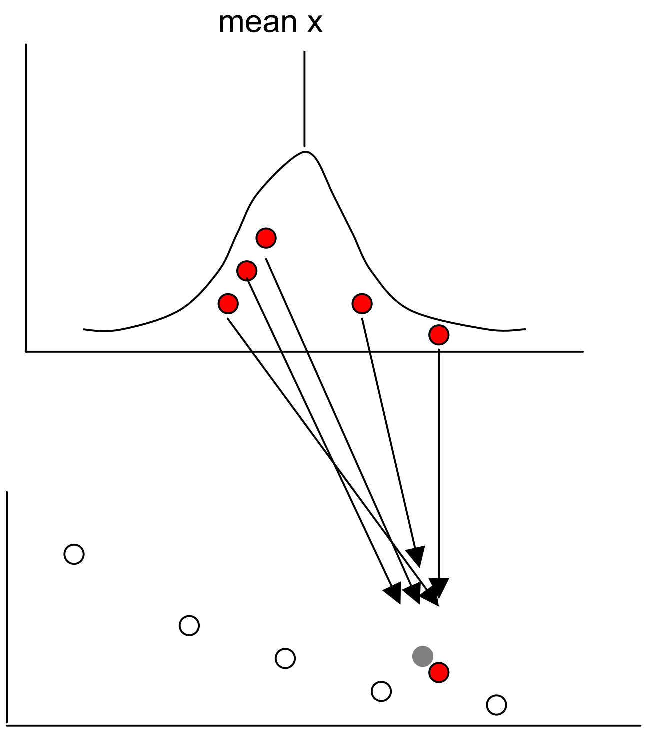
1. Generate ideal model curve using ROTH-C and Kemper and Koch equation for aggregate stability



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y

Sample that population either once, five or fifteen times to regenerate measured mean value.

Hence, reconstitute a 'roughed-up' curve.



The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the mean value).

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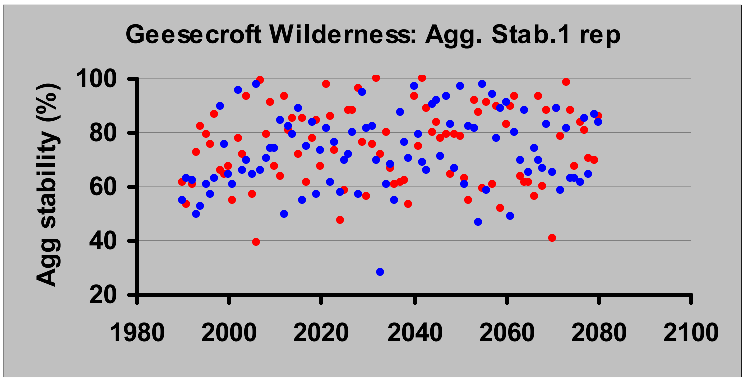
Geesecroft Wilderness

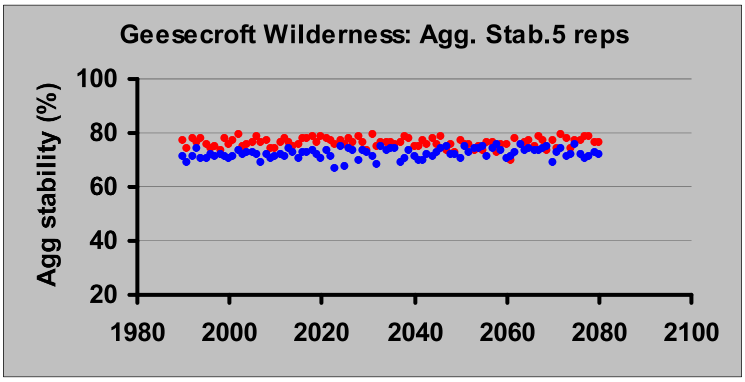
Land use change: arable to woodland

Climate change scenario: none (red points); or med-high scenario (blue points).

One replicate per site taken annually



Five replicates per site taken annually

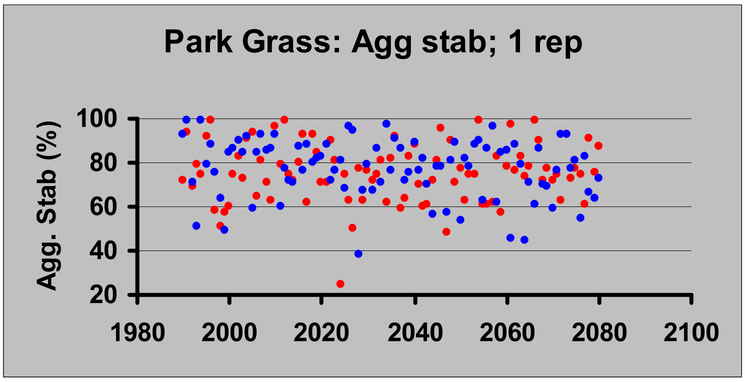


Park Grass

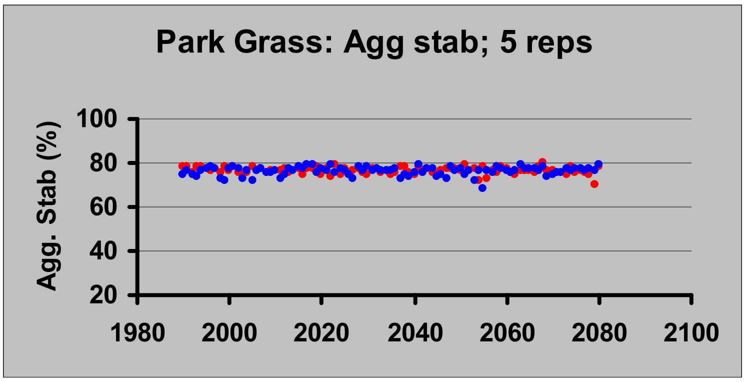
Land use change: none, permanent grass

Climate change scenario: none (red points); or med-high scenario (blue points)

One replicate per site taken annually



Five replicates per site taken annually



Questions

Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is aggregate stability a relevant environmental indicator?
- Is it sufficiently sensitive to changes in environmental pressures (such as climate change)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce the effects of climate change?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Pete Smith and Michael Haft of the University of Aberdeen for generating the ROTH-C simulations.

References

1. Loveland, P. J. & Thompson, T. R. E. 41 (Environment Agency, Bristol, 2002).
2. Kemper, W. D. & Koch, E. J. (USDA, Washington DC, 1966).
3. Hulme, M. & Jenkins, G. J. 61 (Climate Reserach Unit, Norwich, 1998).
4. Clement, C. R. & Williams, T. E. An examination of the method of aggregate analysis by wet sieving in relation to the influence of diverse leys on arable soils. *Journal of Soil Science* **9**, 252-266 (1958).

Appendix

The med-high climate change scenario is set out in detail in Hulme and Jenkins. One way of summarising its predictions is to set out the percentage of years for future thirty year periods which would experience climate extremes. For temperature and rainfall, these are:

	1961-1990	2020s	2050s	2080s
Mean temperature				
hot august (+3.4 °C)	*2*	*15*	*32*	*40*
warm august (+1.06 °C)	*6*	*59*	*85*	*99*
Rainfall				
<50% average summer rainfall	*1*	*7*	*12*	*10*
2 year total<90% of average	*12*	*11*	*14*	*6*

Note: differences are with respect to 1961-1990 actual, not modelled climate.

Soil Macroporosity: Soil Quality Indicator Evaluation

Introduction

Soil macoporosity was **not** one of the headline indicators identified in phase I of a project to identify potential indicators of soil quality[1]. We are including it for consideration as one of the indicators we are not proposing to include and the challenge, therefore, is to champion its inclusion.

The criteria employed are:
1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator?
5. What rate of change in the indicator is important?
6. What should trigger a closer analysis of this indicator?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on soil macroporosity for use in the meeting on 28/29th June. If you have other relevant soil macroporosity data, please bring it on 28/29th.

Challenging proposed indicators

The macroporosity of a soil layer is a parameter that increases and decreases in response to land use activities and environmental conditions. These include:

- agricultural tillage and trafficking by vehicles, animals or humans;
- soil organic matter change;
- physical processes such as consolidation and shrink-swell;
- weather-related factors, such as frost (disaggregation)and rainfall (slaking).

Some of the induced changes are ephemeral, some are seasonal but others may result in long-term changes to macroporosity.

Information sources and assumptions

There are few, if any, datasets on long-term changes in macroporosity so the information in this paper is based on a variety of published and unpublished sources from research over a 30-year period on natural and grossly disturbed soils (Hall et al, 1977; Reeve et al., 1998, 2000).

Macroporosity as measured in the laboratory at specific negative potentials (such as 5 kPa) is reasonably well correlated with bulk density in soil layers that have not undergone gross disturbance by cultivation (Hall et al 1977; Reeve 1986; Douglas et al, 1986). Using a regression equation, macroporosity has been calculated for the top of the subsoil from the bulk density dataset generated for the bulk density SQI. A similar approach might be possible in grassland topsoils but, in cultivated topsoils, macroporosity can vary considerably at a given density according to the cultivation method employed.

Results

Macroporosity spreadsheet.xls includes the data and one chart (Figure 1 below), representing macroporosity over a 50-year period as measured in April – this would be well after autumn cultivations but only shortly after any spring cultivations that might be made.

Questions

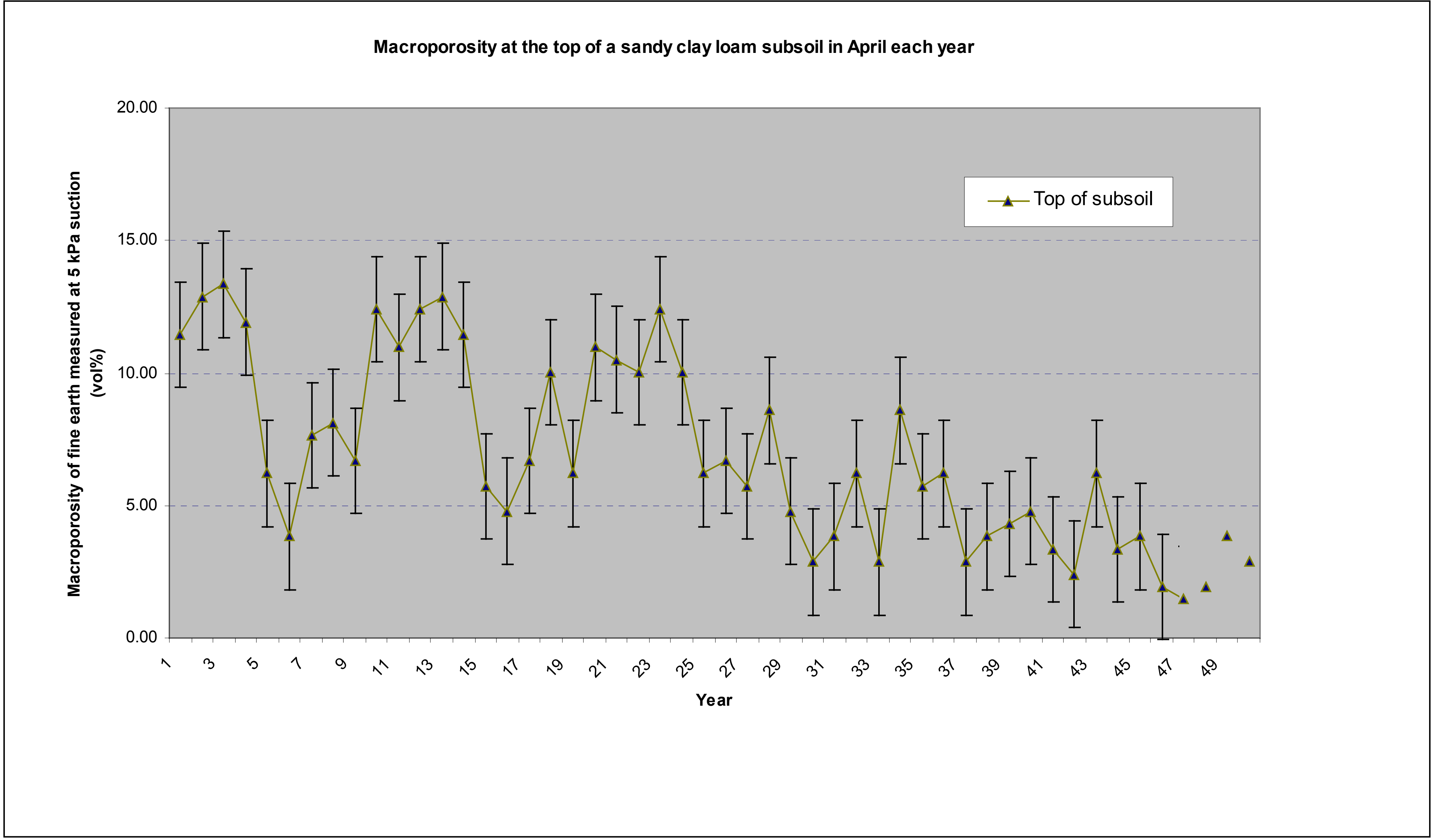
Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is macroporosity a relevant environmental indicator?
- Is it sufficiently sensitive to changes in environmental pressures (such as climate change)?
- Would it be sufficient to infer this indicator from bulk density data rather than measuring it? For topsoils? For subsoils?
- Is it possible to discern, *and interpret*, long-term trends in measurements that could inform policy decisions?
- Is it possible to identify trigger values, either general or for different soil types, and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Malcolm Reeve of Land Research Associates Ltd for the data generation and scenario.

Figure 1: Eight replicates per site taken annually



References

Douglas, J.T, Jarvis, M.G, Howse, K.R & Goss, M.J. (1986) Structure of a silty soil in relation to management. *Journal of Soil Science*. 37, 137

Hall, D.G.M, Reeve, M.J, Thomasson, A.J and Wright, V.F (1977). *Water Retention, Porosity and Density of Field Soils*. Soil Survey Technical Monograph 9. Harpenden.

Reeve, M.J. (1986). Water retention, porosity and composition interrelationships of alluvial soils in mid Hawke's Bay. *New Zealand Journal of Agricultural Research*. 29, 457-468

Reeve, M.J., Heaven, F.W. and Duncan, N.A (1998) The Bush Farm Experiment 15 years on: recovery in soil and agricultural quality. In *Land Reclamation: Achieving sustainable benefits*, eds. H.R. Fox, H.M. Moore & A.D. McInstosh. Balkema, Rotterdam.

Reeve, M.J., Heaven, F.W. and Duncan, N.A. (2000). *Evaluation of Mineral Sites Restored to Agriculture*. DEFRA (www.defra.gov.uk/environ/landuse/restore)

Soil Olsen P: Soil Quality Indicator Evaluation

Introduction

Soil Olsen P was one of the speculative indicators identified in phase I of a project to identify potential indicators of soil quality[1]. Indicators were classified according to the key soil function they pointed to. A measure of extractable P was highlighted as an indictor of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:

1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator (Olsen P)?
5. What rate of change in the indicator (Olsen P) is important?
6. What should trigger a closer analysis of this indicator (Olsen P)?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on Olsen P for use in the meeting on 28/29th June. We welcome any other data on Olsen P you may be aware of, particularly medium-term temporal data.

Challenging proposed indicators

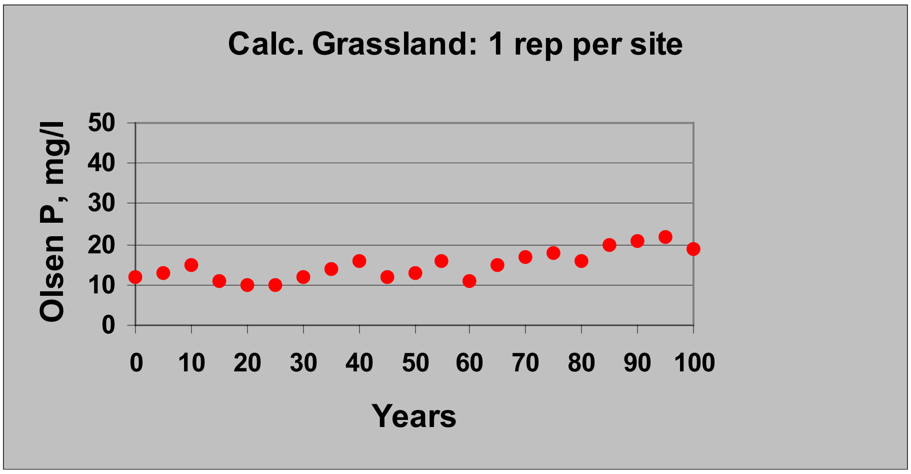
The approach has been to generate 100 years. Estimates of historic N and S deposition, current values and predicted future trend scenarios have been used to estimate topsoil pH.

For Olsen P, Rob Parkinson at Plymouth University has generated data for the effect of two land use and soil type scenarios:

- Scenario 1 is for a calcareous grassland with a progressive increase in grazing pressure from sheep and cattle (9 months per year) and hence returns over the period. The soil is a clay loam, with relatively high organic matter status, pH of 7.5 and is moderately well drained. The annual rainfall is 700 mm, the slopes are low/medium and erosion loss is low.
- Scenario 2 is for productive arable land, which has a long history of over use of P (50-70 kg P_2O_5/year) (immobilisation problem). A recent decline in OM and other pressures has lead to high erosional risk although the slope is low to medium. The soil pH at the site is 6.5, the OM status relatively low and the drainage good. The annual rainfall in the scenario is 700 mm.

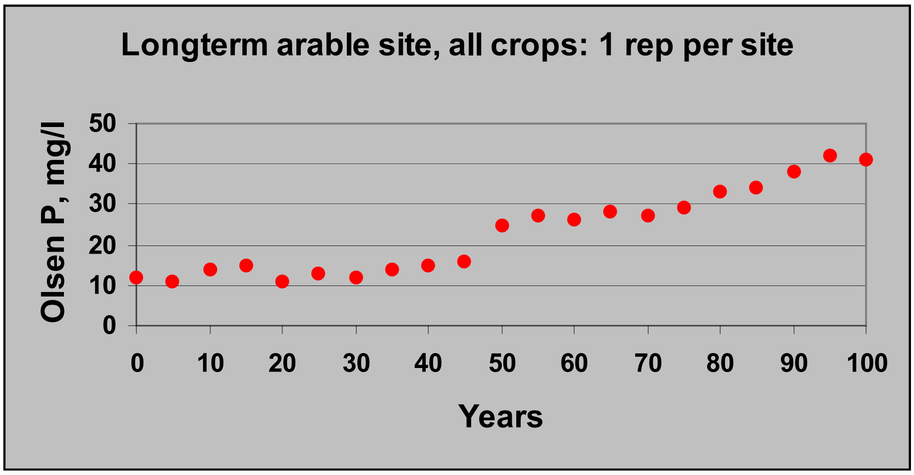
Scenario 1

One replicate per site taken every 5 years



Scenario 2

One replicate per site taken every 5 years



Questions

Using this data (graphs) and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is topsoil Olsen P a relevant environmental indicator?
- Is it sensitive to changes in environmental pressures (such as climate change)?
- Is it possible to discern, *and interpret*, long-term trends in measurements that could inform policy decisions?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Rob Parkinson for providing the data simulations and scenarios.

References

1. Loveland, P. J. & Thompson, T. R. E. 41 (Environment Agency, Bristol, 2002).

Soil Organic Contaminants (Poly-Brominated Diphenyl Ether or PBDE congeners): Soil Quality Indicator Evaluation

Introduction

The concentration of organic pollutants was one of the supporting indicators identified in phase I of the Environment Agency project to identify potential indicators of soil quality (Loveland and Thompson 2002). Indicators were classified according to the key soil function they pointed to. The concentration of organic pollutants in the topsoil was highlighted as a key indicator of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:

1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5 or 10 years?
4. Can we interpret the changes in the indicator (soil organic contaminant concentrations)?
5. What rate of change in the indicator (soil organic contaminant concentrations) is important?
6. What should trigger a closer analysis of this indicator (soil organic contaminant concentrations)?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out some data on soil organic contaminant concentrations for an organic chemical of current regulatory interest for use in the meeting on 28/29th June. We welcome any other data on soil organic contaminant concentrations you may be aware of that would be relevant to our discussions.

Challenging proposed indicators

The approach has been to use data from the literature to project a situation where 100 years of monitoring data is available and is being examined for signs of significant environmental change.

For soil organic contaminant concentrations, we have simulated the effect of adding three polybrominated diphenyl ether (PBDE) congeners (numbers 47, 99 and 153) to soil background concentrations via annual atmospheric deposition and addition of sewage sludge every other year.

Information sources used were:

- Background concentrations (Hassanin et al. 2004);
- Atmospheric addition rates (Hassanin et al. 2005);
- Sewage sludge concentrations (Oberg et al. 2002);
- First order half-lives (150 days for all congeners) were estimated using EPIWIN software (Wania and Dugani 2003).

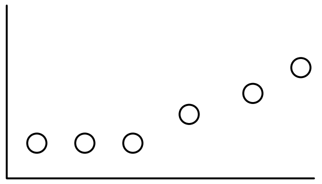
The scenario assumes a soil with a bulk density of 1.3 and mixing to 25 cm depth.

Adding spatial variability

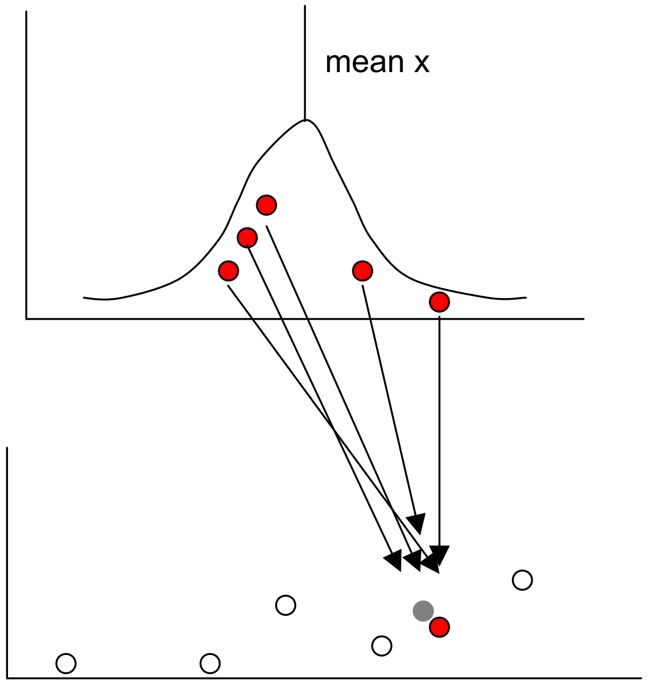
In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by ‘fuzzying’ the modelled curves by assuming that measurements on replicate soil samples will follow a lognormal distribution with a standard deviation of 20% of the mean value, and sampling from this distribution with Crystal Ball software.

The procedure is shown diagrammatically below.

1. Generate model curve using mean background, addition, and half-life dissipation values



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y



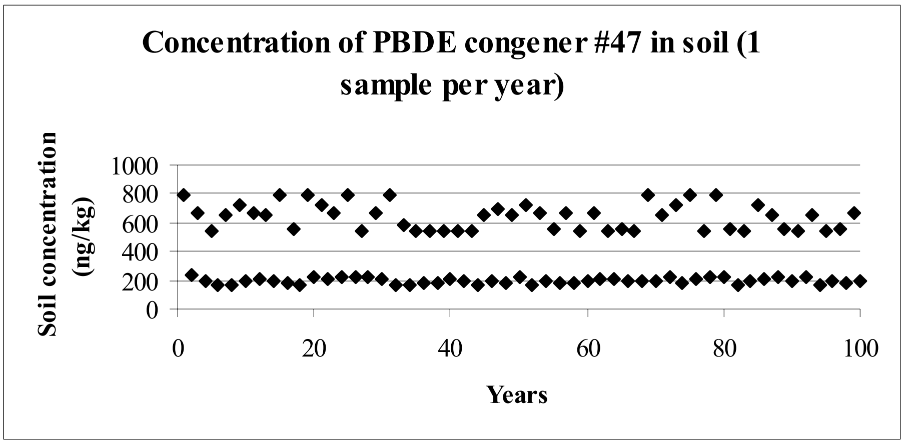
Sample that population either once, five or fifteen times to regenerate measured mean value.

Hence, reconstitute a ’roughed-up’ curve.

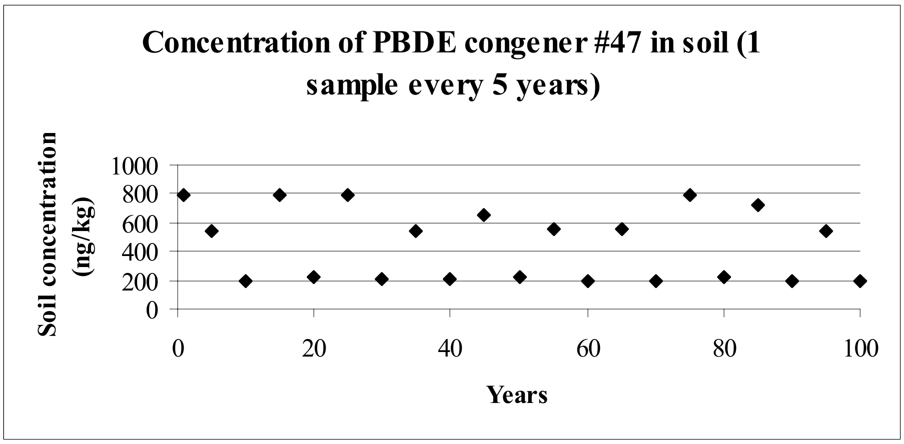
The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the 'mean' value).

PBDE congener #47 (Atmospheric deposition plus sludge additions)

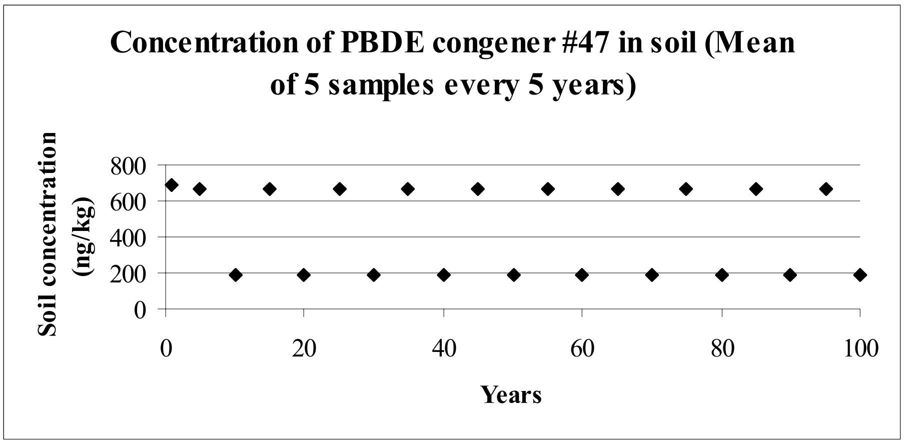
One replicate per site taken annually



One replicate per site taken every 5 years

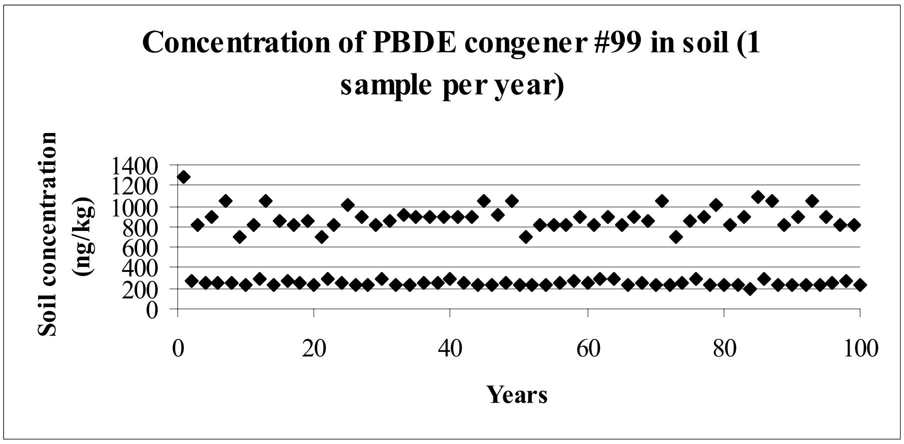


Five replicates per site taken every five years

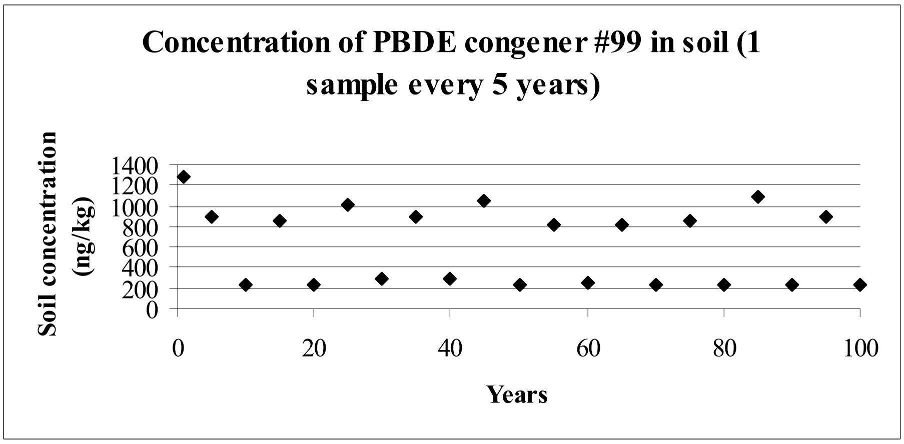


PBDE congener #99 (Atmospheric deposition plus sludge additions)

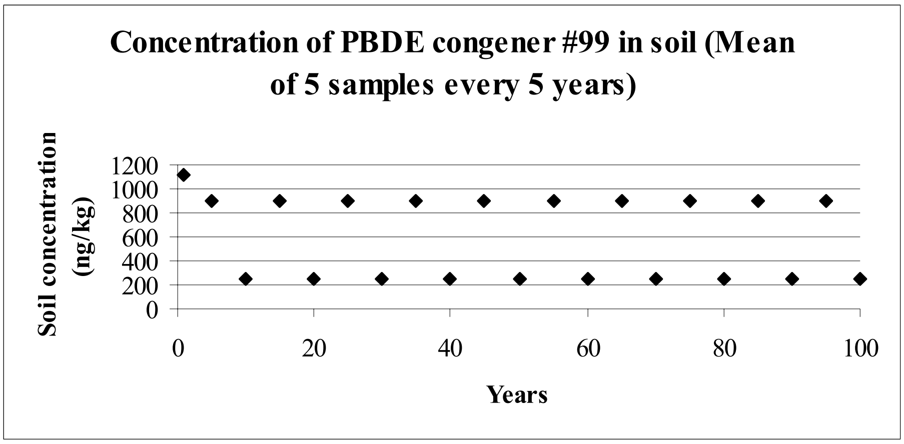
One replicate per site taken annually



One replicate per site taken every 5 years

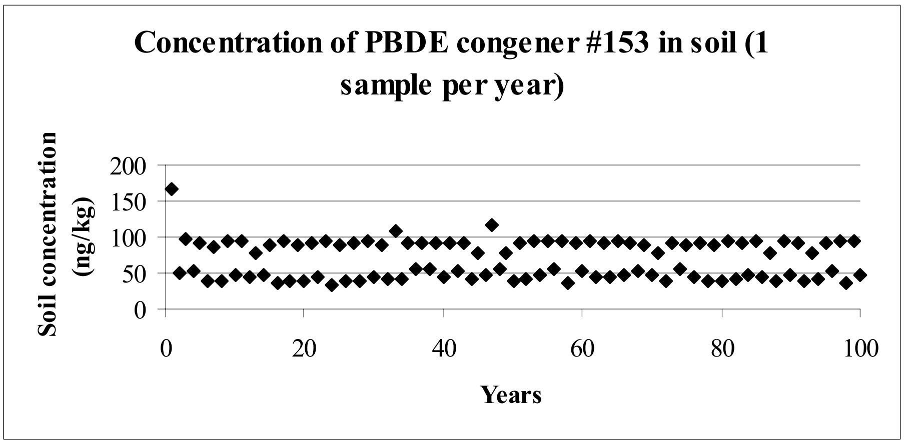


Five replicates per site taken every five years

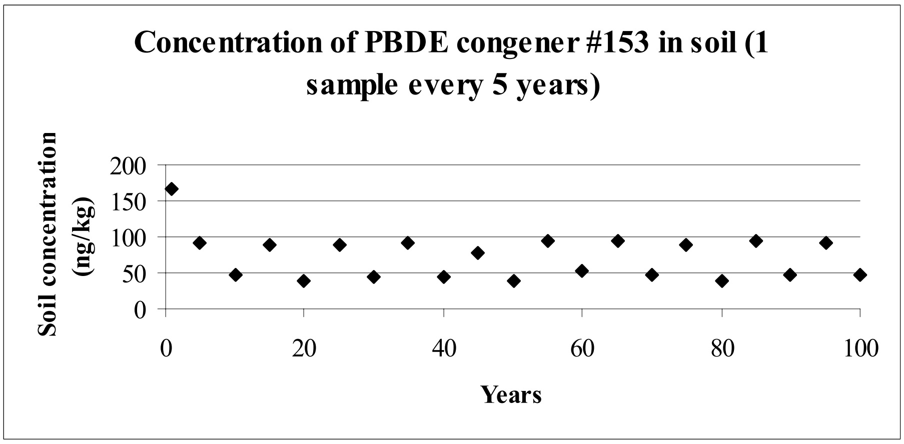


PBDE congener #153 (Atmospheric deposition plus sludge additions)

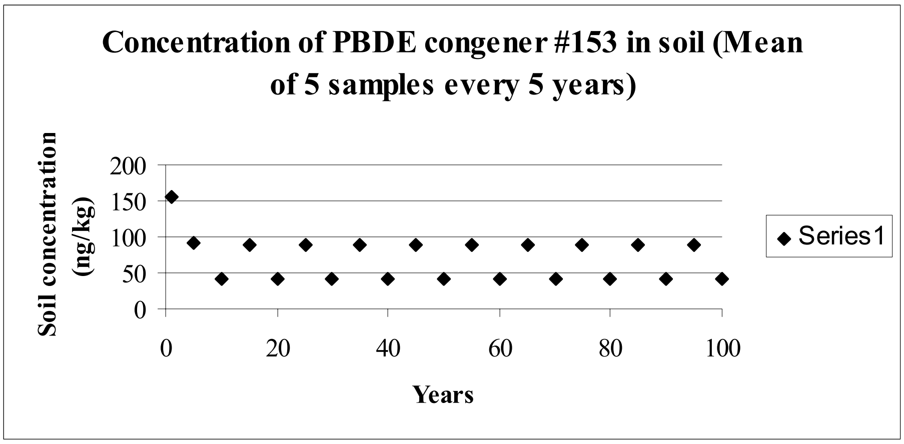
One replicate per site taken annually



One replicate per site taken every 5 years



Five replicates per site taken every five years



Questions

Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is topsoil PBDE concentration a relevant environmental indicator?
- Is its environmental chemistry, fate and behaviour representative of other typical organic chemical contaminants of concern? If not, what other substances should be chosen as representatives?
- Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural or industrial practice)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce metal addition?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is especially grateful to Mark Crane of Watts & Crane Associates for undertaking the data simulations.

References

Hassanin A, Breivik K, Meijer SN, Steinnes E, Thomas GO, Jones KC. 2004. PBDEs in European background soils: levels and factors controlling their distribution. *Environmental Science and Technology*, 38:738-745.

Hassanin A, Johnston AE, Thomas GO, Jones KC. 2005. Time trends of atmospheric PBDEs inferred from archived UK herbage. *Environmental Science and Technology*, 39:2436-2441.

Oberg K, Warman K, Oberg T. 2002. Distribution and levels of brominated flame retardants in sewage sludge. *Chemosphere*, 48:805-809.

Wania F, Dugani CB. 2003. Assessing the long-range transport potential of polybrominated diphenyl ethers: a comparison of four multimedia models. *Environmental Toxicology and Chemistry*,22:1252-1261.

Soil pH: Soil Quality Indicator Evaluation

Introduction

Soil pH was one of the headline indicators identified in phase I of a project to identify potential indicators of soil quality[1]. Indicators were classified according to the key soil function they pointed to. Topsoil pH was highlighted as a key indictor of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:
1. Is the parameter sensitive to changes in environmental drivers (such as temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5, 10 years?
4. Can we interpret the changes in the indicator (soil pH)?
5. What rate of change in the indicator (soil pH) is important?
6. What should trigger a closer analysis of this indicator (soil pH)?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out data on soil pH for use in the meeting on 28/29th June. We welcome any other data on soil pH you may be aware of.

Challenging proposed indicators

The approach has been to use the MAGIC model to mimic a situation where 100 years of monitoring data is available and is being examined for signs of significant environmental change. Estimates of historic N and S deposition, current values and predicted future trend scenarios have been used to estimate topsoil pH.

For soil pH, CEH Lancaster (Helaina Black, Ed Tipping, Andy Scott and Christopher Evans) have simulated the effect of two deposition scenarios:

- A linear reduction, starting in 2000, in both N and S deposition to approximately 40% of current values – the so-called **Gothenberg Protocol**;
- A linear reduction, also starting 2000, in only N deposition, with S deposition staying at current levels.

The deposition scenarios were applied to two soil types: a peaty podzol from Budworth in Cheshire; and a humo-ferric podzol from Plynlimon. In this exercise we will consider only the humo-ferric podzol.

For the purposes of this exercise, the model outputs are taken to represent the actual changes in soil pH which would result from reductions in either S and N deposition, or a reduction in only N deposition.

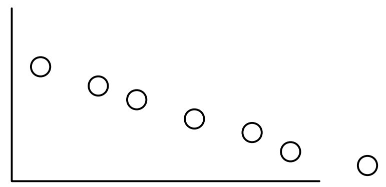
Adding spatial variability to the modelled curves

The model curves present an idealised picture of trends. In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by 'fuzzying' the modelled curves using estimates of the spatial variability in soil pH derived from measurements on replicate soil samples. Data from the ECN network suggest replicate measurements of soil pH over a small area show a standard deviation equivalent to approximately 7% of the mean. This is at the lower end of standard deviations reported in Adamson *et al* [2] who obtained standard deviations of soil pH ranging from 7-14%. In this exercise we will assume a standard deviation of 7%.

Thus, we assume that each point on the modelled ideal curve represents the mean value of a population which has a standard deviation equivalent to 7% of that mean value. We then regenerate that population using a probability package such as @Risk and sample it either once or five times to mimic the taking of one or five replicates at each sampling site (note the accompanying spreadsheet also has data for 10 replicates).

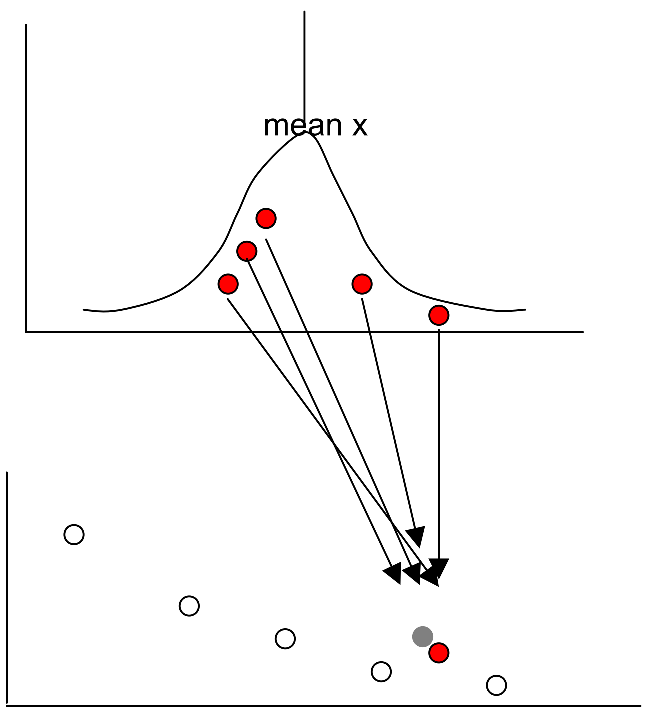
The procedure is shown diagrammatically below.

1. Generate ideal model curve using MAGIC



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y. Sample that population either once, five or fifteen times to regenerate measured mean value.

Hence, reconstitute a 'roughed-up' curve.

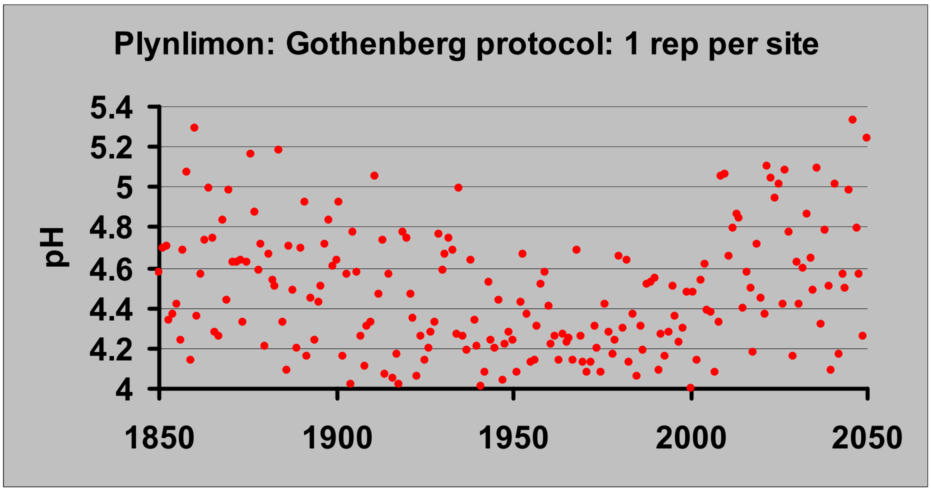


The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the mean value).

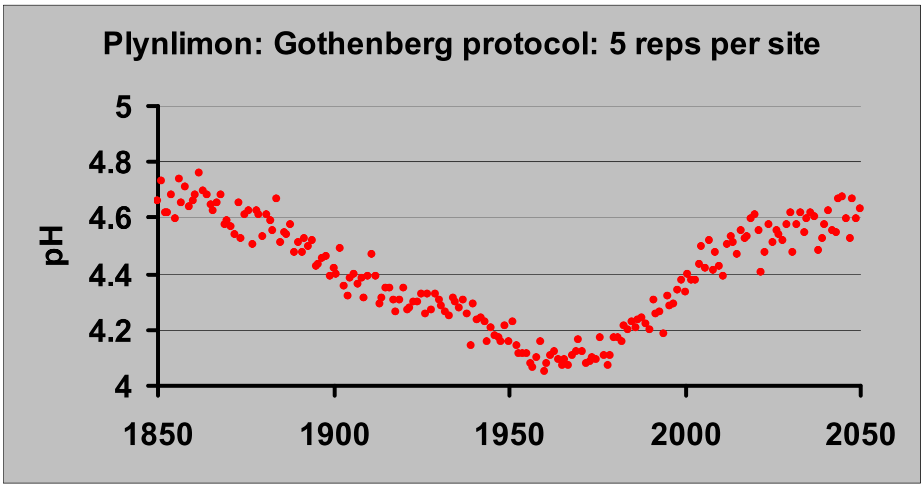
Note: the effects of the reduction protocols (Gothenberg or just N reduction) are only relevant after 2000.

Plynlimon

One replicate per site taken annually

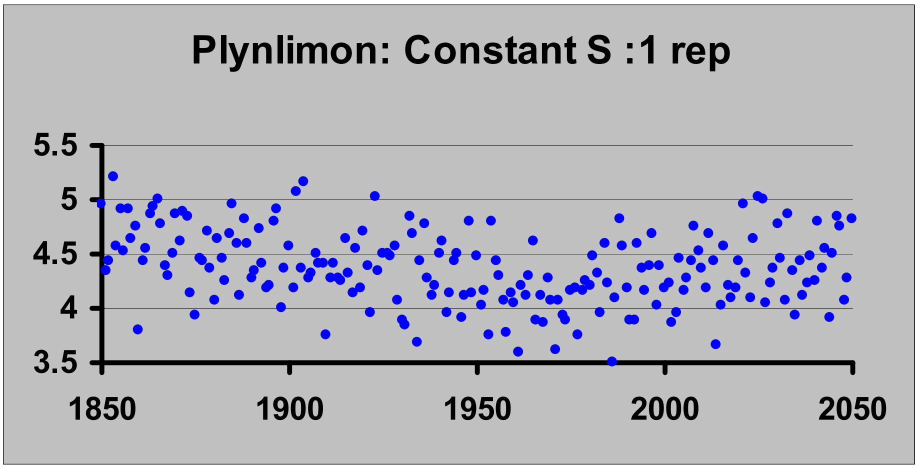


Five replicates per site taken annually



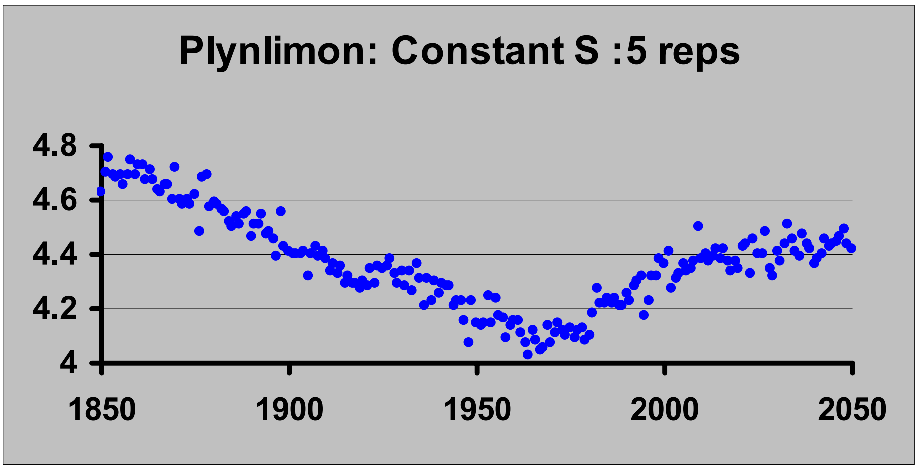
Plynlimon: N reduction; S deposition constant

One replicate per site taken annually



Five replicates per site taken annually

Five replicates per site taken annually



Using this data and any of your own, together with your own expertise in soils, we want to consider the following questions:

- Is topsoil pH a relevant environmental indicator?
- Is it sensitive to changes in environmental pressures (such as acid deposition)?
- Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce acid deposition?
- Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

Acknowledgements

The Environment Agency is grateful to Helena Black, Ed Tipping, Andy Scott and Christopher Evans of CEH Lancaster for generating the MAGIC simulations.

References

1. Loveland, P. J. & Thompson, T. R. E. 41 (Environment Agency, Bristol, 2002).
2. Adamson, J. K., Rowland, A. P., Scott, W. A. & Hornung, M. Changes in soil acidity and related variables over 25 years in the North Pennine Uplands, UK. *Soil Use and Management* **12**, 55-61 (1996).

Appendix iii

Soil Heavy Metals (Cu, Zn & Ni): Soil Quality Indicator Evaluation

Introduction

Soil heavy metal concentration was one of the supporting indicators identified in phase I of the Environment Agency project to identify potential indicators of soil quality (Loveland and Thompson, 2002). Indicators were classified according to the key soil function they pointed to. Topsoil heavy metal concentrations were highlighted as an indicator of **environmental interaction**.

We are now refining that original list of 67 potential indicators by challenging them using the following criteria:

1. Is the parameter sensitive to changes in environmental drivers (e.g. temperature, climate change)?
2. Is it possible to detect changes in the indicator against the noise of spatial variability, measurement error and so on?
3. At what frequency should the parameter be monitored – annually, every 5 or 10 years?
4. Can we interpret the changes in the indicator (soil heavy metals)?
5. What rate of change in the indicator (soil heavy metals) is important?
6. What should trigger a closer analysis of this indicator (soil heavy metals)?
7. Is this indicator fit for use in a cost-effective, broad-scale monitoring scheme?

This part of the delegate pack sets out some data on soil heavy metals for use in the meeting on 28/29th June. We welcome any other data on soil heavy metals you may be aware of.

Challenging proposed indicators

The approach has been to use data from the literature to project a situation where 100 years of monitoring data are available and being examined for signs of significant environmental change.

For soil heavy metals, we have simulated the effect of two scenarios:

- Scenario 1: Background concentration plus annual atmospheric deposition and addition of N and P fertilisers containing trace concentrations of Cu, Zn and Ni, with crop off-take and leaching;
- Scenario 2: Background concentration plus annual atmospheric deposition and addition of pig manure containing realistic concentrations of Cu, Zn and Ni, with crop off-take and leaching.

Information sources used were:

- background concentrations (Jones et al., 1987);
- metal addition rates (Nicholson et al., 1999, 2003);
- leaching losses (Keller et al., 2002);
- crop off-take (Chambers pers. comm.).

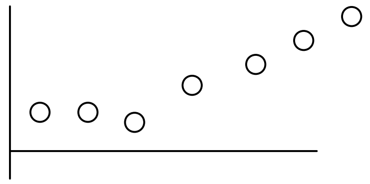
The scenarios assume a soil with a bulk density of 1.3 and mixing to 25 cm depth.

Adding spatial variability

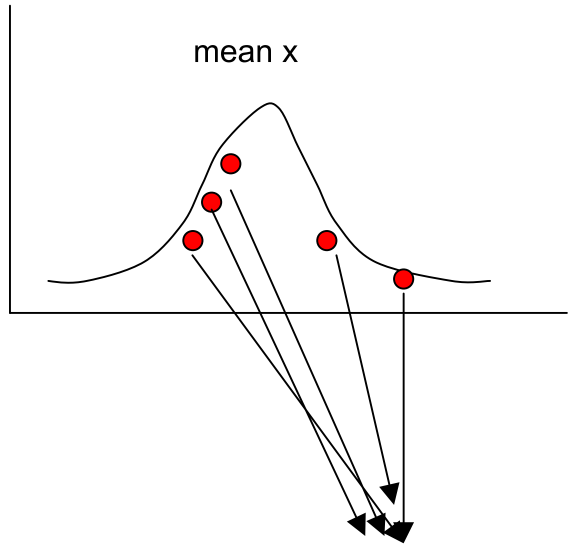
In reality, measurements will be subject to noise resulting from spatial variability and sampling error. We mimic this noise by ‘fuzzying’ the modelled curves by assuming that measurements on replicate soil samples will follow a normal distribution with a standard deviation of 20% of the mean value (except for background concentrations and leaching losses, for which measured inter-sample standard deviations were available in Jones et al., 1987, and Keller et al., 2002) and sampling from this distribution with Crystal Ball software.

The procedure is shown diagrammatically below.

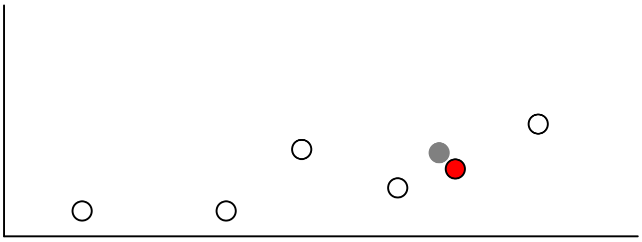
1. Generate model curve using mean background, addition, off-take and leaching values



2. Take each point as mean of a population with mean x (equal to the value of the point) and stdev y



Sample that population either once, five or fifteen times to regenerate measured mean value.

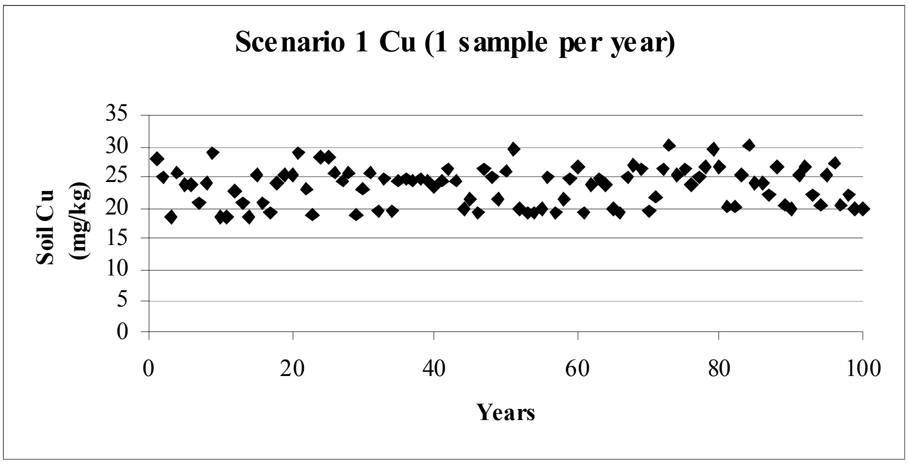


Hence, reconstitute a ’roughed-up’ curve.

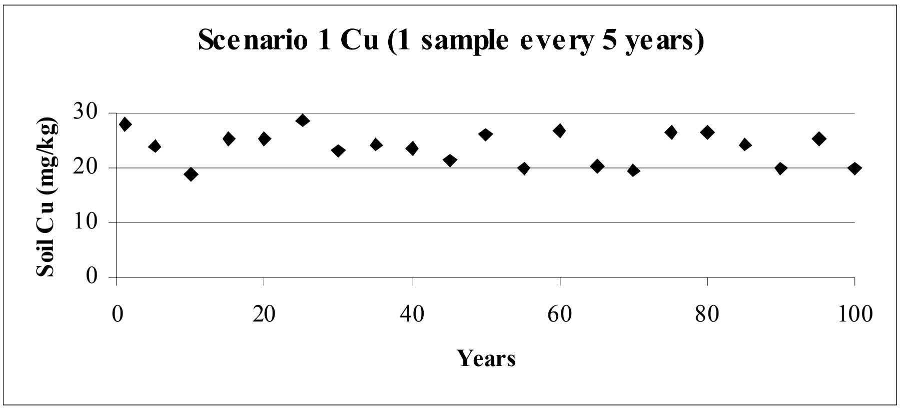
The graphs below summarise the results of such a procedure. Each point represents the mean annual value for a site derived from the number of replicates shown (where one replicate is taken, that is the ‘mean’ value).

Copper: Scenario 1 (Atmospheric deposition plus fertiliser additions)

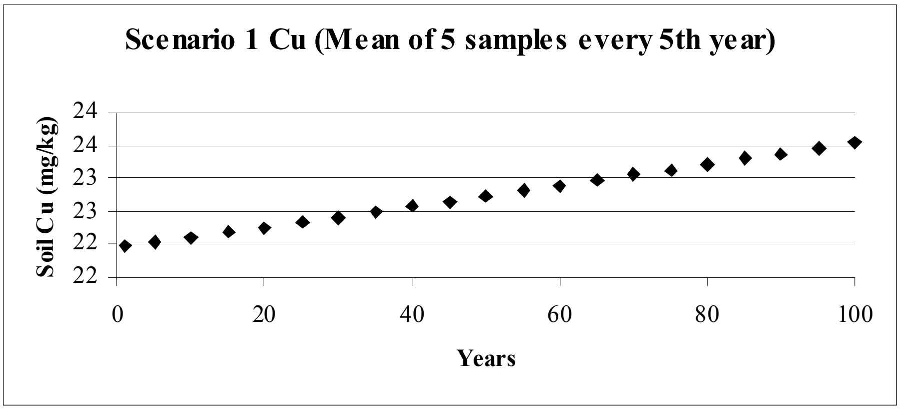
One replicate per site taken annually



One replicate per site taken every 5 years

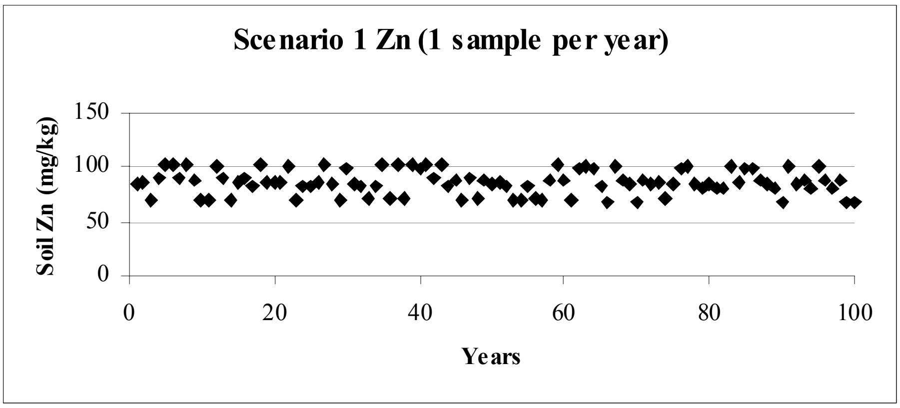


Five replicates per site taken every five years

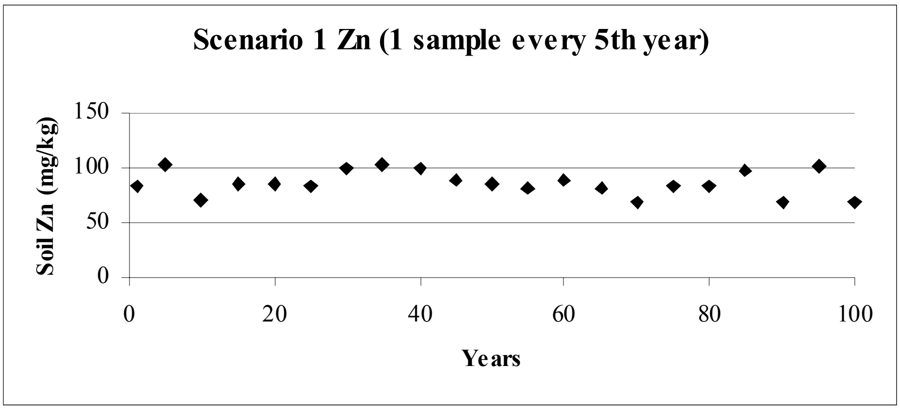


Zinc: Scenario 1 (Atmospheric deposition plus fertiliser additions)

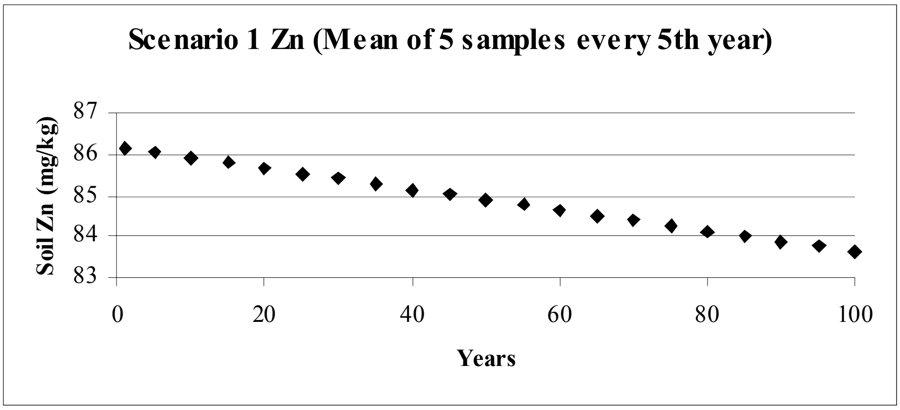
One replicate per site taken annually



One replicate per site taken every 5 years

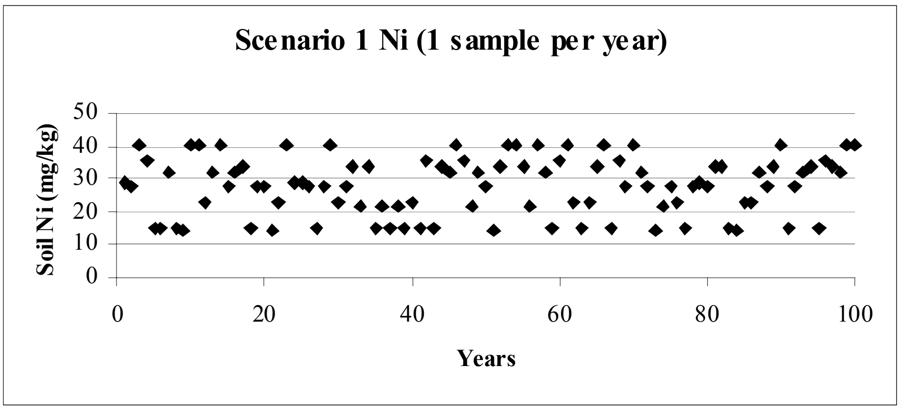


Five replicates per site taken every five years

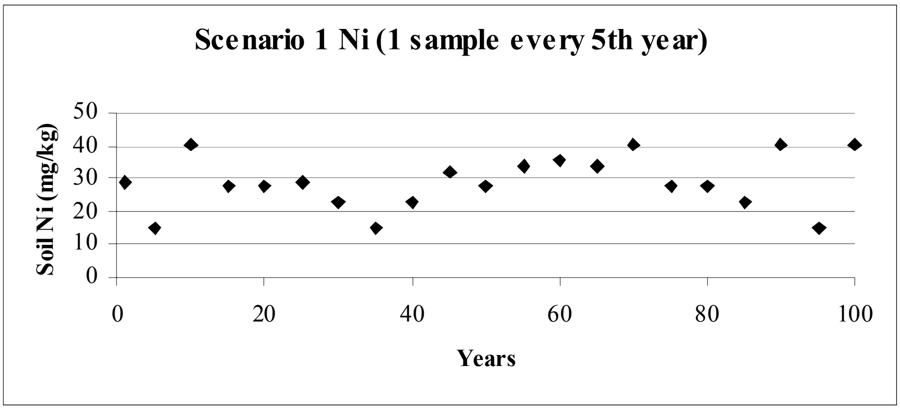


Nickel: Scenario 1 (Atmospheric deposition plus fertiliser additions)

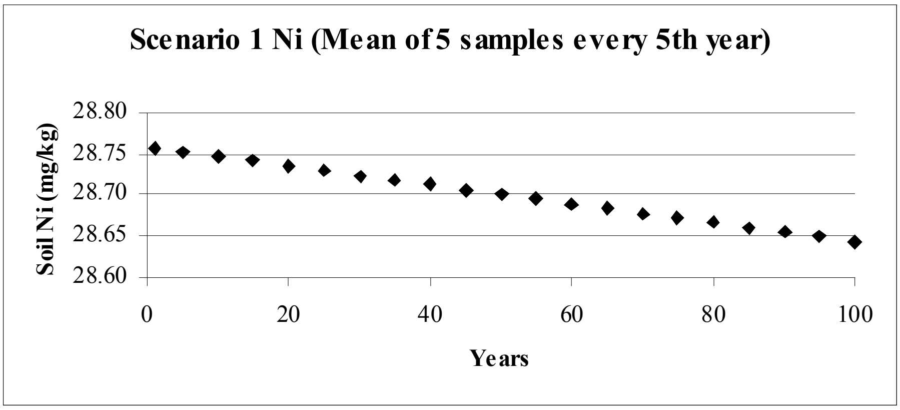
One replicate per site taken annually



One replicate per site taken every 5 years

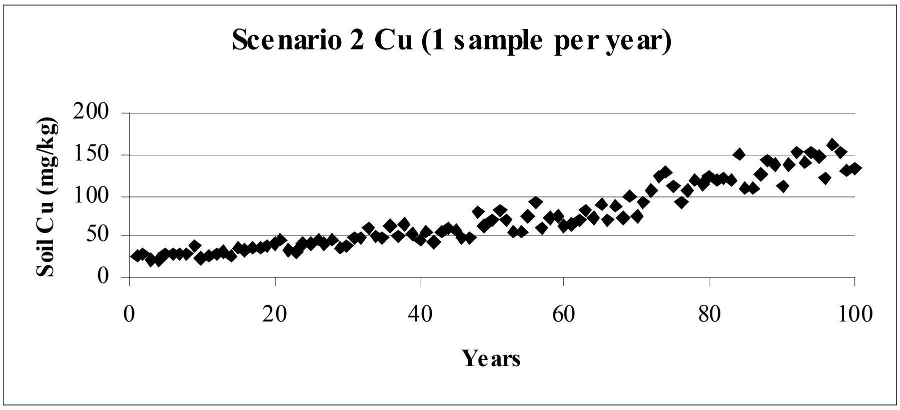


Five replicates per site taken every five years

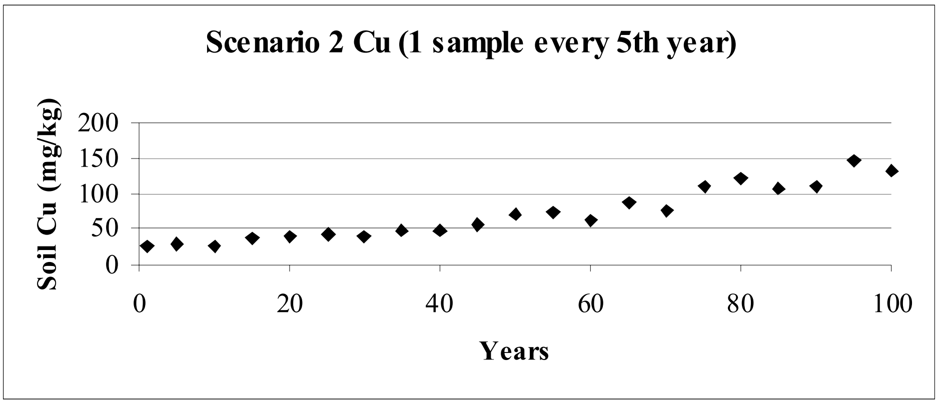


Copper: Scenario 2 (Atmospheric deposition plus pig manure additions)

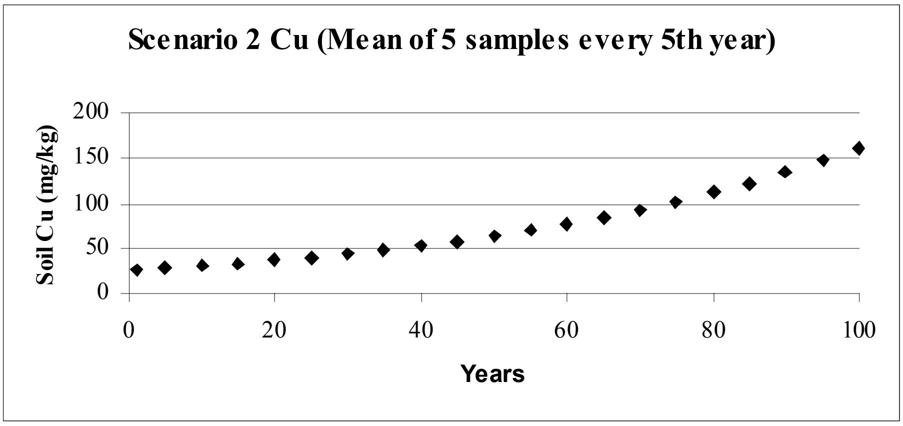
One replicate per site taken annually



One replicate per site taken every 5 years

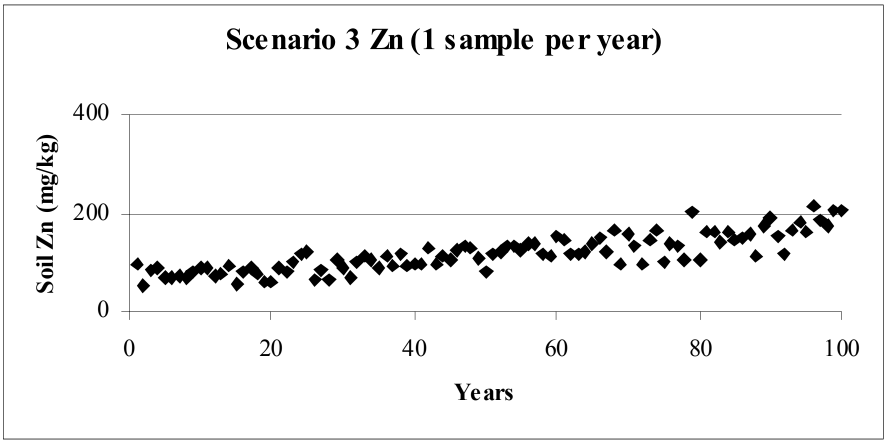


Five replicates per site taken every five years

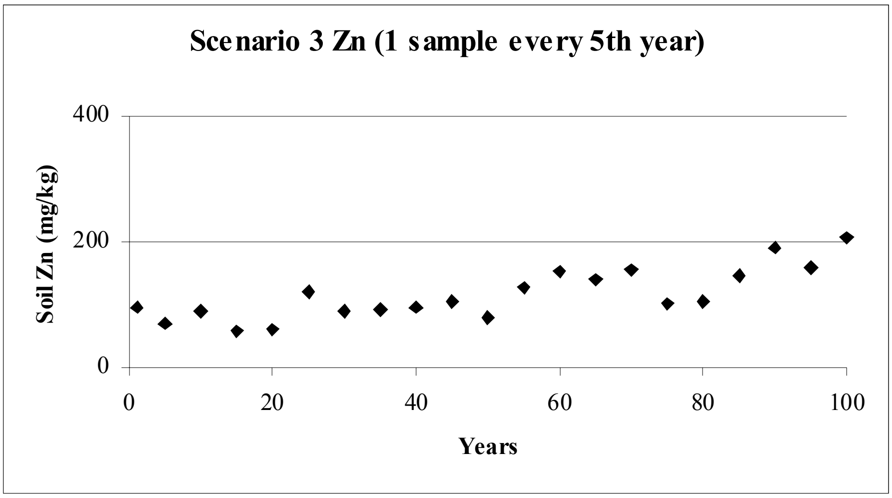


Zinc: Scenario 2 (Atmospheric deposition plus pig manure additions)

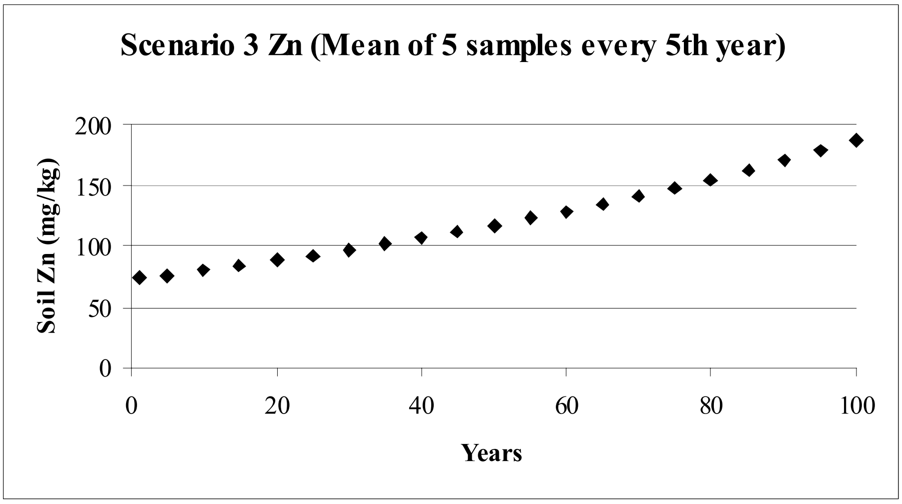
One replicate per site taken annually



One replicate per site taken every 5 years

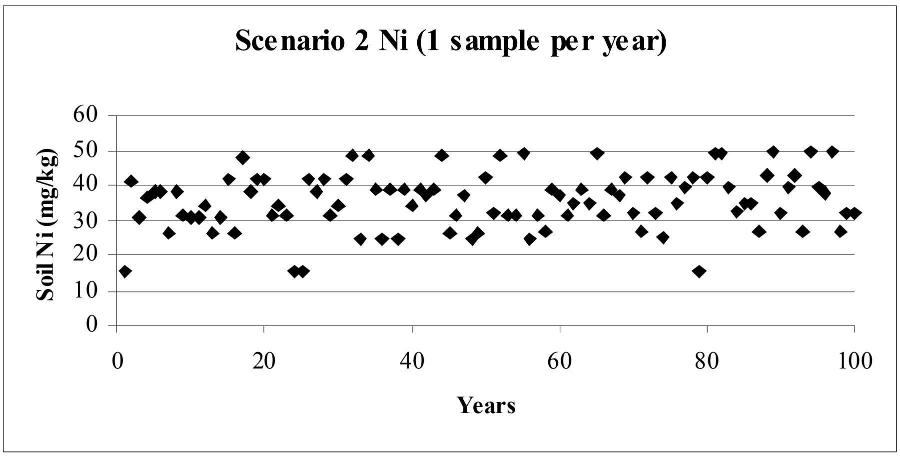


Five replicates per site taken every five years

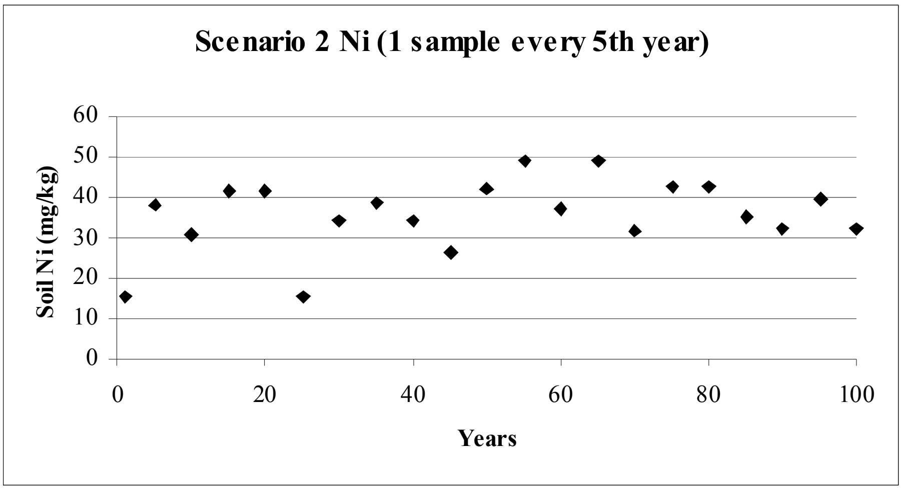


Nickel: Scenario 2 (Atmospheric deposition plus pig manure additions)

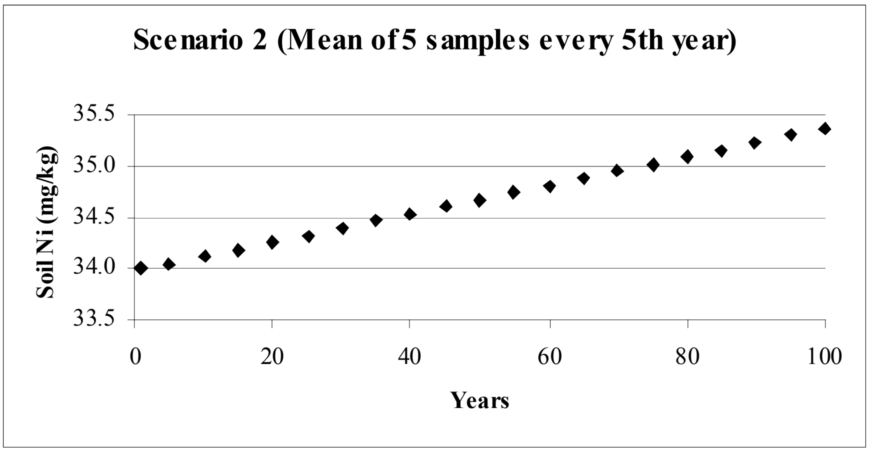
One replicate per site taken annually



One replicate per site taken every 5 years



Five replicates per site taken every five years



Questions

Using these data and any of your own, together with your own expertise in soils, we want to consider the following questions:

7. Is topsoil Cu, Zn or Ni concentration a relevant environmental indicator?
8. Is it sensitive to changes in environmental pressures (such as atmospheric deposition, climate change, agricultural practice)?
9. Is it possible to discern, *and interpret*, long-term trends in measurements which could inform policy decisions? Or is our ability to understand long-term trends inadequate to justify changes in policy which could reduce metal addition?
10. Is it possible to identify trigger values for different soil types and use them in the interpretation of long-term trends?

References

Jones KC, Symon CJ, Johnston. 1987. Retrospective analysis of an archived soil collection. I. Metals. *The Science of the Total Environment* 61: 131-144.

Keller C, McGrath SP, Dunham SJ. 2002. Trace metal leaching through a soil-grassland system after sewage sludge application. *Journal of Environmental Quality* 31:1550-1560.

Loveland PJ, Thompson. T.R.E. 2002. *Technical Report E41*, Environment Agency, Bristol, UK.

Nicholson FA, Chambers BJ, Williams JR, Unwin RJ. 1999. Heavy metal contents of livestock feeds and animal manures in England and Wales. *Bioresource Technology* 70:23-31.

Nicholson FA, Smith SR, Alloway BJ, Carlton-Smith C, Chambers BJ. 2003. An inventory of heavy metals inputs to agricultural soils in England and Wales. *The Science of the Total Environment* 311:205-219.