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Road Testing of ‘Trigger Values’ for Assessing Site Specific Soil Quality. Phase 2 – Other Soil Quality Indicators

Science Report – SC050054SR2

SCHO1008BOSV-E-P

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Steve Killeen

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Head of Science

Executive summary

The developing policy context, including the need to develop a national soil monitoring network, led the Environment Agency to undertake a project (SC030265) under the auspices of the UK Soil Indicators Consortium to identify a number of soil quality indicators (SQIs) and related 'prompt values' to assess soil quality (Environment Agency, 2006a & b). The SQIs were primarily chosen for their role in indicating adverse effects upon the soil functions of environmental interaction and habitat support viz.:

- Extractable phosphorus (Olsen P)
- Total nitrogen / carbon:nitrogen ratio
- pH
- Bulk density
- Soil organic carbon

This project aimed to validate ('road-test') the proposed SQI 'prompt values' in terms of their effectiveness and practicality on a range of soils. The 'effectiveness' was gauged in terms of whether the 'prompt values' were predictive of risk conditions for soils for which additional biological or ecological data were available – in effect the project compared measured biological effects with the risk predicted by the quality indicators. An important part of the project was also to enhance the scientific evidence base underpinning the existing 'prompt values' and, if appropriate, to suggest revisions to the values. The SQIs needed to be validated beyond modelled scenarios and the academic literature to determine whether they fulfilled their requirement as appropriate 'prompt values' for potential ecological risk in the context of soil quality assessment and monitoring.

Where appropriate data were available, the methodology developed in Phase 1 of this study (metals – see Appendix I), was extended to allow a preliminary assessment of the performance of the 'prompt values' in protecting soil quality, and to highlight instances where they may be under-, over- or sufficiently protective of soil quality.

Based on the data available, it was possible to road-test the prompt values for extractable phosphorus, pH and carbon:nitrogen ratio, although the quantity and quality of the data available were not always adequate to support unequivocal statements about the performance of the 'prompt values'. It was not possible to evaluate the performance of bulk density as an SQI, or even to make a statement regarding its suitability as an indicator of soil condition or health. Similarly, no prompt values for environmental interaction or habitat support were identified that could be used in a road-testing assessment of organic carbon status.

Extractable phosphorus. The results of the road-testing assessment suggested that the 'prompt value' for environmental interaction of 60 mg/l soil extractable phosphorus may not be sufficiently protective of water quality, recognising that the relationship between soil extractable phosphorus and surface water phosphorus concentration does not always exhibit a clearly defined 'threshold break-point' in all soil types/situations. Moreover, there was evidence from the literature review that the environmental interaction 'prompt value' for extractable phosphorus may need to recognise differences in soil types, management, and climate conditions.

A single prompt value for habitat support of 10 mg/l was proposed for all 'mesotrophic grasslands'[1]. Results from this study suggested that it might be appropriate to further subdivide this category of grasslands and provide prompt values for each subdivision.

[1] Mesotrophic (or neutral) grasslands are grasslands on moderately fertile to nutrient-rich soils that are neither strongly acid or too basic.

The habitat support ‘prompt value’ for calcareous grasslands[2] (currently 16 mg/l) was assessed to be too high as most species-rich calcareous grasslands have extractable phosphorus concentrations <10 mg/l. A more appropriate prompt value would be 10 mg/l.

Carbon:nitrogen ratio. The number of soils assessed as over-protected in the road-testing exercise, strongly suggests that the carbon:nitrogen ratio ‘prompt values’ for grassland communities are set too high (or the ranges are too narrow). The field experimental data show that valuable habitats are able to flourish where the soil carbon:nitrogen ratio is considerably lower than those suggested by the prompt value ranges.

Topsoil pH. From the datasets available, road-testing of the soil pH prompt value for environmental interaction on mineral soils, indicates that the value is robust and probably set at about the correct level.

For the habitat support function, the topsoil pH ‘prompt values’ were reasonably successful at protecting soils (for all but the mesotrophic grasslands on peaty soils), with approximately 15 per cent of the soils in each category under-protected. The heathland soils dataset showed that the pH was generally >4.5 (but <5), so the distinction between mineral/organic and peaty soil types for the habitat support function is probably not necessary.

Bulk density. Based on the data available, it was not possible to evaluate the performance of bulk density as an SQI, or even to make a statement regarding the suitability of bulk density as an indicator of soil condition or health.

Soil organic carbon. This study found no evidence to contradict the conclusion of previous work that there is little consistent evidence that there are critical thresholds of soil organic carbon above or below which soil properties change significantly. A national monitoring programme should focus on the detection of the magnitude and direction of changes in soil organic carbon levels, rather than comparing soils against an absolute ‘prompt value’. However, there is still a debate to be had over what constitutes an acceptable or tolerable change in soil organic carbon for different soils and their environmental interaction or habitat support functions.

In conclusion, this study has demonstrated that despite some limitations, it was possible to use the ‘road-testing’ methodology to validate the ‘prompt values’ for some SQIs, and in some cases to suggest revisions to the ‘prompt values’. However, the method was not appropriate for use where no specific ‘prompt values’ were defined (for example with soil organic carbon) or where there was a lack of suitable data relating the SQI to response variables (for example with bulk density).

[2] Calcareous grasslands are developed on shallow lime-rich soils most often derived from chalk and limestone rocks

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1 Policy context

The Environment Agency has long recognised the need for proportionate, consistent, evidence-based environmental protection across regulatory regimes. This has arguably been achieved for environmental media that have historically been extensively regulated, such as surface waters and air. However, relatively limited direct regulation has previously existed for soils. The multifunctional role of soil (for example as a growing medium for food crops, a sink for organic 'wastes', a repository of industrial emissions) is one that requires significant skill and understanding to balance.

However, the policy context is currently in a state of change. The EU Thematic Strategy for Soil Protection (European Commission (EC), 2002), Defra's (Department for Environment, Food and Rural Affairs) "First Soil Action Plan for England 2004-2006" (Defra, 2004) and the Environment Agency's Soil Strategy (Environment Agency, 2007) have all highlighted the need for information on the status of and recent changes in soil properties to ensure the long-term protection of soil quality and fertility. As a modern regulator, we must ensure that the standards used to assess soil quality are employed as part of a risk-based approach. In this project, we assessed not only whether the standards are protective of soils, but also whether they are scientifically robust. We seek to avoid standards that are inconsistent across regimes or industries. We should also avoid overly protective standards, which can act as a burden on industry or landowners and can cause 'pollutant swapping' or other environmentally detrimental effects. Standards should be simple to use and their basis transparent, as set out in the Royal Commission's 21st Report on Setting Environmental Standards (http://www.rcep.org.uk/standards.htm).

In addition, the UK, like many other EU member states, is committed to delivering a monitoring network to assess soil quality. Indeed, action 11 of "The First Soil Action Plan for England: 2004-2006" (Defra 2004) states that Defra will work "to identify the indicators which should be built into a national soil monitoring scheme ……. which meets both national and European requirements",. Furthermore, the Environment Agency has made a commitment to deliver robust indicators and 'prompt values' to assess soil quality via a national soil monitoring scheme, which the Consultation on the Draft Soil Strategy for England (Defra, 2008b) states is planned for 2008-10.

2 Introduction

2.1 Background

The developing policy context, including the need to develop a national soil monitoring network, led the Environment Agency to undertake a project under the auspices of the UK Soil Indicators Consortium (UKSIC) to identify a number of soil quality indicators and related 'prompt values' to be used to assess soil quality against a range of soil related functions (Table 1). Soil quality in this context can be defined as: 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. 'Prompt values' were derived for a number of soil quality indicators to be used within a tiered hierarchy. These indicators were chosen and assessed with consideration given to their costs and benefits. 'Prompt values' can be considered as 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. The 'prompt values' were based on scientific evidence, modelled datasets and, in some cases, expert opinion (Environment Agency, 2006a).

Table 1. The six functions of soil (from Blum, 1993)

Function	SQIs and 'prompt values' developed by UKSIC (brackets indicate that the values were not tested in this project)
Support of ecological habitat and biodiversity (habitat support)	YES
Food and fibre production	(YES)
Environmental interaction	YES
Providing a platform	NO
Providing raw materials	NO
Protecting cultural heritage	(YES)

There was a clear need to assess the effectiveness of the proposed 'prompt values' to fulfil their soil protection requirement in terms of soil fertility, biology and sustainability. In Phase 1 of this project, metal (zinc, copper, nickel, cadmium) soil 'prompt values' were tested against field experimental soil quality data (such as biomass size, respiration rate), to assess their performance in protecting soils in comparison with existing regulatory and guideline limit values (EA, 2008; Appendix I). This process needed to be repeated for the other soil quality indicators, which were primarily chosen for their role in indicating adverse effects upon the soil functions of environmental interaction (EI) and habitat support (HS), namely:

- Extractable phosphorus (Olsen P)
- Total nitrogen /carbon:nitrogen ratio (HS only)
- pH
- Bulk density
- Soil organic carbon

An important part of this project was also to enhance the scientific evidence base underpinning the existing ‘prompt values’ and, if appropriate, to suggest revisions to the values in the light of the objective of protecting the environmental interaction and habitat support functions of soils.

2.2 Soil monitoring

The Environment Agency is committed to delivering robust indicators and ‘prompt values’ to assess soil quality within the context of a national soil monitoring scheme by 2009. Soil quality has traditionally been assessed within an agricultural context, with little application to soils in semi-natural or low nutrient systems. The soil indicators developed by the Environment Agency (Environment Agency, 2006a) needed to be validated beyond the modelled scenarios and academic literature from which they were derived, and tested in non-agricultural situations, to determine whether they fulfilled the requirement of being appropriate ‘prompt values’ for assessing potential ecological risk.

3 Aims and objectives

3.1 Aims

This project aimed to validate ('road-test') proposed soil 'prompt values' in terms of their effectiveness and practicality on a range of soils. The 'effectiveness' was gauged in terms of whether the 'prompt values' were predictive of risk conditions for soils for which secondary response variable data (measurements which could determine whether there had been a detrimental effect on soil function) were available. The information collected through this work will be used to facilitate the development of evidence-based policy and guidance to help ensure the sustainable use of soil resources in the UK.

3.2 Objectives

The principal objectives of the project were:

- To test proposed 'prompt values' for the soil quality indicators (SQIs) extractable phosphorus, total nitrogen/carbon:nitrogen ratio, pH, bulk density, and soil organic carbon against field experimental data to determine whether they were sufficiently protective of the environmental interaction (EI) and habitat support (HS) soil functions.

- To enhance the scientific evidence base underpinning the existing 'prompt values' and, if appropriate, to suggest revisions to the values in light of the objective to protect the environmental interaction and habitat support functions of soils.

4 Approach and methodology

The project was divided into two phases. In Phase 1, the project methodology was developed, and validation ('road-testing') of the regimes and limit values for metals was undertaken (Environment Agency, 2008; Appendix I). In Phase 2, the 'road-testing' methodology was extended, where appropriate, to cover the other identified soil quality indicators (Environment Agency, 2006a). This report covers the work undertaken during Phase 2 of the study on other soil quality indicators.

4.1 Approach

The approach adopted in Phase 2 of the project was as follows:

- Undertake a literature search to identify field experimental data that could potentially be used to test the SQIs and to enhance the scientific robustness of the proposed 'prompt values'.

- Where possible, to obtain and collate the field experimental data into a single database.

- Agree suitable environmental and habitat measurements (response variables) that were indicative of changes in soil function (such as drainage water phosphorus concentrations, species richness, and so on)

- Develop a suitable methodology for testing SQI 'prompt values' for EI and HS against field experimental data.

- Assess the performance of the proposed SQI 'prompt values' in relation to protecting the soil functions of EI and HS.

4.2 Methodology

For some of the SQIs (such as extractable phosphorus, pH and carbon:nitrogen ratio), it was possible to test the 'prompt values' using a similar methodology to that adopted for metals in Phase 1 of this project (Environment Agency, 2008). For these SQIs, a database of all the identified field experimental data was created containing details of the measured SQI values and the soil data needed to assign a 'prompt value' (such as soil type, habitat type). This was supplemented with information on response variables (measurements that could determine whether there had been a detrimental effect on soil function). The response variables were different for the selected SQIs and for the identified soil functions. For example, the response variable for extractable phosphorus for the EI function was drainage water phosphorus concentration, whereas for the HS function the response variable was species richness.

The assessment of whether a soil had 'passed' or 'failed' based on the response variable differed for each SQI and soil function. In some situations, there was a defined threshold that a response variable must not exceed. For example, for extractable phosphorus (EI), the soil would 'fail' the response variable assessment if the drainage water phosphorus concentration was above the threshold value of 0.1 mg/l total reactive phosphorus (Environment Agency, 2006). Similarly, for the HS function, the soil would 'fail' if the measured species richness was below the threshold value assigned for the particular habitat type. For other response variables, the only way of testing whether a change in the SQI had unacceptable effects was to use data from controlled experiments (where the SQI was the only variable) and determine whether there had been any change in the response variable compared with a 'control' or baseline soil (the reference value). For example, for soil pH, some data were available from controlled experiments where pH was the only variable altered. In this case, changes in the response variable (extractable metal concentrations) that were significantly ($p<0.05$; analysis of variance - ANOVA) different from the untreated control were used to indicate failure. Full details of how the response variables were used in the road-testing process are given in the results section for each SQI.

For each SQI/soil function, an assessment was also made of whether the soil had 'passed' or 'failed' the relevant prompt value. These results were matched against the assessments of whether the soils had 'passed' or 'failed' based on the observed response variables. This enabled the development of performance tables and graphs to illustrate how successful the prompt values were at protecting the soil.

One of the following four performance categories was assigned to each soil:

- **PASS/PASS** – the soil met the 'prompt value' criteria and no adverse effects on the response variable were observed
- **FAIL/FAIL** – the soil did not meet the 'prompt value' criteria and some adverse effects on the response variable were observed

For soils in these two performance categories, the prompt value was *performing correctly*; that is, a correct prediction was made by the prompt value as to whether there had been harmful effects on soil function.

- **PASS/FAIL**– the soil met the 'prompt value' criteria but some adverse effects on the response variable were observed

For soils in the category, the prompt value was *not sufficiently protective* of the soil.

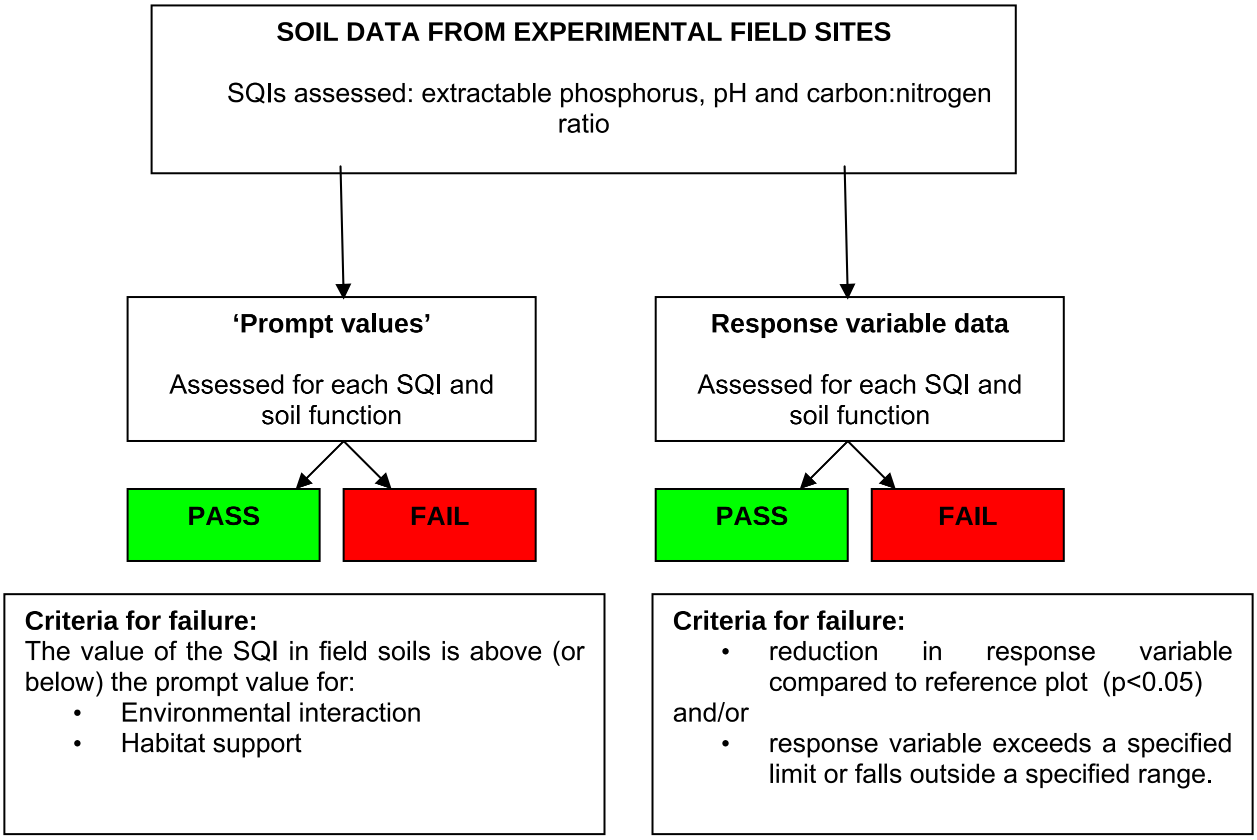
- **FAIL/PASS** – the soil did not meet the 'prompt value' criteria but no adverse effects on the response variable were observed.

For soils in this category, the prompt value was *overly protective* of the soil.

The assessment process is summarised in Figure 1.

It was not possible to test the bulk density and soil organic carbon SQIs using the above methodology. For bulk density, this was due to the lack of supplementary data on other response variables that would indicate whether there was a detrimental effect on soil function. For soil organic carbon, this was due to the absence of 'prompt values'.

Figure 1. Schematic showing the steps taken to assess ‘prompt value’ performance



5 Results

5.1 Review of datasets

One of the key objectives of the project was to identify and obtain field experimental data against which the proposed SQI ‘prompt values’ could be road-tested. A potential source of such data were ‘national’ datasets where one or more of the SQIs had been measured. These included previous or ongoing soil monitoring programmes or studies where soils and corresponding biological/environmental data (response variables) were collected at a number of sites. A review of existing ‘national’ datasets was undertaken to highlight where the selected SQIs had been measured (along with the method and interval between measurements) and where corresponding environmental and habitat response variables were also measured. A summary of the data available in these ‘national’ datasets is given in Table 2, with more complete details available in the final report of SNIFFER project LQ09 ‘National Soil Monitoring Network: Review and Assessment Study’ (Emmett, 2006). A discussion of data suitability for use in the road-testing process is included in the results section for each SQI.

5.2 Extractable

5.2.1 Reasons for the selection of extractable phosphorus as an SQI

Soil extractable phosphorus (Olsen P) has been shown to be an indicator that responds directly to increased inputs of phosphorus to soils and which is linked to increases in phosphorus concentrations in surface waters, an important driver of surface water eutrophication. Thus, the Environment Agency (2006a) recommended its inclusion as an SQI for the EI soil function. Moreover, research work has related soil Olsen phosphorus with alterations in plant successional dynamics, community composition and species diversity (for example, see Roem *et al.*, 2002), suggesting that it could also be a useful SQI in terms of the HS soil function.

5.2.2 Prompt values for extractable phosphorus

The Environment Agency (2006a) proposed ‘no increase’ soil extractable phosphorus prompt values for the soil functions of EI and HS (Table 3), with the proviso that such prompt values needed to be contextualised, particularly with reference to soluble phosphorus leaching, taking into account catchment hydrology, sensitivity of receiving waters, soil erosion risk and so on.

Table 2. Summary of the measurements in the national datasets considered for use in the road-testing assessments

Project title	Date	Geog. area	Land use	No of sites	pH	Extractable P	Organic C	Total N	Bulk Density	Surface water P	Aggregate stability	Infiltration rate	Soluble metals	Available N	Respiration rate	mineralis-able N	Invertebrates	Available water capacity	Air capacity.	Root extension	Species. richness
Effects of organic carbon inputs on soil quality (SOIL-QC)	Ongoing	GB	Arable	7	■	■	■	■	■		■	■		■	■	■	■	■	■		
Countryside Survey (CS)	1978,84,90, 2000	GB	All (ex urban)	256	■	■	■	■							■		■				
Environmental Change Network (ECN)	Every 5 or 20 yrs	UK	Various	12	■	■	■	■	■				■				■				■
Representative Soil Survey (RSSS)	Started 1969	E&W	Farms	1400	■	■		■	■					■							
National Soil Inventory (NSI)	1978-83	E&W	All (ex urban)	5662	■	■	■						■	■							
ESA Grasslands	1995/6	England	Grassland	571	■	■	■	■													■
Bunce Woodland Survey	2000/3	GB	Woodland	103	■		■	■													
AFBI 5K	1995, 2005	NI	All	435	■	■	■	■	■										■		
BIOSOIL	2008	UK	Forest	12	■		■	■	■												
Level II Intensive monitoring of forest ecosystems	2002	UK	Forest	67	■		■	■	■												

Table 3 Proposed ‘prompt values’ for Olsen extractable phosphorus (mg/l) for the soil functions of environmental interaction and habitat support suggested in Environment Agency (2006a)

Function	Soil Type		
	Mineral and Organic	Peaty	Calcareous
Environmental interaction			
Soluble P leaching	>60	>60	>60
Habitat support			
Calcareous grassland	-	-	>16
Mesotrophic grassland	>10	>10	-
Acid grassland	>10	>10	-
Dwarf shrub heath	>10	>10	-

The ‘prompt values’ for environmental interaction were primarily derived from the work of Heckrath *et al.* (1995) on the Broadbalk Continuous Wheat Experiment at Rothamsted where the soils had received either no phosphorus, phosphorus in farmyard manure or superphosphate fertiliser annually for more than 150 years. The authors related topsoil extractable phosphorus concentrations to surface water concentrations of total phosphorus (TP) and dissolved reactive phosphorus (DRP) in tile drains. From this, a ‘change point’ of 60 mg/kg[3] extractable phosphorus was identified; below this, concentrations of dissolved reactive phosphorus (DRP) in the tile drains were <0.1 mg/l (below the level suggested for minimising eutrophication impacts). Similarly Smith *et al.* (1998) identified a change point at approximately 70 mg/l Olsen extractable phosphorus.

Data to support the development of ‘prompt values’ for soil extractable phosphorus in extensive grassland habitats were not so readily available. However, values of extractable phosphorus greater than 5 mg/kg had been suggested to indicate species-poor, nutrient-enriched pastures (Goodwin *et al*., 1998). The ‘prompt values’ for habitat support in Table 3 were derived for habitats identified by the Biodiversity Action Plan (UK Biodiversity Steering Group, 1994) as likely to be affected by excessive values of extractable phosphorus and were based on expert opinion (Environment Agency, 2006a). Moreover, the Environment Agency (2006b) suggested that the ‘prompt value’ for the soil function of ‘ecological habitat and diversity’ (habitat support) should be a *change* of 5 mg/l in extractable phosphorus, rather than the absolute values given in Table 3. This was not tested in the present study.

5.2.3 Response variables

The response variables that were considered as measures of changes in soil function as a result of increases in extractable phosphorus are shown in Table 4. However, the response variables shown in italics (stress radius, invertebrates) were not used in the road-testing assessment either due to the lack of suitable field experimental data or because it was not possible to set an appropriate limit value against which to judge whether the response variables were at an unacceptable level (for example, whilst changes in soil extractable phosphorus may lead to changes in the numbers of soil invertebrates, there is no agreement on what an ‘acceptable’ number of invertebrates should be for a particular habitat).

Table 4. Response variables for the environmental interaction and habitat support functions for soil extractable phosphorus

Environmental interaction	Habitat support
Phosphorus concentration in surface waters	Species richness[1] *Stress radius*[2] *Invertebrate numbers/diversity*

[1]Species richness is a measure of the number of plant species in a sub-plot or quadrat of a specified size.
[2]Stress radius is a measure of the incidence of stress-tolerant plant species in the vegetation cover

i) Environmental interaction

The scientific literature showed a clear link between soil Olsen extractable phosphorus and soluble phosphorus concentrations in surface waters, hence an obvious environmental interaction response variable was the concentration of phosphorus in surface waters. The Environment Agency uses a General Quality Assessment for Rivers (GQA), which takes into account phosphorus (and nitrate) concentrations. These are graded on a scale of 1 (very low) to 6 (extremely high). In the case of phosphorus (measured as total reactive phosphorus on unfiltered samples) a mean level of 0.1 mg P/litre (class 4) is regarded as high and is considered indicative of potential existing or future problems of eutrophication (Environment Agency, 2006), although clearly a number of factors influence whether a river will be eutrophic. In most research studies, phosphorus concentrations are measured in the drainage water (for example in leachate, surface runoff or drains) from fields or small plots prior to dilution and hence may be higher than those in the receiving waters. Moreover, the limit relates to the total reactive phosphorus (TRP) concentration in waters, whilst many of the experiments for which data were available measured molybdate reactive phosphorus (MRP), dissolved reactive phosphorus (DRP), total dissolved phosphorus (TDP) and total phosphorus (TP) (see Table 5). It was decided for practical reasons that this road-testing work would use the limit of 0.1 mg/l MRP/DRP/TDP as one value equating to total reactive phosphorus used in the Environment Agency's GQA assessment.

If further research suggested that 0.1 mg/l is not sufficiently protective (for example algal blooms still occur), the soil quality prompt values reported here would have to be reviewed.

Table 5. Forms of phosphorus commonly measured in drainage waters

Form	Abbreviation	Explanation
Total phosphorus	TP	Total amount of P *before* filtration to remove the particulate fraction
Total dissolved phosphorus	TDP	Total amount of P *after* filtration to remove the particulate fraction
Total reactive phosphorus	TRP	Total inorganic P *before* filtration to remove the particulate fraction
Dissolved/soluble reactive phosphorus	DRP/SRP	Dissolved inorganic P
Molybdate reactive phosphorus	MRP	Dissolved inorganic P

[3] Note that soil concentrations may be reported as mg/kg or mg/l, and can be converted between these two units if the soil density (g/cm^3) is known.

ii) Habitat support

Species richness was identified as one of the most sensitive variables for the detection of changes in grassland botanical status, and has been associated with soil extractable phosphorus concentrations (Critchley *et al.*, 1999). For this reason, it was selected as the response variable indicative of changes in the soil habitat support function. Stress radius was also shown to be related to changes in soil extractable phosphorus (Critchley *et al.*, 2002a). However, as typical stress radius ranges for different habitats have not been developed, it was not possible to make use of these data in this study.

5.2.4 Literature and data search

The purpose of the literature search was to identify field experimental data that could potentially be used to test the 'prompt values' for soil extractable phosphorus or, if this was not possible, to report on work not included in Environment Agency (2006a) that could be used to enhance the scientific robustness of the proposed 'prompt values'.

i) Environmental interaction.

The identified national datasets had not measured soluble phosphorus losses in relation to topsoil extractable phosphorus concentrations (Table 2). However, there have been a number of field studies that related topsoil extractable phosphorus concentrations to drainage water phosphorus concentrations (for example see Heckrath *et al.*, 1995, Smith *et al.*, 1998; Jordan *et al.*, 2000). In all of these studies, a 'change point' was identified, above which the rate of phosphorus loss increased rapidly with increasing topsoil extractable phosphorus concentrations. Below the change point, phosphorus concentrations in drainage waters were largely independent of the soil extractable phosphorus concentration.

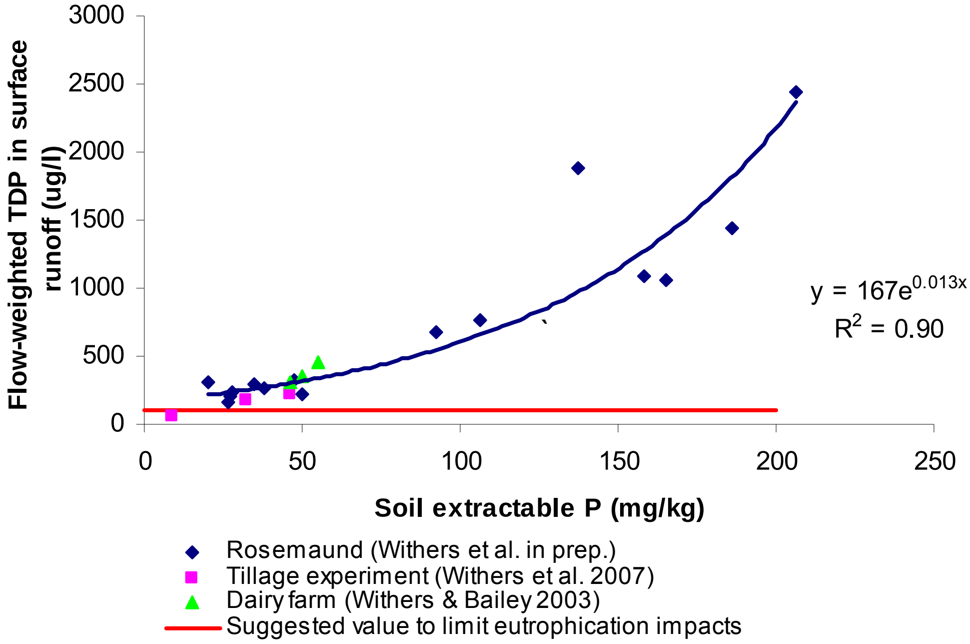
The data from these studies have been summarised in Table 6 , together with other studies where topsoil extractable phosphorus concentrations have been measured alongside drainage water phosphorus concentrations. The 'change points' identified in the two studies on arable topsoils in England, were similar at approximately 60 mg/kg and approximately 70 mg/l extractable phosphorus, and it is the former value (assuming a soil laboratory density of 1 g/cm^3) that was used as the SQI 'prompt value' (Table 3). However, a study in Northern Ireland (using modelled phosphorus loss rates) gave a much lower change point at 22 mg/l extractable phosphorus (Jordan *et al.*, 2000). The lower change point was attributed to wetter and more organic soils typically under permanent pasture in Northern Ireland rather than arable cultivation. McDowell *et al.* (2000) also measured a similar relationship between Olsen extractable phosphorus and calcium chloride-extractable phosphorus (a measure of soil solution phosphorus) in topsoils from an agricultural catchment in Devon. Here, a change point was observed at approximately 31 mg/kg Olsen extractable phosphorus, above which soil calcium chloride-extractable phosphorus concentrations increased rapidly. Topsoil calcium chloride-extractable phosphorus concentrations were also significantly correlated with mean monthly stream phosphorus discharge.

Table 6. Experiments measuring topsoil extractable phosphorus concentrations and phosphorus concentrations/losses in drainage waters

Data source	Details	Olsen P range	P concentration in drainage waters	Comments
Heckrath *et al.*,1995	11 treatments, Broadbalk	7-90 mg/kg	0.03-2.75 mg/l TP and <0.15-1.75 DRP in tile drains	'Change point' identified at 60 mg/kg Olsen P
Smith *et al.*,1998	6 sites, receiving manures	33-217 mg/l	0.2-12 mg/l MRP in leachates @ 30cm	'Change point' identified at 70 mg/l Olsen P
Shepherd and Withers, 2001	5 treatments, receiving biosolids	31-55 mg/l	0.02-0.03 mg/l MRP in leachates @ 1.2m	Insufficient datapoints to define a relationship or change point
Withers and Bailey, 2003	Dairy farm, measured for 3 yrs	46-55 mg/kg	0.28-0.36 mg/l MRP in surface runoff	Insufficient datapoints to define a relationship or change point
Withers *et al.*, 2007a	3 sites, traditional cultivation	9-46 mg/kg	0.06-0.22 mg/l TDP in surface runoff	Insufficient datapoints to define a relationship or change point
Withers *et al.*, in prep	Rosemaund, 15 inorganic P treatments	21-207 mg/kg	0.15-2.2 mg/l MRP in surface run-off	Relationship between Olsen P and TDP. No 'change point'.
Jordan *et al.*, 2000	Modelling of P loss rates from 56 catchments in Northern Ireland	10-35 mg/l	SRP loss rate: 0.1-1.6 kg/ha/yr TP loss rate: 0.1-3.5 kg/ha/yr	'Change point' identified at 22 mg/l Olsen P. Based on P load not concentration so not suitable for roadtesting
McDowell *et al.*, 2000	Topsoil P concentrations and stream P discharge, Slapton catchment, Devon	8-55 mg/kg	Stream SRP concentrations: *c*. 0.05-0.2 mg/l	'Change point' in $CaCl_2$-P concentrations identified at 31 mg/kg Olsen P
Flynn and Withers, 2001	5 sites in E&W where biosolids were applied	10-60 mg/kg	0-0.45 mg/l MRP in surface runoff	Relationship between Olsen P and MRP varies depending on soil type/management/climate
Withers *et al.*, 2007b	24 field soils from across Europe. Lab study.	0-100 mg/kg	0-1.2 mg/l TDP in surface runoff	Relationship between Olsen P and DP in runoff. No 'change point'.

Data from a study at ADAS Rosemaund (Herefordshire) where a range of topsoil phosphorus concentrations were achieved by differential rates of inorganic phosphorus fertiliser application (Withers *et al.*, in prep) are plotted in Figure 2 against flow-weighted total dissolved phosphorus concentration. At Rosemaund (where no crops were sown), total dissolved phosphorus concentrations in surface run off increased rapidly at extractable phosphorus concentrations >60-70 mg/kg. Further measurements taken from a dairy farm in Devon and three tillage experiments (Withers *et al.*, 2003, 2007a) also appeared to fit the relationship derived in Figure 2.

Figure 2. Relationship between Olsen extractable phosphorus and total dissolved phosphorus in surface runoff



Molybdate reactive phosphorus losses in surface runoff at five sites in England and Wales where sewage sludge had been applied were reported by Flynn and Withers (2001) (Figure 3). However, the field applicability of these data should be treated with caution as they were based on simulated rainfall application to boxes with air-dried soils rewetted to field capacity. The water used for the rainfall simulation was borehole water which contained high concentrations of calcium, which would have reduced phosphorus release. Nevertheless, it does provide further evidence that the 'prompt values' for soil extractable phosphorus may need to recognise differences in soil types, management and climate conditions. Tentative extractable phosphorus 'threshold values' at which the molybdate reactive phosphorus each site exceeded the 0.1 mg/l limit are shown in Table 7.

Figure 3. Relationship between soil extractable phosphorus and molybdate reactive phosphorus in surface runoff at five sites in England and Wales where sewage sludge had been applied.

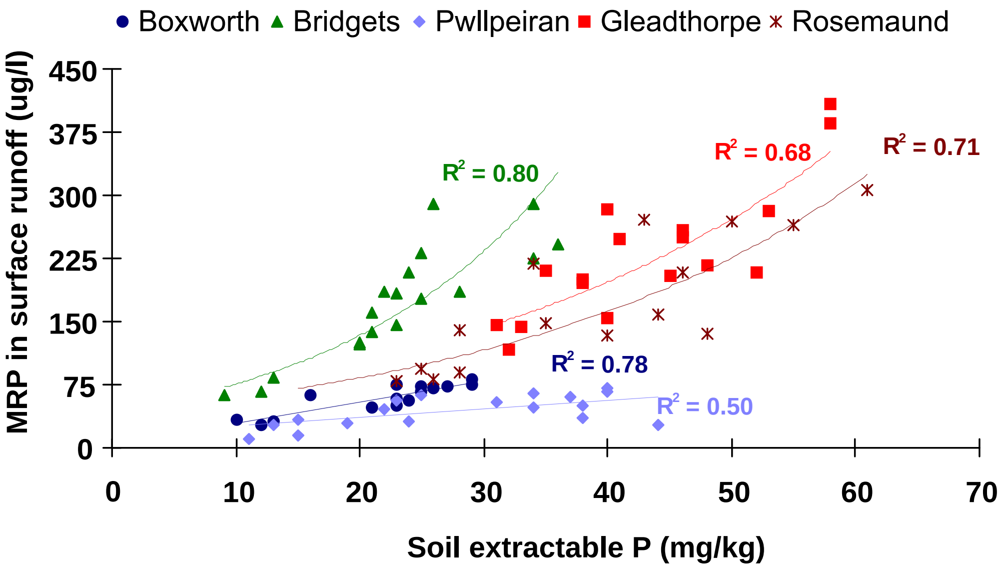
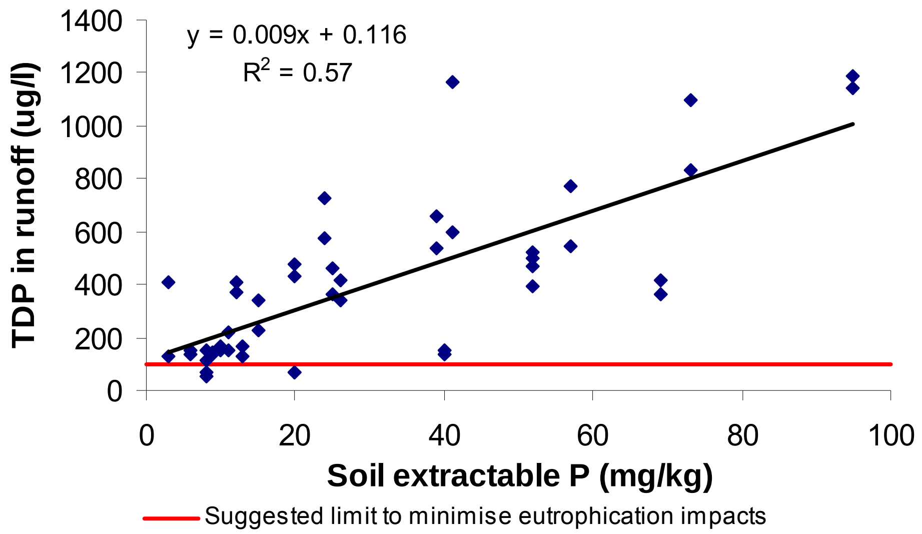


Table 7. Approximate soil extractable phosphorus levels at which surface runoff molybdate reactive phosphorus exceeded 0.1 mg/l for each site (derived from data used to produce Figure 3).

Site	Cropping	Topsoil texture group	Soil extractable P (mg/kg)
Boxworth	Arable	Clay	>30
Bridgets	Arable	Calcareous medium silt	15
Pwllpeiran	Grass	Medium loam	>45
Gleadthorpe	Arable	Sand	<30
Rosemaund	Grass	Medium silt	25

Additionally, data from the EU funded DESPRAL project (Withers *et al.*, 2007b) relating soil Olsen extractable phosphorus to total dissolved phosphorus concentrations in surface runoff showed no clear evidence of a 'change point' (Figure 4). However, the study used benchmark soils from various European countries and was a laboratory-based experiment, so the surface runoff values will not be comparable with field experimental results.

Figure 4. Relationship between soil extractable phosphorus and total dissolved phosphorus concentrations in surface runoff from laboratory based studies – two runoff events (Withers *et al.*, 2007b).



ii) Habitat Support

Species richness has been identified as one of the most sensitive variables for the detection of changes in grassland botanical status, although it does not convey any information about the nature of the change (Critchley *et al.*, 1999).

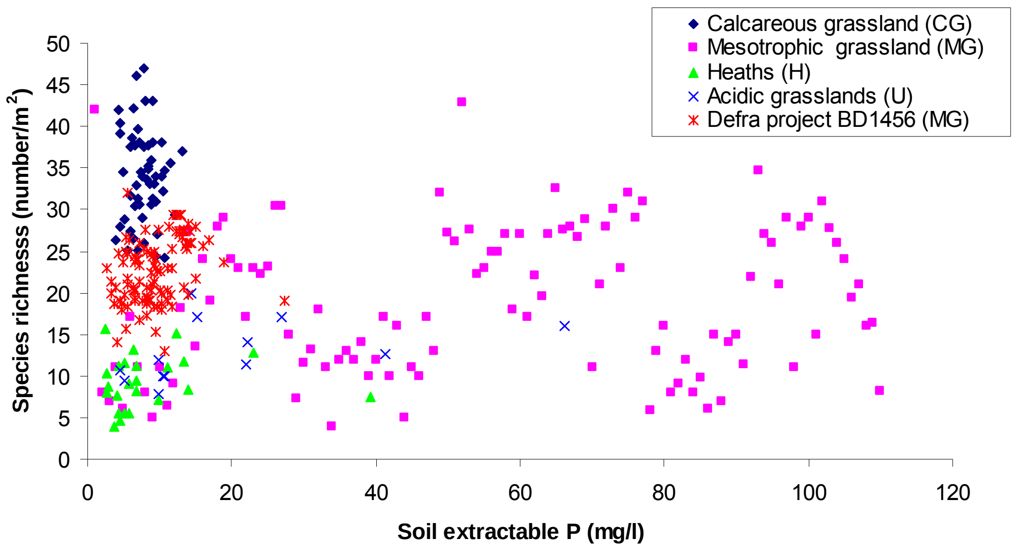
In the national datasets (Table 2), species richness was measured in the Defra-funded Environmentally Sensitive Areas (ESA) grassland survey and the Environmental Change Network (ECN). Data from the ECN were obtained from the Centre for Ecology and Hydrology (CEH) for all 12 monitoring sites, of which two were woodland and 10 were grassland (including heather moorland, permanent pasture and semi-natural chalk grassland sites). Measurements of soil extractable phosphorus were made at the start of monitoring in 1993 and are due to be repeated after 20 years (in 2013). In addition, there are data at each site on the vegetation present (that is, the species recorded in a series of quadrats) at the start of monitoring in 1993 and also at regular intervals since. There is potential for these data to be converted to a species richness measurement than could be used for road-testing. However, there were not sufficient resources available in the project to undertake the data manipulation required. In addition, Defra projects BD1504 ('Assessment of the success of schemes to recreate lowland heath') and BD1506 ('Management of lowland heath to sustain and enhance biodiversity') included 10 samples taken from heath/acid grassland, although there was no corresponding species richness data and the data could not be used for road-testing.

We therefore concentrated on data from the ESA grasslands (Defra project BD1429 'Soil nutrient status and botanical composition of grasslands in ESAs'). Table 8 gives details of the typical range of species richness associated with the main grassland types. In this survey of lowland grasslands at 571 sites in 14 ESAs in England, species richness tended to increase with decreasing topsoil extractable phosphorus concentrations, with high species richness (>30/m^2) only occurring where extractable phosphorus concentrations were <15mg/l (Critchley *et al.*, 2002a) (Figure 5). However, there was a range of species richness (from <5 to 50 m/^{2}) even at these low soil extractable phosphorus concentrations. The results therefore suggested that rather than these being an optimum extractable phosphorus concentration, there was potential for grasslands of high botanical value to develop across a range of low soil extractable phosphorus concentrations (<15 mg/l) and that in many cases these only occurred with appropriate management.

Table 8. Typical range of species richness associated with grassland types (Critchley *et al.*, 1999; Fowbert and Critchley, 2000; Kirkham *et al.*, 2008)

Community		Species richness (values from Kirkham *et al.*, 2008 shown in brackets)
Calcareous grassland	- CG2	27-32 (30-34)
	- CG5	20-23
Mesotrophic grasslands	-MG3	22-28 (22-30)
	-MG5	16-24 (15-33)
Acid grasslands	-U2-U5	6-12 (10-12)
Heaths	-H9/10	2-6

Figure 5. Relationship between species richness and extractable phosphorus for different grassland communities (data from Defra projects BD1429 'Soil nutrient status and botanical composition of grasslands in ESAs' and BD1456 'The impact of organic and inorganic fertilisers on semi-natural neutral grasslands')



Broad grassland types tend to be associated with particular soil properties, although the importance of soil properties in differentiating plant communities varies with grassland type and management practice. From the same survey of ESA grasslands, Critchley *et al.*, (2002b) demonstrated clear differences in soil properties between grassland communities as defined by the National Vegetation Classification-NVC (Rodwell,1992) (Table 9). In general, grasslands of high botanical value were associated with low soil extractable phosphorus and potassium levels, with these communities likely to be more sensitive to changes in soil properties.

Table 9. Soil characteristics of major grassland communities in English ESAs (mean with standard deviation). Data from Critchley *et al.*, (2002b)

Grassland type (NVC)	No. samples	pH	Olsen-P (mg/l)	Total N (%)	OM (%)	C:N
Mesotrophic (MG)	394	6.3 (0.9)	14 (11)	1.04 (0.63)	17.4 (12.7)	9.4 (2.83)
MG3 (unimproved)	6	6.4 (0.9)	8 (4)	0.90 (0.42)	14.6 (4.22)	10.1 (2.09)
MG7 (improved)	100	6.6 (0.9)	19 (14)	0.90 (0.44)	14.3 (8.02)	9.2 (2.33)
Calcareous (CG)	53	7.8 (0.3)	8 (2)	0.98 (0.33)	14.3 (4.92)	8.8 (2.72)
Acidic (U)	12	5.8 (1.3)	21 (18)	0.56 (0.44)	6.6 (6.48)	5.1 (2.85)

Note, Goodwin *et al.*, (1998) reported seasonality in soil extractable phosphorus concentration measurements, with higher values recorded during the warmer summer months, although the variation was only ±c.3 mg/kg. This indicates pragmatic judgement should be used in the interpretation of soil extractable phosphorus results.

Defra project BD1456 ('The impact of organic and inorganic fertilisers on semi-natural neutral grasslands') also measured topsoil Olsen extractable phosphorus and species richness at two un-improved and semi-improved hay meadow sites in England and Wales, subject to different inorganic and organic (farmyard manure) fertiliser treatments. Results from the un-improved meadows (NVC communities MG3 & 5) were assessed to be suitable for roadtesting as the species richness measurements would not have been reduced by phosphorus additions from farmyard manure or fertiliser.

5.2.5 Data selected for road-testing

i) Environmental interaction

Data from the following experiments were used to test the soil extractable phosphorus 'prompt value' for EI:

- Shepherd and Withers, 2001
- Withers *et al.*, (in prep)
- Withers *et al.*, 2007a
- Withers and Bailey, 2003
- Withers *et al.*, 2007b

Measurements of total dissolved phosphorus concentrations in the drainage waters were used as this was available for all the experiments, and the units were converted to mg/l where appropriate. Data from Smith *et al.* (1998) and Heckrath *et al.* (1995) were not used as they provided the main body of evidence upon which the soil extractable phosphorus 'prompt value' was based.

For the road-testing exercise, the measured soil extractable phosphorus concentrations were compared with the 'prompt value' for soluble phosphorus leaching (60 mg/l). If the soil extractable phosphorus exceeded the 'prompt value', then the soil failed the assessment. Also, it was assumed that if the measured total dissolved phosphorus concentration in the drainage water was above the limit of 0.1 mg/l to minimise the risk of eutrophication in surface waters, then the soil would fail the assessment based on the response variable.

ii) Habitat support

Two datasets were selected to test the habitat support 'prompt values':

- Environmentally Sensitive Area (ESA) Grasslands database (BD1429 'Soil Nutrient Status and Botanical Composition of Grasslands in ESAs'; Critchley *et al.*, 2002a,b) - 201 samples
- Defra project BD1456 ('The Impact of Organic and Inorganic Fertilisers on Semi-Natural Neutral Grasslands') - 102 samples.

'Prompt values' (Table 3) have been proposed for the following major grassland types classified according to the National Vegetation Classification (NVC) (Rodwell 1991; 1992):

- Calcareous (NVC classification: CG)
- Mesotrophic (NVC classification: MG)
- Acid grassland (NVC classification: U)
- Dwarf shrub heath (NVC classification: H)

However, within these broad grassland types there are a wide range of grassland communities and sub-communities (as classified by the NVC). These will usually be associated with different types of management (for example nutrient inputs, grazing/cutting regimes), topography, soil wetness and so on. They will also vary in botanical value and species richness. This is particularly the case for mesotrophic grasslands. As a result, only mesotrophic grasslands of high botanical value (as defined by Jefferson and Robertson, 1996) were included in the road-testing database. These were:

- MG3 – unimproved *Anthoxanthum odoratum – Geranium sylvaticum* grassland (51 samples)
- MG4 – *Alopecurus pratensis – Sanguisorba officinalis* (wet) grassland (3 samples)
- MG5 – *Cynosurus cristatus – Centaurea nigra* grassland (86 samples)
- MG8 – *Cynosurus cristatus – Caltha palustris* (wet) grassland (29 samples)
- MG11 – *Festuca rubra – Agrostis stonlonifera – Potentilla anserina* grassland (31 samples)
- MG13 – *Agrostis stonlonifera – Alopecurus* geniculatus grassland (11 samples)

It was assumed that these high botanical value grasslands would typically have a species richness >15/m^2 (see Table 8). The semi-improved and improved MG6 and MG7 communities were excluded from the evaluation, because phosphorus was likely to have been applied to these grasslands as fertiliser and manure. These MG6 and MG7 communities represented 23 per cent (90 samples) and 25 per cent (100 samples) of the whole ESA grassland survey, respectively.

Calcareous grasslands tend to have a higher species richness than mesotrophic grasslands (Table 8 and Figure 5) and it was therefore assumed that they would typically have a species richness >20/m^2. In contrast, acid grasslands and heaths usually have lower species richness (Figure 5) and are valued more for individual plant species present. Species richness may therefore not be a wholly appropriate biological indicator against which to evaluate the 'prompt values' for these two grassland types. However, in the absence of a suitable alternative indicator, it was assumed these communities would typically have a species richness <12/m^2.

For the road-testing exercise, measured soil extractable phosphorus concentrations were compared with the habitat support 'prompt values' for the selected grassland and soil types (16 mg/l for calcareous grassland and 10 mg/l for all other grassland and soil types, Table 3). If the soil extractable phosphorus concentration exceeded the prompt value, then the soil failed the assessment. It was assumed that if the recorded species richness fell below the limit specified for the mesotrophic and calcareous grasslands (15 and 20/m^2, respectively) or above the limit specified for the species-poor acid grasslands and heaths (12/m^2), then the soil would fail the assessment based on the response variable.

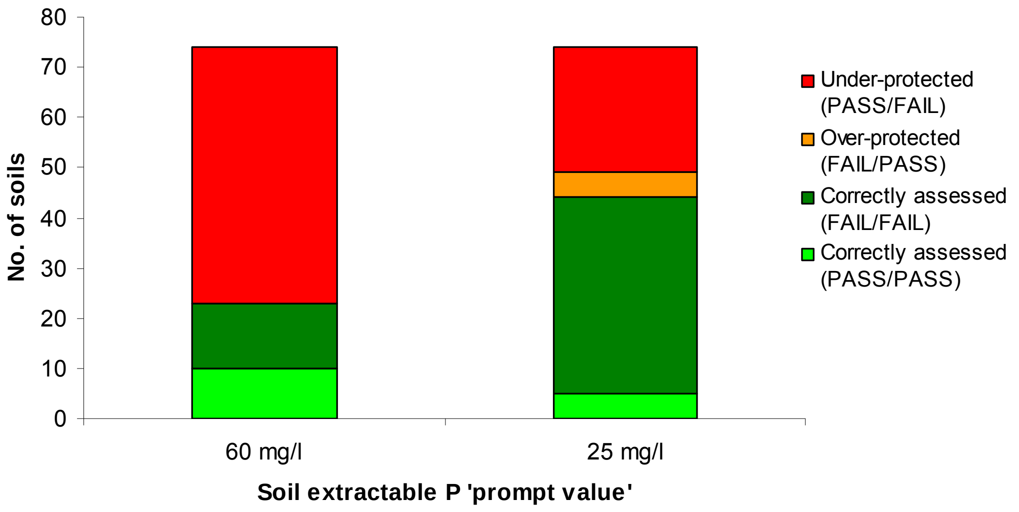
5.2.6 Results of the road-testing assessment

i) Environmental interaction

The results of the road-testing assessment of soil extractable phosphorus 'prompt values' for the environmental interaction function are shown in Figure 6. Using the 'prompt value' of 60 mg/l extractable phosphorus, 51 of the 74 selected soils were under-protected (that is, PASS/FAIL - the soil extractable phosphorus was <60 mg/l, but the measured drainage water phosphorus concentration exceeded the limit of 0.1 mg/l suggested to minimise the risk of eutrophication). Correct assessments (PASS/PASS, FAIL/FAIL) were made for 23 soils and none of the selected soils was over-protected (FAIL/PASS). As using the 60 mg/l 'prompt value' led to a large number of under-protected soils, we also tested what would happen if it was reduced to 25 mg/l. This had the effect of reducing the number of under-protected soils to 25, with 44 correct assessments and only five soils over-protected (Figure 6).

These results suggest that the prompt value of 60 mg/l soil extractable phosphorus may not be sufficiently protective of water quality, recognising that the relationship between soil extractable phosphorus and surface water phosphorus concentrations does not always exhibit a clearly defined 'threshold/break point' on all soil types/situations. Moreover, our assessments have been at the field level and not at the catchment, which is where the 0.1 mg/l limit would be applied.

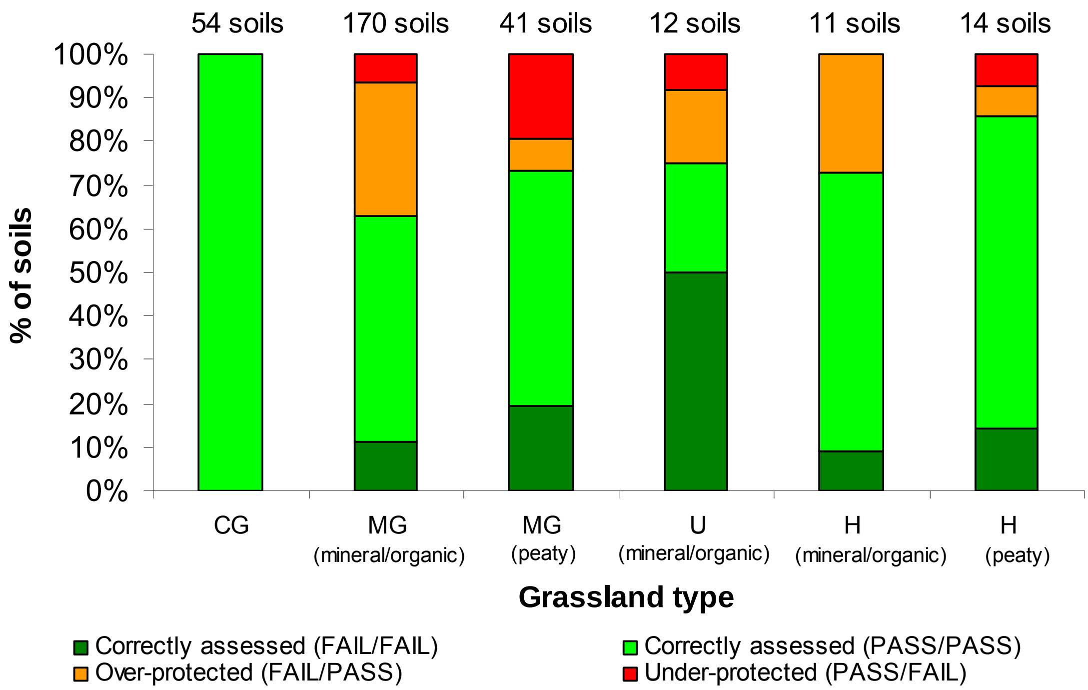
Figure 6. Performance of soil extractable phosphorus 'prompt values' for environmental interaction



ii) Habitat support

The results of the road-testing assessment of soil extractable phosphorus 'prompt values' for the habitat support function on the 303 selected soils are shown by grassland community type and soil type in Figure 7. As there was only one acid grassland (U) on peaty soil, this has not been shown separately on the graph.

Figure 7. Performance of the soil extractable phosphorus habitat support 'prompt values' for different grassland communities and soil types.



All of the calcareous grasslands (CG) had soil extractable phosphorus concentrations below the proposed 'prompt value' of 16 mg/l and species richness was always greater than 20/m^2. Indeed, extractable phosphorus was always <14 mg/l and usually <10 mg/l. Thus, all the soils passed both the 'prompt value' and the species richness threshold (PASS/PASS), indicating that the 'prompt value' of 16 mg/l for these grasslands may be too high.

For the mesotrophic grasslands (MG) on mineral soils, 11 of the 170 soils (seven per cent) were under-protected (PASS/FAIL); that is, the extractable phosphorus concentration was below the 10 mg/l prompt value, but the species richness was nevertheless less than 15/m^2, indicating that the plant community on these soils was not as diverse as would typically be expected. However, 55 soils were over-protected (FAIL/PASS); that is, even though the extractable phosphorus concentration was greater than the 10 mg/l 'prompt value', the species richness was still above the 15/m^2 threshold. On the peaty soils, eight of the 41 soils (20 per cent) were under-protected, with just three soils (seven per cent) over-protected. A large proportion of 'fails' in terms of the response variable (species richness) were MG11 and MG13 communities. These are species-poor wet grasslands and valued more for the types of species present rather than their number *per se*, with only three of the 42 soils having a species richness >15/m^2 and many having extractable phosphorus concentrations >10 mg/l. Other response variable 'fails' due to low species richness tended to be within the MG8 community, even though these are defined as being species-rich wet grasslands. Thus these results should be treated with a degree of caution, as it may not be appropriate to include MG11 and MG13 grasslands in this methodology.

Of the 25 heath soils (H), only one (peaty) soil was under-protected and four soils were over-protected, with all of these in H7 (*Calluna vulgaris – Scilla verna* heath) or H8 (*Calluna vulgaris – Ules galli* heath) communities, which tend to be more species-rich. Within the 12 acid grassland mineral/organic soils (U), only one soil was under-protected and three soils were over-protected.

5.2.7 Summary of the performance of extractable phosphorus as an SQI

i) Environmental interaction

This study has shown that it is not straightforward to define a drainage water phosphorus concentration for use in a PASS/FAIL 'road-testing' assessment. This is because accepted limits to minimise the risk of eutrophication relate to surface water phosphorus concentration at a catchment level, whilst the field experimental data collected in this project have measured phosphorus concentrations in drainage waters prior to any dilution within receiving waters. Nevertheless, it is probably fair to say that if drainage water concentrations of more than 0.10 -0.15 mg/l P are recorded, then there is probably a cause for concern and further investigation is required into potential impacts on the catchment.

Despite the limitations of the approach, the results of the road-testing assessment using data from 74 field experimental soils suggested that the 'prompt value' of 60 mg/l soil extractable phosphorus may not be sufficiently protective of water quality. Reducing the 'prompt value' to 25 mg/l decreased the number of under-protected soils and increased the number of correctly assessed soils. However, it is important to recognise that the relationship between soil extractable phosphorus and surface water phosphorus concentrations is very much a continuum that commonly does not exhibit a clearly defined 'threshold/break-point' on all soil types/situations. Moreover, the literature review provided evidence that 'prompt values' for extractable phosphorus may need to recognise differences in soil types, management and climate conditions. For example, the 'break-point' for organic soils in wetter conditions under permanent pasture in Northern Ireland was found to be lower (22 mg/l) than the 'change point' for a soil under arable cultivation in central England (60 mg/l).

ii) Habitat support

This work has shown that it is problematic to define species richness 'thresholds' for use in a PASS/FAIL 'road-testing' assessment. This is partly because setting typical species richness ranges for particular communities is a subjective exercise (see Table 8), and further work would be required to refine the species richness thresholds for the different grassland sub-communities. Also, some communities are of importance not merely for the number of species they support, but because species of particular value are present. For example, 30 per cent of the heathland soils and 69 per cent of the acid grassland soils used in this study had soil extractable phosphorus concentrations >10 mg/l; that is, they exceeded the 'prompt value' but were nevertheless defined as valuable plant communities under the NVC classification scheme. It is also apparent that to have a single 'prompt value' for soil extractable phosphorus to cover such a diverse range of communities as those encompassed by the term 'mesotrophic grasslands' is not very useful.

In terms of the performance of the 'prompt values' for the different grassland types, it would appear that for calcareous grasslands the value of 16 mg/l is too high, as most of the CG grasslands in this study had extractable phosphorus concentrations <10 mg/l. Indeed, a more appropriate 'prompt value' may be 10 mg/l for all grassland types, since the UK Soil Monitoring design may only include one grassland category.

Species richness can be affected by factors other than soil extractable phosphorus, for example soil pH, nitrogen content, moisture. It is therefore difficult to be sure that in situations where a site has a lower species richness than expected, it is because of the soil extractable phosphorus concentration and not some other soil, management or environmental factor(s).

5.2.8 Recommendations for further data collection or research required

The results of the road-testing assessment undertaken during this study would suggest that further evaluation of the extractable phosphorus 'prompt value' for EI is required in light of catchment scale dilution effects and the relative contribution of urban and rural sources, and water body sensitivity (for example running vs. still waters).

In terms of the habitat support function, for very low phosphorus systems, it is possible that Olsen extractable phosphorus may not provide a sufficiently sensitive measure of changes in phosphorus status (Environment Agency, 2006a) and an alternative measurement technique may need to be investigated.

5.3 Total nitrogen (carbon:nitrogen ratio)

5.3.1 Reasons for the selection of total nitrogen as an SQI

The Environment Agency (2006a) concluded that in regard to environmental interaction, soil total nitrogen "does not provide sensitive, interpretable information to gauge risks and threats to surface water and groundwater quality", and so 'prompt values' were not set for this soil function.

However, soil total nitrogen was selected as an SQI for the soil function of habitat support, based on its use in combination with organic carbon to derive a carbon:nitrogen ratio (Environment Agency, 2006a). The carbon:nitrogen ratio is a useful indicator of soil nitrogen mineralisation potential; that is, the potential of a soil to convert organic nitrogen compounds to inorganic forms. As the carbon:nitrogen ratio increases so nitrogen mineralisation decreases. Also, the carbon:nitrogen ratio was felt to be a useful indicator of acidification in upland soils, with soil total nitrogen content a useful indication of the potential for nitrate leaching losses to occur as a result of soil organic nitrogen mineralisation.In addition, adverse effects of increased soil nitrogen supply in low nitrogen habitats can be manifested through diversity loss and increased dominance of non-indigenous, more competitive species..

5.3.2 'Prompt values' for carbon:nitrogen ratio

The Environment Agency (2006a) proposed normal carbon:nitrogen ratio 'prompt values' for different habitats, prompting further investigation if values for the specified habitats fell outside the range (Table 10).

The values in Table 10 were based on broad habitats from the Joint Nature Conservation Committee and were derived from 1,200 soil samples taken from 0-15cm as part of the Countryside Survey (CS, 2000), together with expert opinion and interpretation (Environment Agency, 2006a). Note that the Environment Agency (2006b) suggested that the 'prompt value' for the soil function of 'ecological habitat and diversity' (habitat support) should be a *change* of 3 in the carbon:nitrogen ratio, rather than the absolute values given in Table 10. This was not tested in the present study.

Table 10. Proposed 'prompt values' for carbon:nitrogen ratio for the soil function of habitat support (Environment Agency, 2006a)

Function	Range of C:N ratios
Habitat support	
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

5.3.3 Response variables

The carbon:nitrogen ratio of a soil could impact on the botanical status of associated vegetation communities, nutrient cycling and biomass size and health. The response variables that were selected as the most appropriate measures of changes in soil function as a result of changes in soil carbon:nitrogen ratio are shown in Table 11. The response variables shown in italics (potentially mineralisable nitrogen/available nitrogen; biomass/respiration rate, invertebrate populations and bacteria/fungi ratio) were not used in the road-testing assessment either due to the lack of suitable field experimental data or because it was not possible to set an appropriate 'threshold value'.

Table 11. Potential response variables for the habitat support function of carbon:nitrogen ratio

Habitat support
Species richness[1]
PMN[2]/ available nitrogen
Biomass/respiration rate
Invertebrate populations
Bacteria/fungi ratio

[1]Species richness is a measure of the number of plant species in a sub-plot or quadrat of a specified size.
[2] Potentially Mineralisable Nitrogen

5.3.4 Literature and data search

Links between forest floor carbon:nitrogen ratios and nitrate leaching into surface waters have been reported for forest ecosystems by Gundersen *et al.* (1998), and Curtis *et al.* (2004) showed that the carbon:nitrogen ratio of surface soil organic matter had potential to act as an indicator of nitrogen saturation and leaching in moorland systems. However, nitrate leaching is a response variable related to the soil environmental interaction function, and since no prompt values were set for this by the Environment Agency (2006a), we did not pursue this relationship further.

As discussed in Section 5.2.4, species richness has been identified as one of the most sensitive indicators of changes in grassland botanical status (Critchley *et al.*, 1999), with details of typical ranges of species richness associated with the main UK grassland types given in Table 8. From a survey of ESA grasslands in England, Critchley *et al.* (2002b) demonstrated clear differences in some soil properties (including carbon:nitrogen ratio) between grassland communities as defined by the NVC (Rodwell, 1992). The carbon:nitrogen ratios for most of the grassland types were approximately10, except for the acidic grasslands (U) where the ratio was approximately five (Table 9). There was less variability between grassland types than suggested by the range of ‘prompt values’ (Table 10). Soil carbon:nitrogen ratio and species richness were also measured as part of Defra project BD1456 (‘The Impact of Organic and Inorganic Fertilisers on Semi-Natural Neutral Grasslands’).

5.3.5 Data selected for road-testing

The same data that were used to road-test the habitat ‘support prompt’ value for soil extractable phosphorus (see Section 5.2.5) were used to test the carbon:nitrogen ratio ‘prompt values’.

Using this database, we were able to assess calcareous grasslands, acid grassland and dwarf shrub heath habitat types. The ESA data contained information on MG6 and MG7 grasslands (semi-improved/improved grasslands) for which species richness was not considered an appropriate response variable (see Section 5.2.5). We have interpreted the term ‘neutral grasslands’ in Table 10 for this road-testing assessment to apply to the species-rich mesotrophic grasslands (MG3, MG4, MG5, MG8, MG11 and MG13).

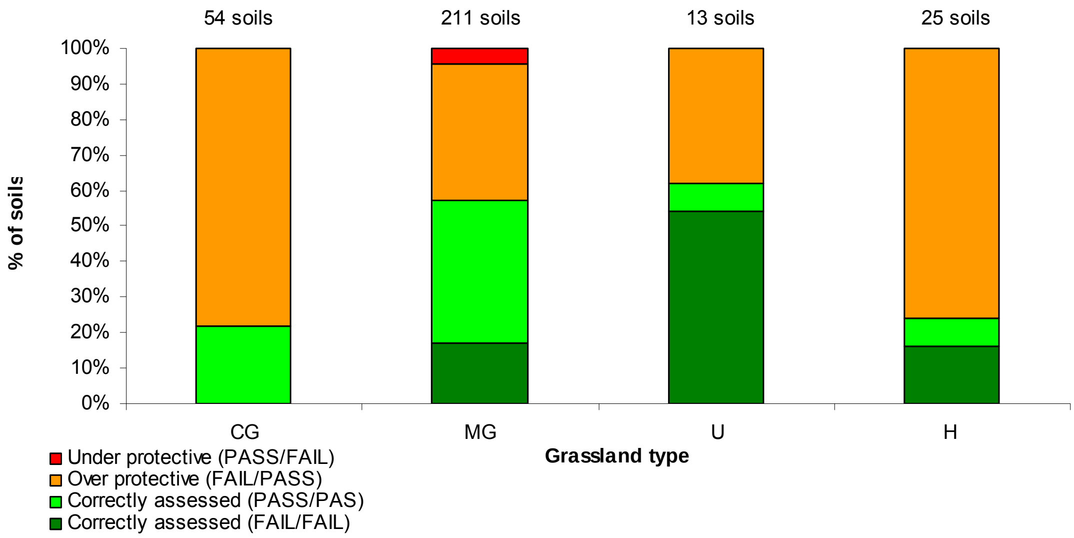
For this road-testing exercise, the measured soil carbon:nitrogen ratio was compared with the habitat support ‘prompt value’ range for the relevant grassland type (Table 10). If the soil carbon:nitrogen ratio fell outside the ‘prompt value ‘range, then the soil failed the assessment. Also, it was assumed that if the recorded species richness for a soil fell below the limit specified above for mesotrophic and calcareous grasslands (15 and 20/m^2, respectively) or was above the limit specified for acid grasslands and heaths (12/m^2), then the soil would fail the assessment based on this response variable (see Section 5.2.5).

There were no data available for woodland (broadleaf and coniferous), arable and horticultural, bog or bracken soils, so these prompt values could not be tested.

5.3.6 Results of the road-testing assessment

The results of the road-testing assessment of carbon:nitrogen ratio ‘prompt values’ for the habitat support function for the 303 selected soils are shown by grassland community type in Figure 8.

Figure 8. Performance of the carbon:nitrogen ratio 'prompt values' for habitat support for different grassland communities.



For the calcareous grasslands (CG), the 'prompt value' range of carbon:nitrogen ratios was 11-14, compared with an actual range for the 54 samples of 4-15. The road-testing assessment found that 42 soils (78 per cent) were over-protected (FAIL/PASS); that is, the soil carbon:nitrogen ratio fell outside the 'prompt value' range, although the species richness was above the limit value.

For the neutral grasslands (MG), the 'prompt value' carbon:nitrogen ratio range was 10-14, compared with an actual range for the 211 soils of 3-19 (with one soil having a carbon:nitrogen ratio of 25). For these communities, 81 soils (38 per cent) were over-protected (FAIL/PASS). Of the nine under-protected (PASS/FAIL) soils, most were species-poor wet grasslands (MG11 and MG13 communities).

On the acid grasslands (U), the 'prompt value' carbon:nitrogen ratio range was 14-21 compared with an actual range for the 13 soils of 2-16[4]. All except one soil fell outside the 'prompt value' range, with five soils (39 per cent) over-protected (FAIL/PASS). Similarly for heaths (H) where the 'prompt value' range for carbon:nitrogen ratio was 19-29, the range measured in the soils was 4-21. All but two of these soils had carbon:nitrogen ratios below the 'prompt value' range, and 19 soils (76 per cent) were over-protected (FAIL/PASS).

5.3.7 Summary of the performance of carbon-nitrogen ratio as an SQI

The number of soils assessed as over-protected by the road-testing exercise, strongly suggests that the 'prompt values' for carbon:nitrogen ratio for these grassland communities are set too high (or the ranges are too narrow). The field experimental data show that valuable habitats are able to flourish where the soil carbon:nitrogen ratio is considerably lower than those suggested by the prompt value ranges. For example, the carbon:nitrogen ratio prompt value range for calcareous grasslands was 10-14, but 41 of the 54 calcareous grasslands (76 per cent) in our database had a

[4] Soils with a low C:N ratio (<4) were sands or loamy sands with very low organic matter (<0.5%) and total N (<0.2%) contents

carbon:nitrogen ratio <11. Similarly, the carbon:nitrogen ratio 'prompt value' range for heaths was 19-29, but 23 of the 25 heaths (92 per cent) in our database had a carbon:nitrogen ratio <19.

In practical terms, the evidence here suggests that C:N ratios may be insufficiently sensitive to detect gradual habitat change (for example the long-term deterioration of calcareous grassland).

5.3.8 Recommendations for further data collection or research required

The Environment Agency (2006a) suggested that total nitrogen was excluded from the minimum data set of SQIs. Further consideration and justification is required for the continued inclusion of soil carbon:nitrogen ratio as a key SQI for the habitat support function as at present the values seem only to be useful for nitrogen sensitive habitats.

There would be some merit in revisiting the Countryside Survey (CS, 2000) dataset from which the 'prompt values' were derived to see how the habitats described compare with the ESA data set used in this study for road-testing.

5.4 Topsoil pH

5.4.1 Reasons for the selection of topsoil pH as an SQI

Topsoil pH has a significant influence on many soil processes, including nutrient cycling and availability, biogeochemical cycling, contaminant sorption, structural stability and biological activity – properties which underpin its inclusion by the Environment Agency (2006a) as an SQI for both the environmental interaction and habitat support functions. In addition, topsoil pH has been shown to be relatively sensitive to changes in soil properties, practical to measure and interpret, as well as already being measured in most relevant scientific research and monitoring schemes (Environment Agency, 2006a).

5.4.2 'Prompt values' for topsoil pH

The Environment Agency (2006a) proposed topsoil pH (measured in water) 'prompt values' for the soil functions of environmental interaction and habitat support (Table 12), based on soil type and soil function (for the environmental interaction function) or soil type and habitat type (for the habitat support function) classifications. Note that the Environment Agency (2006b) suggested that the 'prompt value' for the soil function of 'ecological habitat and diversity' (habitat support) should be a *change* of 0.5 pH units, rather than the absolute values given in Table 12. This was not tested in the present study.

The Environment Agency (2006a) reported that the 'prompt values' for environmental interaction were derived from a range of sources including reference work on heavy metal behaviour in soils (Alloway, 1995; ECI, 2003; MAFF, 1993) and guidelines for maintaining soil pH in agricultural systems (MAFF, 1988). The pH values for microbial function and biofiltering and for the habitat support function were in both cases based upon expert judgement.

Table 12. Proposed 'prompt values' for topsoil pH (measured in water) for the soil functions of environmental interaction and habitat support. (Environment Agency 2006a)

Function	Soil type		
	Mineral	Peaty	Calcareous
Environmental interaction			
Metal retention	<6	<5.5	-
Microbial function/biofiltering	<5	<4.5	-
Habitat support			
Calcareous grassland	-	-	<7
Mesotrophic grassland	<5>7	<5>7	-
Acid grassland	>5	>5	-
Dwarf shrub heath	>4.5	>5	-

Adams and Evans (1989) showed that soil exchangeable aluminium concentrations were closely correlated with soil pH and showed a very steep rise over the pH range 5.5 to 4.5, for peaty soils in an upland catchment in Wales. Below pH 4.5 exchangeable aluminium seemed to become independent of soil pH, which supports the values for peaty soils in Table 12.

5.4.3 Response variables

Table 13 shows the response variables that could be indicative of changes in soil function in response to topsoil pH. The response variables shown in italics (invertebrate numbers/diversity, potentially mineralisable nitrogen/available nutrients, biomass/respiration rate) were not used in the road-testing assessment either due to the lack of suitable field experimental data or because it was not possible to set an appropriate 'threshold value'.

Table 13. Potential response variables for the environmental interaction and habitat support functions of topsoil pH

Environmental interaction	Habitat support
Soluble Al, Mn, Fe, Zn, Cu concentration	Species richness[1]
Biomass/Respiration rate	*Invertebrate numbers/diversity*
	PMN[2]/available nutrients
	Biomass/respiration rate

[1] Species richness is a measure of the number of plant species in a sub-plot or quadrat of a specified size.
[2] Potentially mineralisable nitrogen

Adverse environmental effects linked with low soil pH include increased soil solution concentrations of metals such as aluminium, manganese and iron with related effects on the chemistry of streams and drainage waters and hence aquatic biology. Low soil pH also increases the bioavailability and soil solution concentrations of zinc and to a lesser extent copper, which may lead to phytotoxicity or adverse effects on the soil microbial populations, as well as changes in the chemistry of receiving waters which may impact on their ability to support aquatic organisms. The Freshwater Fish Directive (2006/44/EC) sets imperative and guideline standards for total zinc and dissolved copper in salmonid and cyprinid waters, although these are now recognised to be a poor metric for environmental protection and will be superceded by the quality standards in the Water Framework Directive.

Changes in topsoil pH can also affect the soil microbial population and structure, with related effects on bacterially mediated processes such as nitrification, denitrification and nutrient cycling. Optimum soil pH ranges for different microbial groups and process are given in Environment Agency (2006a). In terms of response variables, biomass carbon and/or respiration rate measurements are commonly used to assess the general size and health of the soil microbial population. However, both biomass carbon and respiration rates vary depending on the soil type, climatic conditions and so on, and no standard or guideline value indicative of 'good' soil health have been published against which the field experimental values could be compared.

For habitat support, pH has been related to species richness (Roem *et al.*, 2002; Critchley *et al.*, 2002a), so this was selected as the response variable for road-testing.

5.4.4 Literature and data search

The purpose of the literature search was to identify field experimental data that could potentially be used to test the 'prompt values' for topsoil pH or, if this was not possible, to enhance the scientific robustness of the proposed 'prompt values'.

i) Environmental interaction

Potential response variables for the soil EI function were identified in Table 13 as soluble Al, Mn, Fe, Zn, Cu concentrations in the soil. Soluble metal concentrations can be considered as the metal form that is likely to be leached from the soil into receiving ground or surface waters. It also measures how strongly metals are retained by a soil, hence the lower the soluble metal concentration, the more strongly retained is the metal by the soil. Soluble metal concentrations are measured in some monitoring programmes and experimental work, however in many cases measurements are made of extractable metals instead. The use of extractants such as ammonium nitrate is thought to provide an indication of the fraction of soil metals that is bioavailable, and as such is often used in studies where crop/plant uptake of metals in being investigated. In this study it has been assumed that extractable metal concentrations can be used a surrogate for soluble metal concentrations and can hence be used to infer how strongly metals are retained by soils, although we acknowledge that there is some debate over the validity of this assumption.

While pH is measured in many national soil monitoring programmes (Table 2), the only programme that measured both topsoil pH and soluble metal concentrations was the Environmental Change Network (ECN). As part of the ECN, data on surface water chemistry is collected on a weekly basis and soil solution chemistry on a two-weekly basis at the 12 terrestrial sites, with both including measurements of pH, aluminium and iron. The ECN did not measure soil solution zinc or copper concentrations so there were no data to assess against zinc or copper limits in freshwaters.

Soil EDTA extractable metal concentrations (zinc, copper, nickel, cadmium) were measured as part of the National Soil Inventory (NSI). However, it was not possible to relate these concentrations to limit values for freshwaters; these data could therefore not be used for road-testing.

As part of the "Long-term Sludge Experiments" (Defra project SP0130), the relationship between topsoil pH and ammonium nitrate extractable metal (zinc, copper, cadmium) concentrations was measured for nine sites, where topsoil pH on the untreated control plots at each site ranged from between 5.4 and 8.0 (Gibbs *et al.*, 2006). For both zinc and cadmium, there was a strong inverse relationship between the extractable concentration (expressed as a percentage of total metal) and pH, whilst copper extractability was most strongly related to the soil total iron content. Similarly, extractable metals and pH were measured as part of a project assessing "Soil Vulnerability to Heavy Metal Pollution" (Defra project OC9325; Alloway *et al.*, 1999) where the pH of five sewage sludge treated soils was adjusted by the addition of lime or sulphur to create a range from approximately 3.0 to 7.5 after equilibration in the field for *c*.18 months). Extractable zinc, nickel and cadmium concentrations increased as pH decreased in all five soils, whilst extractable copper concentrations were not as strongly affected by soil pH. For most soils, the sharpest increase in extractable metal (zinc, cadmium, nickel) concentrations was seen when soil pH fell below approximately 6.0.

It has previously been shown that there is a strong relationship between both ammonium nitrate extractable and soil solution metal concentrations and plant metal uptake (Alloway *et al.*, 1999). In theory, a relationship could be derived between extractable metal concentrations (for which there are many data available) and soil solution metal concentrations (which could then be compared with zinc and copper limits for receiving waters). However, such relationships between extractable and soil solution concentrations would probably vary between soils (particularly where 'by-pass' flow occurs), so the conversion would not be straightforward and would probably introduce many errors into the process, thus minimising its value in a road-testing approach. The data from Alloway *et al.* (1999) can be used to test if there are significant changes in extractable metals on pH treated plots compared with the untreated control plot (pH 6.6-7.0), assuming that extractable metals can be used as a 'surrogate' measure for leachable metals. Therefore, additional statistical analysis (analysis of variance - ANOVA) was undertaken as part of this project to enable these data to be used in the road-testing approach.

Smith *et al.* (1999) reported additional work using four of the soils described above in Defra project OC9325 (two uncontaminated and two metal-contaminated) where biomass carbon and respiration rate were measured. Soil pH was found to have a strong effect on the size of the microbial biomass in all the soils, with the biomass carbon content decreasing significantly as the soil pH declined and, in general, the lowest biomass carbon was recorded in soils where the pH was <*c*.5.5. In contrast, the metabolic quotient (that is, respiration rate divided by biomass carbon) was not strongly affected by soil pH.

The Soil-QC project (Table 2) included measurements of soil pH and biomass carbon at seven arable sites in Britain where medium to long-term organic material applications had been made. Biomass varied greatly between the sites depending on soil type, cropping, organic carbon content and so on (Bhogal *et al.*, 2007). However, due to the narrow range of pH values at each site, it was not possible to establish any relationship between soil pH and biomass size/respiration rate.

ii) Habitat Support

As previously discussed (Section 5.2.5), in a study of ESA grasslands Critchley *et al.* (2002a) showed that species richness was significantly ($p<0.001$) related to soil pH in semi-natural grasslands. The highest species richness (>30/m^2) occurred at pH >6, whereas at pH <5 species richness was low (<20/m^2). Stress radius (a measure of the incidence of stress-tolerant species) was also related to pH, with the highest values on calcareous and heath grasslands at pH of approximately 8 and <5.0, respectively.

Other potential soil response variables identified included potentially mineralisable nitrogen, ammonium nitrate and invertebrate numbers. As nitrification is largely inhibited at pH <4.5 (Paul and Clark, 1989), this can result in a build-up of soil ammonium nitrate and low organic nitrogen mineralisation rates. Most earthworm species thrive in a soil environment, which is not too acidic (Brady, 1974). However, *Lumbricus terrestris* can still be present at a pH of 5.4, *Dendrobaena octaedra* at a pH of 4.3 and some Megascolecidae are present in extremely acid humic soils. Soil pH may also influence the numbers of worms that go into diapause (dormancy) (see Edwards, 2004). However, we could not identify any suitable datasets that had measured these response variables and had a sufficiently wide range of soil pH values, against which to road-test the responses.

5.4.5 Data selected for roadtesting

i) Environmental interaction

Data from Alloway *et al.* (1999) were selected for an initial test of the EI 'prompt value' for metal retention on mineral soils. It was assumed that ammonium nitrate extractable metals provided a robust measure of the ability of a soil to retain metals (that is, the higher the extractable metal concentration the more likely the metal was to be leached from the soil and enter a watercourse). For each of the five soils used in the experiments, only treatments with pH lower than the untreated control were selected, giving a total of 21 data points that could be used for road-testing.

For the road-testing exercise, the measured soil pH was compared with the environmental interaction 'prompt value' (pH <6). If the soil pH was below the 'prompt value', then the soil failed the assessment. A soil failed based on the response data if the measured concentration of one or more of the extractable metals (zinc, copper, nickel, cadmium) was significantly ($p<0.05$) greater than the control treatment.

The original data on biomass carbon and respiration rates from the same experiment (Smith *et al.*, 1999) were not available, so we were unable to test the environmental interaction prompt value for microbial function/biofiltering on mineral soils (pH <5). Since there were no data for peaty soils, we were not able to test these 'prompt values' either.

ii) Habitat support

The same data that were used to road-test the habitat support 'prompt values' for soil extractable phosphorus and carbon:nitrogen ratio (see Section 5.2.5) were used to test the soil pH 'prompt values' for habitat support.

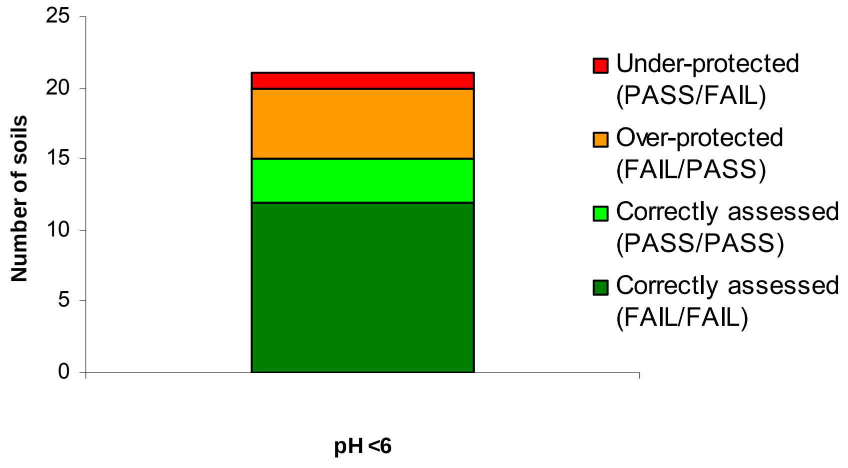
For the road-testing exercise, the measured soil pH was compared with the habitat support 'prompt values' for the relevant grassland and soil type (Table 12). If the soil pH fell outside the 'prompt value' range for mesotrophic grasslands or if it exceeded the 'prompt values' for acid grasslands and dwarf shrub heaths, or if it fell below the value for calcareous grasslands, then the soil failed the assessment. It was assumed that if the recorded species richness for a soil fell below the limit specified above for mesotrophic and calcareous grasslands (15 and 20/m^2, respectively) or above the limit specified for acid grasslands and heaths (12/m^2), then the soil would fail the assessment based on the response variable (see Section 5.2.5).

5.4.6 Results of the road-testing assessment

i) Environmental interaction

Using the 'prompt value' of topsoil pH<6 for metal retention in mineral soils, there were correct predictions for 15 of the 21 soils tested (Figure 9). This 'prompt value' was only under-protective for one of the soils and was over-protective for five soils.

Figure 9. Performance of the topsoil pH environmental interaction ‘prompt value’ for metal retention on mineral soils.



ii) Habitat support

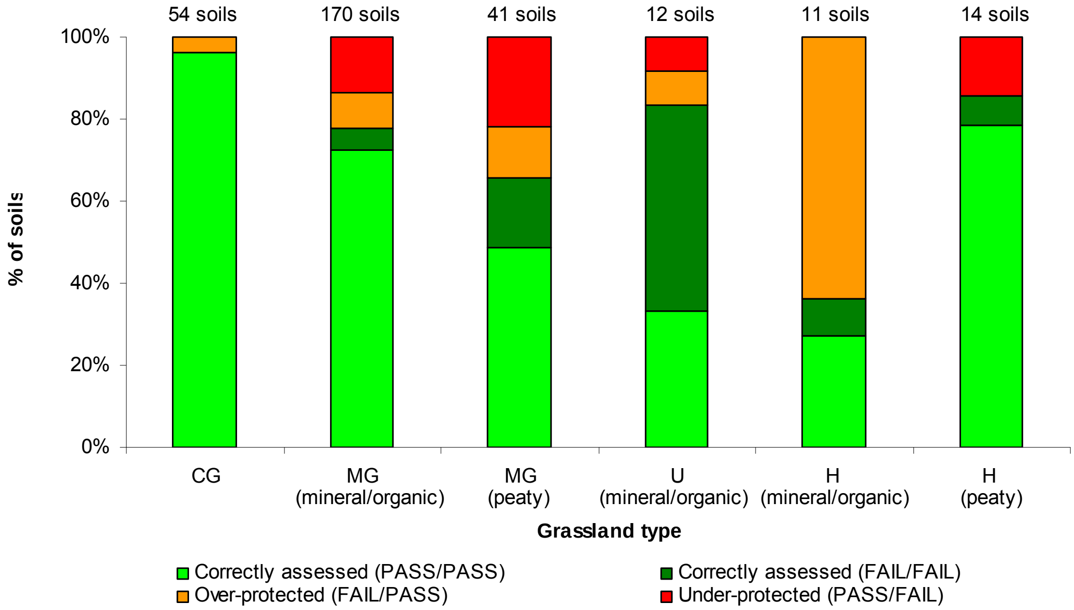
The results of the road-testing assessment of the soil pH ‘prompt values’ for the habitat support function for the 303 selected soils are shown by grassland community and soil type in Figure 10. As there was only one acid grassland (U) on peaty soil, this has not been shown separately in Figure 10.

Nearly all the calcareous grasslands (CG) had a topsoil pH greater than the ‘prompt value’ of seven and species richness >20/m^2, and hence were correctly assessed (PASS/PASS). Two soils were over-protected (with topsoil pH of 6.4 and 6.8). These were from upland calcareous grasslands (CG9 and CG10) and were the only two soils from the survey data representing these grassland types (the rest of the soils were lowland calcareous grasslands).

For the mesotrophic (MG) grasslands on mineral/organic soils, 23 of the 170 soils (14 per cent) were under-protected and 15 soils (nine per cent) were over-protected. For the peaty soils, nine of the 41 soils (22 per cent) were under-protected. The peaty soil results were influenced once again by the inclusion of MG11 and MG13 grasslands, which were typically species poor and may not be appropriate targets for this methodology.

For the acid grasslands (U), only one soil was under-protected and one soil was over-protected. For the 11 heathland mineral soils, the ‘prompt value’ of pH >4.5 was over-protective for seven (64 per cent). The dataset showed that the pH of these soils was generally >4.5 (but <5), so it may be that a distinction between soil types is not necessary. In terms of the 14 peaty soils, the ‘prompt value’ of pH>5 was under-protective of two soils, but over-protective of none.

Figure 10. Performance of the topsoil pH prompt values for habitat support for different grassland communities and soil types



5.4.7 Summary of the performance of pH as an SQI

Road-testing of the soil pH EI 'prompt value' for metal retention in mineral soils, indicates that the value of pH 6 is performing effectively , although only a very limited dataset of 21 soils was available for testing.

For the HS function, the topsoil pH 'prompt values' were reasonably successful at protecting soils (with the exception of mesotrophic grasslands on peaty soils), with less than 15 per cent of the soils in each category under-protected.

The dataset of 25 heathland soils showed that the pH was generally >4.5 (but <5), so the distinction between mineral/organic and peaty soil types is probably not appropriate.

Note that there are some instances where the soil pH 'prompt values' for EI and HS are very different and may be contradictory. For example, for dwarf shrub heath the HS 'prompt value' is >4.5 (i.e. to preserve the plant/animal community the soil pH should not exceed 4.5) whereas the EI metal retention prompt value is <6 (i.e. to minimise leaching of metals the soil pH should not be less than 6). Careful attention should be given to potential anomalies in the design of a soil monitoring programme where these 'prompt values' are used.

5.4.8 Recommendations for further data collection or research required

There would be merit in obtaining further data on relationships between soil pH and soil biological activity (biomass size and respiration rates, and invertebrate populations including earthworm and collembolla numbers). For example, at low pH earthworms are replaced in their function of SOM breakdown by enchytaeides. However it is a matter of judgement whether the non-presence of earthworms in acidic soils is a 'bad' thing since it is 'natural'.

5.5 Bulk density

5.5.1 Reasons for the selection of bulk density as an SQI

Soil bulk density was the only physical parameter selected for inclusion in the minimum data set of SQIs. Topsoil bulk density (that is, the mass of dry soil per unit volume) has a direct impact on a number of key soil physical and biological processes, notably water infiltration rate, gaseous exchange, root access and soil faunal activity. As a transient property it can be modified by soil management practices, hence changes in bulk density can be related to recent soil and land use history, and can infer soil degradation or amelioration. A full justification for the selection of soil bulk density as an SQI and derivation of the threshold values is given in Environment Agency (2006a). The interpretation of bulk density data presents many challenges, some of which are discussed below. For more background to this discussion, see Chapter 3 of the Soil Quality Indicators Report (Environment Agency, 2006a).

5.5.2 'Prompt values' for bulk density

The Environment Agency (2006a) proposed topsoil bulk density 'prompt values' for the soil functions of environmental interaction and habitat support (Table 14), based on soil function (for the environmental interaction function) or habitat type (for the habitat support function). Two critical thresholds were identified depending on whether the soil was mineral or peaty. The Environment Agency (2006a) referred to primary relationships (with macroporosity, aggregate stability, hydraulic conductivity) and secondary factors that could be used to infer changes in soil chemical and biological factors (organic carbon, microbial biomass/activity) that justified the thresholds.

Table 14. Proposed 'prompt values' for bulk density (mg/m^3) for environmental interaction and habitat support. Soil function may be impaired above stated values (Environment Agency 2006a).

Function	**Soil type**	
	Mineral*	**Peaty**
Environmental interaction		
Arable and horticultural	>1.3	>1.0
Improved grassland	>1.3	>1.0**
Habitat support		
Calcareous grassland	>1.3	
Mesotrophic grassland	>1.3	>1.0
Acid grassland	>1.3	>1.0
Dwarf shrub heath	>1.3	>1.0

*Includes calcareous soils
**Value amended from that published in Environment Agency (2006a)

5.5.3 Response variables

The Environment Agency (2006a) and the wider literature review allowed the identification of key potential response variables that could be used for road testing this indicator (see Table 15).

Successful road testing of an SQI is dependent upon the establishment of clear cause and effect relationships that allow interpretation of changes in terms of either environmental interaction or habitat support. Bulk density is problematic in that it was selected by the Environment Agency (2006a) as a broad physical indicator of soil quality, and like pH can impact on a range of soil physical, biological and chemical responses. The response variables listed in Table 15 are those identified from the literature as most likely to display a close relationship to bulk density.

Table 15. Response variables for bulk density (with indicative interpretation in italics)

Environmental Interaction	Habitat support
Physical: infiltration rate, hydraulic conductivity (*susceptibility to flooding; pollutant transfer to water*) Aggregate stability/size (*erosion risk*) Biological: microbial respiration rate (*aeration status*)	Physical: available water capacity (*drought tolerance*), total porosity, air filled porosity Biological: Root extension (*plant growth*); soil invertebrate numbers/diversity/functional groups (such as earthworms, carabid beetles, springtails) (*soil health*)

5.5.4 Literature and data search

The purpose of the literature search was to identify field experimental data that could potentially be used to test the 'prompt values' for bulk density or, if this was not possible, to enhance the scientific robustness of the proposed 'prompt values'.

i) Environmental interaction

Soil compaction due to land use pressures continues to be a significant issue, resulting in the degradation of soil quality and impairment of soil function. The primary focus of research has been on the soil function of biomass production. Reynolds *et al.* (2002) identified optimum values for density and water/air storage in arable soils, while Reynolds *et al.* (2007) observed that: "…for fine-textured soils, the optimum bulk density range for field crop production appears to be in the order of 0.9–1.2 mg/m^3. Bulk density values <0.80 may provide insufficient root–soil contact, water retention and plant anchoring, while bulk density values >1.20 may impede root elongation or reduce soil aeration. The upper bulk density limit for adequate aeration of fine-textured soils appears to be in the order of 1.25–1.30, while mechanical resistance to root elongation in fine-textured soil often becomes excessive for bulk density >1.40–1.60" (Reynolds *et al.*, 2007).

An increase in bulk density (compaction) not only restricts biomass production in arable systems, but is an on-going issue affecting the health of grassland soils. A Defra review concluded that "'little is known about the extent and severity of soil compaction and its impacts on the vital functions (ecosystem services) supported by soil. We know a little about the impacts of soil compaction on above ground biodiversity, but almost nothing on below ground flora and fauna" (Defra, 2008a).

Bulk density has been included in several National SQI minimum data sets. Schipper and Sparling (2000) and Sparling *et al.* (2000, 2004) justified the inclusion of this indicator in terms of bulk density relationship with soil compaction, the physical environment for roots and soil organisms. Many cultivation experiments have been conducted that establish a relationship between bulk density and infiltration rate, saturated hydraulic conductivity, porosity and so on. All were orientated towards biomass production, from which some aspect of environmental interaction can be inferred, such as the potential for saturated overland flow to be generated. Recent work by Reynolds *et al.* (2007) noted that bulk density is often used in soil quality studies as an index of the soil's mechanical resistance to root growth. These workers observed a texture-controlled optimum bulk density in a range of arable and grass ley production systems. This observation reinforces the work of Reeve *et al*. (1973) who highlighted that an on-going problem associated with the use of bulk density as an SQI was the texture dependence of critical values. The discussion in Environment Agency (2006a) continues this debate, with proposed alternative bulk density thresholds depending on clay content.

ii) Habitat support

Habitat support functions are expressed primarily in terms of the impact of changing bulk density on physical and biological soil conditions. 'Prompt values' have been less well researched than those for biomass production, but nevertheless the same general principles and limits can be considered to apply. For example, just as bulk density will influence available water capacity and hence drought tolerance in crop production, so too will plants in semi-natural habitats be affected by changing water availability. In the case of sensitive plants adapted to specific conditions, the relationship can be expected to be closer than for some crop production plants. Soil biological responses, such as root extension and soil-living invertebrate numbers/diversity/functional groups (earthworms, carabid beetles, springtails) are not well understood (Defra, 2008a). For example, earthworm populations vary according to disturbance in arable systems (Blackshaw *et al.*, 2007), but relationships with bulk density *per se* are unclear.

5.5.5 Data selected for roadtesting

Robust datasets to road-test the influence of bulk density on environmental interaction or habitat support would ideally be either:

(i) Focused on individual locations where soil variability is reduced to a minimum, such as the Soil-QC experiments

(ii) Based on a wide range of geographical locations, with many observations allowing multivariate analysis to be undertaken from which the strength of relationships with a range of soil properties including bulk density could be inferred, such as the Countryside Survey (CS, 2000).

The analysis presented here is based on unpublished data that conforms to the first point above, namely the Soil-QC experiment (Bhogal *et al.*, 2007) and the University of Plymouth Long-term Cultivation trial (Donovan, Hazarika and Parkinson, unpublished). At present, no national database of soil properties that includes bulk density and is indicative EI or HS responses is available to allow road-testing of critical bulk density thresholds.

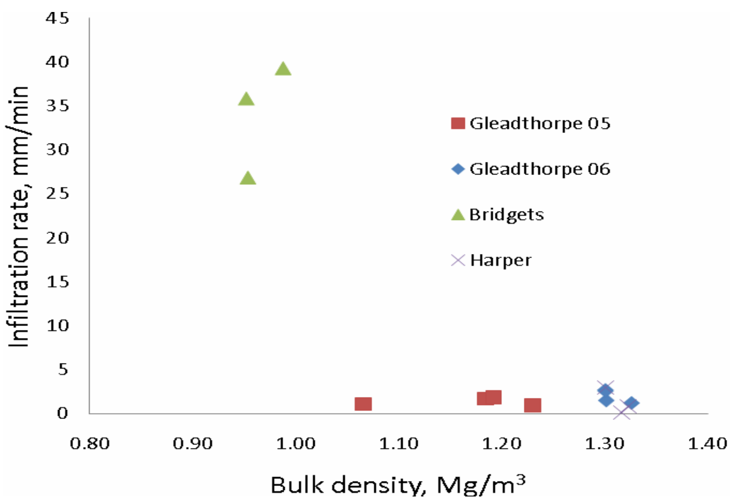
5.5.6 Results of the road-testing assessment

i) Soil-QC experiments

The Soil-QC experiments (Defra project SP0530) seek to develop an improved understanding of the processes and linkages through which organic carbon additions influence soil quality and fertility, and sustainable crop production. Using a network of seven experimental sites, on contrasting soil types, with a history of repeated farm manure or differential rates of inorganic fertiliser nitrogen application, the study has measured a range of soil bio-physical and physico-chemical properties. Data from three of the sites (Gleadthorpe: 5 per cent clay, 1.1 per cent soil organic carbon on the control treatments; Harper Adams: 12 per cent clay, 1.4 per cent soil organic carbon and Bridgets: 23 per cent clay, 3.0 per cent soil organic carbon) were used to assess any relationships between bulk density and two response variables, infiltration rate and earthworm (numbers) mass.

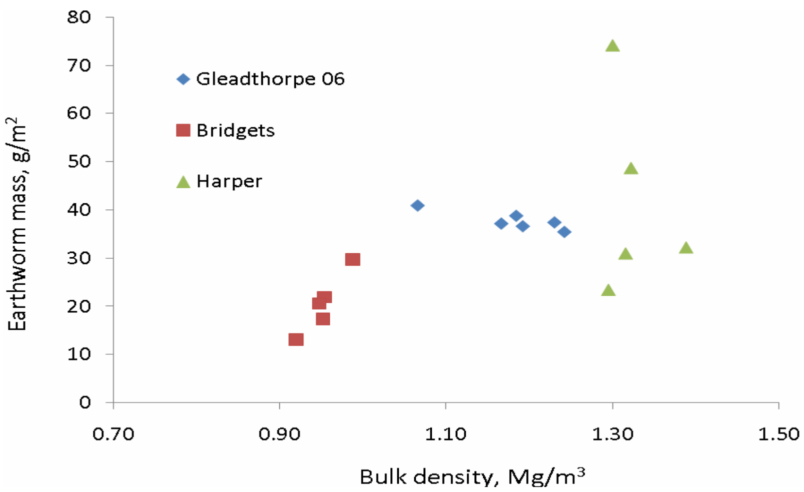
Water infiltration rate was found to be primarily a function of soil texture, with the highest values measured at Bridgets (Figure 11). There was no consistent relationship between bulk density and water infiltration rate, highlighting the significance of macropores in water movement, a factor that is well known to confound relationships between bulk density and water movement (Reynolds *et al.*, 2002). Bulk density is related to total porosity, but whilst hydrologically effective porosity represents only a small fraction of the total volume it is of great importance in terms of EI functions such as infiltration rate.

Figure 11. Influence of bulk density on infiltration rate, Soil-QC sites



Previous research has indicated inverse relationships between earthworm mass and bulk density, although methods of cultivation are known to confound this relationship (Blackshaw *et al.*, 2007). Moreover, the cause and effect of these relationships are not clear - a high abundance of earthworms may reduce bulk density due to their burrowing activity; conversely there may be more earthworms because the soil isn't as compacted. The Soil-QC data showed no clear relationship between bulk density and earthworm mass (Figure 12). At Harper Adams the bulk density approached or exceeded the 1.3 mg/m^3 'prompt value' for mineral soils, with earthworm mass and infiltration rate higher at this site than the other three sites. This analysis reinforces the view of the Environment Agency (2006a) that bulk density is only of use as a broad SQI indicative of general soil health status.

Figure 12. Relationship between earthworm mass and bulk density at three Soil-QC sites.



ii) University of Plymouth long-term cultivation trial

A second data set was identified for investigation from the long term (23-year) winter cereal cultivation trial at the University of Plymouth farm, Seale-Hayne, South Devon (silt loam textured soil). There were two main treatments: soil disturbance (ploughed, tined and direct drilled) and crop residue management (straw residues left *in situ* or removed), with three replicates of each treatment. Earthworms were collected by hand sorting soil monoliths of 6.3 dm^3 volume, taken from the 0-10 cm soil layer. Triplicate bulk density samples (0.22 dm^3 volume) were taken from the same soil layer.

Figure 13. Earthworm biomass versus topsoil bulk density, University of Plymouth long-term cultivation trial.

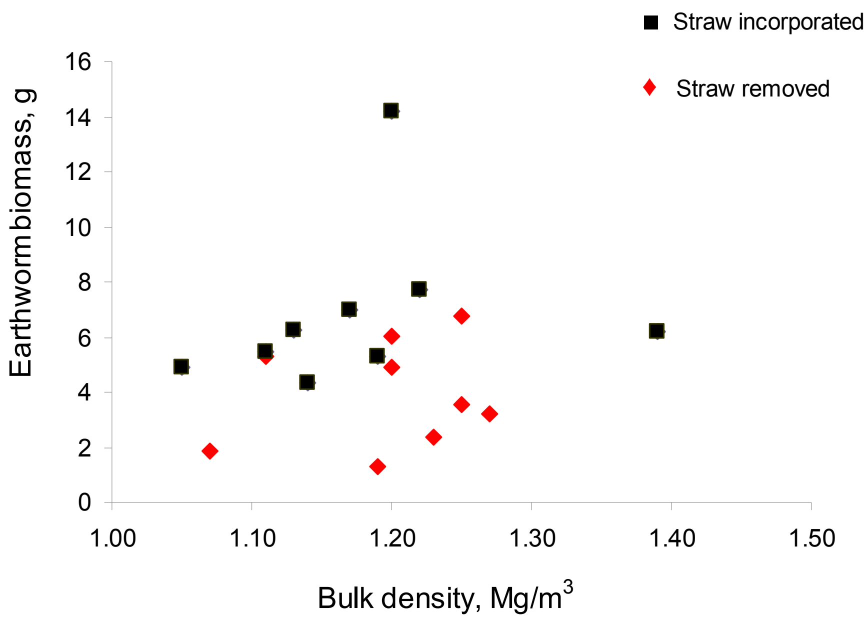
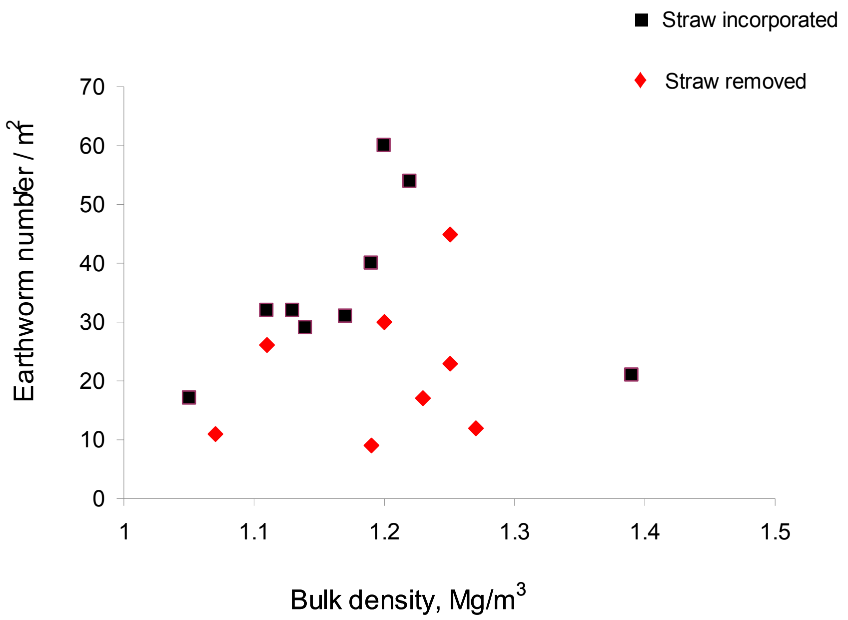


Figure 14. Earthworm number versus topsoil bulk density, University of Plymouth long-term cultivation trial.



Earthworm populations are a potential response function for habitat support (Table 15). In this experiment, the response of earthworms to the increased food provided by the additional organic matter from straw incorporation is clear (Figures 13 and 14). In addition, the earthworm biomass and number data showed a weak, positive correlation with bulk density. However, in neither case could the 1.3 mg/m^3 'prompt value' for mineral soils be tested, as the majority of the data fell below this value.

5.5.7 Summary of the performance of bulk density as an SQI

Based on the data available, it was not possible to evaluate the performance of bulk density as an SQI, or even to make a statement regarding the suitability of bulk density as an indicator of soil condition or health. A much larger dataset is required to assess the robustness of the proposed bulk density 'prompt values'. Hence, it has not been possible to road-test the 1.3 or 1.0 mg/m^3 'prompt values' for mineral or organic soils, respectively.

5.5.8 Recommendations for further data collection or research required

Several national soil, vegetation and environment surveys, which include measurements of bulk density, have been conducted recently in the UK:

- The Countryside Survey (CS2007), which included bulk density measurements for the first time. Work is on-going to investigate relationships between soil properties and invertebrate populations (Spurgeon, *pers. comm.).*
- The UK Intensive Forest Monitoring Programme (BIOSOIL) and Level II survey (Forest Research) contains bulk density data as well as a range of other soil variables. These data were examined to investigate the possibility of correlating bulk density with tree health condition scores, but due to wide variation in other

soil/site conditions between plots, the data were not suitable for evaluating the direct impact of bulk density on tree health.
- Bulk density was measured as a part of other national surveys, as summarised in the SNIFFER LQ09 project (Emmett, 2006), but invariably only focusing on biomass production rather than the EI and HS ecosystem functions under discussion here.

Bulk density data collected during CS2007, for example, will form a baseline, such that when the next Countryside Survey is conducted there will be a nationwide dataset allowing temporal comparisons and assessments to be made, which may also have value in testing the proposed ‘prompt values’. However, there will still be limitations in interpretation, as identified by Sparling *et al.* (2004). Bulk density is a transient soil property that shows temporal variation, particularly for soils with high clay content. In the case of CS2007, samples were collected throughout the year due to the scale of the sampling operation. This is likely to increase variability in the dataset and hence place some limitations on data interpretation.

Schipper and Sparling (2000) and Sparling *et al*. (2004) reported that bulk density can be used in a minimum dataset only to indicate changes in soil health and to make observations related to broad land use classes. The BD2304 review (Defra, 2008a) supported this view, and concluded that for grassland systems, published data on compaction impacts on key ecosystem functions was very scarce, despite the experimental evidence that exists demonstrating a link between compaction, runoff, and flood risk on the one hand, and below ground biodiversity on the other. This study reinforces the recommendation for future work in the BD2304 report that a national survey on the extent, nature and compaction under different grassland management systems and climates is needed. This will allow further detailed investigation of the robustness of the proposed bulk density ‘prompt values’ for the grassland habitats identified by the Environment Agency (2006a).

5.6 Organic carbon

5.6.1 Reasons for the selection of organic carbon as a SQI

Soil organic carbon (SOC) has been identified as a key indicator of soil quality, due to its influence on a wide range of soil physical (for example aggregate stability, water holding capacity, strength), chemical (for example cation exchange capacity and nutrient supply) and biological (for example microbial biomass and respiration) properties (Sikora and Stott, 1996; Carter, 2001; Kemper and Koch, 1966).

Arable cropping has been widely implicated in the deterioration of soil quality through the depletion of soil organic carbon reserves because of oxidation following cultivation (Alison, 1973). Also, reductions in the use of fertiliser nitrogen may result in decreased soil organic carbon levels due to lower crop residue returns (for example see Mihaila and Hera, 1994). Additionally, climate change will affect soil carbon turnover, with higher temperatures potentially increasing rates of organic matter decomposition (Powlson, 2005). It has been estimated that soils in England and Wales are losing carbon at a mean rate of 0.6 per cent per year (Bellamy *et al*., 2005). This not only has implications for the capacity of soils to supply water and nutrients, but their ability to resist erosion and store carbon.

There is growing emphasis being placed on soil carbon storage (sequestration) in the mitigation of climate change, and various measures are being explored to determine how best soil carbon storage and soil organic carbon levels can be increased. For example, Defra's Draft Soil Strategy (priority area 2) is aimed at 'halting the decline in soil carbon' (Defra, 2008b). Moreover, the Sustainable Farming and Food Strategy has a target 'to halt the decline in soil organic matter in vulnerable agricultural soils by 2025, whilst maintaining as a minimum, the soil organic matter of other agricultural soils, taking into account the impacts of climate change' (Defra, 2002).

5.6.2 Prompt values for organic carbon

The Environment Agency (2006a) recommended that soil organic carbon should be measured at all sites in any national soil monitoring programme in order to assess trends in soil organic carbon storage and their impacts on soil quality. However, because there was no clear indication of a change or breakpoint from any of the data reviewed, no 'prompt values' for EI or HS were suggested that could be used in a road-testing assessment *per se*. In addition, the Environment Agency (2006b) reviewed potential SQIs for habitat support and concluded that it was more appropriate to provide information on the direction and magnitude of change from a previous (baseline) sampling, with a change of 20 per cent in the value of soil organic carbon suggested to warrant further investigation.

5.6.3 Response variables

Key potential response variables that could be used for road testing soil organic carbon as an SQI were derived from studies reported by Environment Agency (2006a) and the wider literature (Table 16).

Table 16. Response variables for soil organic carbon (with indicative interpretation in italics)

Environmental Interaction	Habitat support
Aggregate stability/water storage (*surface sealing*, *susceptibility to flooding, pollutant transfer, erosion risk*)	Available water capacity (*drought tolerance*)
Biomass/respiration rate (*carbon dioxide release/carbon storage*)	
Pesticide/metal/nutrient retention (*pollutant transfer)*	

5.6.4 Literature and data search

i) Environmental interaction

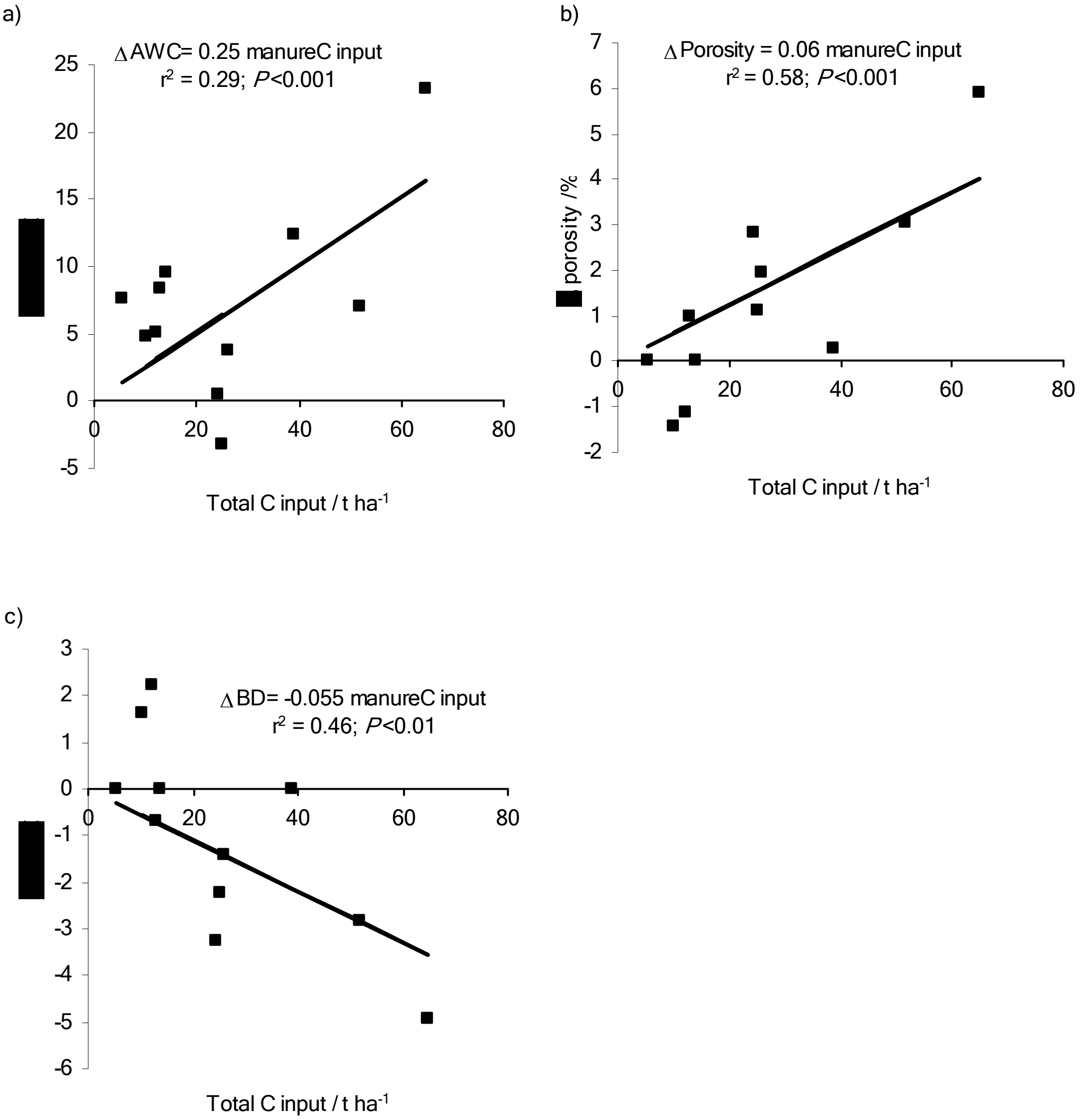
There have been a number of studies that have shown positive relationships between soil organic carbon and potential response variables (for example aggregate stability, water storage potential, pesticide retention, biomass size, respiration rate and so on), which can indicate how well a soil is performing in terms of its environmental interaction function.

Aggregate stability affects the ability of a soil to resist the potentially destructive impacts of cultivation and intense rainfall, and influences the risk of pollutant transfer via soil erosion processes. Kemper and Koch (1966) working with soils from North America reported an increase of approximately 20 per cent in average aggregation as the soil organic carbon content increased from 2.0 to 3.5 per cent. In a later study, the stability of aggregates from 26 British agricultural soils was found to be significantly correlated with soil organic carbon content (Chaney and Swift, 1984). Also, Christensen (1986) and Blair *et al.* (2006) reported a similar correlation between soil organic carbon and aggregate stability, whilst other authors (for example Dormaar, 1983) noted that soil organic carbon fractions (such as humic and fulvic acids and polysaccharides), rather than total soil organic carbon are more important to aggregate stability. Moreover, the effect of soil organic carbon is greater in poorly structured soils - adding organic matter to stable soils may do little to improve their aggregate stabilities (for example see Mbagwu *et al.*, 1993). In a review of aggregate stability and its relation to soil organic carbon content, Teh (1996) stated that soils have a limited capacity for organic matter accumulation, implying that there would be a point where further addition of organic matter would not further improve aggregate stability. Indeed, Greenland *et al.* (1975) reported that for a range of English and Welsh soils, the 'critical' amount of organic matter needed for no slaking and dispersion of aggregates was around four per cent. In summarising the results from these and other studies, the Environment Agency (2006a) concluded that there was no readily identifiable breakpoint in the relationship between soil organic carbon and aggregate stability.

Organic matter can hold up to 20 times its weight in water and can therefore directly affect soil water retention, as well as indirectly through its effects on soil structure (Dick and Gregorich, 2004). Hollis *et al.* (1977) found that the soil moisture content at 5kPa increased from 25 to 50 per cent as the soil organic carbon increased from 0.5 to 4.0 per cent, although there was no further increase in soil moisture at soil organic carbon > five per cent (8.6 per cent soil organic matter). Blair *et al.* (2006) reported a similar correlation between soil organic carbon and water infiltration rate on soils from the Broadbalk Experiment at Rothamsted.

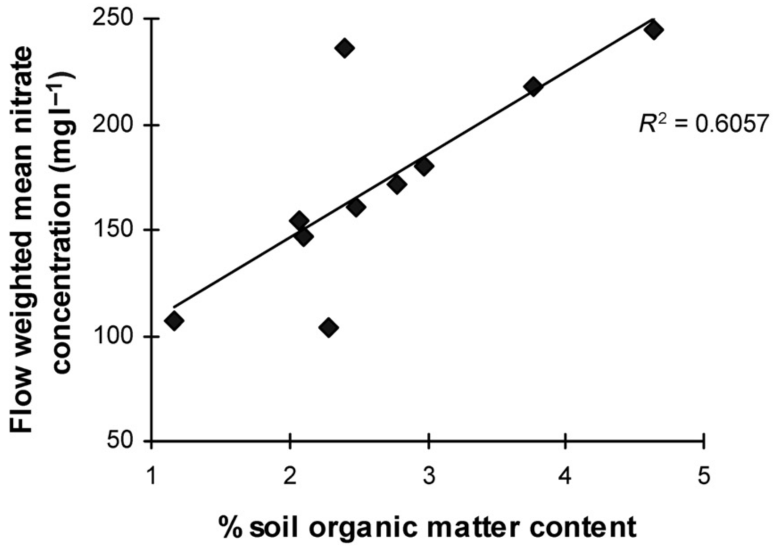
In experiments where the soil organic carbon at six sites in England was increased by the addition of organic matter in manures or crop residues, Bhogal *et al.* (in press) showed that the soil available water capacity increased by approximately 2.5 per cent with every 10 t/ha manure organic carbon applied (Figure 15a). There was also an associated 0.6 per cent increase in topsoil porosity and 0.5 per cent decrease in bulk density with every 10 t/ha manure organic carbon applied (Figures 15b and c). More recent data from these sites suggests it is not the 'fresh' additions, but the total organic carbon load that is responsible for the effects on these soil properties (A. Bhogal, pers. comm.)

Figure 15. Change in topsoil physical properties with total carbon inputs in manure: a) Available water capacity (AWC); b) Porosity; c) Bulk density (BD). Results are expressed as percentage difference from the untreated control treatments (Bhogal *et al*. in press)



The soil organic carbon status of a soil is known to influence the retention of pollutants and nutrients. The Environment Agency (2006a) reported that at soil depths <15 cm pesticide leaching increased where the soil organic carbon was less than 3-4 per cent. Many pesticide studies have shown a clear relationship between declining soil organic carbon and increasing pesticide concentrations in leachates, but with no unambiguous threshold where the soil depth was >30 cm (Pesticides in the Environment Working Group, 2000). Recently published research (Pretty *et al.*, 2007) showed that there was a positive relationship between the soil organic matter (and hence soil organic carbon) content and the nitrate concentrations measured in drainage water after vining pea crops (Figure 16). Again there was no obvious 'break-point' in the relationship. Interestingly, the data appear to challenge the common understanding that higher soil organic carbon levels are always beneficial in term of the soil's EI function of retaining soil nutrients.

Figure 16. Effect of soil organic matter on nitrate leaching after vining pea crops (Pretty *et al*. 2007)



The soil microbial biomass is a component of the soil organic carbon pool which acts not only as an agent for the transformation and cycling of organic matter and nutrients, but also as a source and sink of labile nutrients (Gregorich *et al.,* 1994). Soil microbial biomass measurements therefore give an indication of a soil's ability to store and recycle nutrients and energy. They have also been suggested as a more sensitive indicator of changes in soil organic matter cycling than measurements of total soil organic carbon (Powlson *et al.,* 1987; Omay *et al.,* 1997). Bhogal *et al.* (in press) reported that the soil microbial biomass carbon increased by approximately11 per cent with every 10 t/ha manure or crop residue organic carbon applied. This increase in size of the microbial biomass was associated with an increase in microbial activity (as measured by the soil respiration rate - i.e. CO_2-C production) which increased by approximately16 per cent with every 10 t/ha manure organic carbon applied. The biomass quotient (that is, the ratio of biomass carbon: soil organic carbon) increased with organic carbon inputs, indicating that the microbial biomass formed a higher proportion of the total soil organic carbon pool at higher organic carbon input levels.

For almost all of these relationships there was little clear evidence of an unambiguous change or break-point that could be used as the basis for a 'prompt value'. Indeed, Loveland *et al.* (2001) concluded that there was 'little consistent evidence that there are critical thresholds of soil organic carbon above or below which soil properties change significantly'. In later work, Loveland and Webb (2003) reviewed the evidence for a critical level of organic matter in temperate agricultural soils and discussed the widely accepted threshold of two per cent soil organic carbon (3.4 per cent soil organic matter) below which a potentially serious decline in soil quality will occur. This value (two per cent) was derived from work by Kemper and Koch (1966) and Greenland *et al.* (1975). The latter study was from the UK where soils with < two per cent soil organic carbon were found to be prone to structural deterioration. However, Loveland and Webb (2003) examined the quantitative evidence for such a threshold and again concluded that although many studies have demonstrated numerical relationships between soil organic carbon and various soil properties, firm evidence of a threshold above or below which the contribution of carbon increases or decreases significantly is rare. Identifying ‘prompt values’ for soil organic carbon is further complicated by temporal changes in soil organic carbon driven by changes in land use (for example from grassland to arable) and cropping, which alter the inputs and turnover/loss of soil carbon.

Verheijen *et al.* (2005) using data for arable and ley-arable soils from the National Soil Inventory (excluding calcareous and peaty soils), found that clay content and precipitation were the most important factors controlling the soil organic carbon content. These authors proposed ‘indicative soil organic carbon management ranges’ for different soil clay contents and environmental conditions (Table 17), which reflect the finding that soil organic carbon was generally greater in areas of high precipitation and tended to increase with increasing soil clay content. These ranges could be used as the basis for target soil organic carbon ‘envelopes of normality’ for agricultural soils, although they provide no indication of how well a soil is fulfilling its EI function.

Table 17. Indicative soil organic carbon (per cent) management ranges for arable/ley-arable and permanent grassland soils of different clay content under different precipitation conditions (from Verheijen *et al*. 2005)

	Dry (<650 mm/yr AAP)		Intermediate (650-800 mm/yr AAP)		Wet (800-1100 mm/yr AAP)		Permanent grassland	
Clay content (%)	Lower	Upper	Lower	Upper	Lower	Upper	Lower	Upper
0-10	0.5	1.6	0.5	2.1	0.6	3.4	1.0	4.2
10-20	0.7	2.2	0.7	2.8	1.0	3.8	1.5	4.7
20-30	1.0	2.8	1.1	3.4	1.5	4.3	2.0	5.3
30-40	1.2	3.5	1.5	4.0	1.9	4.7	2.4	5.7
40-50	1.5	4.1	1.8	4.7	2.3	5.3	2.9	6.3

AAP = Average annual precipitation

ii) Habitat support

Broad ‘envelopes of normality’ for soil organic carbon were identified for some habitats using data from the Countryside Survey 2000 (Table 18; Environment Agency, 2006a). However, these may represent extreme ranges in soil organic carbon and include sites which are not functioning well in terms of the ecology supported. In this study, we used data from the Defra ESA grasslands project BD1456 (Critchley *et al.*, 2001a,b) and Defra project BD1456 (‘The Impact of Organic and Inorganic Fertilisers on Semi-Natural Neutral Grasslands’) and compared the soil organic carbon ranges for selected habitats in the Countryside Survey 2000.

Table 18 shows that soil organic carbon values from the two Defra studies were often outside the ‘envelope of normality’ derived from the Countryside Survey. If we assume that the ESA grasslands were all functioning well, this indicates that valuable habitats can exist at what may be considered to be either low or high soil organic carbon values. This supports the conclusion of the Environment Agency (2006a) that the use of soil organic carbon ‘prompt values’ (or ranges) is not appropriate for the habitat support function.

Table 18. Range of soil organic carbon levels (per cent) in semi-natural habitats in the UK

	Countryside Survey 2000*			ESA Grasslands*/Defra project BD1456		
Habitat	No. of soils	Min.	Max.	No. of soils	Min.	Max.
Calcareous grassland	6	13.1	21.2	54	1.7	18.0
Neutral grassland	43	1.16	28.7	211	1.8	37.8
Acid grassland	64	3.7	56.4	13	0.2	32.5
Dwarf shrub heath	93	4.2	56.8	25	2.1	38.7
Broadleaf woodland	62	2.16	56.0	ND	ND	ND
Coniferous woodland	84	1.74	56.6	ND	ND	ND
Improved grassland	310	1.85	53.6	ND	ND	ND
Bog	126	6.6	56.7	ND	ND	ND
Bracken	18	4	55.3	ND	ND	ND

*Measured by loss on ignition and converted to SOC using Ball (1964) equation
ND = No data

5.6.5 Using soil organic carbon as an SQI

There is some evidence that soils in the UK may be losing carbon, probably as a consequence of land-use change, particularly drainage of peat soils and a legacy of ploughing out grasslands, and this could have implications for climate change. For example, Bellamy *et al.* (2005) used data from the National Soil Inventory (NSI) collected between 1978 and 2003, which indicated that carbon was lost from soils across England and Wales at a mean rate of 0.6 per cent/yr (or 13 million tonnes/yr). The relative rate of carbon loss increased with soil organic carbon content, being more than two per cent/yr in soils with soil organic carbon contents greater than 10 per cent irrespective of land use. Similarly, Webb *et al.* (2001) using NSI data from agricultural soils in England and Wales showed that concentrations of soil organic carbon decreased in some soils between 1980 and 1995, especially where soils were ploughed out of grassland and on lowland organic and peaty soils in tillage cropping. The mean soil organic carbon content of soils in arable/ley cultivation in 1980 was 3.4 per cent compared with 2.8 per cent in 1995, a loss rate equivalent to 0.04 per cent/yr. Similarly for permanent (managed) grass the soil organic carbon decreased from 5.0 per cent in 1980 to 4.2 per cent in 1985 (a loss of 0.05 per cent a year). In contrast, data from the Countryside Survey 2000 for a range of semi-natural and agricultural soils suggested that there had been an overall increase (or at least no change) in soil organic carbon between 1978 and 2000 (Black *et al.*, 2002).

The above contrasting conclusions regarding both the direction and size of any changes in soil organic carbon levels reinforce the long-term requirement to continue to measure and monitor changes in soil organic carbon, taking into account the practical challenges that need to be faced when attempting to measure these changes, namely:

- Time required to detect changes
- Spatial variability
- Natural variations in soil organic carbon
- Distinguishing natural changes from management induced changes

Fang and Smith (in Environment Agency, 2006a) concluded that changes in soil organic carbon may not be detectable for many years due to high spatial heterogeneity and because the changes are small relative to the total soil organic carbon content.

It is worth considering 'active carbon' or 'light fraction' organic carbon (LFOC) as a more sensitive indicator to determine the direction of change in soil organic matter content (compared to measurement of the total soil organic carbon pool). LFOC is a transitory pool of soil organic matter between fresh residues and humified, stable organic matter, and is considered to be a labile source of soil carbon, which is more sensitive to changes in soil management practices than the total soil organic carbon pool (Gregorich *et al.*, 1997; Loveland *et al.*, 2001). Malhi *et al.* (2003) observed that the increase in soil LFOC (156-697 per cent increase) was much greater than the increase in total soil organic carbon (17-67 per cent increase), following long-term (27 years) applications of increasing rates of inorganic nitrogen fertiliser. Similarly, Bhogal *et al.* (in press) reported changes in LFOC over four-fold greater than those measured in the total soil organic carbon pool, despite LFOC only comprising 3-12 per cent of total soil organic carbon. This pool could therefore potentially be used as an 'early indicator' of the impact of changed management/environmental conditions on total soil organic carbon.

The development of a national monitoring programme should focus on the detection of the magnitude and direction of changes in soil organic carbon levels, rather than to compare soils against an absolute 'prompt value'. This would be compatible with the requirement in the proposed EU Soil Framework Directive to identify areas where organic matter decline has or is likely to occur (CEC, 2006). However, there is still a debate to be had over what constitutes an acceptable or tolerable change in soil organic carbon for different soils and their EI or HS functions, although Bellamy *et al.* (2005) indicated that a two per cent decline in soil organic carbon on organic soils was 'cause for concern'.

6 Summary and conclusions

6.1 Summary

The UK Soil Indicators Consortium identified a number of soil quality indicators (SQIs) and related ‘prompt values’ that could be used to assess soil quality against a range of soil functions. This project aimed to validate (‘road-test’) the proposed SQI ‘prompt values’ in terms of their effectiveness and practicality on a range of soils. Their effectiveness was gauged in terms of whether the ‘prompt values’ were predictive of risk conditions for soils where secondary response variable data were also available. An important part of the project was also to enhance the scientific evidence base underpinning the existing ‘prompt values’ and, if appropriate, to suggest revisions to the values. The SQIs needed to be validated beyond modelled scenarios and the academic literature to determine whether they fulfilled their requirement as appropriate ‘prompt values’ for potential ecological risk in the context of a survey of soil quality assessment and monitoring

Where appropriate data were available, the methodology developed in Phase 1 of this study (metals – see Appendix I), was extended to allow a preliminary assessment of the performance of the ‘prompt values’ in protecting soil quality, and to highlight instances where they may be under-, over- or sufficiently protective of soil quality.

A summary of the SQIs that were road-tested in this study and an assessment of the quantity and quality of data currently available for road-testing is given in Table 19.

Table 19. Summary of the road-tested SQIs and an assessment of the quantity and quality of data currently available for road-testing

SQI	Suitable for road-testing	Road-testing undertaken		Quantity and quality of data	
		EI	HS	EI	HS
Extractable P	Y	Y	Y	Good	Adequate
C:N ratio	Y	-	Y	-	Adequate
pH	Y	Y	Y	Poor	Adequate
Bulk density	Y	N	N	None	None
OC	N	N	N	-	-

Y=Yes; N=No

6.2 Conclusions

6.2.1 General

- In Phase 1 of this study (metals), it was relatively straightforward to define ‘biological’ response variable thresholds against which the performance of the ‘prompt values’ in different regimes could be tested (for example the legislative limit for grain cadmium). Phase 2 of the study has shown that it is not always straightforward to define similar response variable thresholds for the other SQIs (for example for drainage water phosphorus concentrations or species richness ranges for different habitats) for use in a PASS/FAIL ‘road-testing’ assessment. Indeed in many cases the linkage between cause and effect is complex and uncertain (especially for bulk density and organic carbon), so the road testing methodology can only be used to provide a preliminary indication of the performance of the ‘prompt values’.

- Also in Phase 1 of the project, there was a large amount of data available from controlled experiments where the only variable altered was the metal(s) being studied (such as the Long-term Sludge Experiments). There was little equivalent data available for the other SQIs, usually because the SQI was not the only variable to alter between study sites/soils. In addition, the interpretation of experimental data was often difficult because of the narrow range in SQI values for the sites/soils studied,
- The road-testing methodology can be used with a dataset of soils to assess the performance of ‘prompt values’ in terms of how many of the soil are under-, over- or sufficiently protected. However, this raises the question of what is an acceptable number of under-protected soils in this context, since failing a ‘prompt value’ simply means moving to another tier of investigation.

6.2.2 Extractable phosphorus

- The results of the road-testing assessment suggested that the ‘prompt value’ for EI of 60 mg/l soil extractable phosphorus may not be sufficiently protective of water quality, recognising that the relationship between soil extractable phosphorus and surface water phosphorus concentrations does not always exhibit a clearly defined threshold/’break-point’ in all soil types and situations.
- There was evidence from the literature review that the EI ‘prompt value’ for extractable phosphorus may need to recognise differences in soil types, /management, and climate conditions.
- A single ‘prompt value’ for HS of 10 mg/l was suggested for all mesotrophic grasslands. Results from this study suggest that it may be appropriate to further subdivide this category of grasslands and provide ‘prompt values’ for each subdivision.
- The HS ‘prompt value’ for calcareous grasslands (currently 16 mg/l) was assessed to be too high as most species-rich CG grassland in this study had extractable phosphorus concentrations <10 mg/l. A more appropriate ‘prompt value’ would be 10 mg/l.

6.2.3 Carbon:nitrogen ratio

- The number of soils assessed as over-protected in the road-testing exercise strongly suggests that the carbon:nitrogen ratio ‘prompt values’ for grassland communities are set too high (or the ranges are too narrow). The field experimental data show that valuable habitats are able to flourish where the soil carbon:nitrogen ratio is considerably lower than those suggested by the ‘prompt value’ ranges.

6.2.4 Topsoil pH

- From the data available, road-testing of the soil pH ‘prompt value’ for EI on mineral soils, indicated that the value is robust and probably set at about the correct level.

- For the HS function, the topsoil pH ‘prompt values’ were reasonably successful at protecting soils (for all but the mesotrophic grasslands on peaty soils), with less than 15 per cent of the soils in each category under-protected.
- The heathland soils dataset showed that the pH was generally >4.5 (but <5), so the distinction between mineral/organic and peaty soil types for the HS function is probably not necessary.
- Note that there are some instances where the soil pH ‘prompt values’ for EI and HS are very different and may be contradictory. Careful attention should be given to potential anomalies in the design of a soil monitoring programme where these ‘prompt values’ are used.

6.2.5 Bulk density

- Based on the data available, it was not possible to evaluate the performance of bulk density as an SQI, or even to make a statement regarding the suitability of bulk density as an indicator of soil condition or health.

6.2.6 Organic carbon

- No ‘prompt values’ for EI or HS were suggested that could be used in a road-testing assessment. This study found no evidence to contradict the conclusion of Loveland *et al.* (2001) that there is little consistent evidence that there are critical thresholds of soil organic carbon above or below which soil properties change significantly.
- A national monitoring programme should focus on the detection of the magnitude and direction of changes in soil organic carbon levels, rather than comparing soils against an absolute ‘prompt value’. However, there is still a debate to be had over what constitutes an acceptable or tolerable change in soil organic carbon for different soils and their EI or HS functions.

6.2.7 Overall conclusions

- This study has demonstrated that despite some limitations, it was possible to use the ‘road-testing’ methodology to validate ‘prompt values’ for some SQIs, and in some cases suggest revisions to the ‘prompt values’. However, the method was not appropriate for use where no specific ‘prompt values’ were defined (for example, soil organic carbon) or where there was a lack of suitable data relating the SQI to response variables (for example, bulk density).
- In Phase 1 of this study (metals), it was suggested that within a national monitoring scheme, it may be appropriate to provide an *early warning* that ‘prompt values’ were being approached and if a soil exceeded the ‘early warning’ value, it would trigger the next tier of investigation. For the other SQIs,evaluated in this study, the ‘prompt values’ are less certain and the introduction of additional ‘early warning’ values would probably not be productive. However, in terms of the national dataset as a whole, it would be pertinent to assess whether there has been a change in the number of soils exceeding the ‘prompt values’, as well as changes over time from the baseline measurement. For example, if in the baseline year five per cent of soils in the monitoring scheme fail the ‘prompt value’ for extractable phosphorus and if at the next sampling date this had increased to eight per cent, this would trigger further site specific investigation.

7 Recommendations for further work

- The results of the road-testing assessment undertaken during this study would suggest that further evaluation of the extractable phosphorus ‘prompt value’ for EI is required in light of catchment scale dilution effects and the relative contribution of urban and rural sources, and water body sensitivity (running *vs*. still waters).
- In terms of the habitat support function, for very low phosphorus systems, it is possible that Olsen extractable phosphorus may not provide a sufficiently sensitive measure of changes in phosphorus status and an alternative measurement technique may need to be investigated.
- Further consideration and justification is required for the continued inclusion of soil carbon:nitrogen ratio as a key SQI for the habitat support function. There would be some merit in revisiting the Countryside Survey (CS, 2000) dataset from which the carbon:nitrogen ratio ‘prompt values’ were derived to see how the habitats described compare with the ESA data set used in this study for road-testing.
- There would be merit is obtaining further data on relationships between soil pH and soil biological activity, that is, biomass size and respiration rates, and invertebrate populations (for example earthworm and collembolla numbers). For example, at low pH earthworms are replaced in their function of SOM breakdown by enchytaeides. However it is a matter of judgement whether the non-presence of earthworms in acidic soils is a ‘bad’ thing since it is ‘natural’.
- Bulk density data collected during CS2007, for example, will form a baseline, such that when the next Countryside Survey is conducted there will be a nationwide dataset allowing temporal comparisons and assessments to be made. This may also have value in testing the proposed ‘prompt values’, although there may still be limitations in interpreting the data.
- This study reinforced the recommendation from Defra project BD2304 that a national survey on the extent, nature and compaction under different grassland management systems and climates is needed. This will allow further detailed investigation of the robustness of the proposed bulk density ‘prompt values’ for the grassland habitats.

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9 Appendix I – Phase 1 Report: Executive Summary

The Environment Agency has long recognised the need for proportionate, consistent, evidence-based environmental protection across regulatory regimes. This has arguably been achieved for surface waters and air. However, relatively limited direct regulation has previously existed for soils. As a modern regulator, the Environment Agency must ensure that standards for soils are used as part of a risk-based approach, as well as being protective of soils and scientifically robust. We seek to avoid standards that are inconsistent across regimes or industries, or overly protective standards that are a burden on industry and landowners and can cause 'pollutant swapping' or other environmentally detrimental effects. Standards should be simple to use and their basis transparent.

The developing policy context, including the need to develop a national soil monitoring network, led the Environment Agency to undertake a project under the auspices of the UK Soil Indicators Consortium to identify a number of soil quality indicators and related 'prompt values' to be used to assess soil quality against a range of soil related functions. Soil quality in this context can be defined as 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. 'Prompt values' were recommended for a set of indicators to be used within a tiered hierarchy. 'Prompt values' can be considered as 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. The 'prompt values' were based on scientific evidence, modelled datasets and, in some cases, expert opinion.

The soil indicators, which were primarily chosen for their role in indicating adverse effects upon the soil function of 'environmental interaction'[5], comprised: extractable phosphorus (Olsen P), soil organic carbon (SOC), pH, bulk density, total nitrogen (N), and aqua regia extractable (total) cadmium (Cd), copper (Cu), nickel (Ni), zinc (Zn) and lead (Pb). This report focussed on the performance of the 'prompt values' for metals.

A number of other standards or 'prompt values' for metals in soil currently exist in the scientific, policy and regulatory environments. For example, sewage sludge applications are not permitted on agricultural land where soil metal concentrations exceed specified values and metal additions must not exceed specific loading rates. Indeed, the sludge limits are widely used across many regulatory regimes as *de facto* environmental limit values, covering a range of protection goals for which they were not originally derived. Very few limit values for metals used across the world for environmental and ecosystem protection have been validated. Validation or 'road testing' in this sense means an assessment in the field in order to establish whether the protection goal for which the limit value was set is achieved. There was a clear need to assess the effectiveness of both the established limit values and the proposed 'prompt values' at fulfilling their soil protection goals.

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[5] From the six functions of soil: Support of ecological habitat and biodiversity; Food and fibre production; Environmental interaction; Providing a platform; Providing raw materials; and Protecting cultural heritage (from Blum, 1993)

This project therefore aimed to validate (‘road test’) existing soil metal limit values and proposed soil ‘prompt values’ in terms of their effectiveness and practicality on a range of soils falling under different soil-related regulatory regimes. The ‘effectiveness’ was gauged in terms of whether the limit values were predictive of risk conditions for soils for which secondary data on soil biological process were also available. This is the first time soil limit values for metals have been validated in this way.

A comprehensive database of more than 500 field soils that had received organic material additions (such as sewage sludge, farm manures, compost, paper crumble) was compiled from a number of experiments, with sites representing different soil types and land uses throughout Britain. Each database entry consisted of a suite of soil data (including texture, pH, organic matter content and metal concentrations) together with one or more biological measurements (such as soil microbial biomass, rhizobia numbers, respiration rate, wheat grain cadmium content, earthworm numbers). The biological measurements were used as an indicator of whether there had been a decrease in soil quality due to metals in the organic materials added. A methodology was developed to allow the performance of the different regimes in protecting soil quality to be compared using the soils database, and to highlight instances where the regimes may be under-, over- or sufficiently protective of soil quality.

One of the key findings of the project was that the existing limit values in the Sludge (Use in Agriculture) Regulations and the Code of Practice for Agricultural Use of Sewage Sludge (that is, the existing UK sludge limits) may not be sufficiently protective of soil quality. These values are used not only in these regimes but also ‘read across’ into other legislation and guidelines related to the spreading of materials onto land.

The results also indicated that the Code of Practice for Agricultural Use of Sludge was under-protective of wheat grain cadmium concentrations. This assessment is supported by recently published work, which showed that the current UK soil total cadmium limit of 3 mg/kg was not sufficiently protective against producing grain above the EU grain cadmium maximum permitted concentration of 0.2 mg/kg (fresh weight), unless the soil pH was >6.8. Soil pH has long been known to be a key factor influencing the availability of many metals in soils, although current UK approaches only consider pH as a factor controlling zinc, copper and nickel bioavailability. Given the weight of research evidence and the findings from this project of reduced numbers of under-protected soils where regimes take soil pH into account, we recommend that future changes to UK legislation should embody a pH-based approach for setting soil cadmium limits.

The implementation of lower limits for zinc, copper and cadmium proposed in the EC Working Document on Sludge would reduce the number of under-protected soils compared with current UK legislation. However, the regime had a high number of over-protected soils, which may indicate that the limits for these metals are set too low for some soils. In general, the EU Risk Assessment/SSV ‘prompt values’ performed well in predicting potential risks, but were also over-protective in some situations.

Overall, the project concluded that there was a balance to be struck between environmental protection and regulation and the sustainable recycling of organic material to land. What was clear was that both the science and understanding of metal behaviour in soils has moved on significantly in the last decade and more accurate predictions of metal risks are now possible. Furthermore, it is apparent that the existing limit values for metals in soils, so widely used for the *de facto* assessment of environmental metal risks, may not be wholly protective of soil quality.

It is likely that some soils within a national monitoring scheme may have relatively low metal concentrations, below the ‘prompt values’ specified in all the regimes tested in this project, including the EU Risk Assessment/SSVs that were proposed for this purpose. Nevertheless, increased metal concentrations in such soils could be indicative of a long-term threat to soil quality. Hence, within a national monitoring scheme, it may be more appropriate to look at using ‘prompt values’ in combination with an assessment of changes in soil metal concentrations over time above a specified value. Also, in order to provide an early warning that ‘prompt values’ were being approached, a 75 per cent of the ‘prompt value’ ‘early warning’ limit could be set. This depends on when metals are to be considered in the monitoring scheme. If metal levels are quantified at an early stage of the scheme (that is, at tier one), limited information will be available for each site and an early warning system is advisable. However, at a higher tier (that is, tier 2 or above), information that triggered further investigation and metal analyses will be available and use of the prompt values is advised, in line with the Contaminated Land regime. In addition, in terms of the national dataset, it would be pertinent to assess if there has been a change in the number of soils exceeding the ‘early warning’ and ‘prompt values’ as well as changes over time from the baseline measurement.

Source: Environment Agency (2008)

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