An environmental soil test to estimate the intrinsic risk of sediment and phosphorus mobilization from European soils

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Abstract

Methodologies are required to help identify soils that are vulnerable to both suspended sediment (SS) and phosphorus (P) transfer in land run-off to combat the adverse impacts of agriculture on water quality. A laboratory test that quantifies dispersed particles and associated P in the same suspension was developed to estimate the potential mobilization of SS and P due to rainfall impact from 26 European soils with varied soil physical and chemical properties and P inputs. The test recovers an aliquot of the clay and fine silt (< 20 μm) fraction of soils after gently shaking in distilled water for 1 min at a 1:50 soil-to-solution ratio and measures the dry residue, total P and dissolved (< 0.45 μm) P content. The results of the test correlated well (r^2 = 0.7–0.8) with the amounts of SS, total P and dissolved P in overland flow generated by indoor simulated rainfall (intensity 60 mm h^{-1} for 30 min and a 5° slope). Variation in SS and particulate P mobilization was linked to soil pH, organic matter (or clay) and sesquioxide content, although a multiple regression analysis showed these factors accounted for no more than 55% of this variation. Ranking showed that the soils generating the most sediment did not necessarily generate the most P loss due to variable degrees of P enrichment of the particulate fraction and variable contributions of dissolved P. Particulate P enrichment was related weakly (r^2 = 0.5) to soil total P, while dissolved P fractions were predicted well (r^2 = 0.8–0.9) by conventional soil P tests (water and Olsen). The environmental soil test has a potential role in identifying the comparative risk of sediment and P mobilization from critical source areas connected via both surface and subsurface pathways, and in providing data for incorporation into models predicting sediment and P transfer at the field and catchment scale.

Keywords: Suspended sediment, phosphorus, soil dispersibility, sheet run-off, mobilization

Introduction

Suspended sediment (SS) and phosphorus (P) are two important pollutants transported in run-off from agricultural land that can cause a general deterioration in the quality and biodiversity of surface waters, and threaten the survival of specific species in protected areas (Mainstone *et al.*, 2007). Turbid run-off has a number of direct physical smothering effects on aquatic plant and fish communities and recent evidence suggests that both the quantity and quality of sediment can impair fish egg survival in spawning gravels (Greig *et al.*, 2005). Phosphorus has been linked to eutrophication problems in many standing and flowing waters, and concentrations and loads of P in agricultural run-off from different land uses and land management practices are sufficiently high in some areas to potentially contribute to eutrophication

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(Foy, 2005). Within Europe, the Water Framework Directive (WFD) provides a legislative framework to support the reduction in SS and P loads required to maintain 'good' water quality through effective targeting of programmes of measures as part of river basin management planning. A range of measures are available to reduce sediment and P loads from agricultural land (Withers & Jarvis, 1998; Sims & Kleinman, 2005) and decision support tools are now required to help target these measures most effectively.

Phosphorus in land run-off occurs largely in combination with colloids, primary particles and soil aggregates and so it is appropriate to consider their mobilization and transport together (Edwards & Withers, 1998; Heathwaite *et al*., 2005). Pionke *et al*. (1997) viewed catchments as 'a collection of P sources, storages and sinks tied together by a flow framework' and that 'the interaction between P sources, storages and sinks, and flow pathways defined the key linkages from source to impact area'. The dominance of hydrological controls on P transfer in catchments is now well established, and it is common to observe that the most eroding fields are linked to topographical features that accelerate water movement (e.g. Auzet *et al*., 1993). In some catchments, the framework of flow pathways envisaged by Pionke *et al*. (1997) will have been greatly extended with the introduction of pipe drainage systems and tramlines, the presence of compacted headlands and gateways and/or the proliferation of ditches, tracks and roads. Transport of sediment and P may therefore occur some distance from the water body; Harrod & Theurer (2002) suggested by up to 1 km. Source areas within catchments will therefore be highly variable requiring knowledge of not only hydrological controls, but also the risk of sediment and P mobilization from soils by detachment within a given flow pathway. Fingerprinting studies show that the majority of SS in rivers is derived from the surface soil, including soil particles entering via pipe drainage systems (Walling, 2005). For example, Chapman (2001) found enough ^{7}Be in pipe drain sediments to suggest that they were derived from the uppermost layers of topsoil.

Particle detachment occurs through a dynamic combination of rain splash energy (Ghadiri & Payne, 1988; Sharma *et al*., 1995), the dispersion forces created by changes in ionic strength of the soil solution (Shainberg *et al*., 1992; Curtin *et al*., 1994), the slaking forces of wetting fronts as they advance through the soil (Le Bissonnais *et al*., 1989; Auerswald, 1995), and the shear forces of overland flow (Bryan, 2000). High sediment loads in land run-off occur due to rill and gully erosion during intense storms that generate suspensions with a large range of particle sizes. However, more environmentally significant, chronic transfers of soil particles (up to 1 t SS ha^{-1} yr^{-1}) also occur in shallow unconcentrated surface flow (termed sheet run-off) and in drain flow from arable and grassland fields (Stone & Walling, 1996; Sims *et al*., 1998; Haygarth *et al*., 2006). In sheet run-off, the detachment and subsequent settling of soil particles results in the selective transport of finer silt-sized aggregates and primary particles (Wan & El Swaify, 1988; Hairsine & Rose, 1991). These finer textured aggregates and particles show varying degrees of P enrichment depending on the soil type and history of P inputs in fertilizers and manures (Sharpley, 1985a,b). The greatest risk of P mobilization within both surface and sub-surface flow pathways will therefore occur where there is high vulnerability to surface detachment and selective transport of fine soil particles strongly enriched with P.

A number of routine 'environmental' tests for available P in soils have recently been developed to predict mobilization of soluble (including colloids < 0.45 μm) P in run-off at the field scale (Hooda *et al*., 2000; Heathwaite *et al*., 2005; Vadas *et al*., 2005). Such soil tests have been widely used to set limits of soil P accumulation as a part of regional management plans aimed at reducing the risk of P losses to eutrophic waters (Sharpley *et al*., 2003). There has been no attempt to use 'environmental' soil testing to characterize the risk of particulate P (PP, > 0.45 μm) mobilization at the field scale, despite the close association between sediment and P loss. A number of physically based soil tests (Bryan, 1968) and soil erosion models (WEPP, Nearing *et al*., 1989; RUSLE, Renard *et al*., 1994; EUROSEM, Morgan *et al*., 1998) exist to help predict vulnerability to erosion and soil loss, but they do not have well-developed P transport routines. Those that do (e.g. CREAMS, EPIC and OPUS) are empirical and based on single US data sets (e.g. Foster *et al*., 1985). The scientific basis for modelling of sediment-associated P transfer at the field and catchment scale is therefore very weak.

This study investigated the feasibility of a combined environmental soil test that could distinguish the intrinsic risk of both SS and P mobilization from different soils without crop cover. Simple, routine methods that can quantify the extent of P enrichment of eroding soil particles, and relate these back to soil properties, are potentially useful in further developing modelling approaches to P transfer at a range of scales.

Materials and methods

Characterization of the benchmark soils

Representative bulked samples (0–10 cm) of topsoil, collected from 24 selected experimental field sites across Europe (hereafter called the 'benchmark soils') using a common sampling regime, were allowed to air dry naturally outside, gradually broken up by hand and gently sieved to produce aggregates < 5 mm in size. Some relevant site information for each soil is given in Table 1. With the exception of Pwllpeiran and Rosemaund soils, which were in permanent grass, all soils were under cultivation and receiving variable inputs of P either as inorganic fertilizer or as organic manure. These samples were analysed for selected physical and chemical

Table 1 Origin, classification and history of phosphorus inputs of the benchmark soils

Site name	Country[a]	Soil taxonomy	Annual rainfall (mm)	Land use	Annual P inputs[b] (kg ha^{-1})
Bridgets	UK (1)	Lithic Udorthent	785	Cereals/oilseed rape	0
Boxworth	UK (2)	Aquic Eutrudept	550	Cereals/oilseed rape	0
Rosemaund	UK (3)	Oxyaquic Hapludalf	675	Grass	0
Pwllpeiran	UK (4)	Typic Dystrudept	1800	Grass	0
Gleadthorpe	UK (5)	Typic Udipsamment	630	Cereals/potatoes	0
Olcenengo	Italy (1)	Aquic Haplustalf	900	Rice	30
Villafranca	Italy (2)	Oxyaquic Ustifluvent	900	Maize/wheat/vegetables	30
Mantova	Italy (3)	Vertic Haplustepts	580	Alfalfa/wheat/sugar beet	35
Bonavia	Italy (4)	Udifluventic Haplustept	780	Maize	115 (60 as manure)
Tetto Fratti	Italy (5)	Oxyaquic Ustifluvent	755	Maize for silage	65 (35 as manure)
Riva	Italy (6)	Unclassified	755	Maize for silage	65 (35 as manure)
Sale	Italy (7)	Unclassified	755	Maize for silage	65 (35 as manure)
Jokioinen	Finland (1)	Vertic Eutrudept	650	Ley grassland	0
Turku	Finland (2)	Vertic Eutrudept	715	Cereals	15
Großenzersdorf organic	Austria (1a)	Typic Hapludoll	545	Rye/barley/fallow	35 (as manure[c])
Großenzersdorf mineral	Austria (1b)	Typic Hapludoll	545	Rye/barley/fallow	135
Rottenhaus 5	Austria (2a)	Typic Eutrudept	700	Maize/pea/cereal/potato	44
Rottenhaus 6	Austria (2b)	Typic Eutrudept	700	Maize/pea/cereal/potato	0
Ritzlhof 4	Austria (3a)	Aquic Dystric Eutrudept	750	Maize/cereal/pea	14
Ritzlhof 5	Austria (3b)	Aquic Dystric Eutrudept	750	Maize/cereal/pea	114 (100 as compost)
Keszthely	Hungary (1)	Typic Eutrochrept	687	Wheat/sugarbeet/maize	0
Szentgyorgyvolgy	Hungary (2)	Typic Epiaqualf	803	Set-aside	0
Nagyhorvati	Hungary (3)	Typic Hapludalf	687	Potato/maize/wheat	20 (12 as manure[b])
Somogybabod	Hungary (4)	Typic Udorthent	751	Wheat/barley/maize	3

[a]The site code is given in brackets, e.g. UK1, IT1, FI1, AU1a, HU1, etc. [b]Average over the preceding 5 years. [c]Estimated using an average P content of manure of 1.5 kg t^{-1} fresh material.

properties according to standard analytical methods (Sparks *et al.*, 1996): particle size distribution (PSD) after the removal of organic matter, pH in $CaCl_2$, calcium carbonate ($CaCO_3$), organic matter (OM), cation exchange capacity (CEC) at pH 8.2 using $BaCl_2$ buffered with tri-ethanolamine, and iron (Fe) and aluminium (Al) oxides (Fe_{ox} and Al_{ox}) extracted by acid NH_4-oxalate solution. Total P was determined by inductively coupled plasma-optical emission spectroscopy (ICP-OES) after digestion with concentrated sulphuric and perchloric acid (Sparks *et al.*, 1996) and Olsen P (OP) determined by colour (Murphy & Riley, 1962) after extraction in 0.5 M $NaHCO_3$ at pH 8.5 (Olsen *et al.* 1954). Water-extractable P (WEP) was determined by colour after shaking 1 g of soil with 50 mL deionized water for 1 h on an orbital shaker and allowed to stand for 23 h, re-shaken for 10 min and centrifuged for 10 min at 2500 rpm (1500 g) (Koski-Vähälä & Hartikainen, 2001). The supernatant was immediately filtered through a 0.02-μm sieve prior to analysis.

Indoor rainfall simulation

The potential mobilization of soil particles and P forms in run-off from the benchmark soils was measured under controlled conditions indoors using simulated rainfall (Miller, 2004). Each air-dried, 5-mm-sieved benchmark soil was packed into a metal tray measuring 0.5 m long, 0.25 m wide and 0.085 m deep to a bulk density of 1.3 g cm^3. The trays had perforated bases to allow drainage and were fitted with an outlet tube at one end to allow the collection of sheet run-off (Figure 1a). The soil was wetted by capillary rise for 24 h and allowed to drain for a further 24 h before the trays were placed at an angle of 5° and simulated rainfall applied at a rate of 60 mm h^{-1} for 30 min. To simulate the kinetic energy of raindrop impact, deionized water was fed by gravity through hypodermic needles and randomly dispersed through a wire mesh 9 m above the soil surface. Overland flow was collected for three successive 10-min periods after rainfall was initiated, the run-off volumes measured, and subsamples taken for determination of SS and P fractions. SS was determined as dry residue. Total P (TP) and total dissolved P (TDP, <0.45 μm) were determined by colour after sulphuric acid–persulphate digestion (Eisenreich *et al.*, 1975). Molybdate-reactive P (<0.45 μm) was determined by colour directly without filtering. After the first rainfall event (event 1), the soil trays were left for 5 days and the simulated rainfall experiment was repeated

after initial wetting (event 2). Three replicates of each benchmark soil were tested.

Aggregate stability and dispersion tests

A range of aggregate stability and soil dispersion tests were developed in the laboratory to help explain differences in the amounts of SS and P forms collected under indoor rainfall simulation. Aggregate stability tests included: (1) water stable aggregates (WSA) by a single wet sieving technique adapted from Bryan (1968) and Dexter (1988); (2) the single drop (SD) test of McAlla (1944); (3) the percolation stability (PS) test recommended by Auerswald (1995); and (4) an erodibility test used in Hungary based on soil consistency, the KA test (Sisák *et al*., 2001). Three alternative methods of soil dispersion were compared in the laboratory: (1) gentle dispersion in water (Ministry of Agriculture, Fisheries and Food, 1982); (2) chemical dispersion in sodium chloride (Goldberg & Glaubig, 1987); and (3) ultrasonic dispersion at low amplitude to mimic the kinetic energy of raindrops (Mayer *et al*., 2002). The ultrasonic dispersion methodology delivered too much energy at the vibration amplitudes tested (5 and 10 μm) to be comparable with other methods and is not discussed further here. Further information on this methodology is given in Mentler *et al*. (2004).

Water stable aggregates. This method measured the proportion of soil aggregates >125 μm remaining after wet sieving. Air-dried <5-mm soil (25 g) uniformly distributed on a 125-μm sieve was immersed in a bowl of deionized water (1200 cm^3 with 1 cm of water above the sieve) and horizontally shaken for 5 min. The soil remaining on the sieve was collected, dried at 105 °C, weighed and the percentage of soil remaining on the sieve was calculated. Three replicate measurements were made on each soil.

Single drop. This test counted the number of drops of water required to disperse a single <5-mm aggregate (McAlla, 1944). The greater the number of drops required, the more stable the soil. For each soil, 30 aggregates of the same size were all placed on a sand table at 5-cm tension for 24 h. Each aggregate in turn was then placed on a Perspex cup and exposed to single drops (SD) of deionized water dropped by a hypodermic needle 1 m above the aggregate. The number of drops needed to destroy the aggregate was counted.

Percolation stability. This test measured the amount of deionized water that passed through a column of soil under a constant head of water (Auerswald, 1995). The greater the amount of water collected, the more structurally stable the soil. Ten grams of 1- to 2-mm air-dried aggregates were put in cylindrical glass tubes 10-cm long and of internal diameter 1.5 cm. The lower ends of the tubes were covered with a fine wire mesh and a 1- to 2-mm depth of medium sand grains was placed underneath, and on top of, the aggregates. Deionized water was passed through the aggregate column under a hydrostatic pressure of 20 kPa for 10 min. Three replicate measurements were made on each soil and the volumes of water collected were recorded, and values corrected for total sand content (Mbagwu & Auerswald, 1999).

KA:clay. Soil consistency was determined as the water content of the soil (KA value; cm^3 100 g^{-1} dry soil) when it meets the conditions of the 'yarn test' (broadly analogous to the liquid limit). The yarn test is a simple routine method used in Hungary (Varallyay, 1993) which has been applied to soils for predicting their settling properties in suspension with the purpose of developing indices of erodibility (Sisák *et al*., 2001). In the test, 50 g of air-dried <5-mm soil is

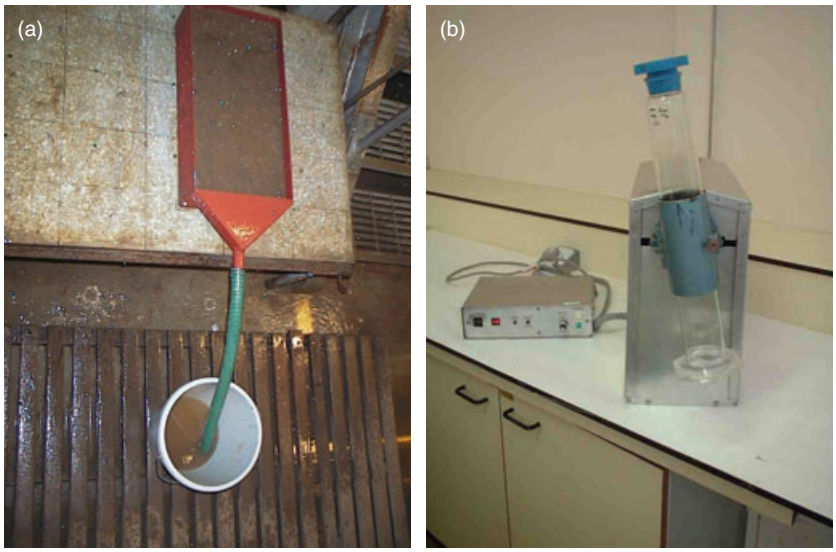

Figure 1 Photographs of (a) the collection of run-off from trays packed with the benchmark soils subjected to two simulated run-off events and (b) the laboratory apparatus designed to ensure even revolutions during the water dispersion test.

thoroughly mixed in a mortar with an increasing amount of deionized water until 'yarns' (threads) can be formed both on the pestle and on the surface of the soil paste by pushing the pestle into the paste and drawing it out quickly. The yarns must lean but must not flow together with the paste. The test represents a function of both colloid content and quality, and the ratio of the KA value to soil clay content is used as an index of soil 'erodibility'. Replicate determinations of KA carried out by an experienced technician are within ± 0.5–1; a single value for each soil was reported. Values for sands are in the range 20–21 and those for clays are in the range 60–80.

Gentle water dispersion. The method for gentle water dispersion was an adaptation of the method for soil 'dispersion ratio' routinely adopted by Ministry of Agriculture, Fisheries and Food (1982) in soil management research and adapted from the method of Middleton (1930). Air-dried < 5-mm soil (20 g) was added to 200 mL of water and left to stand for 1 h. The volume of water was then made up to 1 L and the sample gently shaken end over end for 1 min and allowed to stand for periods corresponding to particle size fractions < 63, 20, 6 and 2 μm, respectively. A custom-built apparatus, as used by Hodgkinson & Thorburn (1996) to study saline soils, was employed to ensure consistency of the energy input into each of the 10 revolutions (20 inversions) of the cylinder over the shaking period (Figure 1b). Seconds before the end of each settling period, a 20-mL aliquot of the suspension was pipetted from a depth of 10 cm below the surface. Half (10 mL) of the pipetted sample was then placed in a preweighed beaker, weighed, oven dried at 105 °C for 24 h and weighed again to determine the mass of SS. The remaining 10 mL sample was used to determine TP, TDP and MRP by the same methods as used in the indoor rainfall simulation experiment. A variant of this method employed single wet sieving (as for WSA) prior to gentle water dispersion to assess the stability of particles > 125 μm (wet sieving + dispersion).

Chemical dispersion in sodium chloride. For the chemical dispersion method, 1 g of soil was shaken for 4 h with 500 mL of 10^{-3} M NaCl (Goldberg & Glaubig, 1987). The suspensions were allowed to settle for fixed times and aliquots of the < 63, < 20 and < 6-μm fractions were collected. The suspensions were then centrifuged to keep the < 2-μm particles suspended and aliquots taken of the clay fraction. For each aliquot, TP and TDP (< 0.45 μm) were determined by sulphuric–perchloric acid digestion and MRP directly by colour using malachite green (Ohno & Zibilske, 1991). For SS, the percentage transmittance (%T) of each aliquot was determined using a spectrophotometer, having obtained good correlations ($r^2 = 0.86$–0.94; $n = 52$) between transmittance and dry residues of sediment for the different particle sizes.

Data handling and statistical analysis

The results of the indoor rainfall simulation are presented as the total amounts of SS, TP and TDP collected over each of the two 30-min rainfall events. A small number of concentration values were interpolated to allow for statistical comparison of total amounts eroded between the soils and events. Variation in the concentrations and amounts of SS and P collected within each 10-min interval are given by Miller (2004) and are not reported here. To allow for the different soil weights and sedimentation vessels used in the laboratory dispersion tests, the amounts of SS, TP and TDP (< 20 μm) dispersed from a 20-g soil sample are presented. This size fraction was most comparable with the PSD of the eroding particles (Miller, 2004). Suitability of the candidate laboratory dispersion tests for predicting the results of the indoor rainfall simulation were assessed by a single regression. Relationships between the results of all indoor and laboratory test data with the soil variables measured were assessed by a stepwise multiple regression using Genstat 5. All regression analyses involving the indoor rainfall simulation results excluded the Tetto Frati soil due to the large bias introduced by the exceptionally large amount of SS collected from this soil.

Results

Physical and chemical properties of the soils

Soil texture determined according to the USDA classification varied from sandy loam to clay with clay contents ranging from 10 to 61% (Table 2). Soil pH ranged from 4.8 to 7.7 and five soils contained appreciable amounts of $CaCO_3$ (15–22%). Organic matter content (range 1.3–5.4%) tended to increase as the content of Al oxides increased ($r^2 = 0.7$) and CEC values (6–29 mol kg^{-1}) were largely explained by differences in clay content and OM (CEC = 0.31 clay % + 2.66 OM % + 0.36; $r^2 = 0.84$, $n = 24$). There was large variation in Al_{ox} (36–130 mmol kg^{-1}) and Fe_{ox} (range 12–108 mmol kg^{-1}) which are the main P-sorbing surfaces in soils, with Al_{ox} dominating over Fe_{ox} in 13 of the 24 soils. Total P concentrations ranged from 404 to 1868 mg kg^{-1}, OP concentrations from 3 to 95 mg kg^{-1} and WEP concentrations from 0.3 to 65 mg kg^{-1} (Table 2). Hence, 50% of the soils exceeded a typical critical value of OP for arable soils of 20 mg kg^{-1} (Sibbesen & Sharpley, 1997). These analyses suggest that the selected soils were sufficiently diverse in their physical and chemical characteristics, and P status, to evaluate the suitability of an environmental soil test for estimating the risk of sediment and P mobilization.

Laboratory soil tests

The WSA, SD and PS tests produced a large range of values; WSA, 28–86%; SD 22–1978 drops and PS, 20–503 cm^3

Table 2 Selected physical and chemical properties of the benchmark soils

Site name	Sand (%)	Silt (%)	Clay (%)	pH $CaCl_2$	$CaCO_3$ (%)	OM (%)	CEC (mol kg^{-1})	Fe_{ox} (mmol kg^{-1})	Al_{ox} (mmol kg^{-1})	Total P (mg kg^{-1})	Olsen P (mg kg^{-1})	Water P (mg kg^{-1})
Bridgets	17	58	25	7.5	20	4.5	23	43	128	1433	10	3.8
Boxworth	27	34	39	6.5	0	3.2	22	27	50	502	9	2.8
Rosemaund	9	66	25	5.3	0	3.7	15	46	50	715	15	12.8
Pwllpeiran	35	40	25	6.1	0	5.4	24	108	130	1486	8	2.5
Gleadthorpe	65	25	10	6	0	2.0	8	54	57	878	25	21.4
Olcenengo	27	55	18	4.8	0	2.3	9	66	41	652	20	0.3
Villafranca	41	46	13	7.3	0	1.7	9	47	36	1452	73	44.7
Mantova	14	53	33	7.6	18	2.4	22	53	71	1336	52	20.5
Bonavia	24	52	24	7.2	2	3.3	15	45	49	955	26	10.3
Tetto Fratti	27	61	12	7.7	2	1.9	6	40	40	1062	41	20.1
Riva	11	69	20	6.3	0	1.5	9	62	52	767	39	31.9
Sale	10	53	37	7.5	0	2.1	19	47	49	954	52	38.4
Jokioinen	20	19	61	5.1	0	5.1	29	38	108	1530	40	13.1
Turku	11	41	48	5.3	0	3.4	21	48	71	1868	69	28.6
Gross (organic)	30	48	22	7.8	21	2.4	14	14	58	952	24	19.3
Gross (mineral)	36	44	20	7.8	22	1.6	11	12	46	1098	20	13.4
Rottenhaus 5	4	57	39	6.4	0	2.8	21	56	51	1075	13	6.5
Rottenhaus 6	7	60	33	6.5	0	2.7	23	55	49	909	8	3.0
Ritzlhof 4	17	60	23	7.3	1	2.2	16	62	53	1314	57	30.7
Ritzlhof 5	16	61	23	7.3	1	2.9	17	59	49	1613	95	64.7
Keszthely	38	40	22	6.9	1	1.3	10	16	36	404	6	1.1
Szentgyorgyvolgy	8	71	21	5.9	0	1.4	11	65	47	692	11	2.7
Nagyhorvati	30	50	20	6.8	1	1.7	9	37	37	682	12	5.6
Somogybabod	27	56	17	7.7	15	1.5	9	15	37	605	3	1.0

10 min^{-1} (Table 3). Ka:clay ratios fell within the comparatively narrow range 0.8–4.5. Both the water dispersion tests (with and without wet sieving) produced a similar order of magnitude variation in sediment export 0.3–3 g, while NaCl dispersed significantly more sediment (range 2–10 g, Table 3). Apart from an expected correlation between WSA and water dispersion ($r = 0.75$, $P < 0.001$), these laboratory tests did not correlate well with each other, neither was the variation in the test results satisfactorily explained by soil variables (Table 4). While WSA, SD and water dispersion tests correlated well ($P < 0.01$) with pH, OM and Al oxides, the Ka:clay test and NaCl dispersion values were related more to clay content and there was no obvious pattern in the values of PS. Correlations between soil test results and OM or clay were similar to those for CEC but coefficients (r) never exceeded 0.7 (Table 4). A multiple regression analysis (data not shown) also indicated that no more than 55% of the variation in sediment dispersed by water action (WSA or water dispersion tests) could be explained; the two most important soil variables were organic matter (37–42%) and pH (12–13%). $CaCO_3$ often substituted for pH in this multiple regression analysis. Whilst the relationship between sediment dispersed by water and OM was negative (positive for WSA), that with pH was positive (negative for WSA).

The dispersion tests provided suspensions for different sedimentation times (corresponding to <63, <20, <6 and <2 μm) that could be analysed for P. Amounts of TP, TDP and MRP dispersed by water ranged from 0.3 to 3.6, 0.02–0.5 and 0.01–0.4 mg, respectively. Corresponding amounts of TP dispersed by NaCl ranged from 1 to 15 mg (TDP and MRP were not measured). While the amounts of dispersed TP were very strongly correlated to the amounts of sediment dispersed, the amounts of dissolved P dispersed were largely independent. Data for MRP and SS for the different particle size classes of water-dispersed suspensions from soils of low, medium and high soil P fertility are shown in Figure 2. Values decrease slightly between <63 and <6 μm and then increase slightly for the <2-μm fraction, but remain essentially constant in relation to the much larger changes in SS. For water dispersion, the amounts of MRP were approximately 85% of TDP amounts ($r^2 = 0.99$), with a constant term of 0.01 mg. The variation in dissolved P dispersed by either water, or NaCl, were largely explained by soil-labile P (OP, $r^2 = 0.8$, $P < 0.001$) or desorbable P (WP, $r^2 = 0.9$, $P < 0.001$). When this independent dissolved P fraction was excluded, the relationship between the remaining PP fraction and the SS provided an estimate of the sediment P concentration. Sediment P concentrations for the <20-μm fraction ranged from 627 to 1754 mg kg^{-1} (mean 1048 mg kg^{-1}, SE 66.2, median 1025).

Table 3 Results of the laboratory tests

Site	Water stable aggregates (>125 μm) (% of soil)	Single drop test (no. of drops)	Percolation stability (cm^3 10 min^{-1})	KA/clay (ratio)	Wet sieving + water dispersion (<20 μm) (g)	Gentle water dispersion (<20 μm) (g)	NaCl dispersion (<20 μm) (g)
Bridgets	76 (0.6)	750 (92)	230 (23)	1.84	1.02 (0.03)	1.11 (0.10)	4.27 (0.02)
Boxworth	84 (0.4)	548 (56)	164 (9)	1.23	0.57 (0.04)	0.74 (0.12)	7.34 (0.17)
Rosemaund	85 (1.0)	452 (59)	503 (42)	2.32	0.69 (0.02)	0.72 (0.07)	6.73 (0.05)
Pwllpeiran	82 (0.5)	1978 (93)	174 (8)	2.52	0.28 (0.04)	0.28 (0.12)	2.17 (0.05)
Gleadthorpe	82 (1.2)	122 (8)	239 (36)	2.30	0.78 (0.03)	0.88 (0.07)	1.97 (0.02)
Olcenengo	59 (1.1)	461 (71)	398 (35)	2.89	1.34 (0.03)	1.17 (0.12)	4.20 (0.08)
Villafranca	41 (1.9)	24 (2)	39 (<1)	3.54	2.30 (0.22)	2.07 (0.05)	4.38 (0.07)
Mantova	63 (1.3)	1164 (92)	32 (2)	1.82	1.96 (0.19)	1.86 (0.02)	8.95 (0.06)
Bonavia	64 (0.5)	138 (21)	158 (24)	2.63	1.65 (0.09)	1.62 (0.14)	6.43 (0.03)
Tetto Frati	28 (2.7)	22 (2)	206 (10)	4.50	3.37 (0.14)	2.68 (0.06)	6.63 (0.03)
Riva	55 (1.9)	38 (5)	53 (5)	1.51	1.78 (0.06)	1.64 (0.10)	4.70 (0.04)
Sale	33 (1.2)	273 (40)	38 (5)	2.15	1.96 (0.09)	1.35 (0.10)	8.70 (0.17)
Jokioinen	77 (0.9)	316 (87)	115 (2)	0.82	0.76 (0.03)	0.64 (0.04)	7.95 (0.04)
Turku	86 (1.3)	697 (122)	29 (4)	0.85	0.62 (0.08)	0.88 (0.01)	9.83 (0.07)
Gross (organic)	48 (1.6)	80 (12)	34 (2)	1.77	1.81 (0.29)	1.70 (0.18)	7.00 (0.07)
Gross (mineral)	41 (1.5)	47 (8)	20 (4)	1.80	1.66 (0.03)	1.94 (0.16)	6.73 (0.07)
Rottenhaus 5	79 (1.0)	376 (39)	150 (7)	1.31	0.64 (0.03)	0.95 (0.06)	9.70 (0.03)
Rottenhaus 6	74 (0.5)	600 (84)	142 (45)	1.64	0.79 (0.04)	0.79 (0.09)	9.23 (0.01)
Ritzlhof 4	70 (1.5)	238 (40)	190 (9)	2.00	0.94 (0.04)	0.81 (0.02)	5.80 (0.07)
Ritzlhof 5	70 (0.7)	225 (17)	207 (16)	2.00	0.99 (0.04)	0.88 (0.02)	6.27 (0.00)
Keszthely	56 (0.3)	138 (18)	70 (1)	1.45	1.18 (0.03)	0.99 (0.10)	3.90 (0.03)
Szentgyorgyvolgy	56 (1.1)	51 (7)	123 (7)	2.10	2.00 (0.09)	1.94 (0.27)	5.60 (0.07)
Nagyhorvati	51 (1.6)	144 (19)	114 (12)	2.15	1.53 (0.08)	1.56 (0.06)	4.90 (0.17)
Somogybabod	30 (4.5)	203 (28)	147 (4)	0.93	1.86 (0.23)	1.21 (0.19)	5.03 (0.03)

Standard errors of the means are given in brackets.

Table 4 Regression coefficients (r) for the linear relationships between the laboratory test data and selected soil variables

Soil variable	WSA	SD	PS	Ka:clay	Wet sieving + water dispersion (<20 μm)	Gentle water dispersion (<20 μm)	NaCl dispersion (<20 μm)
Sand	ns	ns	ns	ns	ns	ns	−0.67***
Silt	ns	ns	ns	ns	ns	ns	ns
Clay	0.42*	ns	ns	−0.60***	−0.44*	−0.41*	0.67***
pH	−0.57**	ns	−0.41*	ns	0.50**	0.47**	ns
$CaCO_3$	ns	ns	ns	ns	ns	ns	ns
OM	0.65***	0.65***	ns	ns	−0.58**	−0.58**	ns
CEC	0.59***	0.61***	ns	−0.45*	−0.57**	−0.57**	0.46*
Fe_{ox}	0.37*	0.56**	ns	ns	ns	ns	ns
Al_{ox}	0.46*	0.69***	ns	ns	−0.40*	−0.39*	ns

WSA, water stable aggregates; SD, single drop test; PS, Percolation stability test. ns, not significant. *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

Indoor rainfall simulation

Run-off volumes generated by indoor rainfall simulation varied from 7 to 27 mm, representing 25–88% of the rainfall applied (Table 5). Most soils generated between 15 and 25 mm of run-off, but with notably smaller volumes recorded for Pwllpeiran, Jokioinen and Rottenhaus 5 soils, which tended to have higher clay and OM contents. There was an order of magnitude variation in the cumulative amounts of SS collected over the two 30-min storm events ranging from 0.2 to 6.6 g, except for the Tetto Frati soil which lost (16–17 g) and was to be excluded from the regression analysis due to its disproportionate influence. The smallest loss of SS was from the Pwllpeiran soil for both events. Amounts of

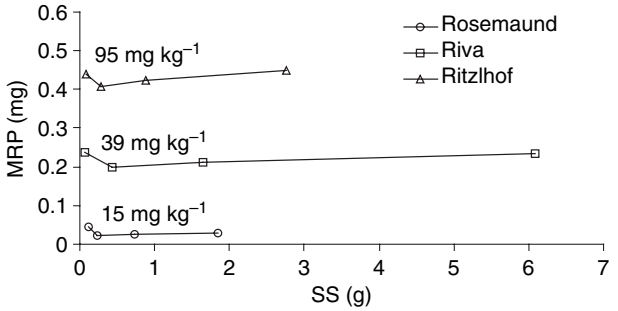

Figure 2 Changes in the amounts of molybdate-reactive P (MRP) as a function of the increasing amounts of suspended sediment (SS) collected for <2-, <6-, <20- and <63-μm sedimentation periods in the laboratory water dispersion test on three soils of contrasting soil P fertility (Olsen P).

TP collected ranged from 0.1 to 20 mg, with the largest loss obtained from the Tetto Frati soil, and the smallest loss from the Pwllpeiran soil. Excluding the Tetto Frati soil, mean amounts of SS, TP and TDP collected in run-off from the first simulated rainfall event were very slightly greater ($P < 0.01$) than those obtained in the second event (Table 5). The majority of the P transported was in particulate (>0.45 μm) form, and amounts of SS in the run-off accounted for about 75% of the variation in the amounts of PP collected from the soils for both events (Figure 3). Sediment P concentrations ranged from 716 to 2805 mg kg^{-1} (mean 1589 mg kg^{-1}, SE 67.0, median 1528). The rainfall simulation experiment therefore provided a sufficiently wide variation in soil particle entrainment and P mobilization in run-off to enable comparison with the data from the aggregate stability and dispersion tests undertaken in the laboratory.

Development of an environmental soil test to estimate sediment and P mobilization

The aggregate stability tests are only potential indicators of erodibility because they do not estimate the mass of sediment detached, neither do they provide material for determination of associated P content. However, the Ka/clay ratio and

Table 5 Amounts of run-off and entrained suspended sediment (SS), total P (TP) and total dissolved P (TDP) in the run-off generated from the benchmark soils by two consecutive simulated rainfall events each lasting 30 min

	Event 1				Event 2			
Site	Run-off (mm)	SS (g)	TP (mg)	TDP (% of TP)	Run-off (mm)	SS (g)	TP (mg)	TDP (% of TP)
Bridgets	19.9 (2.5)	2.4 (0.5)	3.8 (0.8)	4	22.6 (1.6)	1.6 (0.1)	3.3 (0.4)	5
Boxworth	13.9 (1.3)	1.7 (0.6)	2.4 (0.6)	6	21.9 (0.4)	2.1 (0.2)	3.5 (0.3)	4
Rosemaund	20.6 (4.0)	2.7 (1.0)	3.1 (1.2)	11	21.5 (3.4)	2.1 (0.7)	2.5 (0.9)	9
Pwllpeiran	6.7 (3.5)	0.3 (0.2)	0.4 (0.2)	14	10.6 (4.7)	0.3 (0.1)	0.5 (0.2)	14
Gleadthorpe	25.9 (1.0)	2.1 (0.1)	6.1 (0.4)	6	25.8 (0.9)	2.0 (0.7)	5.8 (0.8)	8
Olcenengo	22.7 (1.8)	2.5 (0.4)	3.3 (0.6)	2	23.5 (0.8)	2.1 (0.3)	3.5 (0.5)	2
Villafranca	25.3 (0.4)	6.6 (0.6)	15.7 (1.2)	7	22.9 (1.9)	5.5 (0.7)	11.9 (1.1)	7
Mantova	25.9 (1.3)	5.0 (1.1)	7.1 (1.2)	7	24.7 (0.5)	3.1 (0.3)	5.2 (0.3)	10
Bonavia	24.2 (1.5)	5.2 (0.4)	5.2 (0.5)	8	22.5 (2.2)	3.1 (0.4)	3.8 (0.5)	9
Tetto Frati	25.7 (0.2)	16.9 (3.5)	20.0 (3.2)	3	25.2 (1,4)	16.0 (1.4)	16.7 (2.4)	7
Riva	22.5 (1.0)	4.6 (0.3)	9.4 (0.6)	7	24.5 (0.8)	4.2 (0.5)	6.7 (0.7)	8
Sale	13.3 (1.4)	3.0 (0.6)	4.9 (1.0)	8	19.4 (1.1)	3.6 (0.3)	5.9 (0.9)	8
Jokioinen	9.1 (0.4)	0.8 (0.2)	2.2 (0.2)	7	11.6 (3.3)	1.1 (0.4)	1.9 (1.4)	7
Turku	15.5 (2.0)	2.4 (0.5)	4.6 (0.3)	9	15.5 (1.8)	1.9 (0.5)	4.0 (0.7)	9
Gross (organic)	23.7 (0.8)	3.6 (0.2)	6.6 (1.1)	11	21.7 (0.7)	3.4 (0.5)	6.4 (1.1)	9
Gross (mineral)	21.6 (0.8)	4.9 (0.4)	9.5 (0.5)	5	23.3 (2.2)	6.2 (1.4)	10.8 (2.5)	4
Rottenhaus 5	7.2 (–)[1]	0.6 (–)[1]	1.3 (–)[1]	13	7.5 (–)[1]	0.4 (–)[1]	0.8 (–)[1]	16
Rottenhaus 6	14.7 (2.0)	2.1 (0.9)	3.8 (1.2)	4	12.6 (6.1)	0.8 (0.6)	1.3 (2.6)	9
Ritzlhof 4	14.3 (0.8)	1.7 (0.2)	3.1 (0.4)	25	16.3 (2.7)	1.6 (0.3)	3.4 (0.3)	16
Ritzlhof 5	13.5 (2.8)	1.5 (0.3)	5.2 (1.2)	22	15.0 (3.5)	1.1 (0.3)	3.5 (1.1)	34
Keszthely	22.2 (1.2)	3.0 (0.4)	3.8 (0.5)	4	23.0 (1.2)	2.1 (0.3)	2.7 (0.3)	5
Szentgyorgyvolgy	24.6 (1.0)	3.4 (0.3)	5.5 (0.5)	4	24.6 (1.7)	2.5 (0.1)	3.7 (0.1)	4
Nagyhorvati	26.9 (0.3)	3.4 (0.4)	5.3 (0.5)	7	26.3 (0.2)	2.6 (0.4)	4.5 (0.4)	9
Somogybabod	24.6 (0.8)	6.1 (1.7)	5.1 (0.7)	8	25.6 (0.8)	6.4 (1.9)	4.3 (0.7)	3
Mean	19.4 (1.3)	3.6 (0.7)	5.7 (0.9)	8	20.3 (1.1)	3.2 (0.7)	4.9 (0.8)	9
Mean[a]	19.1 (1.3)	3.0 (0.4)	5.0 (0.7)	9	20.1 (1.1)	2.6 (0.4)	4.3 (0.6)	9

Standard errors of the means are given in brackets. [a]Mean excluding the Tetto Frati soil.

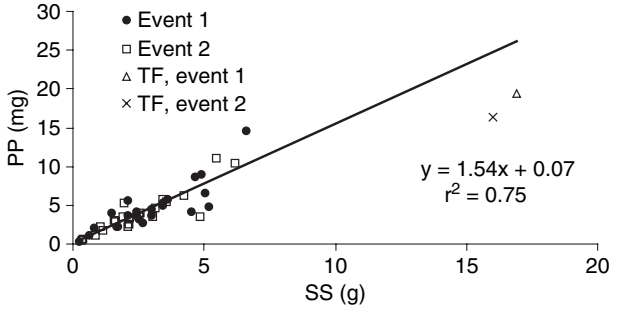

Figure 3 Relationship between suspended sediment (SS) and particulate phosphorus (PP) collected from the benchmark soils under indoor rainfall simulation. Data for the Tetto Frati (TF) soil are not included in the regression analysis.

percentage WSA >125 μm both accounted for a significant ($P < 0.001$, $r^2 = 0.4$) amount of the variation in SS loss in run-off between the soils. The laboratory dispersion tests produced actual amounts of SS and P fractions that could be directly compared with the corresponding amounts obtained in run-off under simulated rainfall. Amounts of SS and TP obtained by the NaCl dispersion test did not show significant ($P > 0.05$) correlations with the amounts of SS and TP in run-off. By contrast, both the water dispersion tests (with and without prior wet sieving) explained between approximately 40 and 75% of the variation in SS and TP in run-off depending on the particle size range selected for measurement during sedimentation in the laboratory. The particle size range that gave the best correlations was the <20-μm fraction and the common regression relationships obtained for SS, TP and TDP over the two events for this fraction are shown in Figure 4. The additional wet sieving prior to dispersion did not significantly improve correlation coefficients and is less practical for routine use. There was no statistically significant ($P > 0.05$) difference in the parameters of the regression equations for the two run-off events, although gradients for event two were always slightly less than those for event one. The amounts of dissolved $P < 0.45$ μm (TDP and MRP) dispersed by both water and NaCl in the laboratory were very highly correlated ($P < 0.001$, $r > 0.9$) with the same fractions measured in run-off. The water dispersion test without prior wet sieving was therefore selected as the most pragmatic and reliable laboratory method to estimate the intrinsic risk of sediment, PP and dissolved P mobilization from different soil types.

Phosphorus enrichment ratio

The P enrichment ratios (PER) of the dispersed sediment both in the run-off and in the laboratory test were calculated from the ratio of the sediment P concentration (estimated as the ratio of PP:SS in mg kg^{-1}) to the original soil total P concentration. Enrichment ratio values for the run-off varied from 0.9 to 3.1 and those obtained from the test ranged from

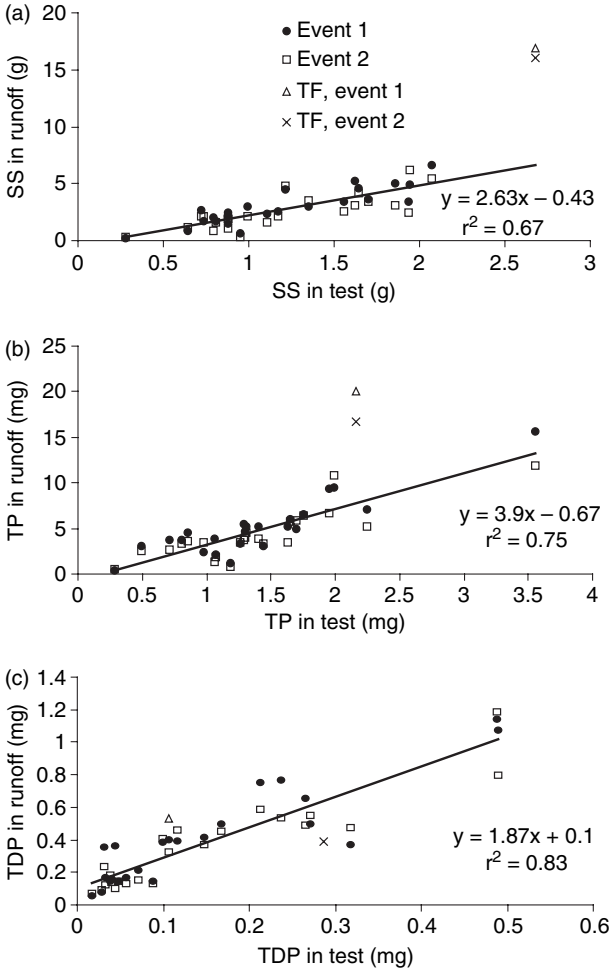

Figure 4 The relationship between suspended solids (SS, g), total P (TP, mg) and total dissolved (<0.45 μm) P (TDP, mg) collected in surface run-off under simulated rainfall and by gentle water dispersion in the laboratory. Data for the Tetto Frati (TF) GL are not included in the regression analysis.

0.5 to 2.5. The range in values for the run-off and test data were highly correlated ($r > 0.75$), but run-off enrichment ratio values were mostly higher than those obtained by the laboratory test due to differences in sediment P concentrations (Figure 5). This difference tended to diminish as the clay content of the soil increased. PER values showed a weak negative correlation only with soil total P content (test $r = 0.67$; run-off $r = 0.74$).

Trends in sediment and P mobilization

The results of the gentle water dispersion test were used to rank the soils in terms of their potential vulnerability to both soil particle and total P mobilization (Figure 6a,b). Comparison between the two figures reveals that the soil with the highest dispersibility (IT5) did not lose the most TP, and some of the less dispersive soils lost more P (AU3b, UK5

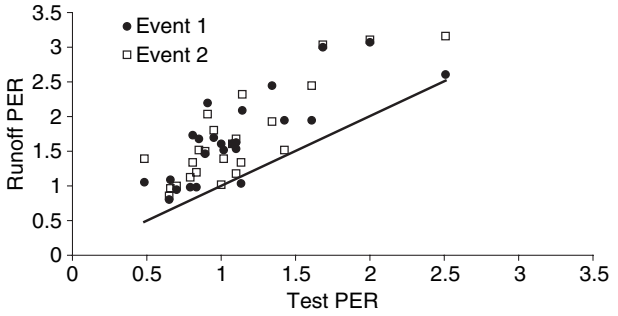

Figure 5 Phosphorus enrichment ratios (PER) obtained in simulated run-off from events 1 and 2 compared with those obtained by the gentle water dispersion test. The 1:1 line is shown.

and IT7) than some of the more dispersive soils (HU2 and IT4). Soil IT2 lost the most TP and was readily dispersed making it a high-risk soil. Differences between vulnerability to sediment loss and vulnerability to TP loss in run-off are due to differences between the dissolved P fractions (Figure 6c) and to variable P enrichment of the particulate fractions (Figure 6d). Some soils which were not dispersive showed very high PER values (UK2, UK5, HU1 and IT1). Similarly, some soils which were not very dispersive lost large amounts of dissolved P (AU3a, AU3b and IT7). These results show that an intrinsic vulnerability to sediment mobilization in sheet run-off does not necessarily present the greatest environmental risk in terms of P loss.

Discussion

The data obtained from the rainfall simulation experiment were assumed to represent the sediment and P mobilization that would occur in sheet run-off from bare soil under the storm intensity and slope conditions selected for study. The erosional processes operating under these experimental conditions include rain splash, ionic dispersion and selective entrainment of the soil particles that are sufficiently fine to reach the river, either in shallow unconcentrated surface flow or via pipe drainage systems in fields (Walling *et al.*, 2000). Transfer of P by sheet run-off in surface or sub-surface hydrological pathways is arguably a more endemic environmental problem than the more dramatic rill and gully erosion which generates coarse-textured sediment lower in P and which may occur on only a few fields in some years (Harrod and Theurer, 2002; Haygarth *et al.*, 2006). The large amount of work required to produce these data prohibited rainfall simulation testing at other rainfall intensities, different slope angles, soil moisture contents or degrees of crop cover. Altering these factors would have influenced the amounts of sediment and P produced; for example rain splash and sheet run-off are known to be sensitive to rainfall intensity (Bryan, 2000), and differences in the particle size signature of eroding sediment will influence the amount of PP eroding due to preferential enrichment of finer particles (Sharpley, 1985a,b; Quinton *et al.*, 2001). Nevertheless, the standardized experimental conditions facilitated a robust comparison of the relative intrinsic risk of sediment and P mobilization across a range of soils with different physical and chemical properties.

Because soil erodibility is a function of the properties of the soil, a number of physically based soil tests related to soil aggregate stability, dispersibility and shear strength have been developed to help characterize vulnerability to soil erosion under different site conditions (Bryan, 1968). A selection

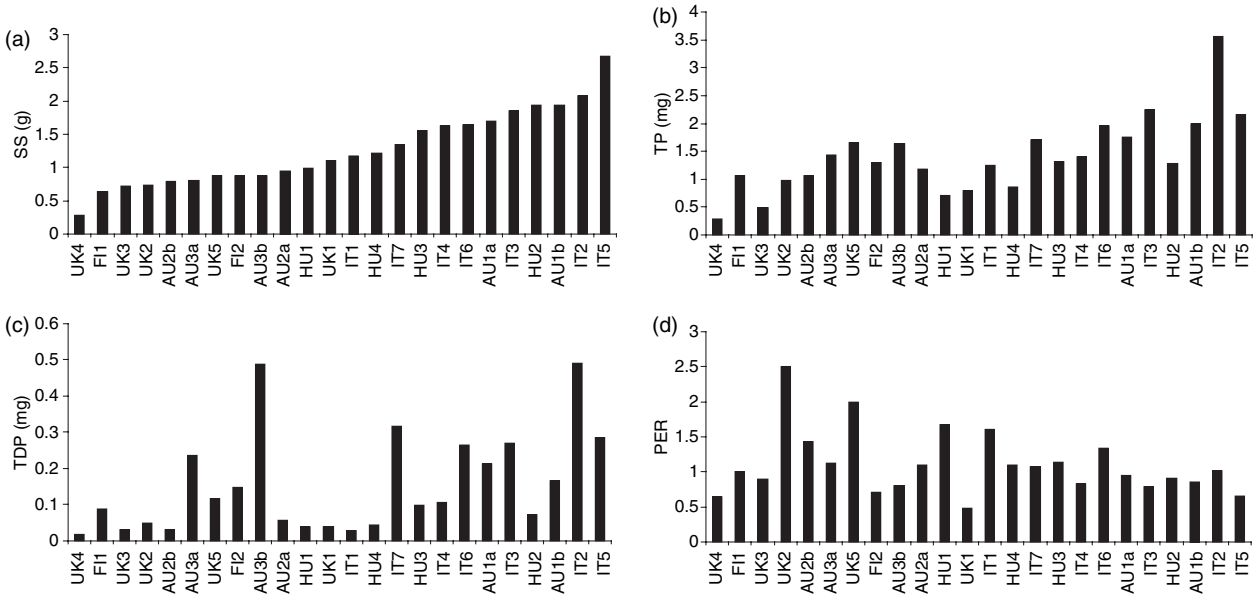

Figure 6 Ranking of the benchmark soils according to susceptibility to dispersion of suspended sediment (SS) as determined by laboratory test and the resulting ranked variation in total P (TP), total dissolved P (TDP) and P enrichment ratio (PER) of the sediment.

of these soil tests were used in this study, but they did not relate well with each other, with the run-off data or with soil variables, and therefore appear to measure different aspects of erodibility. The WSA and SD tests are related more to aggregate strength, PS to susceptibility to slaking, and soil consistency tests – similar to the Ka:clay ratio – to resistance to particle entrainment on cohesive soils (Mbagwu & Auerswald, 1999; Bryan, 2000). The gentle water dispersion test gave both good prediction of particle mobilization under simulated rainfall and provided a suspension for simultaneous determination of the dissolved and PP fractions. The slow end-over-end rotation of the cylinder generates a kinetic energy which disrupts the soil aggregates, and the use of a low soil:solution ratio encourages particle abrasion whilst reducing experimental errors introduced by greater dilutions (So *et al*., 1997). Therefore, although the test does not simulate rainfall, it provides a useful analogue. Various workers have adopted different shaking times of up to 30 min for heavy clay soils, but we found there was little difference in soil dispersion beyond 5 min, and the use of 1 min provided greater sensitivity (Quirk, 1950; Ministry of Agriculture, Fisheries and Food, 1982). The method proved relatively simple to use routinely with good reproducibility between replicates. Dong *et al*. (1983) concluded that soil dispersibilty in water could potentially be used to rank soils according to risk of soil loss.

Good correlation between the results of the indoor rainfall simulation and the gentle water dispersion test was obtained for both <63- and <20-μm particle sizes recovered, although the <20-μm size fraction accounted for most of the variation. These results are fully consistent with other data showing that it is predominantly the silt-sized aggregates, containing aggregates of silt and clay primary particles, that are preferentially mobilized by rain splash detachment and shallow flow in sheet run-off, and which are important for pollutant transfer (Alberts *et al*., 1980; Loch & Donnollan, 1983; Foster *et al*., 1985; Wan & El Swaify, 1988). These authors found that only small proportions ($<5\%$ of detached soil) were transported as primary clay particles. Variation in sediment mobilization was related (albeit poorly) to soil properties which influence the degree of aggregation and resistance to aggregate breakdown (OM, clay and sesquioxides) and to properties which influence soil dispersibility (pH). Organic matter was the soil property with the highest correlation coefficient, as was found by Greenland *et al*. (1975). Links between erodibility and aggregate stability would be expected and have been widely observed in previous work (e.g. Coughlan & Loch, 1984; Mbagwu & Auerswald, 1999), especially on soils which do not have a coherent structural bonding framework, e.g. soils that are frequently disturbed by cultivation (Bryan, 2000). Links to soil pH have also been widely reported and it seems likely that as pH rises, especially above 7, edge to face clay bonding and clay bonding to Fe/Al variable charge surfaces decreases, resulting in increased vulnerability to clay dispersion (Suarez *et al*., 1984; Barberis & Withers, 2002). Free calcium carbonate, which is more likely above pH 7, increases the concentration of Ca^{2+} ions and would be expected to inhibit dispersion (Greenland *et al*., 1975). However, in our study, the calcareous soils did not show reduced dispersibility.

Variation in the PP fraction reflected differences in the amounts of soil mobilized and their degree of P enrichment. The PER values varied by a factor of five and were significantly related to total P content but not to the amount of SS mobilized. This contrasts with Sharpley (1985b) who suggested that the PER of particles in run-off could be predicted by the amount of SS transported. This most probably relates to the lack of detachment of coarser soil particles in our study which would tend to have a lower P content. The greater P concentrations in the SS entrained under indoor rainfall simulation than measured in the test, suggests that the median particle size of SS in the run-off was probably less than that in the 20-μm fraction of the laboratory test. The lack of a difference in PER on the heavier textured soils is also consistent with the observations by Foster *et al*. (1985) who found clay enrichment ratios closer to one on clayey soils due to a more even distribution of clay within larger sized aggregates.

During development of the test, the dissolved (<0.45 μm) P fraction proved to be largely independent of the amounts, or particle size, of SS produced, for example by adopting different shaking times, or by removing different sized fractions during sedimentation. This observation suggests this fraction is largely colloidal in nature because P sorption reactions during transport would effectively move P from P-rich to P-poor soil particles which might be expected to minimize the concentration of truly soluble P in solution. This fraction was clearly highly influenced by the history of P inputs for the different soils, as indicated by its dependence on water-extractable and Olsen P tests, as presented by Quinton *et al*. (2003).

The environmental soil test developed here predicts only the *relative* risk of initial mobilization of sediment and P in run-off, and requires integration with an assessment of the risk of mobilized particles reaching the watercourse. Risk assessment procedures that incorporate mobilization and delivery aspects of P transport have been developed in a number of countries (Heathwaite *et al*., 2003; Sharpley *et al*., 2003). These P index procedures assess risk of soil loss and of dissolved P loss separately based on soil erodibility indices (e.g. soil texture, k value) and routine soil tests, but do not quantify the extent of P enrichment of the soil particles eroded (the PP fraction). The environmental soil test allows assessment of soil, PP and dissolved P simultaneously and therefore has potential to more accurately assess field sites vulnerable to SS and P mobilization. This should allow better ranking of fields for targeting and implementation of

mitigation options either as part of P indexing procedures or through catchment modelling. Process-based erosion models that incorporate rain splash detachment and inter-rill erosion (EUROSEM and WEPP) may also use these data to develop and incorporate associated P transport routines.

Application of the test to fields with variable land-use and P application histories can also provide information on the contribution of agriculture to SS and P loss risk, for example the effects of cultivation systems (e.g. no till) on soil erodibility and the impact of fertilizer and manure P on PP enrichment relative to the intrinsic risk of P loss before agricultural intensification. Different geological regions are known to have different 'background' signals representative of pristine conditions, and it is important to understand the extent to which agriculture has modified background P losses so that appropriate P loss reduction targets can be identified and effective measures selected. Hence, potentially the test can be used to quantify this deviation from more pristine land uses, provided the latter still exist in the landscape, perhaps as ancient forest. The environmental soil test also has potential as a reference point to help quantify the magnitude of sediment and P retention in the landscape, or through the soil. Comparison of the amounts of SS and P initially mobilized with the amounts measured in flow at appropriate monitoring points; for example in drain flow, at the edge of fields or at catchment outlets, provides a means of assessing the delivery factor in P transport (Beven *et al.*, 2005). The very short length of slope (50 cm), used in the indoor rainfall simulation to calibrate the test, ensured that any 'delivery' component of sediment and P transport was kept to a minimum.

Conclusions

The adverse ecological impacts of sediment and P from agricultural sources are of increasing concern in a number of countries. Routine methodologies with some predictive value for assessing the degree to which field run-off becomes entrained with eroding particles, and enriched in P are potentially useful. A laboratory test, based on gentle water dispersion of the <20-μm particle fraction in soils, was found to provide a reliable estimate of the intrinsic risk of sediment and P mobilization in unconcentrated surface flow from a wide range of bare European soils, under simulated rainfall. The most erodible soils were not always the soils that generated the most P, due to differences in the dissolved P (<0.45 μm) component and in the P enrichment of the silt and clay soil particles selectively transported. The test has a number of potential applications in field risk assessment, catchment appraisal and modelling of sediment-associated P transfer in both surface and sub-surface flow. Field evaluation of the value of the test in assessing vulnerability to sediment and P mobilization under various soil and crop conditions is recommended. Application of the methodology will clearly need to take account of the spatial variability in soil texture, land-use history and P inputs within and between fields on farms.

Acknowledgements

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