Effects of Farming Practices on the Transfer of Phosphorus in Rural Catchments

Thesis

UNRESTRICTED

Effects of farming practices on the transfer of phosphorus in rural catchments

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I certify that all material in this thesis which is not my own work has been identified and that no material is included for which a degree has previously been conferred upon me.

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ABSTRACT

To better understand the role of agriculture in the eutrophication of surface waters, the spatial and temporal variation in P concentrations in streams and storm runoff was examined in relation to historic and current farming practices (upland pasture improvement, soil P accumulation, soil cultivations and fertiliser and manure inputs) and other sources of P (farmyards, road runoff and wastewater discharge) in a number of small rural catchments in England.

In an upland pasture catchment and a lowland mixed farming catchment, soil P was the major source of P export rather than annual field operations which generally did not coincide with the main periods of hydrological connectivity. The majority of P transfer was in particulate form (>0.45 μm), although field plot investigations suggested this form of P was relatively insensitive to differences in soil P fertility as compared to the dissolved P fraction, and may release P to the watercolumn even at low soil P concentrations. Three mitigation options (early sowing, reduced cultivations and tramline removal) for reducing soil particulate P transfer were compared at the field scale. All three options had positive effects. Early crop establishment was the most effective, but not consistently so, suggesting a combination of measures is required on farms.

Analysis of 118 storm runoff samples showed that farmyard and road runoff, and runoff receiving wastewater (septic tank) discharges were more concentrated in P than surface and sub-surface runoff from agricultural fields. These sources also arrive more continuously through the year and therefore are potentially more ecologically damaging than runoff from farmed land. It is concluded that urbanisation maybe a greater threat to eutrophication than agricultural intensification even in small rural catchments and that effective, integrated catchment management requires collective social responsibility to tackle the multiple sources of P entering water and not just the targeting of the farming community.

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FARMING, PHOSPHORUS AND THE ENVIRONMENT

1.1 Introduction

The enrichment of surface waters with anthropogenic sources of phosphorus (P) causing eutrophication and the resulting impairment of aquatic habitats and water use for recreation, industry and drinking has become a major environmental issue in recent decades (Moss, 1996; de Jonge et al., 2002). Anthropogenic sources of P include the 'point-source' discharges of municipal and industrial wastewater associated with an expanding population and those of more diffuse origin arising from agricultural intensification and urban development in the wider catchment area (Novotny and Olem, 1994; Sharpley et al., 2001). The need to reduce these anthropogenic P inputs to acceptable levels by voluntary and legislative means is widely recognised; for example in Europe with the recent introduction of the Water Framework Directive (WFD).

The WFD adopts an holistic, catchment-based approach to water management encompassing all types of waterbody and addresses the sources of pollutants and not just the symptoms (Pollard and Huxham, 1998; European Commission, 2000). The concentrations of P in the water column required to inhibit eutrophication are so much lower than the ambient concentrations in many waterbodies that a reduction in major point sources alone may not always be effective (e.g. Foy et al., 2003). Additional assessment and controls over diffuse P inputs in catchments will also be required to achieve the target concentrations considered necessary to restore good ecological status. For example, Carpenter (2005) suggests that drastic reductions in soil P contents, or soil erosion rates, are required to halt the slow flux of P in runoff from over-fertilised soils in intensive agricultural regions which he considers is sustaining lake eutrophication.

Since 1992, the Department for Environment, Food and Rural Affairs (Defra) has funded a programme of research in England and Wales on 'Phosphorus Loss (or Emissions) from

Agriculture'. The programme aims 'to better understand the contribution of agricultural practices to P loss, and to develop proportionate and cost-effective measures and guidelines to control the loss of P to inland and coastal waters to meet national and international requirements'. Much of this research has focused on the processes of P transfer in runoff from agricultural land and the controlling factors operating at the field scale. Whilst the research output from this programme has been regularly reviewed (Department for Environment, Food and Rural Affairs, 1995; 1999; 2004), there has been limited evaluation and integration of data at the catchment scale, or in terms of ecological relevance.

Understanding the relative importance of different farming systems and practices influencing P exports and ambient water P concentrations in catchments is critical to the development and targeting of cost-effective measures to help control diffuse P inputs. Since controls over agricultural practices may have large social and economic consequences for rural communities, it is also important to assess the ecological relevance of agricultural P sources relative to other P sources in rural catchments (Neal and Jarvie, 2005). The lack of any provision for interpreting P supply within an ecological context might result in poor relationships between reductions in P loading and environmental benefit (Edwards and Withers, 2007). Monitoring of small rural headwater catchments with no major point source inputs provides an opportunity to research the spatial and temporal variability in stream P concentrations and P export in relation to farming systems and practices, and relevance to ecological impacts.

The work presented in this thesis makes a contribution to Defra's policy objective through a selected evaluation of their research data collected at a range of spatial and temporal scales in English rural catchments by ADAS UK Limited. This data evaluation was undertaken to:

1. Understand better the impact of farming practices on P transfer in runoff from agricultural land within an upland and lowland catchment.
2. Assess the relative significance of a range of P sources in rural catchments and their potential contribution to river eutrophication.
3. Compare selected best management practices to reduce runoff, soil erosion and P transfer in a priority catchment suffering from siltation and eutrophication.

1.2 Eutrophication of surface waters

A supply of clean potable water and access to healthy, ecologically diverse waterbodies is a primary requirement for quality of life. In recent decades, a deterioration in the quality of water in lakes, reservoirs, aquifers, canals and rivers has become an increasing environmental problem for developed countries. Among the many causal factors has been the enrichment of water with reactive forms of nitrogen (N) and P resulting in the eutrophication of surface waters with threats to human health, habitat diversity, species survival and amenity value (Conley, 2000; Mainstone et al., 2000; Bennett et al., 2001). The UK Eutrophication Forum has defined eutrophication as 'the enrichment of water by nutrients (normally N and P) causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned' (UK Eutrophication Forum, 2005).

This undesirable disturbance manifests itself in a range of symptoms including increased algal growth, reduced water clarity, loss of submerged plants, production of algal toxins, deoxygenation, fish kills, and increased water treatment costs. Foy (2005) describes these symptoms as a succession of impacts that relate back to the increased primary production following nutrient enrichment (Table 1.1). Typically, plant and animal species become fewer and more vigorous, and algal growth proliferates. For example, in East Anglian reservoirs, Hayes and Greene (1984) report a typical succession of diatom growth in early spring, green algal blooms in early summer and blue-green algal blooms in late summer and early autumn. In faster flowing river systems, increased growth of filamentous algae (e.g. *Cladophera spp.*), and periphyton (e.g. diatoms and epiphytic algae), and reduced abundance of shallow rooting submerged plants (e.g. *Ranunculus* spp., *Challitrace spp.*) are of primary concern (Mainstone et al., 2000; Hilton et al., 2006).

Only small increases in ambient nutrient concentrations are required for these eutrophication symptoms to occur depending on the type of waterbody, and other site factors, such as flow regime, water depth, water clarity, temperature, shading and grazing pressures (Søballe and Kimmel, 1987; Foy and Withers, 1995).

Table 1.1 Linking potential impacts of eutrophication (From Foy, 2005).

Eutrophication causes an: • Increase in primary production either as algae or macrophytes *but in many cases the* • Reduction in water clarity *is such that there is a* • Replacement of submerged macrophytes by phytoplankton *which exhibits an* • Increased dominance by blue-green algae *which are liable to* • Form surface algal blooms and produce algal toxins. Eutrophication increases fish production *but increased primary production leads to* • An increasing respiratory demand for dissolved oxygen demand *reflected by* • Increased de-oxygenation in deep water, under ice or at night *causing an* • Increased potential for fish kills *and a* • Decline fish species that are intolerant of low oxygen levels *which is part of* • A reduced biodiversity observed at all trophic levels following enrichment Water treatment costs are increased *by* • Requirement to remove algae from water: • Taste and odor problems caused by algae and macrophytes *and the* • Production and release of algal toxins *which* • Threaten public and animal health • Release of manganese and iron from lake sediments *(reflecting low oxygen status)* Other economic costs include: • Damage to recreation potential and amenity • Reduced valuation of shoreline properties

Public awareness of eutrophication issues has increased worldwide as particular health problems, or even deaths, have occurred as a result of algal growth and associated release of toxins in eutrophic waters (Moss, 1996; Gratton et al., 1998). Surveys indicate that eutrophication is quite widespread with particularly severe problems occurring in localised areas (e.g. McGarrigle, 1993; Carvalho and Moss, 1995). Although the incidence and severity of eutrophication can be quite variable from year to year the costs of cleaning eutrophied waters for drinking, amenity and recreational use are considerable. Pretty et al. (2003) calculated that freshwater eutrophication costs the U.K. £75-114 million per annum.

Controls over the major sources of nutrient emissions to water are therefore required to provide sustainable solutions to the maintenance and enhancement of good quality water. Nutrient criteria for distinguishing ecologically impacted waters have been developed (Organisation for Economic Co-operation and Development, 1982; Dodds et al., 1998; Duncan et al., 2006). These are largely based on P due to the strong relationships observed between algal biomass and total, and or dissolved, P in the watercolumn, especially for lakes. These relationships suggest that reducing the external nutrient load remains the most pragmatic means of reducing the severity and frequency of eutrophication, although additional interventions such as biomanipulation may also be required for some badly impacted systems to fully recover (Ryding and Rast, 1989; Moss, 1990; Jeppesen et al., 2000),

1.3 Role of Phosphorus

Whilst carbon (C), N and P are all required for eutrophication to occur, there is a general consensus that the main focus for reducing eutrophication should remain with P, particularly in freshwaters (Schindler, 1977; Foy, 2005). Certain algal communities can compensate for C and N limitation through acquisition of these elements from the atmosphere. Phosphorus only becomes volatile in small quantities under very rare conditions (as phosphine gas) and therefore can more easily be controlled to limit algal growth. In the natural environment, P present in solution as reactive orthophosphate ($H_2PO_4^{2-}$ and HPO_4^{-}) is rapidly immobilised by chemical reactions and biological processes in soils, or utilised by living organisms for a range of essential functions. Most notable of these is the formation of adenosine tri-phosphate (ATP) which provides the main energy source for the synthesis of organic sugars, nucleic acids, proteins, phospholipids and phytin required for the growth and development of plants and animals (Mengel and Kirkby, 1987).

In most pristine environments, P availability is therefore naturally very low and dependent on the activity of phosphatase enzymes and plant root exudates to hydrolyse P bound in organic forms in soils, sediments and waters (Shand and Smith, 1997; Condron et al., 2005). In eutrophied environments, soils and sediments become more P-saturated and other factors (e.g. other nutrients, light) become limiting to growth resulting in an excess of P supply over demand and larger measured concentrations of P in the water phase. Due to the highly reactive nature of the orthophosphate ion, the majority of P present in water is part of, or attached to, inorganic and organic particles of varying sizes (Shand et al., 2000). This is in contrast to N, which is not readily adsorbed onto soil particles and is largely transported in solution as nitrate. For analytical convenience, a total dissolved (<0.45 μm) P (TDP) fraction is usually distinguished from a particulate P (PP) fraction (>0.45 μm) in research work, but in reality water contains a continuum of particle sizes varying in origin and P bioavailability (Fig. 1.1).

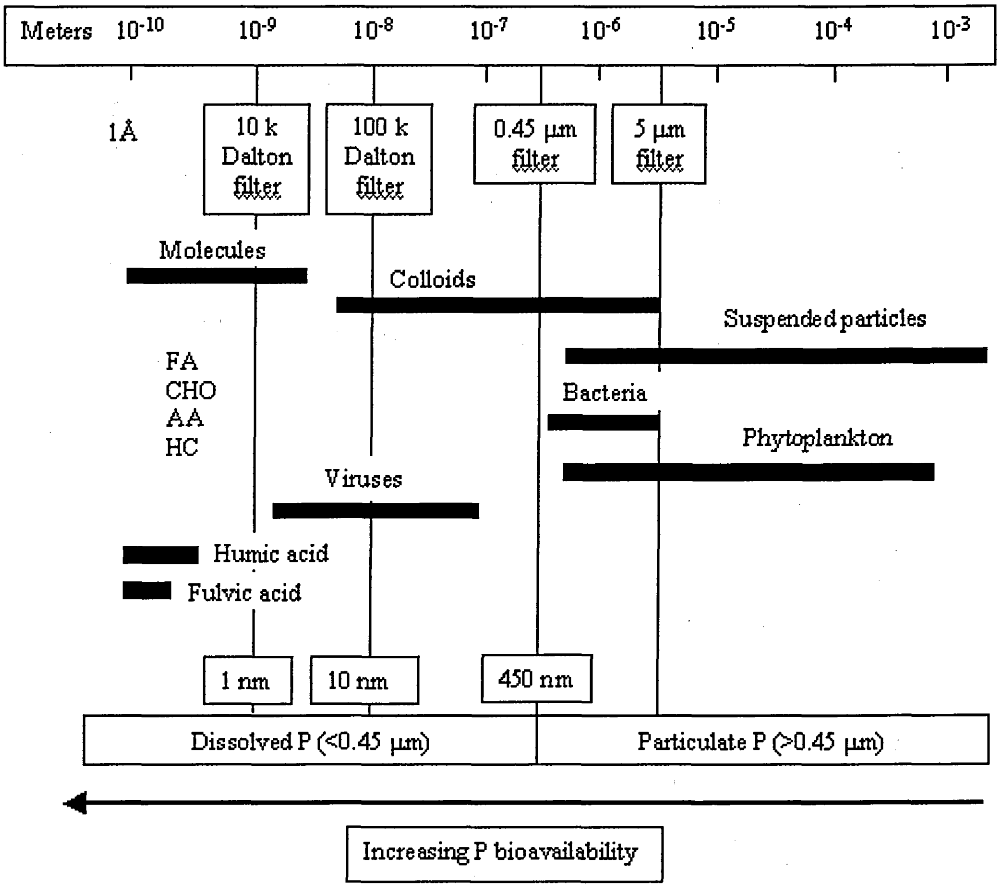


Figure 1.1 Particulate matter found in water in relation to the analytically-defined dissolved and particulate P fractions, and P bioavailability. (Adapted from Aiken and Leenheer, 1993).

The TDP fraction, which will include much colloidal P, is considered to be fully available for algal growth, whilst the PP fraction has been shown to be only partly available to algae depending on the source geochemistry (Hatch et al., 1999; Reynolds and Davies, 2001; Ekholm and Krogerus, 2003). For monitoring in the UK, orthophosphate-P is usually measured in the supernatant of an unfiltered sample (termed total reactive P, TRP) after allowing it to stand. This measurement attempts to combine the molybdate-reactive components of the dissolved and particulate fractions. Appropriate procedures for sampling of waters for P fractions are summarised by Jarvie et al. (2002).

Links between the amount and concentration of P in the water phase and eutrophication response are therefore related to the bioavailability of the source P present, the time of year it arrives in the water column and the residence time available for internal transformations and uptake by the aquatic community. Fundamental differences exist between standing and flowing waters in terms of there residence time with reduced opportunity for changes in bioavailability, internal cycling and uptake of P within the watercolumn. For flowing waters, there are large temporal and spatial separations between the source of P and the point of potential impact, leading Edwards et al. (2000b) to suggest that P concentration is ecologically more relevant than P load in rivers compared to lakes. With less opportunity for internal recycling of sediment-bound P in flowing waters, the role of PP in river eutrophication remains questionable. In some cases, ecological impacts may relate to suspended solids (SS) rather than the P attached to them (Heaney et al., 2001). Hence, recently defined P targets for good ecological status in different types of waterbody in the UK are based on total P (8-49 μg TP L^{-1}) for lakes and on the reactive TRP fraction (40-120 μg L^{-1}) for rivers (Duncan et al., 2006).

These targets are very challenging in relation to the widespread P enrichment found in lakes and rivers in the UK. In a survey of 902 lakes, Foy and Bailey-Watts (1998) found that TP concentrations in 70% of English lakes (subset of 128 lakes) exceeded 100 μg L^{-1}. Muscutt and Withers (1996) found that TRP concentrations in 79% of the 98 largest rivers in England and Wales exceeded 100 μg L^{-1} and 16% exceeded 1 mg L^{-1}. More recent data suggests up to 80% of the total river lengths in different regions in England monitored by the Environment Agency (EA) have average annual TRP concentrations of > 100 μg L^{-1} as shown in Fig. 1.2. Similar P contamination has been reported for European rivers (van Dijk et al., 1994), although the European Environment Agency (2005) report that river P (but not N) concentrations are now declining as municipal and industrial wastewater discharges of P are being better controlled under the Urban Waste Water Treatment Directive (UWWTD). Interestingly, EA regions with the highest proportion of P-enriched rivers also have the highest proportion of nitrate-contaminated rivers, and the proportion of rivers impaired by TRP is always greater than for nitrate (Fig. 1.2).

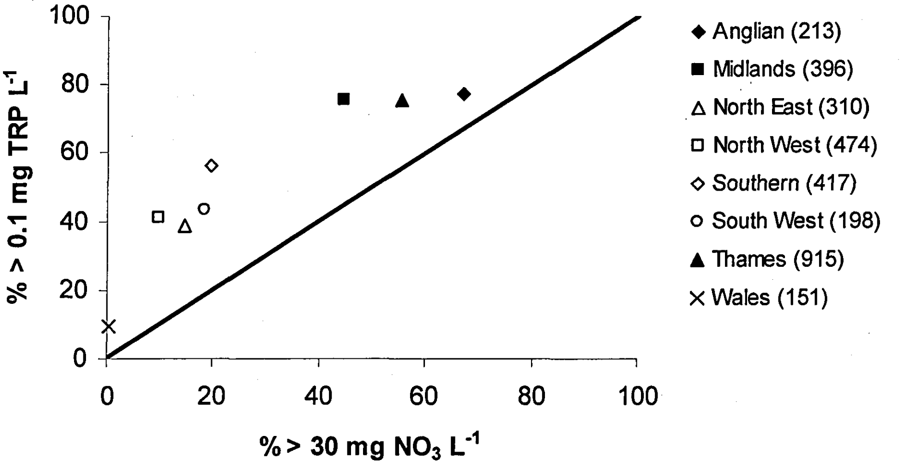


Figure 1.2 The percentages of total river length with concentrations of total reactive P (TRP) > 0.1 mg L^{-1} and of nitrate (NO_3) > 30 mg L^{-1} in each of the regions monitored by the Environment Agency (EA) in England and Wales in 2005. Figures in brackets are the population densities (numbers km^{-2}) within each region. A 1:1 line is also drawn. The TRP target of 0.1 mg L^{-1} is that used in the EA eutrophication strategy (Environment Agency, 2000) and the NO_3 target of 30 mg L^{-1} is representative of the 95 percentile limit of the 50 mg NO_3 L^{-1} standard set in the EU Nitrates Directive.

1.4 Sources of phosphorus

A variety of sources contribute P to freshwaters via a number of different pathways and at different times of the year. Sources differ with respect to their composition, mobilisation and transport and point and timing of delivery (Dorioz et al., 1998; Edwards and Withers, 2008; Jordan et al., 2007). Conventionally, a distinction is made between point sources which are highly concentrated in P, bioavailable and discharged continuously from discrete points and diffuse sources which are more particulate in nature, arrive only during storm events in surface and sub-surface flow and from a number of different areas within the catchment. Although diffuse sources are invariably linked with agriculture, Edwards and Withers (2008) identified a mixed group of rural P sources that had properties intermediate between those of point and diffuse sources (Table 1.2). These intermediate rural sources of P include road and track runoff, septic tank discharges and farmyard runoff and maybe more ecologically relevant than runoff from traditionally farmed land, even though they may not constitute a large proportion of the total annual load (Hively et al., 2005; Douglas et al., 2007). Little information exists on their relative importance in rural catchments.

Table 1.2 Various sources of phosphorus and their chemical and hydrological attributes (From Edwards and Withers, 2008).

Source type/source	Hydrological		Chemical
	Discharge	Rainfall dependency	Composition
Point			
STW[1]/industry	Continuous	Low	Concentrated
CSOs[2]	Episodic	High	Concentrated
Intermediate			
Septic tanks	Semi-continuous	Low	Variable
Field drains	Semi-continuous	Low-high	Variable
Road/track runoff	Episodic	High	Variable (high SS)[3]
Farmyards	Episodic to semi-continuous	Low-high	Variable
Diffuse			
Surface runoff	Episodic	High	Variable (high SS)[3]
Sub-surface runoff	Episodic	High	Dilute
Groundwater	Continuous	Low	Dilute

[1]STW, Sewage Treatment Works; [2]CSOs, Combined Sewer Overflows; [3]SS, Suspended Solids.

Unlike point sources, diffuse sources show a strong seasonal trend in P loadings which relates to the frequency of storm events as well as certain features of land management (e.g. cultivation operations, fertiliser and manure inputs and stock grazing pressures), (e.g. Munn and Prepas, 1986; Lennox et al., 1997; Monaghan et al., 2007). However, at the catchment scale, the relative contributions of individual farming operations are often difficult to quantify separately from those P inputs arising from intermediate and point sources, especially as catchment size increases. Once in the receiving waterbody, various abiotic and biotic processes further modify P transport, bioavailability and potential impacts (Droppo, 2001; Edwards and Wetzel, 2005). These processes become more significant as catchment size increases and water residence time decreases due to hydrological damping. Combining the wide range in source composition (an order of magnitude variation in P concentrations (Edwards and Withers, 2008)), with the large temporal and spatial aspects of P delivery and subsequent cycling in different types of waterbody makes it difficult to accurately assess agriculture's contribution to eutrophication.

These complications are largely ignored in assessing source contributions which are largely based on catchment annual P export loads.

Moss (1996) concluded that the UK was 'awash with nutrients' due to the ubiquitous inputs of P from wastewater discharges, although agriculture was also implicated in some areas. White and Hammond (2007) recently estimated an annual UK loading of 42 kt of P of which 28% originates from agriculture. This contrasts with the situation for N, where agriculture is estimated to contribute ca. 70% of the 571 kt entering UK waters (Hunt et al., 2004). Recent estimates for Europe suggest that agriculture contributes 50-80% of the total N load to water, and between 25-75% of the P load depending on population density and the extent of wastewater treatment (European Environment Agency, 2005). Hence it is perhaps not surprising that river nitrate concentrations correlate better with the proportion of arable land in a catchment, whilst river P concentrations correlate better with indices of urbanisation and population density (Ferrier et al., 2001; Nedwell et al., 2002).

Since there is likely to be a strong linkage between population density and the proportion of intensive agricultural land within catchments, it is often difficult to disentangle the relative contribution and ecological importance of diffuse sources, since they will invariably be masked by highly concentrated point source discharges. In rural areas with lower population density, there is some evidence to indicate a link between intensity of farming practices and P concentrations in streams (Foy and Withers, 1995; Monaghan et al., 2007). As controls over point source discharges of P become more widespread, the relative contribution from diffuse sources will increase (European Environment Agency, 2005) and these linkages may become more evident. Where agriculture does contribute to eutrophication, P inputs from diffuse agricultural sources are likely to be linked to specific farming systems or high-risk practices.

1.5 The nature of farming

As with N, concern over the environmental impacts of P use in agriculture within the developed countries has increased as the linkages between agricultural intensification, increased P concentrations in land runoff and potential eutrophication have become more widely appreciated (e.g. Tunney et al., 1997; Edwards and Withers, 1998). Inputs and cycling of P on farms have increased since the Second World War to optimise livestock and crop production. This has resulted in an accumulation of surplus P in the soil and increased the risk of P transfer in land runoff Some catchment studies suggest a direct linkage between fertiliser P inputs and stream P concentrations (Calhoun et al., 2002).

On intensive livestock farms, faecal deposition and temporary storage of manure P around farm buildings and the need to frequently recycle stored manure to fields has increased the risk of more direct P transfer to watercourses (e.g. Hooda et al., 1999; Edwards and Hooda, 2007). Parallel advances in farm management including increased stocking densities, expansion of the area sown to winter crops at the expense of grassland, conversion from solid to liquid manure handling systems and the installation of underdrainage systems have also contributed to the increased ease with which nutrients are transported in land run-off and drainage from agricultural land to surface waters (Sharpley and Withers, 1994; Chambers et al., 2000; Dougherty et al., 2004). Some examples of farming practices that might influence rates of P export from agricultural land are given in Table 1.3. Aspects of land use, P fertiliser and manure applications and cultivation practices will be investigated in this thesis.

Phosphorus transfer in land runoff can therefore be expected to show considerable spatial and temporal variation. This variation relates to both the suitability of the farming system for the prevailing climate and the particular land use patterns adopted within different farm types. Since P use is largely a function of the type of farming system, the regional distribution of farming systems is therefore an important factor influencing soil P accumulation rates and potential for P loss in relation to climate and landform. For example, in the U.K., there is a clear geographical separation between upland farms which occur above 300 m O.D., lowland arable farms in the east and lowland grass farms in the west (Fig. 1.3). Grassland occupies about 70% of the UK land area and predominates on heavier-textured soils, whilst arable crops, particularly rotations that include high P input crops (e.g. potatoes and horticultural crops) tend to be concentrated on well-drained and more workable soils.

Table 1.3 Some critical farm management operations that influence P transfer in runoff from agricultural land.

Farming operation	Farm type	Timing	Environmental impact	Contributing factors
Land use planning (including pasture improvement)	All	Variable	Loss of soil and nutrients in rapid runoff and during floods.	Siting of high-risk crops close to watercourses
Soil cultivation up and down slope	Arable	Aug-Nov Feb-May	Erosion of soil before crop established	Over-cultivation Late sowing Planting of ridged crops
Establishment of tramlines after sowing	Arable	Sep-Mar	Channelling of runoff and erosion of soil	Trafficking on wet soils leaves deeper channels
Harvesting of root and forage crops	Arable Dairy	Oct-Dec	Erosion of bare compacted soils	Harvesting in wet soil conditions
Fertiliser and manure application	All	Variable	Build-up of soil P P release due to rainfall impact after application	Over-application No allowance for manure P Broadcast rather than incorporated
Grazing of pastures	Dairy, beef and sheep	Mar-Oct	P release from dung pats Erosion of soil from poached areas	Livestock access to streams High stocking densities Grazing wet soils
Grazing of fodder crops/stubbles	Outdoor pigs	All year	Erosion of bare and compacted soils	Grazing during wet soil conditions
	Dairy and sheep	Nov-Feb	Build-up of soil P where animals congregate	High stocking densities
Storage of manure on-farm	All livestock	Nov-Mar	Runoff from manure heaps	Siting manure heaps close to stream
			Overflow from slurry stores	Inadequate storage facilities
Moving livestock between fields and between fields and farm	All livestock	All year	Runoff from dung on tracks and roads	Proximity of tracks and roads to stream
Washing out milking pipes and hosing down yards	Dairy, pigs and poultry	All year	Runoff from farmyard	No separate storage facilities

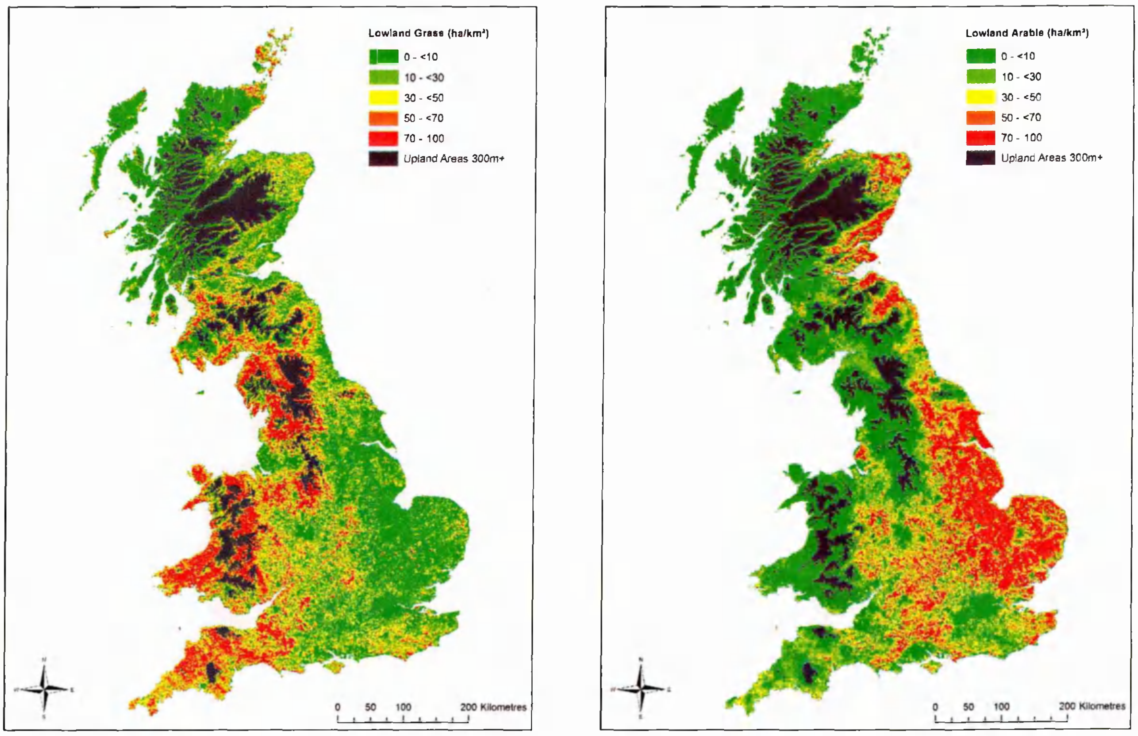


Figure 1.3 The distribution of lowland grass and lowland arable land in relation to upland areas in Great Britain.

1.6 Outline of thesis

1.6.1 Scope of the thesis

To achieve the improvements in water quality required under the WFD, it is important to understand the relative contribution of different P sources and their ecological impact in different types of receiving waters. This will enable investment in cost-effective control action programmes that can reduce P inputs to an acceptable level. Whilst a large amount of research data on the mobilisation and transport of P from agricultural land has been generated worldwide in recent years (Tunney et al., 1997; Sims and Sharpley, 2005), there is still a critical lack of understanding of the role of agriculture in the eutrophication process and where mitigation strategies should best be targeted. This uncertainty relates to the disparity in timing of diffuse P transfer in relation to ecological impacts in flowing waters (Hilton et al., 2006; Jarvie et al., 2006) and the spatial distribution of different sources of P in rural catchments and their ecological relevance (Edwards and Withers, 2007). For agricultural P to cause eutrophication, there must be either contamination of the groundwater that contributes to low summer flows, a significant loss of bioavailable P during the biologically active summer period, or P transported during the winter remains within the waterbody and is remobilised during the summer.

The diffuse nature of agricultural P loss reflects the variable integration of the controlling factors of climate, landform and land management across different spatial and temporal scales. Climate and landform are largely outside the control of the farmer but land management, which encompasses nutrient, soil, crop and livestock management, can be modified to reduce P concentrations in land runoff where these are high (Withers and Jarvis, 1998). Since widespread contamination of watercourses with bioavailable P from agricultural sources is not detected (unlike discharges of municipal wastewater), then particular land uses and farming practices must be responsible for localised eutrophication issues associated with agriculture. The general background hypothesis of this thesis is therefore that the spatial and temporal variation in land management (P inputs, soil P and land management) exerts a major controlling influence over the amount, form, timing and ecological relevance of agricultural P loss to water.

1.6.2 Objectives

This thesis seeks to provide a better understanding of the temporal and spatial patterns of P transfer associated with different farming practices, to place these transfers into context with other sources of P in rural catchments (e.g. farmyards, septic tanks and road runoff) and assess the relative effectiveness of potential options for their control. This understanding will be used to help identify where agricultural practices need to be modified to maintain or improve water quality cost-effectively. The specific objectives are:

Overall objective

To investigate how different farming practices influence the transfer of P in runoff from agricultural land and to appraise the significance of these transfers in relation to other rural sources of P contributing to eutrophication in flowing waters.

Detailed objectives

1. To investigate the impact of farming practices (P inputs, soil P and land management) on stream P concentrations and loads in an upland headwater catchment subject to pasture improvement (Chapter 4).

2. To investigate the impact of farming practices (P inputs, soil P and land management) on stream P concentrations and loads in a lowland headwater catchment under mixed arable and livestock farming (Chapter 5).

3. To determine the impact of soil P accumulation on the mobilisation and potential eutrophication impact of P in land runoff from bare soil (Chapter 6).

4. To compare P concentrations in storm runoff from arable and pasture land with those associated with other potential sources of P in rural catchments and appraise their significance for eutrophication (Chapter 7).

5. To compare the effectiveness of reduced cultivation, time of sowing and presence or absence of tramlines on the mobilisation of P in land runoff from three contrasting soil types in a priority catchment suffering from siltation and eutrophication (Chapter 8).

1.6.3 Structure and outputs

Structure

The thesis is divided into 8 further chapters. Chapter 2 reviews the use of P in UK agriculture, the various sources and sinks of P within catchments, how P is mobilised and delivered from land to water and the fate of P on entering water. Chapter 3 describes the overall experimental approach, the general characteristics of the main study sites, the different land management treatments compared at each site and the methodologies used. Chapters 4-8 present the main experimental results obtained from the field and catchment studies and discuss how they influence our understanding of agriculture's contribution to P export in catchments and role in eutrophication. Finally, the summary, conclusions of the work undertaken in the thesis and recommendations for further work are presented in Chapter 9.

Outputs

A number of peer-reviewed journal papers have been produced from the thesis and these are given below. Papers 1-3 were generated as part of the review process and analysis of the regional distribution of P inputs and outputs in UK agriculture described in Chapter 2. Papers 4 and 5 reported the results of the impact of selected mitigation options on P transfer at the field scale in the Avon catchment (Chapter 8) and paper 6 reported the results of the impact of upland pasture improvement on stream P concentrations at Redesdale (Chapter 4). Further papers will be submitted on the results of data collected in the Wye river basin presented in Chapters 5, 6 and 7.

1. Withers, P.J.A., Edwards, A.C. and Foy, R.H. (2001). Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. Soil Use and Management 17, 139-149.
2. Withers, P.J.A. and Lord, E.I. (2002). Agricultural nutrient inputs to rivers and groundwaters in the UK: policy, environmental management and research needs. The Science of the Total Environment 282-283, 9-24.
3. Withers, P.J.A., Ulen, B., Stamm, C. and Bechmann, M. (2003). Incidental phosphorus loss – is it significant and can it be predicted? Journal of Soil Science and Plant Nutrition 166, 459-468.
4. Withers, P.J.A., Hodgkinson, R.A., Bates, A. and Withers, C.L. (2006). Soil management effects on sediment and phosphorus mobilisation in overland flow from three contrasting soil types. Soil and Tillage Research 93, 438-451.
5. Withers, P.J.A., Hodgkinson, R.A., Bates, A. and Withers, C.M. (2006). Some effects of tramlines on surface runoff, sediment and phosphorus mobilization on an erosion-prone soil. Soil Use and Management 22, 245-255.
6. Withers, P.J.A., Hodgkinson, R.A., Adamson, H. and Green, G. (2007). The impact of pasture improvement on phosphorus concentrations in soils and streams in an upland catchment in Northern England. Agriculture, Ecosystems and the Environment 122, 220-232.

2

PHOSPHORUS TRANSFER FROM AGRICULUTRAL LAND TO WATER

2.1 Introduction

In recent decades, anthropogenic activities have greatly increased the cycling and fluxes of P on a global scale and the incidence of eutrophication, and related impacts, in a range of waterbodies (Allen et al., 1998; Conley, 2000; Smith, 2003). Urbanisation and agricultural intensification are the two key aspects of this anthropogenic activity that have degraded water quality. The concentration of the human population in downstream urban areas has greatly increased the localised discharge of municipal and industrial effluents and the proportion of impervious surfaces in catchments. These changes have altered both the hydrology of stream and river systems, resulted in increased loadings of a range of contaminants and altered the abundance or diversity of aquatic biota (Malmqvist and Rundle, 2000; Paul and Meyer, 2001; Osmundson et al., 2002). For example, Kleine (1979) found that water quality impairment was evident when as little as 12% of the catchment area was occupied by impervious surfaces. In comparing fish community data across 47 small streams in southwestern Wisconsin, Wang et al. (2000) concluded that water quality was degraded more severely by urbanisation than by agriculture on per unit area basis.

Agricultural intensification has also resulted in changes in methods of farming that have potentially adverse environmental consequences. These include trends towards monoculture and polarisation of farm types, increased frequency of cultivation, more productive crops and livestock requiring greater inputs of agrochemicals and feeds, higher stocking rates and dependence on liquid waste handling systems and installation of artificial drainage systems in fields to remove surface wetness. These changes have resulted in uneven accumulation and leakage of nutrients from soils, increased rates of erosion and degradation of soil organic matter (OM) and biomass diversity and increased incidence of acute and chronic pollution associated with handling of livestock wastes (Matson et al., 1997; Hooda et al., 2001; Evans, 2006).

The leakage of P from soils has traditionally been considered to be small due to the soils capacity to bind P and release it slowly into solution. However, there is increasing evidence that widespread soil P accumulation together with increased incidence of erosion has increased the amounts of P moving in dissolved and particulate forms in agricultural runoff in recent decades (Ryden et al., 1973; Edwards and Withers, 1998). Livestock manures in modern farming systems also have a high proportion of their P content in a water-soluble form with a greater risk of mobilisation in runoff if rain follows soon after application (Hooda et al., 1999; Withers et al., 2001). The larger quantities of manure P stored around farm buildings have also been highlighted as an important, localised source of highly concentrated P (Meals, 1993; Wilcock et al., 1999; Edwards and Hooda, 2007). This evidence base suggests actions are required to reduce agriculture's impact on the environment. To manage P sustainably within agricultural systems, it is necessary to identify what components of the agricultural P cycle are to blame, how farming practices are exacerbating the risk of P transfer in runoff waters and what reductions in P fluxes are necessary to restore good ecological quality in different types of waterbody.

This chapter reviews published literature on the nature and magnitude of P transfer from various sources across the land-water interface in rural catchments dominated by agricultural land. A better understanding of the relative importance of P sources and sinks, their spatial and temporal characteristics and their mode and timing of delivery to the watercourse will help identify where P loss reduction strategies to reduce eutrophication impacts might best be targeted within rural catchments. There is currently much emphasis in England on 'Catchment Sensitive Farming (CSF)' on the basis that reductions in point sources alone will not achieve the challenging water quality targets set under the Water Framework Directive (WFD), (Duncan et al., 2006). However, this still requires accurate assessments of the major issues and identification of ecologically-relevant sources in catchments if the water quality improvements required under the WFD are to be achieved by 2015. The review summarises current understanding of firstly, the P cycle in UK agriculture, the P fluxes within it and the spatial and temporal distribution of any surplus P and secondly the transfers of P from the terrestrial to the aquatic environment, how farming practices are influencing these transfers and the relative importance of P sources other than those from farmed land, including those originating within the waterbody.

2.2 Phosphorus cycling in UK agriculture

The accounting of sources, sinks and redistribution pathways in a unit area over a unit time has been encompassed in the term 'cycle', 'budget' or 'mass balance'. Budgets for phosphorus (P) are common to both the terrestrial (e.g. Haygarth et al., 1998a) and the aquatic environment (e.g. Slaymaker, 2003), and are useful as both a conceptual and management framework for understanding P cycling, where losses from the system are likely to occur and how that loss might be reduced. Sources and sinks can be accounted at field, farm, catchment, river basin, national and global scales and operate over storm, seasonal, annual and decade timescales. Sinks may become permanent where they no longer take part in the budget at the spatial and temporal scale being examined (e.g. P fixation in subsoils, or deposition in deep lakes and oceans) but in most cases are temporary and available for remobilisation (termed storages).

Storages help to buffer the impact of primary P sources on our environment but then become a secondary source of P as the store is remobilised. For example, for agricultural production, a certain degree of P storage in the soil is necessary to ensure optimum crop yields, but as P accumulates in the soil, the land surface becomes an increasingly significant source of P to water (Bennett et al., 2001). Within flowing waters, P sources are temporarily stored within bottom sediments but are constantly remobilised and selectively transformed on their journey to a lake or to the ocean affecting their availability (Kronvang et al., 1999; Edwards and Wetzel, 2005). Interactions between different sources and sinks occurring over varying temporal and spatial scales governs the variability in concentrations and loads of P entering water and their ecological impact.

An annual P budget for UK agriculture in 1993 is presented in Fig. 2.1. The year 1993 was chosen on the basis of the availability of data from a number of different sources, which are detailed in Appendix 1. Primary sources of P in this budget are atmospheric deposition and imports of fertilisers, feeds and manures, whilst P sinks are the crop and animal products that are consumed (foods) or used (timber, biofuel) by the human population. For P budgets over much longer timescales, the weathering of soil parent materials may also be taken into account. Atmospheric inputs are almost entirely associated with wind blown soil and dust particles, with amounts varying from ca. 0.2 kg ha^{-1} in upland areas (Gibson et al., 1995) to 0.5 kg ha^{-1} or more in cultivated lowland areas where soil is left exposed (Holton et al., 1998). Atmospheric P inputs alone can exceed P outputs in upland areas (e.g. Crisp, 1966), but in lowland catchments, these inputs are negligible in relation to the large amounts of fertilisers and manures applied to land and P storage in the soil (Fig. 2.1).

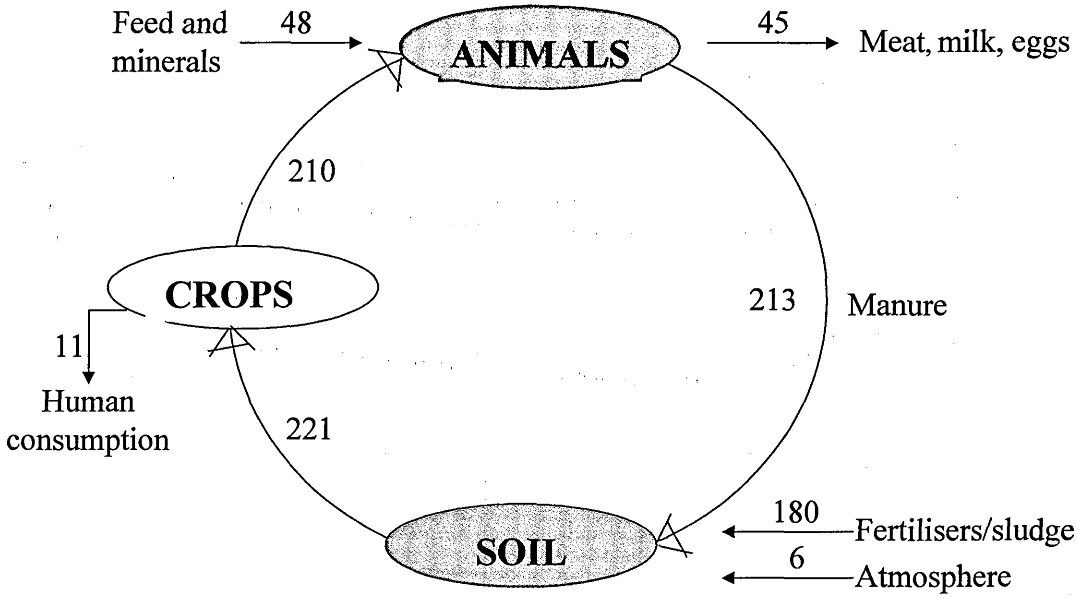


Figure 2.1. Cycling of P in UK agriculture in 1993. Figures are in '000 t.

The cycle shows that P is stored in the soil, in crops and in animals and that the fluxes of P between these stores are far greater than the imports of the primary imports and exports of P. Imports of P totalled 234,000 t in 1993, of which inorganic fertiliser accounted for 77% and feed and minerals (e.g. soya bean meal, maize starch residues, fishmeal and dicalcium phosphate) accounted for 20%. Much of this imported fertiliser P was as ammonium phosphate and highly water-soluble. The amount of P recycled in animal excreta was similar to that removed in crops and grassland at ca. 220,000 t, whilst a comparatively small amount of P was removed in crop and animal products for human consumption (56,000 t). A large proportion (87%) of the P removed in UK arable products was therefore fed to livestock, and a large proportion (82%) of the feed consumed was excreted as dung and urine. Approximately 55% of total excreted P is stored around farm buildings during the winter housing period (November to April) and spread back onto land during the year. The efficiencies of P use (P export/output calculated as a proportion of P import/input) in crop production, livestock production and in the whole of UK agriculture in 1993 were 56%, 18% and 25%, respectively. This leaves a national P surplus of 178,000 t, or ca. 9 kg P ha^{-1} yr^{-1} if spread over the total agricultural land area (18.5 million ha) in the UK, or ca. 15 kg P ha^{-1} yr^{-1} if spread over just the area of cultivated and managed pasture land (11.5 million ha). This surplus is very similar to that calculated for Northern Ireland by Foy et al. (2002) but lower than the surplus calculated in 1973 (Centre for Agricultural Strategy, 1978). This difference is due to lower inputs of fertiliser P than in 1973. This surplus P is stored in the soil and represents a potential source of increased P transfer to the aquatic environment. If one assumes an annual P loss of ca. 1 kg ha^{-1} over the arable and productive grassland area (Department for Environment, Food and Rural Affairs, 2004), the total annual loss of P is ca. 11 kt or 5% of the total P imports in 1993.

2.2.1 Spatial and temporal variability in surplus phosphorus

The data presented in Fig. 2.1 suggest that the P surplus can be calculated as the difference between total P inputs in fertilisers and livestock manures, and total removal of P in grassland and arable crops; i.e. the surface balance approach to nutrient accounting (Oenema et al., 2003). Using this approach, it was found that the distribution of the national P surplus between arable land and productive grassland in 1993 was similar, but that arable land receives a greater average areal loading of surplus P than does grassland due to its smaller total area (Table 2.1). This reflects the combined influence of the substantial applications of inorganic P fertiliser and pig and poultry manure P applied to arable land. Manure inputs to grassland are larger than to arable land, but are spread over a greater land area so that the average loading is lower. Outputs from arable and grassland systems are quite similar (Table 2.1)

Table 2.1. Estimated average annual inputs, outputs and surplus of P (kg ha^{-1}yr^{-1}) on all arable and managed grassland in the UK in 1993.

	Arable land	Grassland[a]	Arable and grassland
Area ('000 ha)	4566	6769	11335
Inputs			
Fertiliser	23	9	15
Manure - housed	14	8	10
- grazing	1	13	8
Outputs	19	18	18
Surplus[b]	19	12	15

[a]Temporary and permanent grassland but excluding rough grazing. [b]Excluding the small imports of P from the atmosphere (0.3 kg ha^{-1}yr^{-1}) and in sewage sludge (0.6 kg ha^{-1}yr^{-1}). Losses in land run-off and drainage are also excluded.

The distribution of manure P is also more variable on arable land: <20% of arable receives manure in a single year (equivalent to a P loading of 80 kg P ha^{-1} according to Smith et al, 1998), while some arable fields never receive manure P. Dairy farms that rely on home-produced grass have lower P surpluses than those that are dependent on imported concentrate feeds, which have twice the P content of grass (Lynch and Caffrey, 1997). Hence the annual P surplus can vary from zero (or deficit) on some farms to 100-200 kg ha^{-1} or more on others; for example, Domburg et al. (2000) calculated surpluses of up to 160 kg P ha^{-1} for individual farms in one Scottish catchment. Average annual P surpluses in the EU calculated by Brouwer et al. (1995) were 24, 12 and 269 for dairy, arable cropping and granivore farms, respectively. Withers et al. (1999) calculated a surplus of up to 120 kg P ha^{-1} on forage maize fields receiving both cow slurry and fertiliser and Watson et al. (2003) found soils soon became P saturated where animals (pigs) preferentially defecated around drinking troughs and fence perimeters. The spatial variability in surplus P is therefore very large at the field and farm scale.

Using the same surface balance approach, temporal trends in the P surplus can be calculated and related back to changes in the magnitude of P sources and sinks. Such calculations show that the national P surplus in the UK increased from a value of ca. 95,000 t (7 kg P ha^{-1}) in 1935 to a peak of ca. 240,000 t (20 kg P ha^{-1}) in 1970 due to a rapid expansion in the use of fertilisers and feeds after the Second World War (Fig. 2.2).

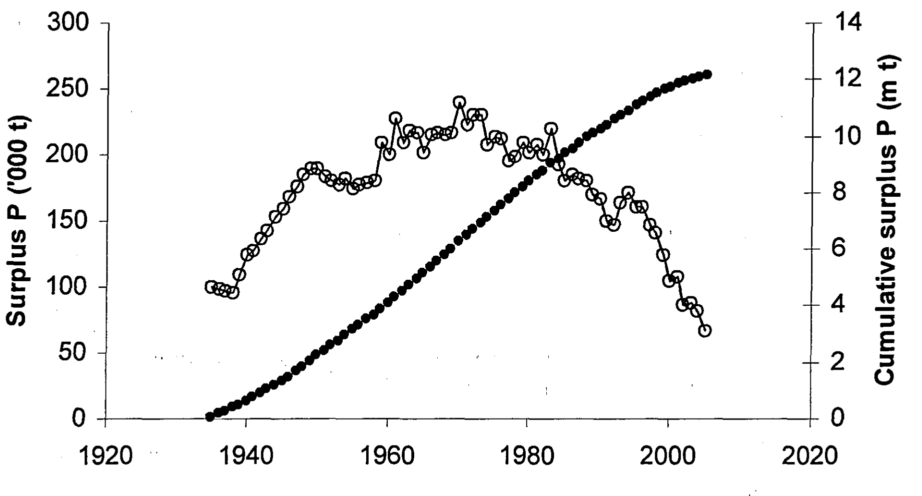


Figure 2.2 Changes in the annual P surplus for the UK between 1935 and 2005 and the cumulative trend.

For example, consumption of purchased feeds increased threefold to 15 million t between the period 1950 to 1970 (Ministry of Agriculture, Fisheries and Food, 2001b), while fertiliser P use increased by 30%. Since 1970, the surplus has declined by over 70% to a value of 66,000 t (4 kg P ha^{-1}across all agricultural land) in 2005. This reflects both a large decrease in fertiliser P use (>50% decline in the last 25 years) and increased crop P removal in higher yielding crops. Although manure P inputs maybe overestimated due to lower volumes excreted, or a lower manure P content, in earlier years, the data suggest that an increasing proportion of total crop P offtake is exported from the cycle for an expanding population. For example, cereal exports to other countries were 335,000 t in 1970 and 5.3 million t in 2000 (Ministry of Agriculture, Fisheries and Food, 2001b). As national surpluses have decreased, the cumulative upward trend is slowing (Fig. 2.2), although surpluses on individual farms and fields can still be very large.

2.2.2 Storage of surplus phosphorus

The soil represents the major store for surplus P in the terrestrial environment and the physical (moisture, temperature and aeration), chemical (pH, ionic strength and redox potential) and biological (microbial, plant and organic residues) processes operating in soils governs the hydrological routing of water and P availability for uptake by crops or release to runoff (Fig. 2.3). Soil P occurs in inorganic and organic forms in varying proportions depending on the environment (climate, soil type) and the type of land management (cultivation frequency, lime and P inputs). An understanding of the cycling of organic and inorganic P in soils is necessary to understand the links between surplus P accumulation in soils and P transfer in land runoff.

Organic P cycling in soils

Organic P is that fraction of soil P that is bound to carbon (C) and can account for up to 90% of total P in upland soils where the harsh environmental conditions favour accumulation of OM. Plant and animal residues recycled to soil are mineralised by microorganisms to a range of organic compounds which include orthophosphate mono and di-esters, phosphonates and organic polyphosphates (Harrison, 1987; Condron et al., 2005). Organic P compounds may therefore be of direct plant origin, direct microbial origin, or decomposition products of both, and have an important function in maintaining P availability in natural environments and in grassland soils that accumulate OM. Monoesters (mostly inositol phosphates) and diesters (nucleic acids and phospholipids) are the most frequently identified forms and are weakly adsorbed by soils through ligand exchange. Inositol phosphate is the most common organic P compound (up to 60% of soil organic P) since it is strongly bound within soil-organic matter complexes which provides a degree of protection against breakdown. In contrast, diesters are broken down in days (Tate, 1984).

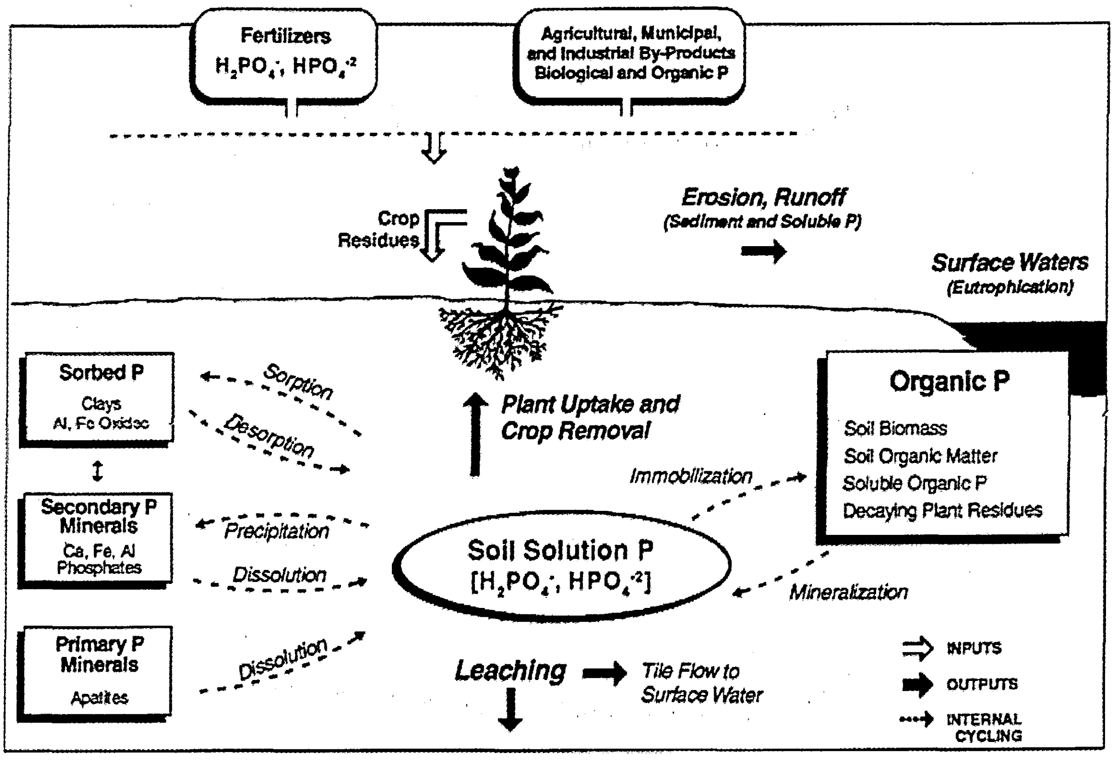


Figure 2.3 The soil P cycle (with permission from Pierzynski et al., 2005b).

Rates of P mineralisation of depend on phosphatase activity, the availability (accessibility) of suitable carbon substrates for the range of microorganisms present and environmental factors such as wetting and drying. Phosphatase enzymes are produced by plant roots, mycorrhizae and microorganisms according to their demand for P relative to the availability of inorganic P in the soil (Tarafdar and Claassen, 1988). Brookes et al. (1984) measured net mineralisation rates ranging between 2-11 kg P ha^{-1} yr^{-1} for arable soils and 20-40 kg P ha^{-1} yr^{-1} for grassland soils in lowland England. In organic and peat soils, mineralisation provides the main source of P for plant uptake and dissolved organic P compounds dominate the soil solution (Shand et al., 1994;

Turner et al., 2003). The capacity of peat soils to adsorb inorganic P is largely dependent on their iron (Fe) content (Cuttle, 1983; Daly et al., 2001).

Inorganic P cycling in soils

In regularly cultivated and fertilised soils, P availability is dependent more on inorganic P reactions than organic reactions (Sharpley, 1985a; Pierzynski et al., 2005a). When land is improved for agriculture through liming, cultivation and/or NPK fertiliser additions, OM is broken down more quickly, crops take up more P, animals deposit larger quantities of P in dung and the cycling of P through the soil, plant and animal is increased. An increasing proportion of soil P is held in inorganic P forms, equilibrium P concentrations in the soil solution increase and phosphatase production is inhibited. After fertilisation, the solubility products of various secondary P minerals (Ca, Fe and Al precipitates) removes excess inorganic P out of solution allowing desorption and adsorption reactions to regulate the diffusive flux of P to solution in line with rates of plant P uptake (Syers and Iskander, 1981; Frossard et al., 2000).

In natural ecosystems, inorganic P is held in the form of primary minerals (largely apatite) and solution P concentrations are governed by slow rates of weathering. Inorganic P in solution is present as the large, weakly hydrated orthophosphate ion ($H_2PO_4^-$ or HPO_4^{2-}) which is rapidly adsorbed onto positively charged surfaces of amorphous Fe and Al oxides and hydroxides in acid and neutral pH soils, and also to calcium carbonate in calcareous soils. The amount of positively charged surfaces largely governs the soil P sorption capacity and is pH dependent, decreasing as pH rises (Barrow, 1985). Orthophosphate may also be specifically adsorbed onto exposed negatively charged clay surfaces by overcoming the electrostatic forces of repulsion.

As plants take up P, the concentration of inorganic P in the soil solution is replenished by the desorption of P from a pool of available P (termed labile P) at rates which depend on the soil P buffering capacity (Holford, 1997). As the soil P sorption capacity is gradually reduced, added P becomes more weakly held and a higher concentration of P is maintained in the soil solution. The changeover from high-energy to low-energy surfaces occurs when the soil P sorption capacity is ca. 25% saturated and this target value has been found to correspond to increased mobility of P in percolating drainage water (Breeuwsma and Silva, 1992; Holford et al., 1997).

2.2.3 Rates of surplus P accumulation in soils

The average cumulative surplus P loading to agricultural land in the UK since 1935 (Fig. 2.2) amounts to over 1000 kg ha^{-1} over the productive grassland and arable area (11.5 million ha). For many soils, this is equivalent to at least a 50% increase in average native soil total and exchangeable P levels (measured as Olsen-extractable P (OP)) since the Second World War. For example, the results of the National Soils Inventory (NSI) in England and Wales show a median total P concentration of 700-800 mg kg^{-1} for arable and productive grassland compared to an average value close to 500 mg kg^{-1} for deciduous and coniferous woodland (Table 2.3). Clearly, the relative impact of cumulative surplus P on soil total P is site specific depending on soil type, the amount of the cumulative P surplus and the depth over which it has accumulated. This has resulted in a wide range in soil total and OP concentrations (Table 2.2).

Table 2.2 Total and Olsen-extractable P concentrations in soils according to selected land uses in 1985. (Data is derived from the National Soils Inventory of England and Wales after Harrod and Fraser, 1999).

Land use	No. of samples	Total P (mg kg^{-1})		Olsen-P (mg kg^{-1})	
		Range	Median	Range	Median
Horticulture	40	280-2312	854	4-160	45
Arable	1880	246-4189	734	2-205	26
Grassland (<5 years)	673	170-4535	822	1-274	20
Grassland (>5 years)	1558	188-4529	847	1-337	17
Upland grass	229	236-2214	764	1-130	12
Rough grazing	385	219-2659	768	1-144	13
Coniferous woodland	207	41-1685	448	1-94	11
Deciduous woodland	229	108-2636	528	1-210	12

As arable and horticultural soils generally receive larger amounts of water-soluble P fertiliser than grassland soils (Table 2.1), they maintain greater average concentrations of OP (Table, 2.2; Fig. 2.4). Soils regularly receiving both manure and fertiliser often exhibit very high OP concentrations: for example those in intensive horticulture, in rotations involving responsive crops such as potatoes and sugar beet, or receiving pig and poultry manure and on dairy farms with high stocking rates and forage maize (Domburg et al., 1998; Skinner and Todd, 1998). Farming practices which increase surplus P accumulation in soils include underestimating the nutrient value of P in organic manures, uneven distribution of manures around the available land area and not matching total P inputs according to crop demand. In many cases, farmers have preferred to apply insurance dressings of P in feeds and fertilisers to ensure that productivity is not reduced in difficult seasons, or animal fertility compromised. This has led to a high proportion of over-fertilised soils in many parts of the developed world and increased risk of P loss in agricultural runoff (Tunney, 1990; Withers et al., 1999).

However, data from the Representative Soil Sampling Scheme (RSSS) in England and Wales suggests the proportion of over-fertilised soils in both arable and grassland soils has declined since the survey began in 1969 (Fig. 2.4). This probably reflects fixation of soil P into non-labile forms over time (Syers and Iskander, 1981), reduced fertiliser use due to increasing costs and better nutrient management on farms. These temporal trends further compound the wide spatial variability in soil P concentrations outlined above. The survey data also shows that there is still a lack of appreciation, especially amongst grassland farmers, of the basic principles and importance of P management as demonstrated by the proportion of soils that are still deficient in P (Fig. 2.4).

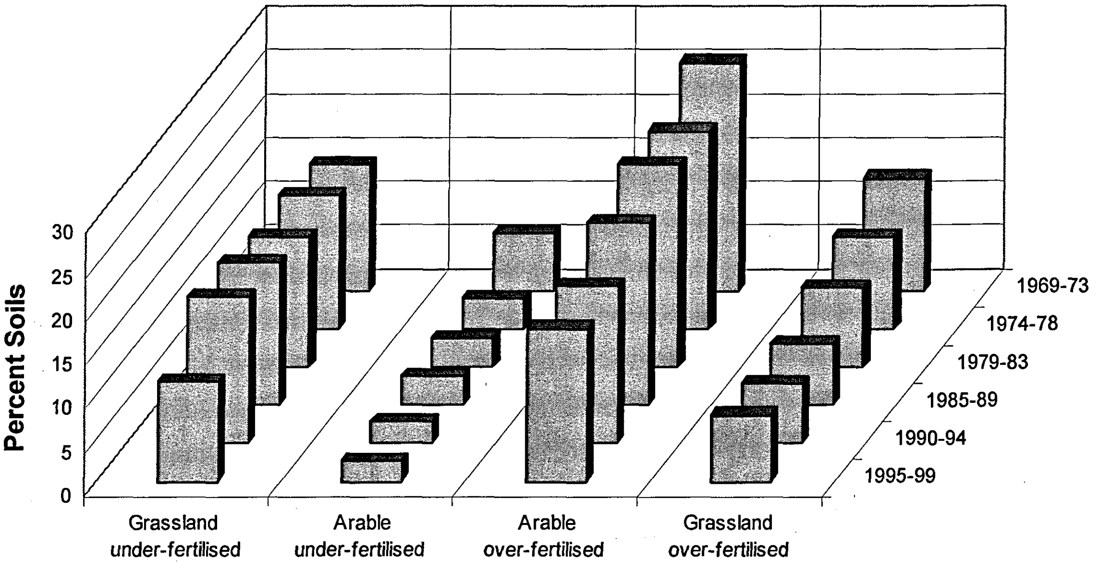


Figure 2.4 Proportions of Olsen-P for arable and grassland soils in England and Wales either under-fertilised (<15 mg L^{-1}), or over-fertilised (>45 mg L^{-1}) with P during 1969-1999. Data are updated from Skinner and Todd, (1998).

2.3 Transfer of phosphorus from terrestrial to aquatic ecosystems

As rainwater passes over, or through, the soil at various rates it mobilises and transports inorganic and organic P along surface and sub-surface hydrological pathways to the watercourse (Fig. 2.3). Sources of P at the land surface which may be released or detached to runoff include soil, vegetation, fertilisers and livestock faeces creating the potential for a wide range in the composition and concentration of any P present (Sharpley et al., 2001; Edwards and Withers, 2008). During storm events, variable combinations of these sources become mobilised and transported from variable areas within fields, farms and catchments providing a large degree of spatial and temporal variability. Typically the majority of the P is transported during a few major events (Johnson et al., 1976; Munn and Prepas, 1986; Pionke and Kunishi, 1992) and seasonality is largely dominated by the pattern of storm events during the year. Rainfall distribution is therefore an important factor governing whether runoff P concentrations are greatest during summer (e.g. Donohue et al., 2005), or during winter (e.g. McDowell and Wilcock, 2007). Phosphorus export in catchments is therefore a function of both water discharge and the ease with which P is mobilised and transported in runoff. This relationship defines the flow-weighted P concentration which provides a useful way of comparing source P availability at different temporal and spatial scales (Foy and Withers, 1995).

In agricultural areas, rates of P transfer from land have increased relative to those from more pristine environments as a direct result of the increased amount and accessibility of P source material at the land surface. As described above, this extra P source material reflects the greater quantities of P applied to agricultural land and the accumulation of surplus P in the soil. Additional, and more indirect, effects of farming on P transfer are associated with changes in crop and livestock production methods as farm systems have intensified (see Table 1.3). These include:

- increased cycling of nutrients during the conversion of marginal hill land to pasture in sensitive upland areas (Harrison and Taylor, 1987; Cuttle and James, 1995).
- increased runoff and soil erosion from arable land due to a switch from spring cropping to winter cropping, increased frequency of cultivation and incidence of compaction (Chambers et al., 2000; Evans, 2006).

- increased runoff and soil erosion from grassland due to increased stocking rates causing poaching of grazed land and stream bank collapse (Sharpley and Syers, 1979; James and Alexander, 1998; McDowell and Wilcock, 2007).
- increased leakage of P due to a switch from solid to liquid-based livestock waste handling systems (Williams and Nicholson, 1995; Smith et al., 2001; Dougherty et al., 2004)
- increased hydrological connectivity between the field and the watercourse due to the introduction of underdrainage systems in fields (Sims et al., 1998; Stamm et al., 1998)

Rates of P transfer in land runoff are therefore highly dependent on the interaction between the environmental factors which are outside the farmers control (climate, soil type and slope) and land management factors which are within the farmers control. These land management factors can be grouped into nutrient management (P inputs and how they are managed), crop and livestock management (crop types, residue management, animal numbers and stocking rates) and soil management (type, method and timing of cultivations). Agriculture's contribution to P loads entering surface waters depends on whether flow-weighted P concentrations in land runoff are affected more by environmental factors or by land management factors.

2.3.1 Farming and phosphorus loss at the catchment scale

In contrast to N, relationships between P exports, or river P concentrations, and land use are not readily detected in large catchments heavily influenced by urbanisation (i.e. impervious surfaces), a variety of point sources or eroding channel banks. Monitoring of larger catchments has invariably shown that riverine P concentrations are dominated by dissolved reactive P (DRP) which (a) typically become diluted at high flow and increase during low flow and (b) show positive relationships with boron (B) which is a conservative ingredient of detergent washing powders (Jarvie et al., 2006). These characteristic features reflect the dominant influence of point sources and their high ecological impact, rather than agricultural P transfers which largely occur during high flow.

In smaller rural catchments, some clear effects of agricultural intensification have been observed. In Finland, Foy and Withers (1995) summarise data for Finnish catchments where stream concentrations of total P (TP) increased with the proportion of cultivated land and for two Northern Ireland grassland catchments where concentrations of dissolved reactive P (DRP) increased under high stocking rates. Maberley et al. (2003) detected a negative relationship between algal P limitation in 30 upland lakes in the UK and the proportion of improved pasture in their catchments. In New Zealand, hill catchments with improved pasture export much greater P loads than forested catchments in the same locality (Cooper and Thomsen, 1988; Quinn and Stroud, 2002).

Conversely, Jordan et al. (1997) found no relationship between river P concentrations (all forms) and land use in 17 small Chesapeake Bay watersheds with low human populations and no sewage outfalls. Concentrations were related more to sediment levels in the rivers which appeared to be unrelated to the proportion of row crops in the catchments. Salvia-Castellvi et al. (2005) also found that in-stream sediment mobilisation accounted for much of the variation in P across several catchments. Kyllmar et al. (2006) found that P exports from 27 small Swedish catchments were more related to soil type than to land use, with highest flow-weighted P concentrations occurring on clay soils. Some correlation existed between land use only within a soil type grouping. Clearly, factors other than land use and land management are important in small catchments.

Some literature data on P export from small rural catchments dominated by different land uses are presented in Tables 2.3-2.6. Contrary to previous reviews which have tended to concentrate on only areal P export rates, P transfer is also presented in terms flow-weighted concentrations. The flow-weighted concentration in mg L^{-1} is equivalent to P export in kg ha^{-1} per 100 mm of flow and is probably a more useful method of assessing the effects of land use across different geographical regions with variable flow regimes.

Forested catchments

The exports of P from forested catchments are usually taken to represent background P concentrations in land runoff and therefore act as a reference point for a particular geomorphology. Several reviews have concluded that forested catchments export on average about 0.1 kg ha^{-1} yr^{-1} (Ryden et al., 1973; Prairie and Kalff, 1986). Data for forested catchments in the UK are rare. Nisbet (2001) quotes P export rates of 0.05-0.32 kg P ha^{-1} yr^{-1} from Scottish forests receiving rock phosphate fertiliser but no flow data are provided.

Kronvang et al. (2005) reported an average (1989-2002) flow-weighted TP concentration of 53 μg L^{-1} (range 42-59 μg L^{-1}) for seven small catchment streams in Denmark with >90% forest (Table 2.3). The corresponding DRP concentration was 21 (range 14-26 μg L^{-1}), or 40% of TP. Rekolainen et al. (1995) reported very similar average (1981-1990) TP values for seven small forest catchments in Finland; mean of 36 μg L^{-1} (range 5-67 μg L^{-1}). In New Zealand, flow-weighted TP and DRP/TDP concentrations in streams draining both hardwood and conifer forests were typically 40-50 μg L^{-1} with ca. 30-40% in dissolved form (Cooper and Thomsen, 1988). Concentrations of TP in a stream draining native scrub were only slightly larger at 57 μg L^{-1} (Table 2.3). On steeper land, Quinn and Stroud (2002) measured larger TP concentrations from native forest (92 μg L^{-1}), but when pasture was introduced into the area the mean flow-weighted concentration rose to 202 μg L^{-1}.

Forest management potentially influences P concentrations in draining streams through P fertiliser application and increased transfer of suspended solids (SS) either following ploughing during afforestation, during clearfelling or from forest roads during these operations (Soutar, 1989; Nisbet, 2001). Transfer of SS can be considerable but no data are provided for P. Harriman (1978) examined the effects of fertiliser application (47 kg P ha^{-1} of ground rock phosphate) to either part or all of small forested catchments in Scotland over a 2-3 year period (Table 2.3). The 'control' treatment accidentally received P and a stream draining a nearby open moorland area was used as the control instead. The streams draining the fertilised forest showed relatively large flow-weighted TP concentrations (100-200 μg L^{-1}) compared to the control, and other values for forested catchments in Table 2.3. Nearly all the transported P was in dissolved form and accounted for up to 15% of the fertiliser P applied. Increased concentrations were measured even after 3 years, although Nisbet et al. (2002) suggest this can be reduced to 6 months by application of smaller more frequent amounts, by hand rather than by helicopter and provision of strategically placed buffer strips.

Table 2.3 Some recent examples of export and flow-weighted (FW) concentrations of total P (TP) and dissolved P (DRP or TDP) in stream flow from forested catchments.

Land use Catchment	Characteristics	Size (km^2)	Monitoring period (yrs)	Flow (mm)	TP export (kg ha^{-1} yr^{-1})	FWTP (μg L^{-1})	FWDRP (% of TP)	Reference
Denmark	7 small catchments with <10% agricultural land	n.d.	1989-2002	184 (112-250)	0.09 (0.04-0.14)	53 (42-59)	39 (33-44)	Kronvang et al. (2005)
Finland	7 small catchments with no cultivated land	0.07-21.7	1981-1990	329 (165-508)	0.1 (0.02-0.16)	36 (18-56)	n.d.	Rekolainen et al. (1995)
Taupo, New Zealand	Puruwai - hardwood forest on volcanic soils	0.28	2 (1983-84)	300	0.12	40	15	Cooper and Thomsen (1988)
	Puriki - pine forest on volcanic soils	0.34		215	0.095	44	39	
Whatawhata, New Zealand	Wkakakai – steep hill land with hardwood forest on brown earth soils	3.0	1 (1996-97)	629	0.58	92	47	Quinn and Stroud (2002)
	Kiripaka – mixed land use (59% forest, 41% pasture) in the same area	2.7	1 (1996-97)	660	1.33	202	20	
Loch Ard Forest, Aberfoyle, UK	Stream 2 (open moorland)	n.d.	2 (1973-74)		0.12	11		Harriman (1978)
	Stream 3 (partially fertilised forest)	0.9		1137	1.51	139	>90	
	Stream 4 (fully fertilized forest)	1.1			2.36	216		

n.d., no data.

Upland catchments

Upland catchments are largely composed of acidic, peaty soils with moorland vegetation (*Nardus and Molinia spp.*) used for rough grazing with variable pockets of pasture improved by addition of lime and NPK fertilisers. Annual flows are much larger than found in lowland areas and are frequently over 1000 mm (Table 2.4). Flow-weighted concentrations of P in upland streams draining granitic parent materials are often close to detection limits (< 10 μg L^{-1}) (Rigler, 1979; Jarvie et al., 2008a), but are in the range 54-61 μg L^{-1} in limestone areas (Crisp, 1966; Bowes et al., 2003), with 20-40% in DRP form and 20-30% in dissolved unreactive form (Table 2.4). These values are very similar to those summarised for forested areas (Table 2.3). Wood (2002) measured a P export of 0.62 kg ha^{-1} from an upland granitic catchment (17.6 km^2) with largely unimproved grassland rather than moorland, equivalent to a flow-weighted concentration of 37 μg L^{-1}.

The effects of pasture improvement on stream P concentrations in the uplands have not been extensively studied in the UK. What little work has been done suggests there is little effect at the catchment scale (e.g. Hornung, 1984), although increased P concentrations in runoff following fertiliser application have been recorded in smaller scale (1ha) field studies but not consistently so (Roberts et al., 1989; Cuttle and James, 1995). In New Zealand, conversion of native forest/scrub to hill pasture has greatly increased the export of both dissolved and particulate P forms (Cooper and Thomsen, 1988; Quinn and Stroud, 2002). Flow-weighted TP concentrations up to nearly 0.4 mg L^{-1} have been measured on steeply sloping land with the majority occurring in particulate form (Table 2.4). This effect is related to both an increase in the proportion of rainfall occurring in overland flow, increased erosion and availability of source material (dung and fertilisers). When steep hill land was reverted back from pasture to forest, a significant reduction in TP export (60%) and flow-weighted concentrations was observed (Table 2.4; Quinn et al., 2007). Further work is required to determine the effects of pasture improvement on transfers of dissolved and particulate P in the UK.

Table 2.4 Some examples of export and flow-weighted (FW) concentrations of total P (TP) and dissolved P (DRP or TDP) in stream flow from upland/hill areas.

Catchment Location	Characteristics	Size (km^2)	Monitoring period (yrs)	Flow (mm)	TP export (kg ha^{-1} yr^{-1})	FWTP (μg L^{-1})	FWDRP (% of TP)	Reference
Rough Sike, Pennines, UK	Limestone moorland grazed by sheep, acidic peat soils.	0.08	1 (1962-63)	1368	0.84	61	48[1]	Crisp (1966)
Hart Tor Brook, Dartmoor, UK	Granitic moorland grazed by sheep, acidic peat soils	<2	1 (1974-75)	1916[2]	0.2	<10	n.d.	Rigler (1979)
Grinton Bridge, Yorkshire, UK	Limestone moorland grazed by sheep, acidic peaty and organic soils	258	2 (1998-00)	978	0.27	62	23[3]	Bowes et al. (2003)
Erwood, Wye Wales, UK	Upland catchment of the R. Wye, with 57% grass, 19% rough grass and 20% woodland	1283	1 (2002-03)	815	0.44	54	39[4]	Jarvie et al. (2008b)
Leeham Ford, Devon, U.K	Upland catchment with 80% grassland (largely unimproved) on peaty podzols.	17.6	1 (1998-99)	1660	0.62	37	37	Wood (2002)
Pwllpeiran Farm, Wales	Unimproved molinia grassland	0.01	4	1469	<0.03	<10	n.d.	Cuttle and James (1995)
	Improved grassland: reseeded + P	0.01	(1984-88)	1522	1.36	89	n.d.	
Taupo, New Zealand	Purutaka – ryegrass/clover hill pasture on volcanic soils, grazed by sheep and cattle and receiving 30 kg P ha^{-1} yr^{-1}	0.11	2 (1983-84)	442	1.67	378	22	Cooper and Thomsen (1988)
Whatawhata, New Zealand	Mangaotama – steep hill land with pasture grazed by sheep receiving 21 kg P ha^{-1} plus drainage from a farmyard and woolshed[5]	2.6	4 (1995-00)	875	3.04	361	8	Quinn et al. (2007)
			5 (2000-05)	818	1.15	142	15	

[1]Probably total dissolved P (TDP). [2]Estimated. [3]TDP was 45%. [4]TDP was 70%. [5]Data are given for years monitored before (1995-2000) and after (2000-2005) a land management improvement programme. n.d., no data.

Lowland grassland catchments

Grassland occupies ca. 70% of the UK land area and is therefore an important land use with respect to P transfer in agricultural runoff. In the UK, grassland tends to be located on heavier-textured soils in wetter areas and generally receives more frequent applications of P in both fertilisers and manures than arable crops. Due to the lack of regular soil inversion, permanent grassland (>5 years old) soils also accumulate a high proportion of applied P within the surface 0-5 cm, which is the zone that interacts most with incoming rainfall and resulting overland flow (Ahuja et al., 1986; Haygarth et al., 1998b). Phosphorus exports from a range of small to medium sized catchments dominated by grassland are given in Table 2.5.

Export of TP varies widely (0.2-6 kg ha^{-1}) depending on soil type and land management, with typically 40-60% in dissolved form. Flow-weighted concentrations are equally variable (0.04 – 1 mg L^{-1}). For example, two contrasting catchments with dairy herds in Devon (Denbrook and Drewston) showed an order of magnitude difference in P export (0.6 – 6 kg ha^{-1}) and flow-weighted concentrations (0.01 v 0.1 mg L^{-1}) despite similar discharge volumes (Haygarth, unpublished; Ulén et al., 2007). The catchment stream with the highest P export was on clay soil (50 overland flow events), and received runoff from both a forage maize field (i.e. cultivated crop receiving large amounts of manure) and a farmyard. The catchment stream with the lower P export was on sandy soil (only nine overland flow events during the same period), no forage maize and no direct connectivity between the farmyard and the stream.

The importance of catchment geology and soil type is also clearly evident in a comparison of P export from three grassland catchments in Ireland with similar intensity of agricultural practices (Jordan et al., 2005). Monitoring of these catchments was carried out over one year at a range of scales with results for the largest scale presented in Table 2.5. Flow-weighted TP concentrations were greatest in the stream draining the poorly drained clay catchment (Oona) due to the predominance of overland flow, and least from the free-draining calcareous catchment (Clarianna) due to the co-precipitation of the dissolved P component with calcite in the water column. The catchment with the sandy and loamy soils produced intermediate P concentrations.

Table 2.5 Some examples of export and flow-weighted (FW) concentrations of total P (TP) and dissolved P (DRP or TDP) in stream flow from lowland grass.

Land use Catchment	Characteristics	Size (km^2)	Monitoring period (yrs)	Flow (mm)	TP export (kg ha^{-1} yr^{-1})	FWTP (mg L^{-1})	FWDRP (% of TP)	Reference
Denbrook, Devon, UK	Dairy farming on clay soils, 64 kg P ha^{-1}, 2.5 LU ha^{-1}, 54 mg kg^{-1} Olsen-P	0.48	2 (2001-03)	610	5.9	0.967	39[1]	Haygarth (unpublished)
Drewston, Devon, UK	Beef and sheep farming on sandy soils, 16 kg P ha^{-1}, 1.75 LU ha^{-1}, 49 mg kg^{-1} Olsen-P	0.22	2 (2001-03)	639	0.6	0.094	53[1]	Haygarth (unpublished)
Dripsey, Ireland	Dairy beef and sheep farming on brown podzolic soils over Old Red sandstone	14	1 (2002)	1037	1.6	0.154	63[1]	Jordan et al. (2005b)
Oona, N. Ireland	Dairy, beef and sheep farming on tile drained glacial clay soils	85	1 (2002)	817	3.13	0.383	47[1]	Jordan et al. (2005b) Douglas et al. (2007)
Clarianna, Ireland	Dairy, beef and sheep farming on free draining limestone soils	13.6	1 (2002)	416	0.17	0.041	47[1]	Jordan et al. (2005b)
Toenepi stream, New Zealand	Dairy farming in 70% of catchment area plus inputs of dairy shed wastewater	15.1	2 (1995-97)	602	1.16	0.193	53	Wilcock et al. (1999)
Toenepi stream, New Zealand	Same catchment but following voluntary implementation of environmental stewardship	15.1	2 (2002-04)	366	0.67	0.183	46	Wilcock et al. (2006)

[1]TDP.

Whilst plot scale work has shown TP export from Irish grassland is dominated by DRP (Tunney et al., 2000; Kurtz et al., 2005), the data for the Oona catchment showed that the majority of the P export at all scales was in particulate form (58-67%). Douglas et al. (2007) suggest this may be de to immobilisation by stream sediments as well as the inherently greater risk of soil erosion and particulate P (PP) loss during grazing on clayey soils.

Phosphorus exports from a range of grassland catchments in New Zealand with beef, sheep and dairy farming and receiving between 4 and 80 kg P ha^{-1} have been recently reviewed by Gillingham and Thorrold (2000) and McDowell et al. (2005). Values for TP range from 0.7-8.5 kg ha^{-1} yr^{-1} with highest P exports associated with dairy farming, which is considered to be a major cause of poor water quality in regions which have seen a 10-fold expansion in cow numbers in recent years. Data from one catchment study (Toenepi stream) before and after voluntary implementation of environmental stewardship are given in Table 2.5. Peak P concentrations in the stream were associated with discharges from farm ponds used to treat dairy wastes rather than direct land runoff. When this dairy waste was recycled to land rather than direct to the stream (via ponds), there were distinct improvements in water clarity and reduced faecal contamination, although stream TP concentrations only reduced slightly (Wilcock et al., 2006). In other grassland catchments in New Zealand, runoff P concentrations are influenced more by stocking intensities, fertiliser and manure applications and soil type (clays) (McDowell et al., 2005; Monaghan et al., 2007).

Lowland arable catchments

Phosphorus exports from a range of small to medium sized rural catchments in the UK and Scandinavia dominated by arable land are given in Table 2.6. Annual export of TP varies from <0.1-4.2 kg ha^{-1} with flow-weighted concentrations of 0.14-0.42 mg L^{-1}, except for the Rosemaund catchment in Herefordshire where land runoff is very turbid and values approach 1 mg L^{-1}. Not all the P export is dominated by particulate P, for example at Colworth where runoff reaches the stream via underdrainage systems and in Denmark, where soil P fertility is high on the sand/loam textured soils. Monitoring frequency does vary; for example, Kyllmar et al. (2006) report data for 27 small arable catchments where sampling was mostly on a fortnightly basis so storm event contributions may be underestimated. The range in TP export was relatively small (0.06-0.92 kg ha^{-1}) with flow-weighted concentrations of up to 0.3 mg L^{-1}. The range in P export is very similar to catchment data reported by others (e.g. Ryden et al., 1973) and very much lower than the large range in P export (up to 30 kg P ha^{-1} yr^{-1} or more) measured from field plots (Ryden et al., 1973; Catt et al., 1998)

It is not readily apparent from the literature what specific management actions on arable farms are responsible for the variation in flow-weighted TP concentrations measured. Clearly soil type (clayey soils) and slope (steeper slopes will generate more runoff) are important factors. For example, Kyllmar et al. (2006) recorded largest TP concentrations on clay soils and data for the UK and Norway which generally have steeper sloping land are greater than for Sweden and Denmark. However, these factors are outside the control of the farmer. Arable cultivations will both loosen soil for entrainment and provide a greater period of exposure to rainfall whilst crops become established. The tendency to over-cultivate in recent years to produce a fine seedbed for effective action of pesticides and the recent introduction of tramlines which tend to run up- and down slopes has increased the amounts and erosive force of runoff and soil susceptibility to particle detachment (Evans, 1990). As shown in Section 2.2, arable crops tend to receive larger amounts of P fertiliser than grassland and generally have larger OP concentrations which may increase both dissolved and particulate P concentration in surface and sub-surface runoff. Further work is required to understand the effects of these specific management actions on P transfer within small catchments in order to prioritise mitigation actions.

These catchment data comparisons show that flow-weighted P concentrations increase from values of <0.1 mg L^{-1} in catchments with forest and unimproved upland pasture to values ranging between 0.1 and 0.4 mg L^{-1} in farmed catchments and up to values of 1 mg L^{-1} in catchments with acute P concentrated discharges connecting directly with the stream (e.g. Denbrook), or particularly dispersive soils (e.g. Rosemaund). Some of this variation is due to environmental factors (soil type and slope). However, the P concentrations measured in farmed catchments are well above those considered necessary to achieve good ecological quality and therefore pose a eutrophication threat. The challenge is therefore to identify which particular sources, farming practices and hydrological pathways are contributing P in rural catchments and how it is delivered to the watercourse. This requires knowledge of the factors controlling the mobilisation and transport of P and the processes involved.

Table 2.6 Some examples of export and flow-weighted (FW) concentrations of total P (TP) and total dissolved P (TDP) in stream flow from lowland arable catchments. Range values are given in parenthesis for long-term databases.

Catchment location	Characteristics	Size (km^2)	Monitoring period (yrs)	Flow (mm)	TP export (kg ha^{-1} yr^{-1})	FWTP (μg L^{-1})	FWTDP (% of TP)	Reference
Rosemaund. Herefordshire, UK	Jubilee station, 61% arable, silty clay soils over Old Red Sandstone	0.03	6 (1994-00)	282 (116-344)	2.3 (0.2-4.2)	0.81 (0.2-1.24)	32 (21-67)	Hodgkinson and Withers (2007)
Cliftonthorpe, Leicestershire, UK	Upper 2 station, 83% arable, silty loam soils over	0.05	2 (1998-00)	498	1.6	0.31	30	Hodgkinson and Withers (2007)
Colworth, Leicestershire, UK	Underdrained calcareous clay soils on boulder clay, intensive arable cropping	1.0	1.7 (1999-01)	707[1]	1.7[1]	0.24	54	Pepper and Groves (2003)
Whittle Dene, Northumberland, UK	Mixed arable (55%)and livestock on drained sandy clay soils (F2 station)	2.3	3 (2003-06)	392	0.94	0.22	29	ADAS (unpublished)
Sweden	18 rural catchments with >60% arable land.	1.8-16.8	6-18	295 (142-455)	0.4 (0.06-0.92)	0.14 (0.03-0.33)	n.d.	Kyllmar et al. (2006)
Finland	Hovi, 100% arable land	0.12	10 (1981-90)	297	1.3	0.42	n.d.	Rekolainen et al. (1995)
	Loytaneenoja, 68% arable land	5.64	10 (1981-90)	326	0.7	0.21	n.d.	Rekolainen et al. (1995)
Denmark	23 rural catchments with >60% arable land	3.7-57.8	1 (2000)	270 (93-510)	0.4 (0.15-0.94)	0.16 (0.09-0.35)	64 (41-86)	Andersen et al. (2005)
Norway	8 rural catchments with 40-85% arable land	1.1-87.1	10-20	633 (269-1285)	1.39 (0.3-2.6)	0.23 (0.09-0.37)	n.d.	Ulen et al. (2007)

[1]total over the 20 months of monitoring. n.d., no data.

2.3.2 Hydrological controls

Pionki et al. (1997) viewed a catchment 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 importance of the hydrological linkages between terrestrial and aquatic environments for P transfer is now well established and provides the basis for identification of contributing areas within catchments and therefore effective mitigation. In this review, the terminology adopted by Ryden et al. (1973) is used to define the different hydrological pathways that deliver P from the field to the watercourse. *Runoff* refers to that part of precipitation which falls onto land and ultimately appears in surface streams, rivers and lakes. *Surface runoff* refers to the overland flow component and *sub-surface runoff* refers to the component that infiltrates the soil and moves to the stream above the main groundwater level in timescales ranging from minutes to days (Haygarth et al., 2000). In many agricultural soils, artificial drainage systems (tile, plastic and mole drains) intercept sub-surface runoff greatly reducing these transit times.

Surface runoff has been further separated into *infiltration-excess runoff* (Hortonian) and *saturated-excess runoff* (Pilgrim et al., 1978). Infiltration-excess runoff occurs when rainfall intensity exceeds the infiltration capacity of the soil, and saturated-excess runoff occurs when the surface layers of the soil become saturated, either due to a slowly rising water-table or to a perched water-table above an impermeable subsoil. *Groundwater runoff* is that component of precipitation that percolates slowly (weeks and months) through the soil matrix and provides the bulk of low flows (baseflow) in streams during summer months. Runoff can be separated from baseflow on a storm hydrograph by a distinct increase in flow rate (Dingman, 1994). In contrast to N, which continues to be transported in baseflow during periods between storms, the transport of the P stored within the terrestrial environment occurs largely in surface and sub-surface runoff (Pionki et al., 2000). Slower percolation of P through the soil matrix can reach shallow groundwater in some areas with P-saturated soils (e.g. Smith et al., 1998; Chardon and Schoumans, 2007), but this is not the major transport pathway in most catchments in the UK.

Differences in rainfall intensity and duration, antecedent soil moisture and routing of runoff through the landscape gives rise to large temporal and spatially variability in the occurrence and rates of surface and sub-surface runoff (Pilgrim et al., 1978). Hortonian runoff is dependent on the variation in the infiltration capacity at the land surface and therefore occurs only in small source areas, termed 'partial areas' and during the more intense storms. Modern farming practices are considered to have reduced soil infiltration capacity through an increased occurrence of soil compaction caused by cultivation and livestock poaching (Heathwaite et al., 1990; Harris et al., 1993). Occurrence of infiltration-excess runoff is therefore highly sensitive to land use and land management. Impervious surfaces around farm buildings and along farm tracks and roads clearly have little or no infiltration capacity and therefore contribute the bulk of infiltration excess runoff in catchments.

Saturation-excess flow typically occurs in areas near the stream, or in hollows, where water tables are closer to the surface and on heavier-textured soils where impermeable clay subsoils restrict water percolation depth causing the soil surface to become saturated even under low intensity storms. Near-stream areas showing saturated-excess runoff expand outwards from the stream in a dynamic spatial and temporal pattern depending on hillslope topography and storm duration. The distribution of slowly permeable soils rather than land use is the primary factor affecting the extent of saturation-excess runoff within catchments (Smith, 1987; Needleman et al., 2004). Needleman et al. (2004) found that the antecedent moisture content of the soil influenced the occurrence of surface runoff under low-intensity storms but not under high intensity storms. Variability in sub-surface runoff relates to the presence of root channels, worm burrows, holes, cracks and fissures in the soil and the presence of impermeable subsoil layers or tile drains which direct the runoff laterally. Sub-surface runoff generally has a longer residence time in the soil allowing nutrient diffusion into the percolating water. Solute levels, or conductance, have therefore been used to separate out the relative contributions of the surface and subsurface runoff (Pilgrim et al., 1978; Haygarth et al., 2000).

Surface runoff therefore occurs from only limited and variable source areas (VSAs) which expand and contract within and between events and the hydrological connectivity between these VSAs and the stream determines the extent to which P export from the catchment is transport limited (Ward, 1984; Gburek et al., 2002). The linkage between VSAs and P sources in catchments further established the concept of Critical Source Areas (CSAs) which hypothesized that the majority of P export from a catchment will occur from only a limited land area where VSAs coincide with source P release (Gburek and Sharpley, 1998; Heathwaite et al., 2000). The CSA concept revolutionised catchment management since P loss mitigation actions needed to be targeted at CSAs rather than across the whole catchment, and provided a fundamental contrast with the nature of N export (Table 2.7).

Table 2.7 Controls on algal-available phosphorus and nitrogen export from the WE-38 watershed, Pennsylvania. (Adapted from Pionki et al., 2000).

Controls	Algal-available phosphorus	Nitrogen
Process	Mostly in surface runoff (90%) and in large part particulate (50-75%)	Mostly in sub-surface runoff as nitrate (70-90%)
Spatial	Primary sources of export (90%) are small in area (10%) and predictable	N balance and land use distribution in the catchment
Temporal	Most export (90%) by stormflow (10%) and mostly (70%) during a few storms (5/7) in late winter/spring	Little to none in stormflow but 70% occurs in winter-spring

Whilst VSA and CSA theory suggests that only near-stream areas will contribute surface runoff P to streams, the potential transport of P from more distant areas has been increased in recent decades through features such as artificial land drainage systems, tramlines introduced to improve spreading accuracy of agrochemicals and reduce trafficking to controlled areas, and urban infrastructures (roads, highways, paving of gardens) which act as conduits or connectivity channels for surface-runoff. Sources of P entering water may therefore originate some distance from the waterbody; Harrod and Theurer (2002) suggested by up to 1 km for sediment-bound P. Source areas within catchments will therefore be highly variable requiring knowledge of not only hydrological controls, but also the risk of sediment and P mobilisation from soils by detachment within a given flow pathway.

2.3.3 Source controls

Soil phosphorus

Ahuja et al. (1986) showed that rainwater interacts with only a shallow layer of soil (2-5 cm) and any P enrichment of surface runoff that occurs will therefore be dependent on the sources of available P in, or on, this shallow layer. In undisturbed soils receiving regular P inputs, large gradients in OP have been measured from the surface downwards (e.g. Haygarth et al., 1998b; Karlen et al., 1991). Increased topsoil P accumulation can also increase sub-surface P losses where there is a preferential flow pathway (macropores, cracks and fissures) from the surface

(Stamm et al., 1988; Schelde et al., 2006). In the absence of preferential pathways, inorganic P migrating in solution downwards through the soil profile would be readily adsorbed by the subsoil. Shepherd and Withers (1999) concluded that exceptionally large amounts of cumulative surplus P were needed to cause subsoil enrichment and leaching of inorganic P from uniformly textured soils. However, leaching of P has clearly occurred in certain areas with very sandy or peaty soils of low P retention capacity; for example in Europe's low-lying deltas (Chardon and Schoumans, 2007). Organic P compounds in solution have been shown to migrate through the soil more quickly than inorganic P due to their size-charge characteristics and weaker adsorption strengths (Rolston et al., 1975; Chardon et al., 1997). Chardon et al. (2007) recently found evidence of such P migration through the soil below large dung pats in grazed fields.

Transfer of soil particulate P

Soil particle detachment occurs through a dynamic combination of rainsplash energy, the dispersion forces created by changes in ionic strength of the soil solution, the slaking forces of wetting fronts as they advance through the soil, and the shear forces of overland flow (Sharma et al., 1995; Bryan, 2000). Largest sediment loads in land runoff occur due to rill and gully erosion during intense storms. However, more environmentally significant chronic transfers of soil particles (up to 1 t SS ha^{-1} a^{-1}) also occur in shallow unconcentrated surface flow (termed sheet runoff) and in drainflow from arable and grassland fields (Fraser et al., 1999; Bilotta et al., 2007; Schelde et al., 2006). For example, James and Alexander (1998) found that sheet runoff was responsible for the transport of fine soil material disturbed by sheep grazing at very low rainfall intensities (< 2 mm hr^{-1}).

The processes of particle detachment, settling out during transit and retention by natural or artificial barriers in the landscape (e.g. hedges; riparian buffer zones) generally leads to the selective transport of fine silt-sized aggregates and primary particles (Foster et al., 1985; Wan and El Swaify, 1998). Evans (1992) found that a large proportion of the soil eroded from a clay catchment was exported from the field, whilst in a sandy catchment the majority of soil remained in the catchment. Using ^{137}Cs fingerprinting techniques, Quine and Walling (1991) report within-field retention rates of 40-70%, suggesting only the finer particles reach the stream. Hence, Walling et al. (2000) found that 95% of fluvially transported particles are <63 μm in size, with median diameters closer to 12 μm and remaining fairly constant over a range of flow regimes.

These finer-textured aggregates and particles show varying degrees of P enrichment compared to the original soils depending on the soil type, the history of P inputs in fertilisers and manures and the range in particle size detached (Sharpley, 1980; Quinton et al., 2001). Where rill and gully erosion delivers coarser-textured particles and aggregates to streams during extreme events, the degree of P enrichment is reduced. There has been surprisingly little research conducted on the effects of soil P on the particulate P fraction transported in runoff, other than to demonstrate that soil P preferentially accumulates on the finer soil particles (e.g. Maguire et al., 1998; Scalenghe et al., 2007).

Surveys of rill and gully erosion by water during the 1980's left little doubt that modern methods of crop and soil management have increased the vulnerability of field soils to erosion and increased the amounts of soil transported across fields during storm events (Spiers and Frost, 1985; Evans, 1990; Chambers et al. 2000). In particular, the increase in the area sown to winter cereals, the introduction of tramlines which concentrate and increase the velocity of water flow, the removal of hedgerows which increases the length of slope, and the reduced soil stability arising from continuous cultivation are major contributing factors. Late-drilled winter cereal crops often fail to provide the groundcover necessary to protect the soil from raindrop impact (25-30%) and in a survey of 80 fields susceptible to erosion monitored during 1989-1994, 80% of the erosion events were on land cropped to winter cereals (Fig. 2.5).

Since 1945, the area of rotational grassland in the UK has declined by 40% in favour of continuously cultivated crops, largely winter cereals. Other susceptible land uses include outdoor pig production, potatoes, sugar beet and forage maize (Chambers et al., 2000). In upland areas, significant amounts of sediment are also transported along drainage dykes installed to improve soil drainage status prior to afforestation (Nisbet, 2001). Increased grazing pressure associated with agricultural intensification in both upland and lowland areas appears to be an important factor in high rainfall areas where PP is transported in surface and sub-surface flow from grassland fields (Sharpley and Smith, 1979; Heathwaite et al., 1990; McDowell et al., 2003).

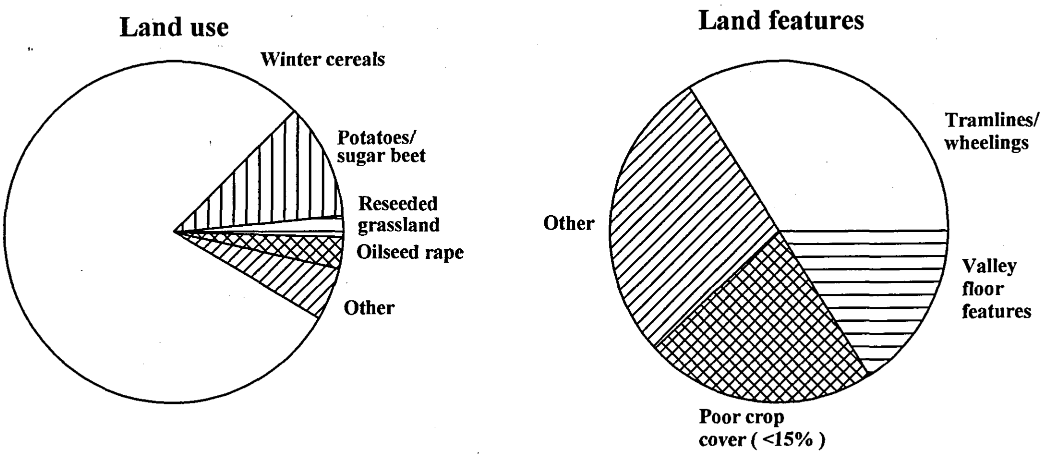


Figure 2.5 Major factors influencing water erosion on farms in England and Wales. Adapted from Chambers et al. (2000).

Transfer of soil dissolved P

The dissolved P (<0.45 μm) fraction in land runoff is conventionally separated into a molybdate-reactive inorganic portion (dissolved reactive P, DRP) and a molybdate-unreactive portion (dissolved unreactive P, DUP) which is composed of organic P and condensed P (Jarvie et al., 2002). Whilst DRP diffuses in and out of solution at rates depending on the concentration gradient between soil and solution phase (a dynamic equilibrium) and hence affected by soil:solution ratios, DUP is controlled more by soil microbial and microfloral activity and typically becomes the main form of P in organic or peaty soils and upland farming systems (Shand et al., 1994; Turner, 2005). In well fertilised soils, the DRP fraction is usually dominant and most research has been conducted on predicting DRP concentrations in runoff following the work of Heckrath et al. (1995) which showed that DRP concentrations in drainflow from field plots increased rapidly above a change-point of 60 mg kg^{-1} OP, Fig. 2.6. In their experiment, DRP constituted about 80% of total P in the runoff, giving the misleading impression that soil available P can predict total P concentrations in runoff.

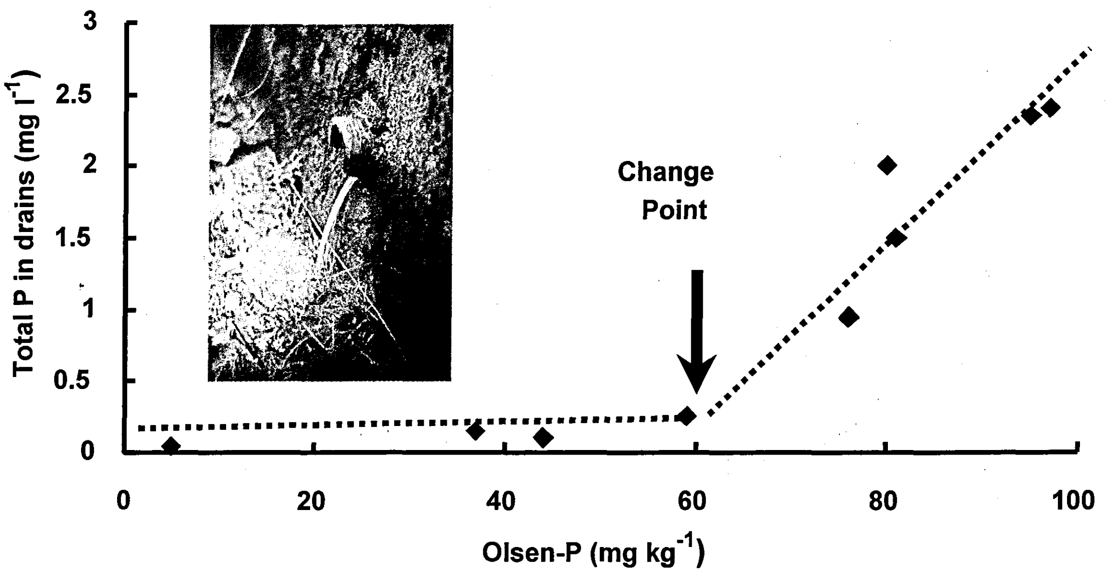


Figure 2.6 Concentrations of P in drainflow from Broadbalk plots at Rothamsted increase sharply above a change-point of 60 mg kg^{-1} Olsen-P. DRP was 80% of TP in this experiment. (Supplied by Dr K. Goulding, Rothamsted by kind permission).

Highly significant linear and non-linear relationships between extractable soil (labile) P and DRP concentrations in runoff have been obtained in subsequent studies depending on soil type, soil:solution ratio, the range in STP levels tested, the particular reagent used and the contact time with the soil (Sharpley, 1995; Koopmans et al., 2002). Relationships are often linear under simulated rainfall, which represents short duration (typically 30 minutes), high intensity precipitation and rapid runoff, and low ionic strength conditions. Under natural conditions and low intensity storms, or where there is a greater opportunity for P equilibration with the soil (e.g. sub-surface flow), then the relationship equates more to a non linear adsorption isotherm. To overcome soil specificity, various indices of soil P saturation and their relationship to runoff DRP, or surrogates for runoff DRP (such as water-extractable P, iron-oxide strip P, $CaCl_2$-P), have been investigated (Hooda et al., 2000; Kleinman et al., 2000; Daly et al., 2001). These relationships are highly dependent on the methodology used and must first be calibrated with runoff P to establish threshold saturation values, or guideline saturation values for a particular target DRP concentration in the runoff water (Casson et al., 2006).

Farming practices which increase concentrations of DRP and DUP in surface, subsurface and leachate from soils are clearly related to nutrient management. As outlined above, rates of soil P accumulation vary widely depending on the amounts of surplus P applied and the depth over which the surplus is distributed. The recycling of organic manures is a particular issue because farmers have little confidence in their nutrient availability and often ignore their nutrient value when calculating fertiliser requirements (e.g. Domburg et al., 2000). Addition of manures also has other implications for P mobility in soils for both organic and inorganic components. Chardon et al. (1997) found significant movement of DUP through sandy soils receiving pig slurry. Regular additions of manure also raise soil pH and soil exchangeable Ca causing an increase in the proportion of Ca-P reaction products relative to Al and Fe bound P in soils. Sharpley et al. (2004) found that this shift reduced the amounts of water-extractable P in manure-amended soils and suggested that some standard soil tests that dissolve Ca-P precipitates during the extraction would overestimate DRP in runoff waters. Similarly, regular application of municipal biosolids and other waste materials which contain P binding elements (Fe/Al/Ca) have been shown to reduce the release of soil inorganic P to runoff water (Penn and Sims, 2002; Elliot et al., 2005). The potential for these waste materials for reducing P leaching in P-saturated soils is now being investigated (Agyin-Birikorang et al., 2007).

Fertilisers and faeces

Fertilisers and livestock faeces are regularly applied to fields and provide additional sources of P for mobilisation when it rains, greatly increasing the range in DRP, DUP and PP concentrations present in the runoff. Phosphorus release from spread manures is a particular concern because the large volumes produced on farms encourage their application at inappropriate rates, or timings, depending on available land area and manure storage facilities. Large application left on the surface of wet, frozen, compacted and intensively underdrained soils in high rainfall areas are particularly vulnerable to P transfer. On grassland, slurry is often spread repeatedly throughout the year without incorporation into the soil. On arable farms, large single fertiliser applications which cover the P requirements of succeeding crops in the rotation and top-dressed after sowing have become more common (Chalmers et al., 1999). The risk of P loss following application of fertilisers and manures is therefore highly management sensitive and in principle can be controlled more easily.

Most research has been undertaken at the small box/plot scale where the processes of mobilisation and the controlling factors can be investigated more easily (Kleinman et al., 2004). The large variability in runoff P following fertiliser or manure application reflects source P release properties (e.g. P content and whether in granule, biosolid or slurry form), spatial distribution (e.g. dung patches from grazing livestock or more uniformly spread housed manure) and exposure time at the surface (e.g. time to incorporation of spread manure or time animals are kept on concrete around farm buildings). These sources may be considered as separate to the soil since P mobilisation only occurs where and when they are present and have been termed 'incidental P' by Haygarth and Jarvis (1999). Fresh applications of fertiliser, housed manure or slurry, and P excreted during grazing influence runoff P through a combination of (a) effects on runoff volume, (b) detachment and transport of particulate material (fertiliser and manure solids) and (c) release of P into solution during runoff (Preedy et al., 2001; Smith et al., 2001; Kleinman et al., 2002; Owens and Shipitalo, 2005). The relative importance of these processes depends on the type of material applied, the rate, timing and method of application, the frequency and timing of rainfall events after application and the rate of water movement across, or through, the land surface to the watercourse.

The amounts of P in surface and sub-surface flow following application of fertilisers and manures to field plots have been widely reported in the literature, usually expressed as a percentage of the amount of total P applied. Values are typically below 5%, but ranging to over 20% in unfavourable conditions (Khaleel et al., 1980; Sharpley et al., 2001; Hart et al., 2004), but the range in concentrations is very large (<1–100 mg L^{-1}). Some recent examples are shown in Table 2.8. Concentrations can, however, remain high for several weeks, or even months after application when incorporated into the soil (Catt et al., 1998; Ulén and Mattsson, 2002). Most of the released P is in DRP form and can be predicted by the amount of water-extractable P (WEP) in the material spread. For example, Kleinman et al. (2005) reports values in the range 0.2-16.8 mg kg^{-1} in a survey of 140 manures with largest values for pig manure (9.2 g kg^{-1}) and lowest for beef cattle manure (2.3 g kg^{-1}). Limited experimental data for manures in England show values of <10% of total P for treated sewage sludge, 15-30% for solid manures and up to 60% for dilute livestock slurries (Withers et al., 2001; Hodgkinson et al., 2002).

Table 2.8 Some recent examples of losses of total P (TP) and total dissolved P (TDP) in overland flow following land application of inorganic P fertilisers as compared to livestock manures.

Experimental Conditions	Treatments	Amount of P applied kg P ha^{-1}	P export: Flow (mm)	TP loss g P ha^{-1}	TDP loss (% of TP)	FWTP conc. mg l^{-1}	P loss (% of P applied)	Reference
Rosemaund, Herefordshire, U.K.	Control (Period 2) - dry winter	0	5.4	32	22	0.6		Withers et al. (2001)
Winter cereals, silty soil, 5° slope,	TSP, incorporated	90	5.0	35	40	0.7	<0.1	
16 m * 2 m plots, cumulative loss in	Cow slurry, incorporated	40	5.1	42	42	0.8	<0.1	
overland flow from natural rainfall								
over 2 winter periods,	Control (Period 3) – wet winter	0	4.2	109	17	2.6		
	TSP, surface-applied	90	5.7	1332	86	23	1.4	
	Cow slurry, surface-applied	49	12.9	2632	47	21	5.1	
Okehampton,Devon, U.K.								
Grassland, clay soil, 5° slope,	Control	0	48	60	36	0.1		Preedy et al. (2001)
16 m * 2 m plots, interflow from	TSP	29	48	1863	74	3.9	6.2	
natural rainfall over 7 days, 7 mg kg^{-1}	Liquid cow slurry (50 m^3 ha^{-1})	29	48	1804	45	3.8	6.0	
Olsen-P. Wet soil conditions	TSP+ cow slurry	29	48	2297	82	4.8	7.7	
Akansas, U.S.A								
Silt loam soil, 5° slope, overland flow	Control	0	7.6	1210	55	16		Edwards and Daniel (1994)
from 4 simulated storm events, 50	Poultry manure	87	7.6	1980	75	26	0.9	
Mm hr^{-1} until runoff	TSP	87	7.6	2680	94	35	1.7	
Pennsylvania, USA								
Mean of three soils, 2 m * 1 m plots,	Control	0	77	320	72	0.4		Shigaki et al., (2007)
50 mm hr^{-1} for 30 min, overland	Triplesuperphosphate	100	91	22710	92	25	22	
flow cumulative loss over 1, 7, 21	Single superphosphate	100	100	13420	94	13	13	
and 42 days.	Rock phosphate	100	79	740	61	0.9	<1	
	Swine manure	100	98	7090	83	7.2	7	
Australia								
Pasture, clay loam soil, 13% slope,	Single superphosphate	22	22	6600	90	30	30	Austin et al. (1996)
8 m * 30 m irrigation bays, overland	Single superphosphate	44	17.5	10100	90	58	23	
flow from 3 * successive irrigation	Single superphosphate	66	14.6	13200	90	90	20	
Events, 17 mg kg^{-1} Olsen-P	Single superphosphate	88	11.3	12300	90	109	14	

The largest P concentrations most frequently occur in the first storm event after application and usually decline rapidly to soil background concentrations within about 20–40 days. The magnitude of the initial P concentration is related to the amount of P applied and how soon rainfall occurs after application (Nash and Halliwell, 1999; Hart et al., 2004). More slowly available P sources maintain a more constant pattern of P release and cumulative P loss rates may therefore not differ greatly where these sources are left on the surface for some time (Smith et al., 2001; Shigaki et al., 2006). Where flow rates are high after heavy rain, whole manure particles as well as soluble P released from the manure can be transported in the runoff water with little difference between fertiliser/manure types (Heathwaite et al., 1990; Preedy et al., 2001).

Vadas (2006) found that about 20% of slurry solids and 40-65% of slurry TP and WEP infiltrated the soil after application rendering this less available for transport. However, the water contained in slurries can act as the principal vector for P and encourages P transport when alternative solid P inputs would not cause P loss, especially when applied to wet soils (e.g. Hodgkinson et al., 2002; Stamm et al., 2002). Slurries can also physically seal the soil surface causing increased runoff volumes and rates and hence P transport (Smith et al., 2001; Withers et al., 2001). Once incorporated into the soil, differences in the pattern of P release between different P amendments becomes less significant (Table 2.8). In simulated rainfall experiments, where soils were packed into boxes of varying slope length and manures were applied at one end only, P concentrations in surface runoff declined by 70-90% over a distance of 10 m due to dilution with runoff from the non-manured area (McDowell and Sharpley, 2002). Where fertilisers and faeces are applied some distance away from the watercourse, and there is no preferential hydrological link, these sources probably become less significant. Even where rapid hydrological linkages exist, the high P concentrations found in box/plot experiments field can be considerably dampened at the catchment scale due to the effect of dilution with runoff from areas not receiving these amendments.

Freshly excreted dung patches are more localised and concentrated sources of P, depending on stocking density, the type of grazing system and any tendency to defecate around field perimeters, feeding areas and water troughs. Data compiled by Whitehead (2000) suggests a dung patch from a grazing cow will contain an equivalent P loading of 350 kg P ha^{-1}, but that only about 6% of a grazed field would be actually covered by dung each year. Leaching of P occurs directly beneath dung patches, or preferential defecating areas (Watson et al., 2003; Chardon et al., 2007), but Khaleel et al. (1980) concluded that it was difficult to detect differences in near surface P losses at the field scale due to grazing activity. Runoff P concentrations are often higher when animals are grazing than when they are not, especially when soil conditions are wet (Sharpley and Syers, 1979; Jordan and Smith, 1985; McDowell et al., 2003). Direct deposition of faeces into streams also occurs where livestock have access causing a range of environmental issues and not just associated with P (Davies-Colley et al., 2004).

Runoff P concentrations in sub-surface runoff following P application are generally much smaller than in overland flow due to the greater opportunity for P adsorption in the subsoil (Culley et al., 1983; Sharpley et al., 2001). Tile drainage systems do, however, encourage rapid by-pass flow in soils, and provide a high degree of hydrological connectivity with the watercourse. Severe contamination of P in runoff through tile drains has been reported (Hooda et al., 1999; Hodgkinson et al., 2002; Schelde et al., 2006). At a demonstration site in England, tile drains in two adjacent fields (Fields G and H) on the same farm were continuously monitored over successive seasons, and water samples taken on a flow-proportional basis. Dilute slurry supplying 18 kg P ha^{-1} was applied to both fields on the 24 February 2000. When rain fell after application, the concentrations of DRP in the drainage water increased up to 6 mg P L^{-1} in Field G and up to 13 mg P L^{-1} in Field H (Fig. 2.7). Corresponding concentrations of total P were ca. 35 and 80 mg P L^{-1}, respectively.

The loss of P in the 7 days after slurry application (1.12 and 2.83 kg TP ha^{-1} in fields G and H, respectively), represented 64% of the total P loss measured over the entire drainage season in Field G, and 56% of total P loss in field H. The three-fold difference in P loss between the two fields shows the large spatial variability in loss that is likely to occur depending on site-specific conditions. In the previous year, the same amount of slurry P applied in March produced much smaller P losses (ca. 0.05 kg P ha^{-1}) in both fields, which was considered to be due to the lack of drainflow within 10 days of application. These data suggest ‘incidental’ P loss will be subject to very large temporal and spatial variability.

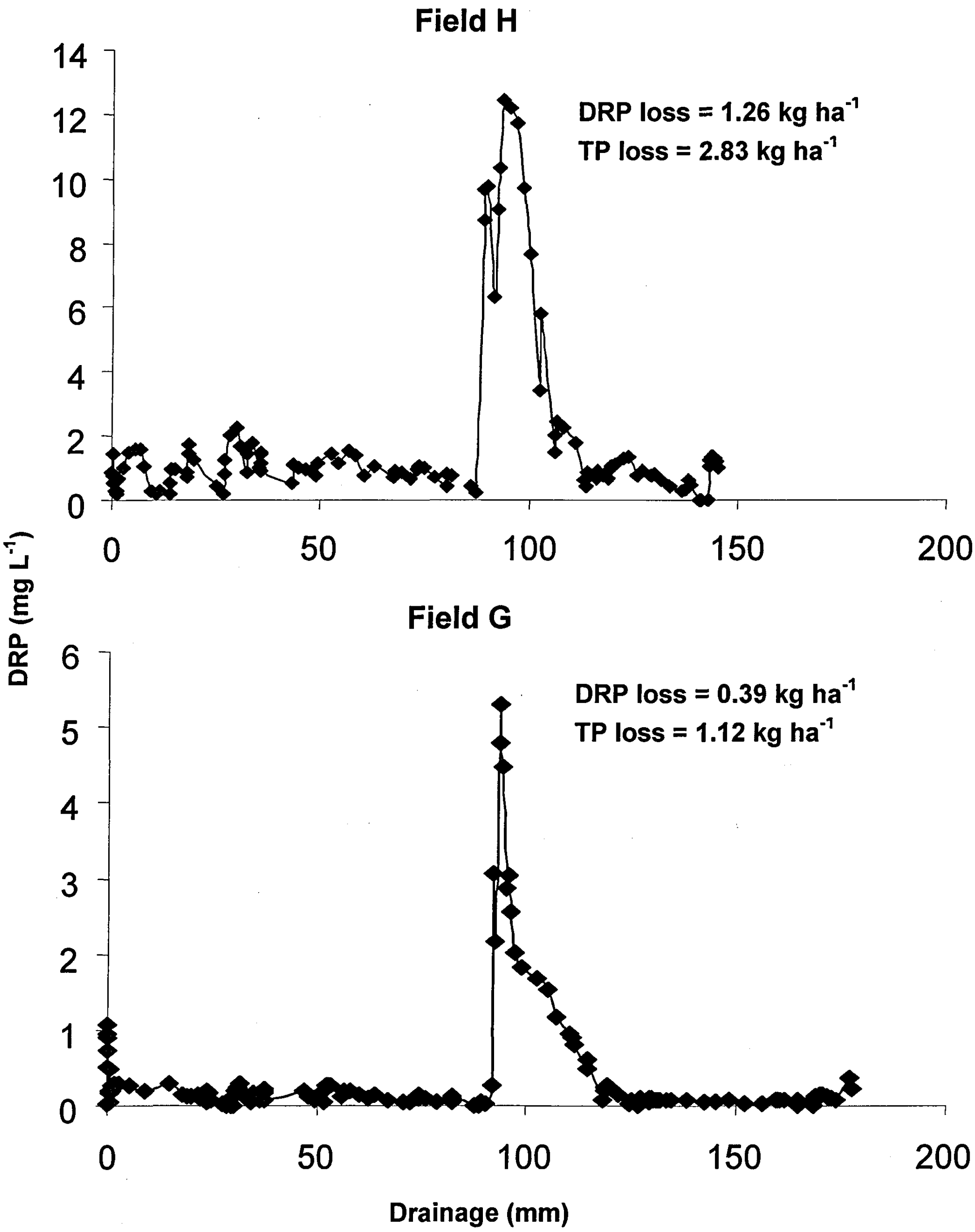


Figure 2.7 Changes in the concentration of dissolved reactive P (DRP) following a single application of separated slurry on two adjacent fields (Field G and Field H) in 2000 in England. The loads (kg ha^{-1}) of DRP and total P (TP) measured in the 7 days following application are also given.

Vegetation

There is relatively little information on vegetation effects on runoff P other than leaching of P from both crop canopies and senescing crop residues accounts at least in part for the seasonal variation in runoff P concentrations (Timmons et al., 1970; Sharpley, 1981; McDowell et al., 2007). In summarising studies in the USA, Pionke et al. (1997) reports runoff DRP concentrations of up to 0.3 mg L^{-1} from standing crops during rainfall whilst desiccating vegetation in riparian buffer strips may also be a source of DRP transport to streams (Uussi-Kamppa et al., 2000; Hodgkinson and Withers, 2007). Much of the 'phosphate' absorbed by plants is stored in plant cell vacuoles for future metabolic reactions within the cytoplasm (Mengel and Kirkby, 1987), and would be readily released as vegetation started to desiccate due to grazing/cutting or was subject to freeze-thaw cycles. McDowell et al. (2007) concluded that defoliation by grazed caused substantial release of DRP from pature plants accounting for 20% of the P export from grazed pasture. Further work is required to understand better the role of DRP loss from vegetation and the circumstances under which this might be transported to the watercourse.

Farmyards

Runoff and piped discharges from farmyards where animals congregate are also highly contaminated with both soluble and particulate phosphorus and have the potential to cause severe localised pollution problems (Dils and Heathwaite, 1996; Wilcocks et al., 1999; Edwards et al., 2007). Farmyard runoff may include parlour washings, runoff from silage and manure stores on concrete, livestock sheds, roof water and domestic wastewater (septic tank) discharges as well as open yard runoff. The range in P concentrations can be very large (Table 2.9). Edwards et al. (2007) monitored roof runoff and farmyard runoff from four dairy farms in Scotland during summer. Roof runoff was largely composed of particulate P (80% of total) and DUP (15% of total) and concentrations became diluted as stormflow increased. In contrast, farmyard runoff was largely composed of DRP and large concentrations were maintained throughout the storm. On one farm, mean TDP concentrations in roof runoff averaged nearly 1 mg L^{-1} and was thought to be due to contamination by bird droppings.

Edwards and Hooda (2007) measured up to 51 mg TP L^{-1} in flow from two farmyard drains (mean 1 and 6 mg L^{-1}) and estimated that the farmyard area contributed 25-30% of annual downstream P loads. These authors review similar contributions from studies in Ireland. Hively et al (2005) concluded that the relatively impervious surfaces on farms such as cow paths and farmyards maybe of minor spatial extent, but the highly concentrated runoff generated from these areas may be much more significant to summertime P loadings. Mean TP concentrations for cowpath and farmyards were 0.99 and 13.2 mg L^{-1}, respectively with associated concentrations of SS up to 500 mg L^{-1}. The relative significance of farmyard sources of P requires further investigation.

Table 2.9 Range, mean and median concentrations (mg L^{-1}) of P forms in runoff collected from roofs and yards on four farms in Scotland. After Edwards et al. (2007). The data for farmyards are compared to the range in P concentrations reviewed by Edwards and Withers, (2008).

	n	Range	Mean (s.e.)	Median
Roof runoff				
TP	10	0.101-3.298	0.767 (0.312)	0.365
TDP	10	0.005-2.468	0.295 (0.243)	0.0345
DRP	10	0.005-1.942	0.203 (0.193)	0.005
Farmyard runoff				
TP	17	0.022-2135	131 (125)	3.83
TDP	22	0.008-19.5	4.73 (1.07)	2.81
DRP	21	0.005-19.5	3.90 (1.09)	1.81
Wider review				
TP	33	0.02-247	31	n.d.
SS	33	690-1284	982	n.d.

n.d., no data.

Urban contributions

Urban storm water is a composite mixture of runoff from roads, roofs, residential driveways, parking lots, construction sites and gardens. The concentrations, forms and loads of P in urban runoff show large seasonal and storm variability with storm concentrations typically following a log-normal distribution. For example, Duncan (2005) found that the band representing plus and minus one standard deviation from the mean typically covered one order of magnitude for a range of urban pollutants. This variation is linked to both the multiple sources of P in urban areas and the differential routing and speed of flow due to the ubiquitous presence of various impervious surfaces and their slopes. Sources of P mobilised in urban runoff include atmospheric deposition, leaf fall, industrial wastes, urban litter, residential activities (e.g. car washing), livestock excreta, domestic and industrial fertilisers, soil particles deposited by vehicles and eroded from roadside verges, detergents and lubricants (Bannerman et al., 1993; Smullen and Cave, 1998). The extent of impervious surfaces greatly alters the proportion of rainfall occurring in overland flow and their proximity to urban P sources greatly influences the forms and concentrations of P mobilised.

Runoff from impervious surfaces is described and modelled using the paired concepts of build-up and wash-off (Duncan, 2005). Build-up is the accumulation of material (dirt) due to dry deposition between storm events whilst wash-off is the removal of this accumulated material by rainfall or street washing. Surfaces apparently rapidly accumulate dirt from a variety of sources (i.e. probably not source limiting), whilst the amounts removed in wash-off are largely a function of rainfall and flow energy. Mitchell (2001) reviewed the range in event mean concentrations (EMC) for 765 separate urban catchment data sources (71 in the UK, 171 in the rest of Europe, 434 in North America and 89 in Australasia and Japan) to develop a screening tool to identify significant pollutant loads from urban areas. The results relating to SS and P forms for various types of urban surface or land use are shown in Table 2.10.

As reported by Duncan (2005) and compiled data for the US (see Smullen and Cave, 1998), average mean EMC values for TP lay between 0.2-0.4 mg L^{-1} with about 40-50% in dissolved P form. Very similar values are reported for a range of highways in California by Kayhanian et al. (2007). Concentrations of TP appear to be slightly greater in runoff from residential areas and less from more open areas. Average concentrations of SS were more variable (50-200 mg L^{-1}), with highest values found for roads (Table 2.10). Duncan (2005) found that SS and TP concentrations in roof runoff (45 and 0.12 mg L^{-1}, respectively) were generally lower than from other urban areas, whilst urban roads tended to have a greater SS (but not TP) concentration.

In rural areas, the amount of impervious urban surface is more restricted and includes a wider variation in the degree of surface impermeability. Hence, major roads, village centres and scattered dwellings are supplemented by minor roads, lanes and a range of tracks; for example through forests and on farms. Concentration data for runoff from semi-impervious or impervious surfaces in rural areas are particularly sparse. Duncan (2005) reports lower concentrations of SS in runoff from rural roads (i.e. fewer vehicles) compared to urban roads, but similar concentrations of TP. However, sample numbers were very low (n = 6). Kayhanian et al. (2007) also found that SS concentrations were greater on roads with average annual daily traffic of >100,000 vehicles per day than on rural roads with <30,000 vehicles/day (159 vs 70 mg L^{-1}).

However, TP and DP concentrations were more influenced by the surrounding land use with significantly greater concentrations in agricultural areas. Roads are also recognised as a major source of SS transfer to watercourses in forested catchments during clearfelling (Nisbet, 2001; Forsythe et al., 2006). The contribution of urban areas to TP loads is rarely quantified separately in national or catchment source apportionment studies due to the lack of monitoring data and the difficulty of accurate representative sampling.

Table 2.10 Event mean concentrations of suspended solids (SS), total P (TP) and total dissolved P (TDP) in runoff from various impervious surfaces.

Impervious surface	SS ($mg\ L^{-1}$)	TP ($mg\ L^{-1}$)	TDP ($mg\ L^{-1}$)	Reference
Urban open	126 (57-280)[1]	0.22 (0.08-0.58)	0.06 (0.02-0.17)	Mitchell (2001)
Ind./comm.	50 (18-140)	0.30 (0.16-0.54)	0.16 (0.07-0.35)	
Residential	85 (38-193)	0.41 (0.24-0.72)	0.20 (0.11-0.36)	
Motorways	195 (110-344)	0.28 (0.15-0.52)	n.d.	
Main roads	157 (62-396)	0.34 (0.17-0.67)	0.18 (0.10-0.31)	
All highways	113 (1-2988)[2]	0.29 (0.03-4.7)	0.11 (0.01-2.4)	Kayhanian et al. (2007).
Non-urban[3]	70	0.2	0.1	
Urban[4]	159	0.3	0.1	
Forest roads				
Gravelled	307	0.06	n.d.	Forsyth et al. (2006)
Ungravelled	394	0.04	n.d.	

n.d. – no data. [1]inter-quartile range Q1-Q3. [2]full range. [3]Non-urban highways were classified as those with <30,000 vehicles per day. [4]>100,000 vehicles per day.

Wastewater effluents

Domestic and industrial wastewater effluents are discharged more or less continuously from STWs, or industrial premises, to surface waters after various degrees of treatment to remove toxic contaminants (metals and organics). Phosphorus in industrial effluents is derived from the manufacture of phosphoric acid, polyphosphate compounds and in metal finishing. Devey and Harkness (1973) report that trade effluent mean TP concentrations generally range from 1-6 mg L^{-1}, although concentrations of over 100 mg L^{-1} have been recorded. The effluent P is likely to be strongly complexed with metal cations and major industrial effluents are now linked up to a STW for treatment. Domestic wastewater effluent is comprised of both excretal (ca. 1.4 g $person^{-1}\ day^{-1}$) and detergent (ca. 1.3 $person^{-1}\ day^{-1}$) contributions (Department of the Environment, 1991), although an increasing proportion of detergents used in domestic homes are now P free. Hence Neal et al. (2005) found small STW effluent TP fluxes to sub-catchments of the Upper Thames river basin were equivalent to a daily per capita flux of only 1.45 g (0.53 kg person yr^{-1}). In larger STW, TP fluxes equivalent to 0.8-0.9 kg person yr^{-1} may be more appropriate (Smith et al., 2005).

After treatment, mean effluent TP concentrations discharged to watercourses are now generally in the range <1 to 10 mg L^{-1}, with lowest values recorded at the few major STW in eutrophic sensitive waters where tertiary treatment (P stripping) is carried out. However, most STW effluent without P stripping contains 4-10 mg L^{-1}. For example, Neal et al. (2005) found mean a TP concentration for 6 STW of 6.6 mg L^{-1} (range 0.2-17.1 mg L^{-1}) of which 91% was in DRP form, 6% in DUP form and 3% in PP form. The highly bioavailable nature of STW effluent is well recognised and a major concern for eutrophication due to its near continuous delivery through the year (Mainstone et al., 2000). Effluent P concentrations also show diurnal variations with peaks occurring at midday and during the evening (Devey and Harkness, 1973; Jordan et al., 2007). The concentrations and loads of P discharged from STW will therefore depend on the per capita population in the catchment area, the extent of industrial activity and the method of treatment adopted at the STW.

Rural catchments contain a range of smaller STW but also have a high proportion of the population that relies on septic tank systems for waste disposal. Potential P losses associated with correctly designed, erected and maintained septic tank systems that discharge to an adequate soakaway were considered by Gold and Sims (2001) to be negligible. However, many systems either discharge directly to the stream, are not emptied regularly enough or do not have adequate soakaway facilities and are thought to make a significant contribution to P concentrations in some rural catchments (Butler and Payne, 1995; May et al., 1998; Day, 2004). In many cases the precise number and distribution of septic tank systems is unknown. By matching up those addresses receiving water bills with the total number of postcode addresses in the Bassenthwaite catchment, May et al. (1998) estimated there were 1100 septic tanks in the catchment area supplying 18% of the TP load to the lake assuming a per capita P export coefficient of 0.7 kg yr^{-1}. Arnscheidt et al. (2007) report septic tank densities of 3-14 km^2 in Ireland.

More recent estimates of P export from rural populations connected to septic tanks are in the range 0.2-0.4 kg person^{-2} yr^{-1} (Foy and Lennox, 2000; Carvalho et al., 2004). In their review paper of potential treatment systems for septic tank effluent, Edwards and Withers (2008) report a range in TP concentrations of 1-22 mg L^{-1} (mean 10.2 mg L^{-1}). The efficiency of P removal by septic tanks is apparently quite low although data is very limited; Gold and Sims (2001) report a value of ca. 40% but Montangero and Belevi (2007) found values of 11-27% for systems in Vietnam. System tank design is clearly a factor, but septic tank effluents do appear to be of equal or greater potency than STW effluent with regard to P and hence of significance for eutrophication (Arnscheidt et al., 2007), despite the much smaller volumes of effluent discharged compared to a STW.

2.3.4 Aquatic sources of phosphorus

In addition to sources of P derived from the catchment area, concentrations of P in flowing and standing waters are also dependent on a number of 'internal' P sources mobilised by the physical, chemical and biological processes occurring within the water column (Fig. 2.8). These processes include the erosion of stream channel banks during high spates (Laubel et al., 2003 McDowell and Wilcock, 2007), remobilisation of P due to physical disturbance of bed sediments or macrophyte communities (Koski-Vahala and Hartikainen, 2001; Gainswin et al., 2006), release of P from bed sediments (Marsden, 1989; Jarvie et al., 2005) or decaying aquatic biota (Chambers and Prepas, 1994; House and Casey, 1989). The balance of these processes under different flow conditions determines the availability of P at any given time, and those relevant to slower flowing waters and standing waterbodies will determine the eutrophication impact together with other environmental factors (e.g. availability of other elements (C, N, Si), turbidity, shading and grazing pressures). The role of in-stream processes on the cycling and availability of P has recently been reviewed by House (2003), Edwards and Wetzel (2005), and Hilton et al. (2006).

The important contribution of channel bank erosion for transfer of SS has been highlighted in a number of catchment studies, with data for the UK suggesting a contribution of between 1 and 55% (Walling, 2005; Walling et al., 2008).

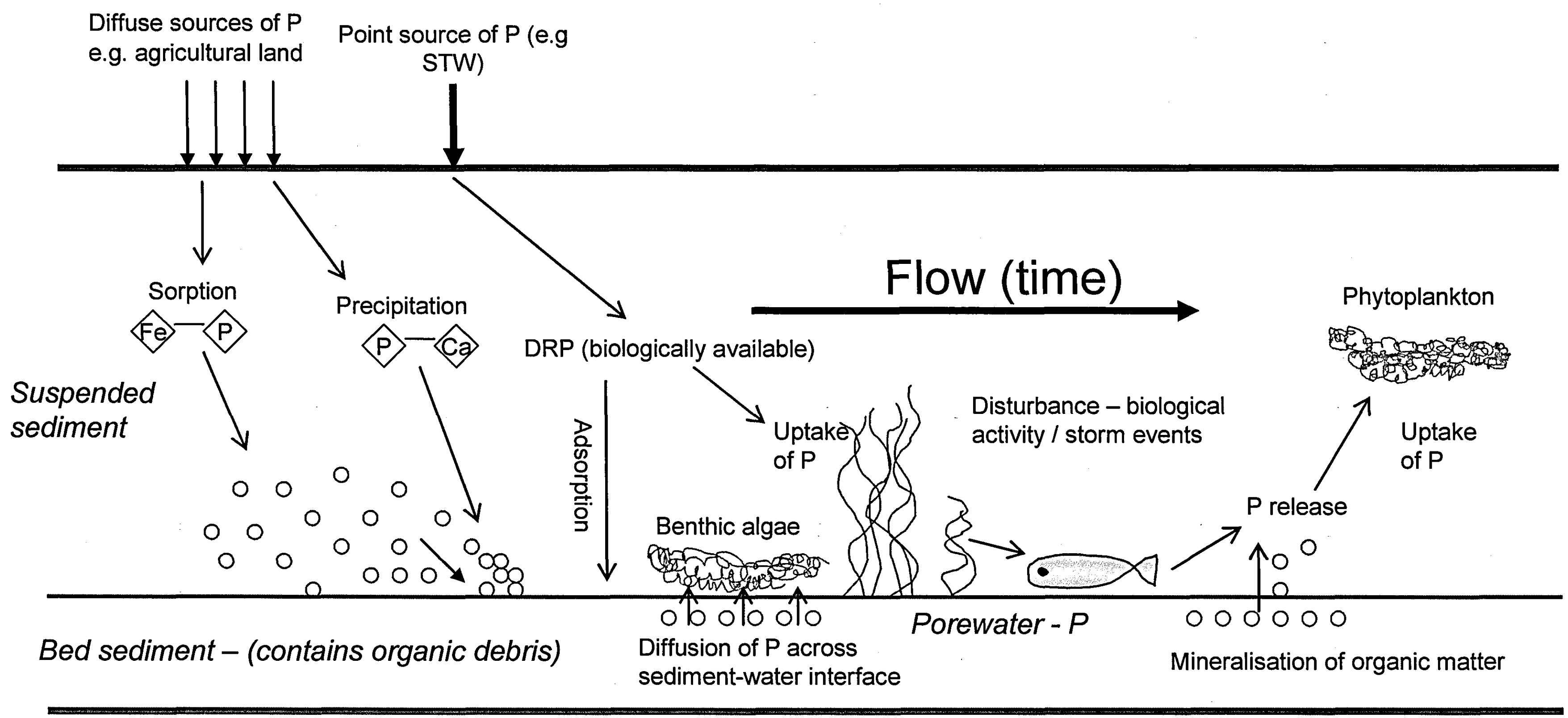


Figure 2.8 Conceptualised diagram of in-stream processes influencing P concentrations in flowing waters.

Recent estimates of the effects of stream bank mobilisation on P export in two major river basins (Wye and the Hampshire Avon) have also been recently published by Walling et al. (2008). Relative contributions to PP fluxes ranged from 5-54% on the Avon and from 21-43% in the Wye, with rates of transfer in the range <0.1-0.1 kg P ha^{-1} yr^{-1}. These values are lower than those measured in Danish streams (0.2-0.3 kg P ha^{-1} yr^{-1}) by Laubel et al. (2003). Comparison of PP and SS fluxes suggested that eroding banks associated with riparian pasture had a greater P content than those in cultivated areas, reflecting the greater surface accumulation of P in undisturbed pasture systems and excretal contributions from livestock. In a New Zealand catchment, river bank collapse due to trampling by dairy cows was a major source of P to the stream channel requiring bank fencing (McDowell and Wilcock, 2007).

Suspended and bottom sediments can rapidly adsorb P entering the watercolumn and release any accumulated P when water column concentrations decline (i.e. source is removed). This has been most frequently demonstrated downstream of a point STW source (e.g. Haggard et al., 2005). Changes in diffusion gradients across the sediment-water interface due to differences in the rate of flow, physical disturbance of bed sediments by macroinvertebrates and microbial activity/mineralisation of sediment OM all contribute to release of DRP and DUP into the watercolumn (Gachter and Meyer, 1993; Koski-Vahala and Hartikainen, 2001; House 2003). Suspended sediments may also be temporarily trapped within macrophytes stands and be remobilised during storm events or due to physical disturbance by fish and grazers (Sand-Jensen, 1998; Evans et al., 2004). Similarly, during active growth phases in spring, DRP is rapidly taken up by stream macrophytes and algal communities within relatively short distances (e.g. < 100 m) from entry into the watercolumn (Keup, 1968; Mulholland et al., 1990). Macrophyte productivity usually reaches a peak in late summer and autumn (Flynn et al., 2002) but P is recycled back into the water column at various rates before decay commences (House and Casey, 1989). Algal life cycles are shorter with P release providing a ready substrate for a succession of communities.

2.4 Conclusions

1. The regionalisation and intensification of arable and livestock farming systems has increased the range and spatial/temporal variability in P surpluses on farms, P

concentrations in field soils and the risk of dissolved and particulate P transfer in land runoff.

2. Phosphorus transfer occurs due to the mobilisation and transport along rapid surface and sub-surface flow pathways of dissolved and particulate P forms from a number of different sources. The hydrological linkages between sources and source areas in catchments and the watercourse is therefore critical to P delivery and has led to the concept of both source and transport limitation. Artificial drainage systems in fields are a major pathway of P delivery.

3. Much research suggests that the magnitude, form and timing of P transfer in land runoff from fields where flow is initiated is highly sensitive to management of P inputs in fertilisers and manures, rates of soil P accumulation and land (soil, crop and livestock) management. At the catchment scale, where flows are larger, the effects of individual field management operations become diluted by runoff from non-agricultural areas, or are masked by a greater variety of point sources downstream. Establishing a link between farming practices and stream P concentrations and loads in small rural catchments remains a high research priority.

4. A comparison of P export in small catchments with different land uses suggest annual P transfer rates of up to 8.5 kg TP ha^{-1} depending on annual flow, catchment characteristics (soil type and slopes) and land use/management. Annual flow-weighted concentrations of TP are typically below 0.1 mg L^{-1} in forested and unimproved upland areas and between 0.1 and 0.4 mg L^{-1} in farmed catchments, although concentrations up to 1 mg L^{-1} have been recorded in vulnerable areas or where runoff is contaminated by farm waste discharges.

5. Whilst some linkages between farming operations and catchment P transfers have been established in grassland catchments, there is a lack of information on (a) the effect of pasture improvement on stream P concentrations in upland regions and (b) the effects of farming practices on P transfer in lowland arable catchments. Both these aspects are investigated in this thesis.

6. Sources of P mobilised in land runoff other than those originating from agricultural fields include farmyards, roads and tracks and municipal/industrial effluent discharges, including those from small village STW and septic tank systems in rural areas. In many cases, specific sources have compositional and hydrological characteristics which are

intermediate between conventionally defined ‘point’ and ‘diffuse’ sources and may be more ecologically relevant than runoff from farmed land. The relative importance of these ‘intermediate’ sources in catchments has not been widely investigated and there is potential to misdiagnose the causes of P enrichment in rural areas with implications for catchment management. This aspect is investigated in this thesis.

3

EXPERIMENTAL APPROACH, SITES, TREATMENTS AND METHODS

3.1 Introduction

In this chapter, the experimental approach, treatment comparisons and general characteristics of the main study sites used to assess P transfer in land runoff associated with different farming systems are described. As far as was possible, common methodologies were used for site characterisation, runoff collection and sampling, stream monitoring and data handling and these are presented here rather than be repeated within individual chapters. Subsequent chapters will evaluate and interpret the data collected for each of the sites in relation to the hypotheses presented and the specific treatments imposed. Site data were largely comprised of individual farm information gathered by survey, historically available field and catchment site data from a range of sources and new site data generated as part of this thesis.

All the data presented were collected as part of laboratory, field and catchment-based projects undertaken by ADAS UK Limited (ADAS) for the Department for Environment, Food and Rural Affairs (Defra) under their 'Phosphorus (P) Loss from Agriculture' research programmes (NT10, PE01, PE02). For some of these projects, additional funding was also provided by the Environment Agency (EA), Natural England (NE) and/or the European Commission (EC). The amount and type of data collected was to some extent therefore constrained by the funding available within the individual projects. The ideas, experimental approaches, methodologies and data evaluation were primarily those of the author but equipment installation, treatment application and site sampling were undertaken with assistance from ADAS colleagues.

The chapter firstly summarises the broad experimental approach, main farming practices studied and data sources gathered, then describes the characteristics of the individual study sites and the specific treatments investigated, and finally presents the materials and methods used, including the analytical and statistical methods employed.

3.2 Experimental approach

The main objective of this thesis was to investigate how different farming practices influence the transfer of P in runoff from agricultural land, to appraise the significance of these transfers in relation to other rural sources of P contributing to river eutrophication and to assess potential mitigation options for their control. To provide a suitable framework to discuss the relative importance of agriculture as a source of P, data are presented for three study catchments (Redesdale, Rosemaund and Avon) that differ in their location (climate and lithology), farming systems (land use and farm practice) and environmental issues (Fig. 3.1).

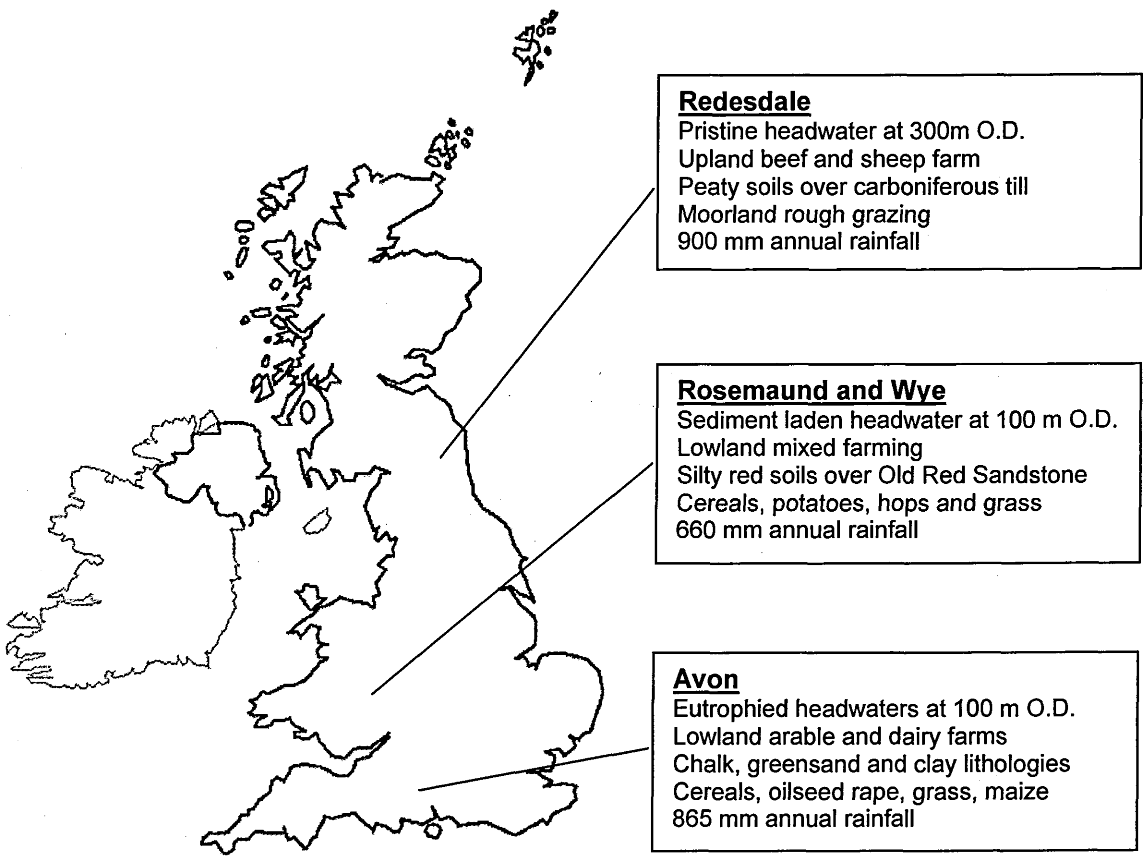


Figure 3.1 Location of main study catchments and their general site characteristics.

The review (Chapter 2) highlighted three main factors controlling the magnitude, form and timing of P transfer from arable and pasture land that are under the farmers control; P input management, soil P status and soil management. These controlling factors are investigated within the context of (a) upland grassland improvement of peaty soils in Northumberland, Northern England (Redesdale, Chapter 4), (b) cultivation of dispersive silty soils in

Herefordshire, Central England (Rosemaund, Chapters 5 and 6) and (c) selection of best management practices (BMPs) in a priority catchment in Wiltshire, Southern England (Avon, Chapter 8). To assess the relative importance of agricultural P transfer in relation to other rural sources of P, the range in P concentrations measured at Rosemaund were compared to those measured in storm runoff from a wider range of sources in three catchments (Whitchurch, Dinedor and Kivernoll) with similar lithology within the Wye river basin (Chapter 7).

Redesdale and Rosemaund are both small headwater catchments (up to 2 km^2), a scale at which any deleterious effects of farming practices on stream quality are most likely to be detected (Jarvie et al., 2006). In these two catchments, the spatial and temporal variation in stream suspended solids (SS) and P concentrations were examined in relation to catchment P inputs and the type of farming practised. The Avon is a much larger catchment (1706 km^2) whose headwater tributaries are impacted by a greater variety of point source P inputs that confounded any separate assessment of diffuse P inputs from agriculture. However, in association with the Environment Agency (EA), the opportunity was taken to compare at the field scale the effects of one soil management options (reduced cultivation) and two crop management option (early drilling and removal of tramlines) on P transfer in land runoff from the three main soil types in the catchment. The data from one of these three field sites (Pewsey) representative of erodible Upper Greensand soils were then compared to stream SS and P concentration data for two tributaries (East Avon, 86 km^2 and West Avon 85 km^2) draining this lithology after accounting for point source inputs. The Whitchurch, Dinedor and Kivernoll catchments in Herefordshire were slightly larger (<10 km^2) than the Rosemaund catchment to enable a wider range of potential sources to be sampled within a rural setting.

The mobilisation and/or delivery of P within each catchment were assessed at the soil profile, field and catchment scales. Field and catchment data were measured over either a two or a three-year period and evaluated across storm, seasonal and/or annual timescales. Phosphorus mobilisation at the soil profile scale was assessed at each site by a laboratory test that estimated the amounts of particulate-associated (>0.45 μm) and dissolved (<0.45 μm) P dispersed at the soil surface due to rainfall impact (Withers et al., 2007). Phosphorus transfer at the field scale was measured directly in surface runoff collected from small (typically 30 m^2) runoff plots (Rosemaund and Avon sites only), and/or from field drain outfalls (Rosemaund), under natural rainfall. At Redesdale and Rosemaund, continuous flow measurements and storm-triggered (automatic) event sampling supplemented by periodic, automated sampling during baseflow quantified P transfer at the catchment scale. In other catchments, daily automated sampling and/or weekly grab samples were combined with continuous flow to provide estimates of catchment P export. Samples of storm runoff were also taken over a two-year period from a range of rural sources including roads, farm tracks, septic tank effluent, farmyards, springs, field surface runoff, field underdrainage systems and streams within the Whitchurch, Dinedor and Kivernoll catchments to determine SS and P concentrations and compare their relative ecological importance.

3.3 Environmental issues, treatment summary and data sources

3.3.1 Redesdale

Redesdale is an upland catchment in Northumberland with peaty topsoils and marginal hill pasture/moorland vegetation used primarily for sheep grazing. Pasture improvement has been historically undertaken in this catchment to increase agricultural output but it is unclear whether improvement may have impaired water quality due to increased P transfer in land runoff. Two adjacent sub-catchment areas at Redesdale with different proportions of previously improved grassland (7 v 47% of total area) were therefore monitored over a three-year period (1994-97) to assess potential P enrichment of their soils (field scale) and draining streams (catchment scale). The two sub-catchments have no anthropogenic influences other than that of pasture improvement and therefore provided a unique opportunity to investigate the impact of P fertiliser inputs and increased grazing pressure on P transfer. Long-term soil analysis data for different soil depths in the improved areas available from 1977-1986, and historic stream sampling data for 1977, were supplemented by further analysis of soil and stream samples taken during, and/or subsequent to, the main catchment monitoring period. Information on annual P inputs and grazing densities at different times of the year was collated from detailed farm records.

3.3.2 Rosemaund

Rosemaund is a lowland catchment in Herefordshire with red, silty soils on slightly undulating slopes (generally <5°) supporting beef, sheep, deer and arable crops. Historically, the farm has been a major producer of hops, a crop that leaves high levels of residual P fertility in the soil. During storm events, the dispersive nature of the soil leads to highly turbid, P-rich runoff, the majority of which enters the stream via intensive tile drainage systems that were installed when grant aid was available in the 1960s and 1970s. The Rosemaund stream feeds into the R. Lugg, which is a main tributary of the R. Wye. The Wye catchment has been designated a Special Area of Conservation (SAC) under the EC Habitats Directive, includes large areas of outstanding natural beauty and is an important habitat for the Atlantic Salmon (*Salmosalar*), (Environment Agency, 1998). There have been concerns over declining fish populations in recent years due to deterioration in water quality (sediment and P), particularly in the R. Lugg, which was designated as sensitive to eutrophication in 1994 under the Urban Waste Water Treatment Directive (UWWTD). This resulted in a programme of P stripping at the major STWs in the lower Wye catchment to reduce sewage effluent P discharges. Attention has now focused on the diffuse contribution of P from agriculture which has become proportionally more significant.

Previous work has quantified a sediment budget for Rosemaund and traced the sources of in-stream SS to the land surface (Russell et al., 2001; Chapman et al., 2005). Previous work has also shown that a high proportion of soil exchangeable P (Olsen-P) is extractable in water form compared to many other soils (Withers et al., 2001). The soils in the catchment therefore represent a high risk of dissolved P transfer due to historic inputs of P when the farm was growing hops, rapid mobilisation of particulate P in runoff from cultivated areas and fast delivery to the stream via pipe drain outfalls. To investigate the effects of soil P fertility on P transfer in overland flow, and on the P sorption properties of the entrained sediment, a range of soil P levels on adjacent plots (Holbach plots) were established in one field within the catchment. The surface runoff generated from these plots under natural rainfall was collected and analysed over three years (2001-2004). These data were then compared to the outflow data from a typical arable field drain (Foxbridge, 0.06 km^2) and stream data for a small sub-catchment (Jubilee, 0.31 km^2) and at the catchment outlet (Belmont, 1.5 km^2) for the period 1997-2000. This catchment therefore provided a unique opportunity to examine P transfer at different spatial scales. Soil P analysis data were collected for fields in the catchment and information on P inputs and farming operations at different times of the year were collated from farm records.

3.3.3 Avon

The Avon is a lowland, chalkland catchment centred around Salisbury in Wiltshire with a mixture of gently undulating land cropped to cereals or in dairy farming, steep chalkland slopes (up to 10°) grazed by sheep and river meadows used for summer grazing. Chalkland rivers are renowned for their biodiversity, including vulnerable species such as water crowfoot (*Ranunculus* spp.), Atlantic salmon (*Salmosalar),* Bullhead (*Ictalurus*), Sea Lamprey (*Petromyzon*) and the Desmoulin's whorl snail (*Vertigo moulinsiana*), (Berrie, 1992; Wheeldon, 2003). The major tributaries and main river in the catchment have been variably suffering from siltation, eutrophication and general 'chalk stream malaise' which has upset the natural balance of aquatic ecology, with a loss of key macrophytes and declines in salmonid, coarse fish species and invertebrates (Acornley and Sear, 1999; Environment Agency, 2002). A number of potential contributory causes have been identified including point source effluent discharges, diffuse inputs from agriculture in the headwaters and water abstraction.

In 1999, a catchment management initiative called *Landcare* was started in the area upstream of Salisbury to help reduce the agriculturally-derived loads of pollutants, particularly SS and P entering the major tributaries (Huggins, 1998; Environment Agency, 2002). As part of this initiative, farmer demonstration plots were established at field sites on the three major lithologies that dominate the catchment: Upper Chalk (Wilton), Upper Greensand (Pewsey) and Kimmeridge Clay (East Knoyle). The demonstration events were organised by the EA for farmers and their advisers to encourage the adoption of more sensitive soil and land management practices required to help reduce the risk of runoff, and erosion in the catchment. To supplement this promotional activity, the demonstration sites were monitored over two winter periods (2002/03 and 2003/04) to provide supporting data on the effects of two specific BMPs (reduced cultivation and early sowing) on runoff generation and associated mobilisation of P. These plots were further used to assess the impact of removing tramlines on SS and P export, representing a third BMP. Information on P inputs and farming operations at each of the three demonstration sites were collated from farm records. Data on catchment export of SS and P for two tributaries on Greensand lithology (East and West Avon) for the same monitoring period were collected under the PSYCHIC project and compared with the P transfer at the Pewsey site.

3.3.4 Sources of phosphorus in rural catchments

A number of sources of P other than from farmed land are sufficiently well distributed in rural catchments to cause eutrophication and it is critical to understand the relative importance of these sources in order that any mitigation actions are targeted effectively. These additional sources include farmyard runoff, road runoff, septic tank overflows or direct discharge from individual dwellings and farm, or village, sewage treatment works (STW) discharging effluent (May et al., 1998; Mitchell, 2001; Edwards et al., 2007). The range in SS and P concentrations associated with these additional sources were compared with the range obtained in surface runoff and drainflow from arable and pasture land in selected catchments (Whitchurch, Dinedor and Klvernoll) within the Wye river basin in Herefordshire. The soil types in these catchments were very similar to those at Rosemaund and these source comparisons provided an opportunity to assess the potential ecological relevance of P transfer from arable and pasture land compared to other rural sources for river eutrophication.

3.4 Site descriptions and treatment details

3.4.1 Redesdale catchment

Redesdale is a 1543 ha upland beef and sheep farm located a few miles north of Otterburn in Northumberland (G.R. NY833961, Fig. 3.2). The farm has pioneered research into land improvement and profitable livestock production in the area for over 40 years. Within the Cleughbrae unit of the farm, SS and P concentrations in streams draining two adjacent sub-catchments (RD3 and RD4) were monitored intensively from 1994-1997. The RD3 sub-catchment (190 ha) was dominated by moorland and unimproved rough grazing, and contained only a very small proportion (7%) of land that had been agriculturally improved by liming, fertiliser application and reseeding. The RD4 sub-catchment (110 ha) contained a much higher proportion (47%) of improved grassland, but also included some plantation woodland, and moorland rough grazing. The catchment areas are typical of unenclosed upland farming on Carboniferous till in Northern England (Plate 3.1).

The soils are dominated by poorly drained stagnohumic gleys (Kielder Series) with variable depths of peat (typically 15-40 cm) overlying glacial boulder clay drift containing local

Carboniferous sandstones and shales (Bradley, 1993). In small areas, the drift may be very sandy, very shallow over rock, or the depth of peat may extend beyond 80 cm. Winter rainfall is not absorbed by the wetter soils and runs off rapidly. Drainage status and depth of peat are therefore the main factors determining the natural vegetation type and the feasibility of land improvement.

In RD3, open drainage ditches (grips) accelerate surface runoff down sloping hillsides to tributaries with steep-sided channels (Plate 3.1). The RD4 catchment area is lower down the hillside and less steeply sloping with fewer grips feeding the one main tributary on its northern boundary (Fig. 3.2; Plate 3.1). Its more favourable topography is one of the reasons this sub-catchment was targeted for land improvement. RD3 and RD4 streams join just below the RD4 monitoring station, and together with other feeder streams form the main Cleughbrae Burn passing a weir (RD2), the farm, and then into the River Rede (Fig. 3.2). Altitude ranges from 365 m at its highest point descending to 220 m at the farm buildings. Average annual rainfall (1970-2000) is ca. 900 mm and the area can be snow bound for extended periods during the winter.

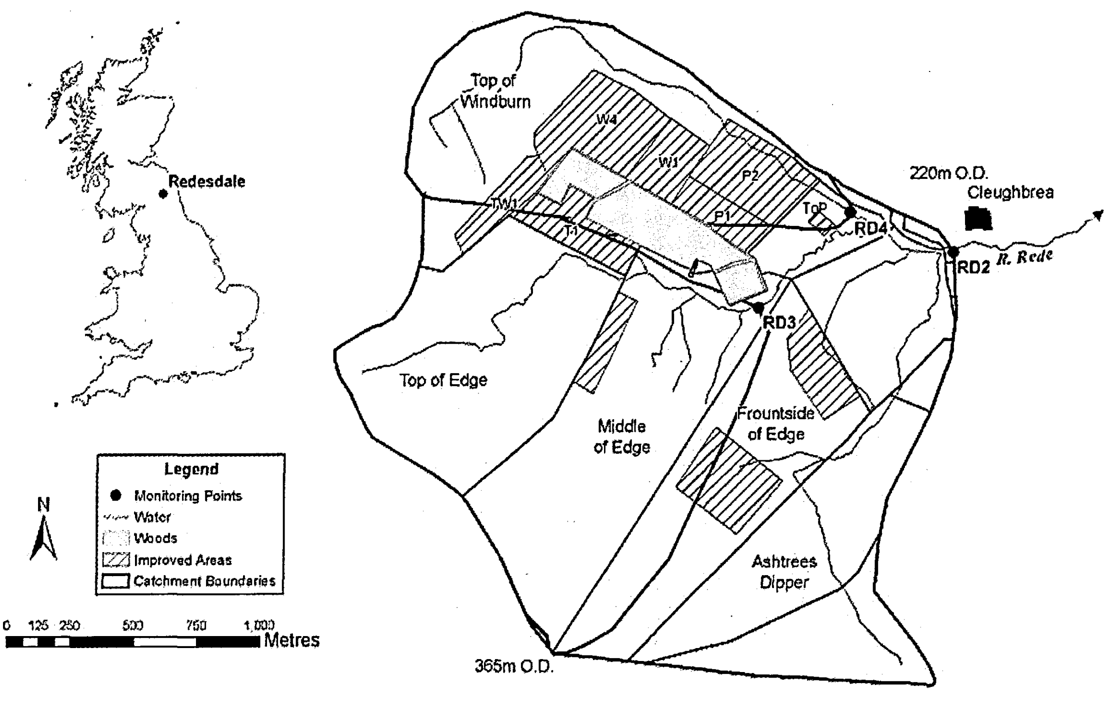


Figure 3.2. Location of pasture improvement areas within the RD3 and RD4 catchments at Redesdale. The raingauge was sited at RD2.

View over the Redesdale catchment in Northumberland with moorland and improved pastures grazed by sheep.

Looking upstream from the RD3 monitoring station onto the incised and marginal rough grazing land on the hill.

The main section of improved grassland within the RD4 catchment. The terrain is flatter with fewer incising streams.

The soil type on the improved area has a peaty topsoil over gleyed glacial clay drift with sand pockets.

Plate 3.1 Various land use aspects of the Redesdale catchment.

The unimproved vegetation at Rededsale as described by the National Vegetation Classification (NVC) mainly comprises degraded wet heathland, M25 (*Molinia caerulea – Potentilla erecta* mire), with areas of degraded blanket mire, M20 (*Eriophorum vaginatum* blanket mire), areas of *Nardus stricta* dominated U5 (*N. stricta – Galium saxatile* grassland) on the more free draining soils, and, where rushes have invaded, M23 (*Juncus effusus/acutiflorus – Galium palustre* rush pasture) (Rodwell, 1991; 1992). The soils are acid (pH 4.0), deficient in P (4-6 mg L^{-1} Olsen-extractable P, Ministry of Agriculture, Fisheries and Food, 2000), largely unploughable and suitable only for rough grazing. However, areas dominated by *Nardus* and, to a lesser extent, *Molinia* with shallower peat topsoil were targeted for hill land improvement research in the early 1970s using surface seeding methods.

History of land improvement

From 1974 onwards, five areas (P1, P2, W1, W4 and T1) were 'improved' using a system of pioneer cropping that became known as the 'Redesdale technique' (Davies, 1978). This technique involved liming (usually the preceding year), destruction of the natural *Nardus* sward with a herbicide and addition of basic slag or partially-acidulated rock phosphate fertiliser (typically 76 kg P ha^{-1}) to raise the soil P status. Stubble turnips were subsequently sown in June in each of two consecutive years with NPK compound fertiliser (110, 24, 46 kg ha^{-1} of N, P and K, respectively). Both turnip crops were grazed by finishing lambs between October and January to help break up the surface grass mat. A perennial ryegrass, Timothy and white clover seeds mixture was then sown with NPK compound fertiliser (110, 24, 46 kg ha^{-1} of N, P and K, respectively) the following May (Davies, 1978). In 1989, an additional area (TW1) was improved by liming, sward destruction and soil P fertilization (49 kg P ha^{-1} as triplesuperphosphate, TSP, 20% P) followed by simply rotaseeding of the grass/clover seeds mixture into the soil with NPK fertiliser (50, 11, 21 kg ha^{-1} of N, P and K, respectively) without pioneer cropping with turnips. A further 74 kg P ha^{-1} as TSP was applied to raise soil P status of TW1 in April 1990. The locations of the six improved areas are shown in Fig. 3.2.

The improved areas have subsequently been fertilised with N (60-80 kg ha^{-1}) in April each year, and then grazed by sheep until about the end of August. The areas are then rested until mid-October when they are grazed by ewes until January and thereafter not grazed until April. Weaned calves also sometimes graze between June and August to fully utilize the grass at the peak of its growth. The areas are then cut (topped) and usually fertilised towards the end of August with NPK compound fertiliser, supplying 40, 9 and 50 kg N, P and K, respectively. Fertiliser application was dependent on the results of soil analysis: in some years only an NK fertiliser was applied, and in others an additional large single application of P fertiliser (rock phosphate or TSP) was applied to prevent any decline in soil P fertility. The aim was to maintain soil available (Olsen) P levels (0-7.5 cm) at 16-25 mg L^{-1} (P Index 2, Ministry of Agriculture, Fisheries and Food, 2000), a threshold above which field experiments on the farm have shown that there is no response to fresh P fertiliser inputs (Merrell and Withers, 1987). All improved areas were fenced off to maximise grass utilization. Other small areas (Frontside of Edge, Middle of Edge and Top of Pasture (ToP) have also been improved over the years, either by rotaseeding or direct drilling, as part of research projects, but have not been fenced off nor subsequently received lime and fertiliser on a routine basis (Fig. 3.2). ToP and other small areas bordering the farm track and stream within the RD4 catchment have received housed farm yard manure (FYM) at regular intervals at rates of ca. 6 t ha^{-1} supplying ca. 8 kg P ha^{-1}.

3.4.2 Rosemaund catchment

ADAS Rosemaund is a small 176 ha beef, sheep, deer and arable farm located just north of Hereford on the Border between England and Wales (G.R. SO565480, Plate 3.2). The farm has conducted research on optimising crop and livestock production systems since 1949 and has grown a wide range of crops including cereals, potatoes, peas, beans, oilseed rape, root crops, maize, fruit and hops. The catchment area (1.5 km^2) includes the majority of the research farm (including woodland) together with a small area (8.7 ha) of low intensity grassland which is outside the farm's northern boundary (Fig. 3.3). The catchment also includes the farm buildings and associated yards, and a small STW that services the resident population of 35 people and a visiting population of ca. 30 people. The STW adopts a simple continuous trickle filter system with sediment tanks and only the overflow is discharged directly to the stream (Fig. 3.3; Plate 3.2). Stream monitoring points included the catchment outlet (Belmont), a small upstream sub-catchment (Jubilee, 0. 31 km^2) and a field drain outfall from a 6 ha field (Foxbridge). Supplementary measurements of instantaneous concentrations and flows from the farmyard and the outfall from the village STW were also collected in 2006/07 to provide some information on the likely contribution of point sources in the main Belmont catchment. The Jubilee catchment has no known inputs of point sources.

View over the Rosemaund catchment in Herefordshire, with both arable and grassland farming.

The farm sewage treatment works services 65 residents with overflow discharging directly into the stream.

View of Foxbridge field and up to the top of the catchment. The topography is gently undulating and all fields are tile-drained.

Jubillee field just after sowing and tramline establishment. The red silty soil disperses easily during storm events.

Plate 3.2. Various land use aspects of the Rosemaund catchment.

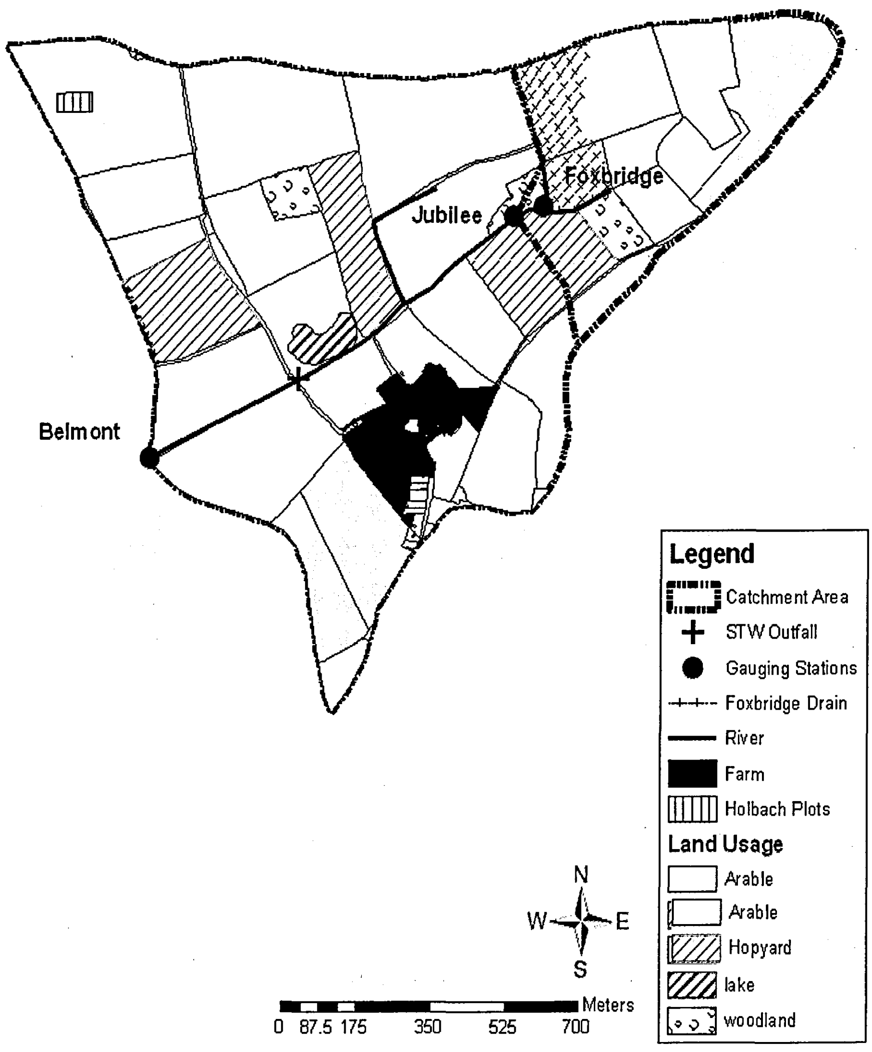


Figure 3.3 Location of flow monitoring points, sewage outfall, Holbach field plots, Foxbridge land drains and land uses in the Rosemaund catchment.

The soils are uniformly red, silty clay loam in texture and developed over Silurian Raglan mudstone which lies ca. 1-3 m below the surface (Plate 3.2). Small areas of alluvium are present in the stream corridor. Soil types range from argillic brown earths (Bromyard Association) which occupy most of the catchment, to heavier-textured stagnogleyic argillic brown earths (Middleton Association) on lower slopes and in hollows, and pelo-alluvial soils (Compton Association) in the river corridor (Hodgson and Palmer, 1971). On the heavier soils, silty clay textures predominate nearer the surface. All the soils are naturally well fissured with extensive earthworm channels and typically crack during prolonged dry weather. Almost the entire catchment was extensively underdrained between 1950 and 1990 with pipes laid at 0.8 m depth and 20 m spacing, back-filled with gravel to within 0.3m of the surface and regularly subsoiled to maintain fissure continuity and reduce any surface wetness.

The silty nature of the soil leads to rapid dispersion of soil particles at the surface when it rains, which then become transported down fissures to the underdrainage pipes and directly into the stream (Chapman et al., 2005). Drain outflow therefore becomes very turbid during storms with sediment concentrations of up to 5 g L^{-1} (Hodgkinson and Withers, 2007). Overland flow occurs largely as unconcentrated shallow flow (sheet runoff) and rill (but not gully) erosion is frequently seen in some areas of the catchment. Average annual rainfall is 660 mm.

The relative areas of the different land uses in each of the monitored catchment areas are given in Table 3.1. The cultivated area is routinely cropped to winter cereals (wheat, barley and oats) with winter oilseed rape as a break crop. Hops were discontinued in 1998 and sown to grass but their legacy of high soil P fertility makes it important that these areas are quantified (Fig. 3.3). Small areas (<10 ha) are cropped to potatoes on outside contract, or to maize, and there is a small orchard (5 ha) of cider apples. There is also a small amount of set-aside (8 ha). Winter crops are usually sown from mid-September to mid-October, although drilling may continue into November in wet years. The soil is usually ploughed, power-harrowed once (twice on heavier soils), drilled and rolled. Subsoiling is also carried out in some years. Phosphorus fertiliser inputs to the arable crops are usually applied on the stubble prior to ploughing as 0:20:30 or 0:24:24 and based on the results of regular soil analysis. Hence not all fields receive fertiliser each year. During the catchment monitoring period, arable fields received P fertiliser more during the first year whilst grassland fields were fertilised in the third year. When fertilised, arable crops and grassland typically received ca. 20 kg P ha^{-1} which is broadly equivalent to crop P offtake.

Table 3.1. Land use distribution, P inputs and soil P concentrations in each of the catchment areas monitored in the Rosemaund catchment.

Monitoring point	Area (km^2)	Land use (% of area)				P inputs[2] (kg P ha^{-1})	Soil P status[3] (mg kg^{-1} OP)
		Pasture	Arable	Hopyard	Other[1]		
Foxbridge	0.06	0	100	0	0	45	21
Jubilee	0.31	28	61	5	6	30[4]	30 (16-88)[4]
Belmont	1.5	17	62	16	5	20	55 (16-125)[5]

[1]Woodland in Jubilee and woodland and farm buildings/yard in Belmont.
[2]Average estimated annual inputs of P during the monitoring period based on farm records.
3Area-weighted average and range in concentrations of Olsen-extractable P (OP) in 0-10 cm.
[4]Excluding the low intensity grassland outside the Rosemaund farm.
[5]Based on 60% of the catchment area sampled covering arable and grassland fields.

The farm supports a 700 ewe flock, 200 deer and 280 finishing cattle. Sheep are housed from January to March inclusive, deer from December to April inclusive and cattle are finished in straw yards over winter. The ewe and deer flock graze on the 54 ha of grassland on the farm, although only a proportion of this is in the catchment area. Manure is all recycled as farmyard manure (FYM) to arable stubbles in autumn, and to spring crops of potatoes and maize. During the catchment monitoring period, manures were recycled largely during the second year at rates up to 65 t ha^{-1}. Pastures are mostly permanent, grazed only and do not receive any manure. Silage is imported onto the farm. A P balance conducted for the farm based on the methodology described in Chapter 2 showed total annual fertiliser and manure P inputs of 1398 and 2596 kg P, respectively and a total crop offtake of 3639 kg. The farm is therefore approximately in balance with a surplus P input of only 2 kg P ha^{-1} yr^{-1}. Estimated average P inputs for the different catchment areas are shown in Table 3.1. Soil fertility is moderate to very high (16-125 mg OP L^{-1}) reflecting both the historic prevalence of hops on the farm and locally high inputs of FYM.

Holbach field plots – treatment details

Within the Rosemaund catchment, fifteen hydrologically isolated plots (15 m * 2m) were established in 1995 in one of the farm fields to investigate the impact of different P amendments (fertiliser, cow slurry and sewage sludge) on P transfer in overland flow (Withers et al., 2001). The arable field has a uniform 5° slope and a silty clay loam soil (30% clay, 66% silt, 4% sand and 2.7% organic matter) which is typical of the catchment area (Bromyard Association). These plots were subsequently used to investigate the impact of soil P accumulation on the mobilisation of P in overland flow and the P sorption properties of the entrained sediment. In spring 1998, varying amounts (250-4000 kg P ha^{-1}) of P fertiliser (triplesuperphosphate, 20% P) were applied to the plots based on the results of soil analysis in autumn 1997 to establish a wide range of soil P concentrations. The fertiliser was incorporated into the soil to a depth of 20 cm using a rotovator and allowed to equilibrate with the soil over a 2-year transition period. The plots were not replicated and each had a different soil P concentration.

The runoff from the plots during storm events was then monitored and sampled over three winter periods: 11 January 2001 to 9 April 2001 (Period 1, storm events 1-5), 17 November 2001 to 30 January 2002 (Period 2, storm events 6-10) and 1 November 2003 to 15 March 2004 (Period 3, storm events 11-22). In Period 1, the plots were left uncultivated although grass weeds were allowed to establish themselves. For the second period, the weeds were sprayed off and the plots cultivated to 20 cm depth in late September 2001 using a spading machine and crumbler roller to simulate seedbed conditions. For period 3, the plots were rotavated twice to 20 cm depth in late October 2003 but were not rolled because soil conditions became too wet. No crops were sown and no P fertiliser was applied during the monitoring periods.

3.4.3 Avon catchment

The River Avon rises just north of Pewsey in Wiltshire (GR SU165601) and, fed by four major tributaries draining the predominantly chalkland catchment area above Fordingbridge, flows to the sea at Christchurch (Fig. 3.4). The geology of the upper Avon above Fordingbridge is dominated by Upper Chalk but with smaller outcrops of Greensand, Purbeck and Portland Sandstones and Kimmeridge Clay. Below Fordingbridge, the R. Avon flows over the sands and gravels of the Reading Beds and London Clay, and then over the acidic clays, sands, silts and gravels of the Barton, Bracklesham, and Bagshot Beds.

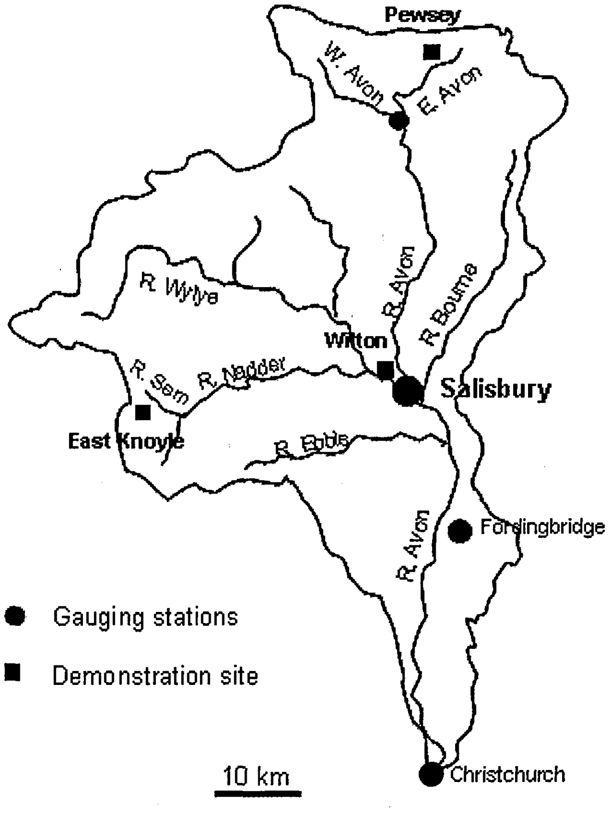


Figure 3.4 The Hampshire Avon catchment showing the location of the demonstration sites (Pewsey, Wilton and East Knoyle) and monitored tributaries (East and West Avon) in relation to other tributaries and towns.

View of the Avon catchment in Wiltshire with variably steep slopes and largely intensive arable farming.

Large sloping fields of free draining soils with direct connectivity to the roads dominate the chalk landscape.

Tramlines running up and down slope cause channelling of runoff and soil erosion on Greensand soils in headwater areas.

Smaller areas of tile drained clay soils on flatter terrain support livestock farming on grass and forage maize.

Plate 3.3. Various land use aspects of the Avon catchment.

The demonstration areas – treatment details

As part of the LANDCARE initiative, demonstration plots were established at field sites on the three major lithologies in the catchment: Upper Chalk (Wilton), Upper Greensand/Gault (Pewsey) and Kimmeridge Clay (East Knoyle), Fig. 3.4. These lithologies produce soils that have very different hydrological characteristics and susceptibility to runoff and erosion. The field sites chosen were representative of the landscape type and contrasted strongly in soil physical and chemical characteristics (Table 3.2). Slope varied from very slight (1°) at East Knoyle to more steeply sloping at Pewsey (5°) and Wilton (8°). Clay content and organic matter were lowest at Pewsey (13% and 2.1%, respectively), intermediate at Wilton (27-30% and 4.1%) and highest at East Knoyle (26-38% and 5.1-8.6%). Wilton was the only strongly calcareous site, although soil pH at East Knoyle and Pewsey was also high (pH $\geq$7). Soil total P (TP) concentrations were high at Wilton and East Knoyle and low at Pewsey, reflecting differences in soil texture and P sorption capacity. OP concentrations were relatively low at Wilton, but well above the level considered satisfactory for crop growth (ca. 20 mg kg^{-1}) at Pewsey and East Knoyle (Table 3.2). The larger OP levels in the two fields at East Knoyle reflect the history of dairy manure application at this site. Demonstration plots were sown to cereals in both years and cultivated/drilled either early (E) or late (L), and were either traditionally cultivated (TC) or reduced cultivated (RC), providing four treatment combinations: E-TC, E-RC, L-TC and L-RC.

Plots were not replicated but were large in size (20 m by 20 m) as required for demonstration purposes, and the farmer used local cultivation practices when establishing the treatments. At East Knoyle in 2003/04 the L-TC treatment was replaced with an E-RC headland plot (hereafter referred to as 'headland') due to field size restrictions. Early drilling was usually at the end of September, but late drilling varied from late October to early January depending on the weather (Table 3.2). Traditional cultivation included ploughing to 20-25 cm (with or without a press), and either tine harrowing or power harrowing before drilling. Reduced cultivation was with either heavy discs (Pewsey and Wilton) or a heavy harrow (East Knoyle) to 5-8 cm (Table 3.2). At Wilton and Pewsey, plots were drilled up and down the slope and tramlines established after drilling by one or more tractor passes. At East Knoyle, plots were drilled across the slope, except for the headland plot which was drilled up and down the slope. Crop residues were returned at all sites and different fields were used each year except at Pewsey.

Table 3.2. Site characteristics and treatment details for the demonstration plots in the Avon catchment.

	Pewsey		East Knoyle		Wilton	
	02/03	03/04	02/03	03/04	02/03	03/04
Site characteristics						
Cropping	WW	WO	WW	WW	WW	WW
Slope °	5	5	1	1	7	9
Soil parent material	Greensand		Kimmeridge clay		Upper Chalk	
Soil association[1]	Ardington		Wickham 2		Upton	
Topsoil texture	Fine sandy loam		Clay loam		Silty clay loam	
Subsoil texture	Sandy clay loam		Clay		Chalk rubble	
Sand (%)	67	67	8	6	17	16
Silt (%)	20	19	66	56	56	54
Clay (%)	13	14	26	38	27	30
pH	7.4	7.1	7.1	6.9	8.0	8.2
O.M. (%)	2.1	2.5	5.1	8.6	4.1	4.2
$CaCO_3$ (%)	<0.5	<0.5	0.5	2.5	72	59
Total P mg kg^{-1}	315	332	1120	1130	985	1000
Olsen-P mg kg^{-1}	33	32	67	46	11	9
Treatment details						
Early drilling dates	24 Sept	28 Sept	26 Sept	2 Oct	20 Sept	24 Sept
Late drilling dates	10 Jan	23 Oct	24 Oct	7 Nov	-	24 Oct
Cultivations – Traditional	Plough (22 cm), press, tine harrow and drill		Plough (15-18 cm), cambridge roller Power harrow (x1) and Combi Drill		Plough (22cm), roll, tine harrow, drill and roll	
Cultivations - Reduced	Heavy discs and press tine harrow and drill		Heavy harrow (5-8 cm), power harrow (x1) and Combi drill		Heavy discs, press, tine harrow, drill and roll	

WW, winter wheat (*Triticum aestivum* L.); WO, winter oats (*Avena sativa* L.). [1]Findlay et al., (1984) In addition to the main treatments, the opportunity was taken to examine the impact of tramlines on SS and P transfer from the demonstration plots established at Pewsey. In the first year, the tramlines on the E-TC treatment had tyre lugs pointing in a downward $\vee$ pattern, while the lugs in the tramlines on the E-RC treatment were pointing in an upward $\wedge$ direction. In the second year, this was reversed on the early-drilled treatments but not the late-drilled treatments. These treatments were not established by design but the consequence of farmer practice in establishing the plots within the rest of the field, which was drilled across the slope with tramlines also running across the slope. Nevertheless the presence of the tramlines allowed comparison of SS and P export from tramlined and non-tramlined areas superimposed on the main experiment (Fig. 3.5). To investigate separately the effect of tramlines, three additional replicate runoff plots without a tramline were established where resources permitted.

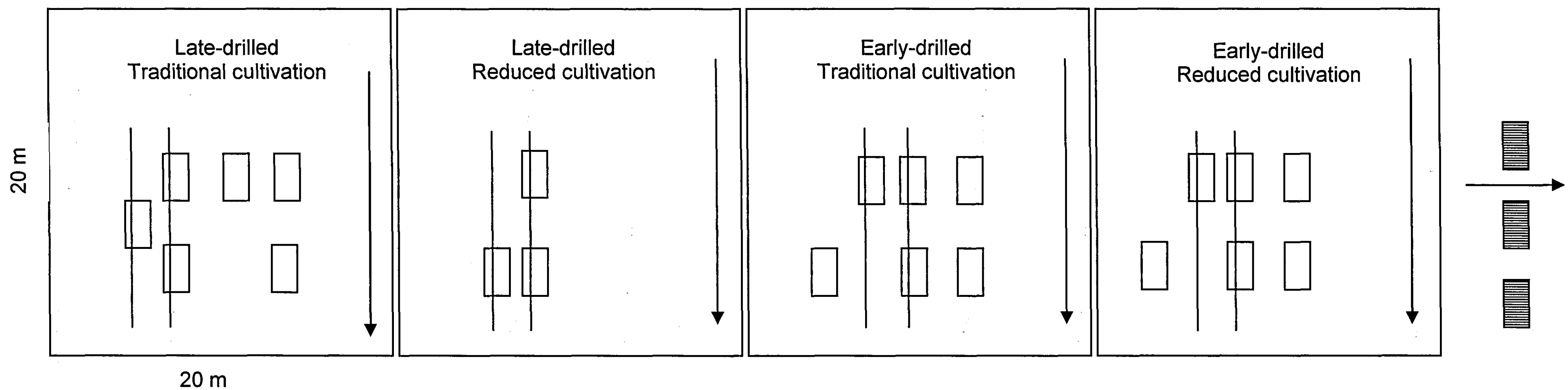
In the first year, plots without tramlines were established on the early-drilled treatments only when it became obvious that the tramlines on the E-TC and E-RC treatments were having a significant impact on surface runoff generation. The monitoring period for comparison was therefore relatively short (5 sampled storm events) from 17 January – 12 March. In the second year, the E-TC, E-RC and L-TC treatments were compared (Fig. 3.5). For the early-drilled treatments, the monitoring period was from 22 October – 25 March (9 sampled events) and for the L-TC treatment, the monitoring period was from 23 December – 25 March (5 sampled events). In the second year, three additional runoff plots were established just outside the main experimental area where the tramlines were running across slope rather than up-and-don slope (Fig. 3.5). The plots were monitored over the same time period as the other plots in the second year.

3.4.4 Wye catchments

Streams and a range of storm runoff sites in three small (<10 km^2) rural catchments (Whitchurch, Dinedor and Kivernoll) were monitored over a two-year period (2005/06 and 2006/07). All thee catchments drain into first-order tributaries of the Wye river and are located on similar Devonian Old Red Sandstone lithology, but varied in the type and intensity of farming system and point source inputs (Fig. 3.6). In each catchment, co-operative farmers were interviewed to collect information on fertiliser and manure P inputs and land use (Table 3.3).

Figure 3.5 Site plan showing layout of runoff traps with and without tramlines within each demonstration area at Pewsey. The arrows indicate the direction of drilling.

The shaded runoff traps were those located outside the demonstration areas where the direction of drilling was across slope.



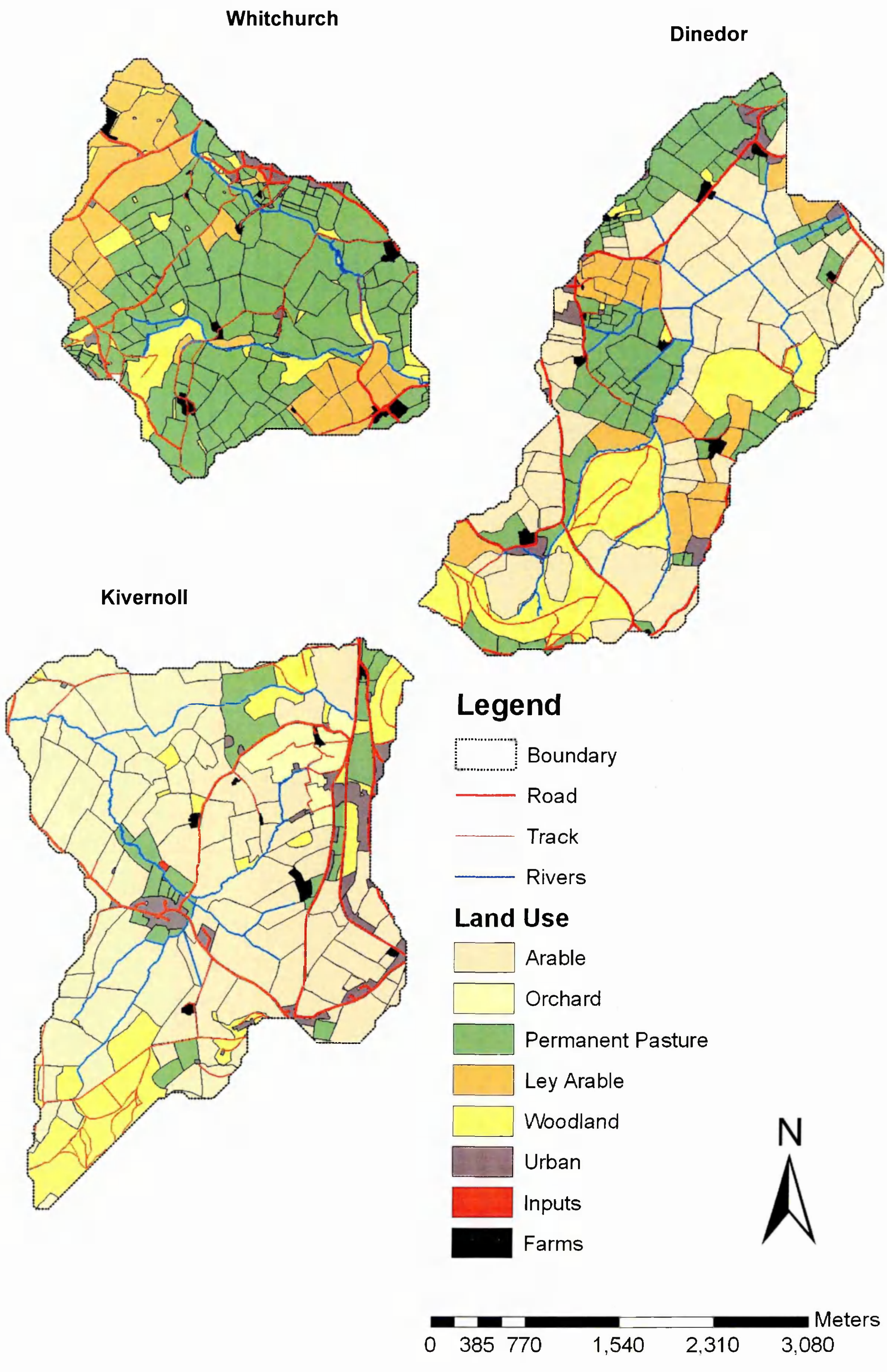


Figure 3.6 Land use within the Whitchurch, Dinedor and Kivernoll catchments.

Table 3.3. General characteristics of the Whitchurch, Dinedor and Kivernoll catchments.

River Catchment	Area km^2	Rainfall[1] mm	Dominant soil types[2]	Farming system	Land use (%)				P inputs[5] $kg\ ha^{-1}$	Soil P[6] $mg\ L^{-1}$
					Woodland	Arable[3]	Pasture	Other[4]		
Wye										
Whitchurch	6.46	812	Eardiston	Low intensity arable, beef and sheep	9	23	60	8	<5-15	14 (3-33)
Dinedor	8.69	694	Bromyard Eardiston	Intensive arable, beef and sheep	19	53	23	5	11-75	15 (3-70)
Kivernoll	9.87	737	Bromyard Eardiston	Intensive arable, and poultry	13	68	11	7	26-92	35 (11-57)

[1]Long-term (30 yr) annual rainfall. [2]Soil Association (Ragg et al., 1984). [3]Including ley-arable. [4]Urban areas and farmyards. [5]Fertilisers and manures where they were applied. [6]Average (range) of Olsen-P concentrations in fields sampled for fertiliser recommendations (Ministry of Agriculture, Fisheries and Food (2000).

The Whitchurch catchment (6.5 km^2) has a combination of steeply sloping riparian pasture land on silt loam soils (Eardiston Association) grazed by cattle and sheep in the centre of the catchment with ley-arable crops receiving farmyard manure (FYM) on perimeter plateau land. Fertiliser P use is very low (average <5 kg ha^{-1}). There are no major point source inputs but there are a large number of farmyards and the Whitchurch stream receives a large amount of runoff down steeply sloping roads. The Dinedor catchment (8.7 km^2) includes a mixture of beef and sheep farming on permanent and ley grassland in the west of the catchment with arable land growing cereals, oilseed rape, beans and potatoes in the east of the catchment. One farm in the catchment has pigs. The soils are heavier-textured (silty clay loam) than those in the Whitchurch catchment and have a greater range of OP concentrations due to greater use of P fertilisers, although average soil P fertility is still low and quite similar to Whitchurch (Table 3.3). The upper part of the catchment has steeper slopes and therefore a significant input of road runoff and the overflow from a village hall septic tank enters a field ditch in the lower part of the catchment. The Kivernoll catchment (9.9 km^2) is almost totally under arable cultivation with intensive winter cereal, oilseed rape, sugar beet and potato production receiving poultry manure and P fertiliser. Average soil P fertility is very much greater than in Whitchurch or Dinedor and there is a STW in the lower part of the catchment (Fig. 3.5, Table 3.3).

3.5 Materials and methods

3.5.1 Soil profile scale

In-situ mobilisation of P

To estimate the intrinsic risk of sediment and P mobilisation in sheet runoff at each study site, a laboratory test (termed the DESPRAL test) that quantifies dispersed particles and associated P in the same suspension was undertaken on soil samples from representative fields (Withers et al., 2007). Spade spit topsoil samples of soil to 10 cm depth from 10-15 locations in the field were bulked, slowly air-dried, gently broken up and sieved by hand to produce aggregates <5 mm in size. 20 g of air-dried <5 mm soil were added to 200ml distilled water and left to stand for 1 hour in a cylindrical flask. The volume of water was then made up to 1 litre and the sample shaken end over end for 1 minute and then allowed to stand for 280 s. This is the period after which particles and aggregates <20 μm are left in suspension in the water column. A custombuilt apparatus, as used by Hodgkinson and Thorburn (1995) 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 (Plate 3.4). Seconds before the end of the settling period, a 20 ml aliquot of the suspension was pipetted out from a depth of 10 cm below the surface. Half (10 ml) of the pipetted sample was placed in a pre-weighed beaker, weighed, oven dried at 105°C for 24 hours and weighed again to determine the mass of SS. The remaining 10 ml sample was used to determine total P (TP), total dissolved P (TDP) and dissolved-reactive P (DRP). The results were expressed as the concentration of the SS (mg L^{-1}), or P forms ($\mu g\ L^{-1}$) that might be potentially mobilised as a result of raindrop impact (Withers et al., 2007). Sub-samples of each <5 mm sieved soil were further ground to pass 2mm and analysed for total P (TP), Olsen-extractable P (OP) and water-extractable P (WEP).

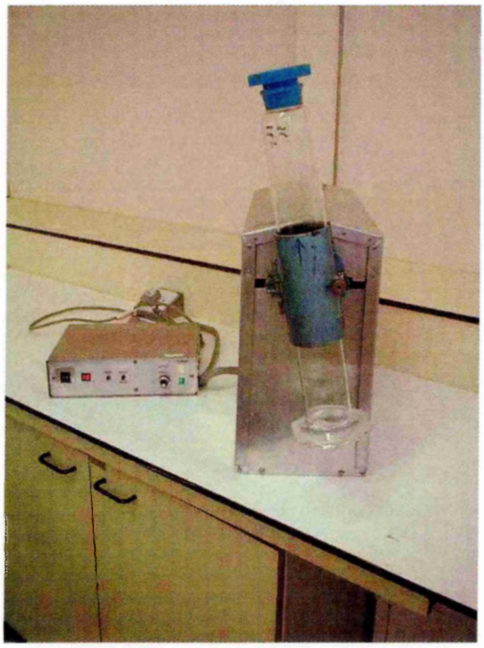


Plate 3.4. Apparatus for delivering end-over-end revolutions at constant speed for the DESPRAL test (from Withers et al., 2007).

3.5.2 Field scale

Soil measurements

Soil samples were collected from all sites to characterise their general physical and chemical properties including particle size distribution (PSD), calcium carbonate ($CaCO_3$), organic matter

(OM) and pH. Additional samples were collected from each site according to the individual treatment comparisons and requirements as follows.

Redesdale

From 1977–1986, temporal changes in OP concentrations in the soils of improved areas P1, W1 and T1 were monitored annually to a depth of 15 cm in 4 depth increments (0-2.5, 2.5-5.0, 5.0-7.5 and 7.5-15 cm). Samples were taken well after, or before, any P fertilisers were applied to avoid contamination. For each depth, twenty-five cores were taken in a 'W' pattern and bulked as recommended by Ministry of Agriculture, Fisheries and Food (2000) for monitoring soil P fertility. Additional topsoil (0-7.5 cm) samples were taken before (January 1992), during (January 1995) and after (December 1997) the 3-year catchment monitoring period to further monitor changes in field OP concentrations.

Rosemaund

Soil samples (25 cores from 0-10, 10-20 and 20-30 cm depths and bulked) were taken from selected fields in the Rosemaund catchment for determination of total P, OP and WEP concentrations. The Holbach plots were sampled (0-20 cm) in the same way at the end of the transition period in April 2000, in January 2003 and in February 2004 to monitor changes in total P, OP and WEP concentrations. Soil samples taken in 2000 were additionally analysed for calcium chloride extractable P ($CaCl_2$-P) to identify a potential change-point, as defined by Hesketh and Brookes (2000). Soil samples taken in 2003 were additionally analysed for soil pH, total calcium (TCa), oxalate-extractable aluminium (Al_{ox}), iron (Fe_{ox}) and P (P_{ox}), and P sorption properties, and also dispersed in sodium chloride to determine the P content of individual particle size fractions.

Avon

In the Avon catchment, soil samples were taken annually (0-20 cm depth, 25 cores and bulked) from each of the demonstration areas for determination of total P, OP and WP. Additionally, a range of soil physical measurements were carried out to help explain any treatment differences in observed runoff. These measurements included: soil infiltration rate, vertical soil penetration resistance by cone penetrometer, horizontal soil shear strength by shear vane meter, bulk density, air capacity and a visual assessment of soil structure. The percentage bare ground was also determined at each site. Not all of these measurements were undertaken in the first year.

Rate of water infiltration was determined using a double ring infiltrometer and a flooding technique (Ministry of Agriculture, Fisheries and Food, 1982). Five (eight at Pewsey) replicate ring measurements were made on each demonstration plot, outside of the runoff plot area. After flooding of the inner and outer rings, the drop in water level in the inner ring was recorded every 10 minutes over a 5 hour period, and the average drop per hour over the this period was calculated as the mean infiltration rate.

Soil strength measurements carried out at constant moisture content can provide quantitative data on the degree of soil compaction (Ministry of Agriculture, Fisheries and Food, 1982). Two assessments of soil strength were carried out: soil resistance to a horizontal shear stress using a Pilcon DR1176 Shear Vane fitted with a 19mm vane, and soil resistance to vertical stress, or penetration, using a Farnell drop cone penetrometer with a cone base area of 129 cm^2. Measurements were taken, where possible, to a minimum depth of 30 cm in either 5 cm (shear vane) or 7.5 cm (penetrometer) increments. In the first year, shear vane measurements were taken from 12 replicate points within each plot, three of which were in the tramline. In the second year, this was increased to 15 replicate points, of which 5 were in the tramline. Penetrometer readings were taken from 30 replicate points within each runoff plot of which 10 were in the tramline.

Intact soil cores of ca. 220 cm^3 volume were taken from within the surface (0-10 cm) and sub-surface (10-25 cm) layers of the topsoil using stainless steel cylinders for determination of dry bulk density and porosity. Five or six replicate samples were taken in May 2003 and January 2004 at each depth from each main demonstration area outside of the runoff plots. Soil structure was visually assessed according to a modification of the Pearlkamp method (Ministry of Agriculture, Fisheries and Food, 1982), where a spadeful of soil was lifted to shoulder height and dropped, and the degree of structural disintegration was scored on a scale of 1-10 (1 = poor structure, 10 = good structure) by two separate people. The modification was based on the visual score assessment methodology developed in New Zealand (Manaaki Whenua Landcare Research, 2005). Ten measurements were taken on each demonstration area outside of the runoff plots. This was an arbitrary but rapid assessment of coarse differences in structural stability between the treatments.

The percentage of bare ground was assessed on 23 January and 8 April 2004 using a 0.25 m^2 quadrat divided into 25 x 0.01 m^2 sections. The assessments were undertaken at nine locations within each runoff plot, of which three were in the tramline and took into account the proportion of the plot area occupied by the tramline.

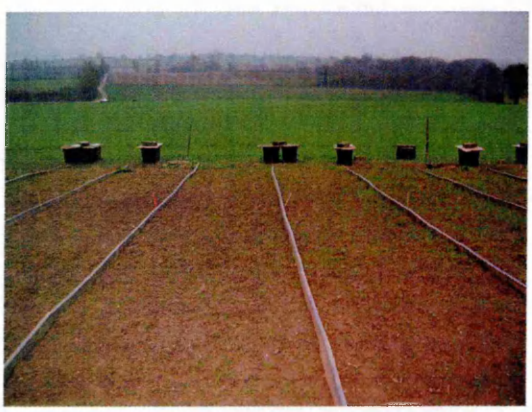
Whitchurch, Dinedor and Kivernoll

Soil samples (0-10 cm) were taken from fields representative of soil type and the range in soil P fertility within each catchment to characterise pH (water), OM, total iron (TFe), total aluminium (TAl), TCa, Fe_{ox}, Al_{ox}, P_{ox}, TP, OP, WEP and degree of P saturation (DPS).

Runoff measurements

At the Holbach (Rosemaund) and the Pewsey, Wilton and East Knoyle (Avon) sites, surface runoff generated by storm events was collected from small plots (15 m by 2 m) each bounded by 30 cm deep stainless steel dividers driven into the ground. At Rosemaund, the upslope plot width was hydrologically isolated by a gravel trench rather than a metal divider. Each plot contained a tramline down one side of the plot. Overland flow was collected via a 110 mm gutter pipe (Holbach) or a metal collecting tray (Avon sites) cemented in place at the bottom of each plot and fed by connecting pipes into a plastic or fibreglass reception tank (Plate. 3.5).

(a)



(b)

Plate 3.5. Hydrologically-isolated field plots were used to monitor sediment and P mobilisation at (a) the Rosemaund Holbach plots and (b) the Avon demonstration sites.

After each major rainfall event, the runoff that had collected in the tank was measured and recorded, and thoroughly agitated prior to collecting a representative 250 ml sub-sample for determination of SS and P concentrations. For the first two monitoring periods at Holbach, only the larger storm events giving 10-15 mm or more of rain over a 24-48 hour period were generally monitored and any runoff from minor events during these periods was discarded and the tanks cleaned in readiness for the next storm event. At the Avon sites, and in the third monitoring period at Holbach, less intense storms were also included to provide a more continuous record of runoff generation and any very small runoff volumes that were not sampled for SS and P content were left in the tank. Sample P stability tests showed there was little difference between filtering in the field, or back in the laboratory, suggesting the P concentrations can be considered in equilibrium with the SS in the collection tank at sampling. At all sites, rainfall amount and intensity were measured at each site with an automatic rain gauge, supplemented as necessary with data from the nearest meteorological station.

The Holbach plots were left in situ for the duration of the experiment (4 years) with annual re-levelling of collecting gutters and tanks. At the Avon sites, the runoff plots had to be established in the farmers fields under a normal cropping year. The runoff collectors were installed as soon as was practicable after drilling and removed in April to allow fertiliser and spraying operations on the field. At East Knoyle in 2002/03, prolonged wet weather prevented the establishment of any runoff plots. The monitoring period for the early-drilled treatments in 2002/03 was from 4 December to 12 March (10 sampled storm events) and in 2003/04 was from 22 October to 31 March (9 sampled storm events). The monitoring period for the late-drilled treatments commenced on 17 January in 2002/03 and on 23 December (7 January for Wilton and East Knoyle) in 2003/04, with typically 5 storm events monitored.

Sediment characterisation

For selected storm events, the SS in runoff from the Holbach plots at Rosemaund and the Avon demonstration sites was collected for determination of PSD, P content and P sorption properties where there was sufficient sample. At Rosemaund, SS was collected from two single, contrasting storm events (events 19 and 21), and the cumulative SS generated from 8 successive storm events (events 11-18) in 2003/04. For events 11-18, the sediment from successive storm events was stored in cleaned dustbins in order to provide sufficient material to undertake P sorption isotherms. To flocculate the sediment, 10 mL of 0.5M magnesium chloride ($MgCl_2$) was added for every litre of runoff. The runoff was stirred for 5 minutes, left to stand for at least 12 hours to clear and the supernatant discarded by siphoning. If the supernatant was not clear, another 10 mL $MgCl_2$ was added per litre of runoff. The resulting suspension was subsequently washed twice with de-ionized water by stirring, settling and siphoning of clear supernatant in order to remove any magnesium salts that might precipitate on drying. Finally, the washed suspension was left to air dry in a warm oven (<30°C) and the sediment residue gently ground by hand in a pestle and mortar to a fine powder. At each of the Avon demonstration sites, large runoff samples were collected for two storms in the second year (5 November and 24 November 2003) from each trap on the early-drilled treatments. The sediments were not flocculated and the runoff samples were simply allowed to air dry in a warm room and the sediments gently ground.

Each sediment sample was gently homogenised with distilled water to form a paste and a sub-sample dispersed with a few drops of sodium hexametaphosphate (3.3 wt %) buffered with sodium carbonate (0.7 wt %) prior to flushing through a Coulter LS 230 Laser Granulometer (Allen and Thornley, 2004) to determine PSD. Organic matter was not removed from the samples prior to homogenisation and dispersal. Repeat analyses were undertaken where distributions appeared skewed, but generally one analysis was carried out on each sediment sample. The ratio of the P in the sediment to the P in the field soil was calculated and termed the P enrichment ratio (PER, Sharpley, 1980). Values of PER were calculated for TP, OP and WP. Clay and silt enrichment ratios (CER and SER) were similarly calculated.

Sources of rural runoff

Representative spot (grab) samples of storm runoff were collected from different sources within each of the three catchments in 2005/06 and 2006/07. Sites were selected on the basis of visual evidence of regular runoff and sampled once during each storm event for determination of SS, P forms (DRP, TDP and TP), N forms (ammonium-N, NH_4-N; nitrate-N, NO_3-N and total N, TN) and boron (B). Boron was used as a marker for domestic detergents (Dyer and Caprara, 1997). In some cases water volumes were too low to take a sediment sample. The sites sampled were field surface runoff (1 site), field drain outfall (8 sites), minor roads (4 sites), farmyards (1 site) and a field ditch receiving septic tank discharge (1 site). A breakdown of the numbers of samples from each runoff type is given in Table 3.4. A total of 118 samples were collected over the two years with additional data on flow rates at some sites in the second year, where this was feasible. Spot flow rates at sampling were taken either by timed collection of the runoff in a container of known volume, or measuring the runoff profile (height and width) and timing the speed of flow between two points using a float (piece of cardboard). These flow rates are only very approximate due to the irregular profiles associated with non-channelled surface runoff and the very fast flow rates during some storms where runoff was channelled.

Table 3.4 Sample numbers of storm runoff types collected in the Wye river basin.

Catchment	Field surface	Field drain	Road	Farmyard	Field Ditch	Total
Whitchurch	-	-	12	-	-	12
Dinedor	13	7		-	13	33
Kivernoll	-	32	17	13	-	62
Rosemaund	-	-	-	11	-	11
Total	13	39	29	24	13	118

In a more detailed study over the 2006/07 winter, storm runoff samples were taken more frequently from three separate tile drain outfalls with variable soil P status but similar soil type. Two of the field drains were in the Kivernoll catchment and one field was in the Dinedor catchment. Samples were taken manually every 1-2 hrs during four or five storm events in January 2007. Similarly, SS and P concentrations in runoff entering two large road drains feeding into the stream channel at Whitchurch were also monitored in the same manner over the 2006/07 winter.

3.5.3 Catchment scale

At all monitoring points in all catchments, transfers of SS and P forms were calculated from simultaneous measurements of flow and concentration. Flow was measured continuously either through thin plate weirs (Rosemaund) or control sections (Redesdale, Avon, Whithcurch, Dinedor and Kivernoll), e.g. Plate 3.6. The stage height through these weirs/sections was monitored electronically using either a rotary potentiometer linked to a float, an ultrasonic device or a Unidata capacitance probe. Calibration of stage height with flow were undertaken using a *Valeport (Model 801)* electromagnetic flow meter. Turbidity meters (*D & A Instrument Company*) were used to continuously monitor SS concentrations.

(a)

(b)

Plate 3.6. Flow was continuously monitored over (a) a thin-plate weir at the Jubilee station at Rosemaund and (b) through a purpose-built control section at the RD3 station at Redesdale.

At Rosemaund and Redesdale, flow data were recorded onto Campbell CR10 dataloggers every 5 minutes and downloaded daily via a telemetry link. A computer program in the datalogger used the stage height data to trigger automatic water sampling on a flow-proportional basis during storm events (1-3 hour intervals), and daily during base flow to determine background concentrations. Not all baseflow samples were analysed (typically one sample was analysed every 3 days) and the sampling programme was therefore biased towards storm events. Tile drain monitoring at Rosemaund was targeted at periods when the drains were actively flowing with a typical sampling interval during storm flow of 1-3 hours depending on the size and duration of the storm. A tipping bucket rain gauge linked to the datalogger provided a continuous record of rainfall at each site. Stream grab samples were taken as necessary to supplement automatic sampling.

At the East and West Avon, Whitchurch, Dinedor and Kivernoll catchments, manual water quality samples were collected on a weekly basis and filtered (using 0.45 µm cellulose nitrate filter membranes) in the field to minimise sample deterioration prior to analysis for P forms within 24-48 hours. Suspended solids were also analysed on these samples. At East and West Avon, daily samples were also collected using an automatic Isco sampler but were only analysed for TP and SS. The baseflow index (BFI, the proportion of total flow occurring as baseflow) was calculated for each of the catchments using the Dingman slope method (Dingman, 1994). This method separates a stormflow component when the gradient of the flow hydrograph exceeds a gradient of 13.12 L s^{-1} km^{-2} day^{-1}, and returning to baseflow when flow falls below this gradient.

3.6 Analytical methods

3.6.1 Soils and sediments

For general characterisation of soils, PSD, $CaCO_3$, OM and pH were determined according to standard laboratory techniques (Ministry of Agriculture, Fisheries and Food, 1986). Total P, Fe, Al and Ca concentrations in soil and in SS were determined by Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) after aqua-regia (nitric and hydrochloric acid) digestion (Methods for the Examination of Waters and Associated Materials, 1986). OP was determined by the method of Olsen et al. (1954), and WEP in soil was determined by colour (Murphy and Riley, 1962) after shaking 5g soil in 50ml distilled water for 16 hours and filtering through a Whatman No. 2 filter. OP and WEP analyses were done in duplicate. Oxalate-extractable P, Fe and Al in soils were determined by ICP-AES after extraction in the dark (Schoumans, 2000) and the molar ratio of P/Fe+Al calculated as the degree of P saturation (DPS) without incorporating an α factor.

At Holbach, adsorption isotherms were conducted on the original soil, and the cumulative sediment collected from storm events 11-18, on each plot. 1g of soil was shaken with 20 ml of 0.01 M $CaCl_2$ containing various concentrations of P (0 to 100 mg L^{-1}) for 16 hours at 20°C (± 2°C). The range in initial P concentrations was varied to produce final solution P concentrations (c) in the range 0-30 mg l^{-1}, and also to allow assessment of the equilibrium P concentration (EPC_0, defined as the point of zero net P sorption). The amounts of P adsorbed by the soil or sediment from solution (Q) were related to c using either a single-surface (high soil P concentrations), or a two-surface (low soil P concentrations) Langmuir function (Holford et al., 1974). Values of EPC_0, the binding energy constant (k) and the maximum P sorption capacity (Q_{max}) were computed from the fitted function. Q_{max} was computed after addition of native P, and k was computed after subtraction of the EPC_0 to represent the average P binding energy over the full range of final P concentrations (Holford et al., 1997). OP was taken to represent the native P and when expressed as a proportion of Q_{max} represented the percentage soil and sediment P saturation (P_{sat}).

3.6.2 Land runoff and streams

Land runoff and stream samples were collected as soon as practicable after storm events and analysed for total P (TP), total dissolved P (TDP) and dissolved-reactive P (DRP). Dissolved fractions were measured after passing the sample through a 0.45 μm cellulose filter. DRP was determined by colour (Murphy and Riley, 1962, Eisenreich et al., 1975) after reaction with phosphomolybdenum blue and with adjustment for silica interference. TDP and TP were determined either by ICP-AES following digestion with nitric and hydrochloric acid (aqua regia), or colorimetrically after persulphate digestion (Methods for the Examination of Waters and Associated Materials, 1980; 1986). A comparison between the two methods showed very good agreement over the range in SS concentrations encountered. The detection limit (DL) was 1μg L^{-1} for DRP, 1 and 2 μg L^{-1} for TDP and TP using the colour method, and 50 and 100 μg L^{-1} for TDP and TP, respectively using ICP. A particulate P fraction (PP) was defined as the difference between TP and TDP, and a dissolved unreactive P fraction (DUP) was defined as the difference between TDP and DRP.

In the Whitchurch, Dinedor and Kivernoll catchments, stream and storm runoff samples were additionally analysed for nitrate-N (NO_3-N), ammonium-N (NH_4-N), total dissolved inorganic N (TDN) and boron (B).Nitrate-N was determined directly using ion chromatography (Neal et al., 1998), ammonium-N was determined colorimetrically using an indophenol blue method (Leeks et al., 1997) and TDN was determined using automated thermal oxidation to NO, which is then measured using a chemoluminescent detector. Boron was determined in hot water (Basset, 1980).

3.7 Data handling and statistical analysis

Soil, runoff and stream concentration data were described by distribution statistics and simple or multiple regression analyses with site parameters. Comparisons of mean concentration values between treatments were made using student t test assuming equal or unequal variance depending on sample numbers. For storm runoff sites, differences in mean concentration values between sites within a P source group, or within a catchment, were assessed only for sites with sample numbers greater than 5. Significant differences were assessed by comparing the confidence intervals of mean values after log transformation for determinands with wide ranging values. For the Avon demonstration plots, data from individual runoff plots were taken as independent treatment replicates (no treatment replication), and treatment effects analysed by a one-way ANOVA (Genstat 8 Committee, 2004). Where more than one stream sample was taken in a day (during storm events), the values were averaged to a daily concentration, and where the data were heavily skewed they were log transformed, prior to statistical analysis.

At Redesdale, a large proportion (75% for RD3 and 40% for RD4) of samples taken during the monitoring period contained TDP and TP concentrations below the DL. As many measured values were close to the DL, and to avoid the large bias that would occur in leaving them out, the impact of a number of scenarios for dealing with values below the DL on mean TDP and TP concentrations were compared. Values of TDP and TP can never be zero, since they must be at least the DRP, or TDP, concentration respectively. Hence the scenarios ranged from giving TDP concentrations below the DL the value of DRP (and then giving concentrations of TP below the DL the value of TDP) to giving all values below the DL the value of the DL. Concentration trends within and between catchments were similar whichever scenario was chosen and the scenario that produced the average in the range was used to calculate P loads of the different P fractions. This scenario gave concentrations below 50% of the DL the value of 50% of the DL, and concentrations between 50% and 100% of the DL were assumed to be correct.

Annual catchment loads were calculated as the cumulative product of the total daily flow and the daily average P concentration. No consistent relationships were found between instantaneous flow and sediment or P concentrations. Daily loads for days when no stream samples were taken were therefore interpolated from the log-log rating curves describing the relationship between daily flow and daily load. These relationships had correlation coefficients (*r*) of 0.85-0.95 for P forms and 0.65-0.90 for SS. To compare average annual P transfers in the different sized catchments, annual flow weighted concentrations (mg L^{-1}) of each P fraction were calculated from the total export of P (kg) and the total flow (mm) over the year. In this content, the flow-weighted concentration in mg L^{-1} is equivalent to kg ha^{-1} per 100 mm of flow.

4

FARMING PRACTICES AND PHOSPHORUS LOSS IN THE UPLANDS: REDESDALE

4.1 Introduction

The uplands of Britain occupy approximately one third of the UK land area and include a number of varied and ecologically unique landscapes comprising hills, moors, mountains and marginal agricultural land above the limits of enclosed farmland (Ratcliffe and Thompson, 1988). They have undergone considerable land use change over the centuries and have evolved as largely acidic environments with nutrient poor soils and oligotrophic streams. These streams provide a valuable source of potable water and play an important role in diluting the nutrient rich waters that occur in more densely populated lowlands (Soulsby et al., 2002; Bowes, et al., 2003). Phosphorus is often (but not always) the nutrient most limiting plant and algal growth, with concentrations of dissolved inorganic P often close to detection limits (Rigler, 1979; Hornung, 1984; Neal et al., 2003). Ecological communities are therefore dependent on seasonal phosphatase activity to utilise the predominant organic P forms in soil solutions and streams (Livingstone and Whitton, 1984; Shand et al., 1994; Turner et al., 2003). Upland ecosystems are therefore sensitive to changes in P status, a sensitivity which has been recently accentuated by increasing atmospheric deposition of N (Edwards et al., 2000a). An understanding of P dynamics in upland environments, and land management pressures that influence these dynamics, is therefore important to maintain the relatively pristine quality of upland streams and their unique ecological communities.

Changes in land use and soil fertility in upland areas over the last century have been associated with fluctuating rural populations, agricultural intensification, afforestation, gamekeeping (grouse/deer) and recreational activities (Eadie, 1985; Hornung and Newson, 1986; Soulsby et al., 2002). Agriculture remains a major land use, although the harsh climate and the predominance of waterlogged and/or impoverished soils severely restrict productivity (Floate, 1977). Opportunities for pasture improvement are consequently limited, but significant areas of marginal upland have nevertheless been reclaimed for agricultural production aided by farm subsidies and improved pasture establishment techniques (Eadie, 1985). A report by the Nature Conservancy Council (1984) estimated that 150,000 ha (8%) of moorland in England and Wales were improved for agriculture between 1950 and 1980. Conversion to, and improvement of, upland pastures has also occurred extensively in other countries to increase agricultural output (e.g. Lambert et al., 2000).

Techniques of pasture improvement have ranged from simple methods of direct seeding with grass/legume mixtures to pioneer cropping with stubble turnips (Ministry of Agriculture, Fisheries and Food, 1984; Newbould, 1985). All methods include initial liming and bulk applications of P fertiliser (basic slag, rock phosphates or superphosphates), to raise the inherently poor soil P status, followed by smaller periodic P fertiliser applications to maintain a satisfactory soil P status for optimum grass production (Hornung, 1984; Merrell and Withers, 1987). Such applications, and consequent higher grazing intensities, are known to increase the cycling of inorganic and organic P in upland soils (Harrison and Taylor, 1987). Increased P loss in land runoff has been reported from field areas receiving soluble P (and N) fertilisers during, or soon after, pasture improvement (Roberts et al., 1989; Williams and Young, 1994; Cuttle and James, 1995). Conversely, Roberts et al. (1989) found no increase in P transfer after P fertiliser (119 kg P ha^{-1}) was cultivated into a peat over clay soil in a 1.5 ha field lysimeter study. Soil type, the method of improvement and the solubility of the fertilisers applied appear to be major factors influencing mobilisation and transport of P in upland ecosystems

At the catchment scale, pasture improvement is not considered to have a major influence on the chemistry and ecology of upland streams in the UK, especially with respect to P, although data are very limited. For example, Hornung (1984) monitored three variably improved catchments in Mid Wales and found orthophosphate concentrations remained below the detection limit (5 $\mu g\ L^{-1}$) in all three streams. Fixation of the historically applied P by Fe-rich sub-surface layers of the stagnopodzolic soils was thought to have reduced soil P release to runoff. More anecdotal evidence has suggested that stream P concentrations in catchments with 15-25% of land improved remain largely unaltered (Roberts et al., 1989; Cuttle and James, 1995). The proportion of improved land in the catchment, hydrological connectivity between the field and watercourse and the extent of dilution by groundwater, will likely determine whether any additional P mobilised in surface runoff from improved areas can be detected or not in draining streams (Hornung, 1984; Soulsby et al., 2002).

None of the previous studies have examined the effects of pasture improvement on transport of particulate P, or P transfer dynamics during storms, even though P loss is actively associated with eroding soil particles. Inputs of particulate P may be of ecological significance if improved areas are a source of sediments with high equilibrium P concentrations that will release dissolved inorganic P into downstream oligotrophic water columns (Jarvie et al., 2005). Transfers of particulate P may also be indirectly increased where higher stock densities on improved areas increase the risk of soil wash and erosion during storms, and by stream bank collapse (Nature Conservancy Council, 1990; Grieve et al., 1995; James and Alexander, 1998). More catchment-based data are therefore required to assess potential water quality impacts associated with pasture improvement and P fertilisation, without the confounding effects of other anthropogenic P sources.

At ADAS Redesdale, a technique of pasture improvement involving cropping with stubble turnips prior to reseeding was developed to improve grass productivity and sustain higher sheep stocking rates on the peaty upland soils in the area. A significant area of the marginal hill pasture at Redesdale was improved in this way during the 1970s and 80s and subsequently managed with increased stocking rates and annual fertiliser inputs. Since the majority of this pasture improvement was undertaken in one part of the Redesdale farm, the opportunity was taken to compare the changes in the P status of peaty soils and streams draining two adjacent sub-catchments with different proportions of improved pasture. Previous monitoring of these streams before extensive application of P fertilisers indicated they had very similar and low P status. The objective of the study was to assess how pasture improvement influenced the magnitude and timing of stream fluxes of dissolved and particulate P forms and their ecological significance. It was hypothesised that pasture improvement would increase stream P fluxes due to the build-up of soil P during the improvement phase rather than any direct release of P from annual maintenance applications of P fertiliser, or increased excretal deposition, which only occurs when rainfall coincides with P application/deposition.

4.2 The Redesdale catchment

The Redesdale catchment is located in the uplands of Northumberland where marginal hill pasture land and moorland provides rough grazing for sheep and beef during summer months. Productivity is limited by the nutrient-depleted and poorly-drained peaty over clay soils and various pasture improvement techniques have been investigated at Redesdale in an attempt to raise productivity and income on upland farms in the area. One technique involving pioneer cropping with stubble turnips to break up the surface mat of Nardus species proved particularly successful and was adopted over a large area of the farm during the 1970s and 80s (Fig. 4.1). Comprehensive details of the technique are given in Chapter 3 and by Davies (1978).

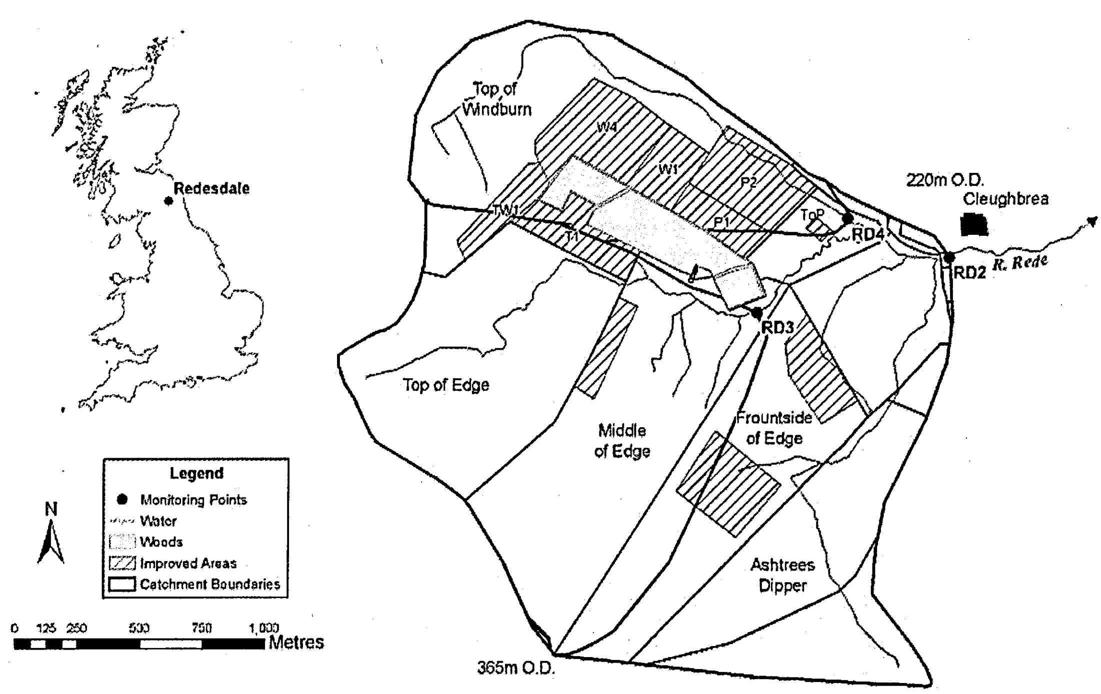


Figure 4.1 Location of the RD3 and RD4 catchments in relation to the distribution of improved and unimproved areas at Redesdale.

The effect of pasture improvement on soil (Olsen) P (OP) concentrations to a depth of 15 cm was monitored between 1977 and 1986 and more irregularly to 7.5 cm depth thereafter to determine the need for additional P inputs to maintain adequate soil P fertility. The effect of pasture improvement on stream P concentrations and loads in two subcatchments (RD3 and

RD4) was monitored between 1994 and 1997. RD3 contained only 7% of improved pasture while RD4 contained 47% of improved pasture. Historic data on DRP concentrations in both RD3 and RD4 streams, and upstream of improved areas, during 1973 and 1974 provided data prior to pasture improvement (Webber and Wadsworth, 1976). Additional samples of both soils and streams were taken during 2006 to determine differences in potential in-situ mobilisation of soil P during raindrop impact, current levels of stream P upstream of improved areas and analytical comparisons. More comprehensive details of the Redesdale catchment, the land use history and methodologies used to measure soil and stream P concentrations and loads are provided in Chapter 3.

4.3 Effect of pasture improvement on soil phosphorus

4.3.1 Soil phosphorus

Total inputs of P to the improved areas prior to the commencement of stream monitoring in 1994 ranged from 143 to 429 kg P ha^{-1} depending on when the area was first improved and the type of improvement (Table 4.1). Amounts applied during the improvement phase were typically ca. 150 kg P ha^{-1} (range of 134 to 199 kg P ha^{-1}), with further periodic maintenance dressings (equivalent to ca. 10 kg P ha^{-1} yr^{-1}) thereafter. The largest P input was to P1/P2, which was directly connected to the RD4 stream (Fig. 4.1).

Table 4.1. History of P inputs to the main improved areas within RD4 at Redesdale, and the results of soil analysis (Olsen-P (OP), 0-7.5 cm) before, during and at the end of the catchment monitoring period (1994-1997).

Site	Area	Year	Total P inputs		OP[b]		
	(ha)	Improved	up to 1993[a]	1994-1997	1992	1995	1997
			(kg P ha^{-1})			(mg L^{-1})	
P1	17.2[c]	1974	429 (148)	26	31[c]	16[c]	13
P2		1979	400 (199)	26			23
W1	6.8	1975	410 (148)	26	23	15	13
W4	10.5	1981	221 (151)	26	18	11	11
T1	7.3	1977	302 (146)	26	n.d.	20	16
TW1	3.4	1989	143 (134)	83[d]	48	21	15

[a]The amounts of P applied during the improvement phase are given in brackets.
[b]Target concentrations (0-7.5 cm) for optimum grass growth are in the range 16-25 mg L^{-1} (Ministry of Agriculture, Fisheries and Food, 2000).
[c]The P1 and P2 areas were not sampled separately until 1997.
[d]Includes 74 kg P ha^{-1} applied as a single dressing of triplesuperphosphate in May 1997.
n.d. – not determined

The improved areas were enclosed so there was minimal P transfer from improved to unimproved areas allowing an approximate P balance to be calculated. Cumulative total P inputs from the date of improvement up to 2005 when soil total P (TP) analysis was undertaken can be estimated at ca. 300-500 kg ha^{-1} assuming similar maintenance P fertilisation rates after 1997. Soil TP concentrations to 15 cm depth (bulk density of 0.5 g cm^{-3}) were very variable with concentrations of 421-734 mg kg^{-1} in unimproved areas and 767-1133 mg kg^{-1} in improved areas. The applied P was therefore largely accounted for in the top 15 cm.

Annual soil analysis of improved areas P1, W1 and T1 from 1977–1986 (1982 for W1) showed preferential accumulation of the applied P in the top 2.5 cm of the soil, with OP concentrations rising to 40-50 mg L^{-1} (P index 3/4) after large single applications (Fig. 4.1). The rates of soil P accumulation in this shallow surface layer were similar for all three areas, increasing by ca. 20 mg L^{-1} for every 100 kg P ha^{-1} applied ($P < 0.001$, r^2 0.92). These levels declined when no P fertiliser was applied, or when smaller annual P applications were made in subsequent years (e.g. in August 1983/84 for P1 and 1984/85 for T1). OP concentrations fluctuated less below 2.5 cm and tended to decrease with increasing soil depth, but with evidence of P movement to the 5.0–7.5 cm and 7.5-15 cm horizons as soil P concentrations in the top 2.5 cm declined. For example, in P1 and W1 in 1979, and in P1 in 1985 (Fig. 4.2).

Averaged over the standard soil sampling depth of 0-7.5 cm used in fertiliser recommendation systems on grassland (Ministry of Agriculture, Fisheries and Food, 2000), OP concentrations were maintained in the range 16-25 mg L^{-1} for optimum grass growth. This is in contrast to the unimproved areas which contained only 4-6 mg L^{-1}. Hence, at the start of stream monitoring, soil P fertility on improved areas was adequate (Index 2, W1 and W4), or above adequate (Index 3, P1/P2 and Index 4, TW1), depending on recent P applications (Table 4.1). As observed in the earlier soil P data, OP concentrations generally declined during the catchment monitoring period despite the annual inputs of 9 kg P ha^{-1}, with a faster rate of decline where soil P levels were initially high. These fluctuations in OP indicate a degree of P mobility within these peaty topsoils, with potential for P mobilisation in surface runoff.

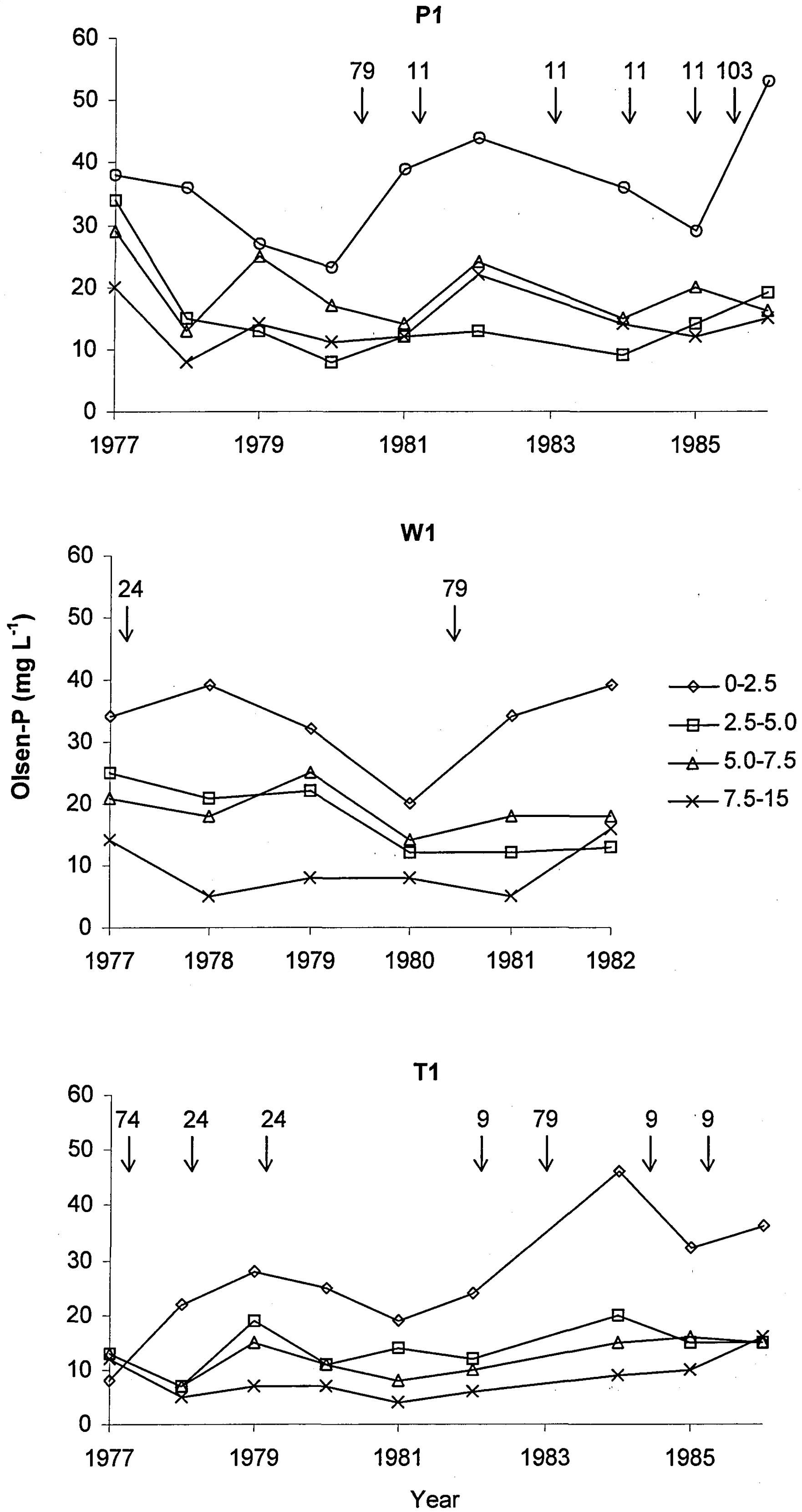


Figure 4.2 Changes in Olsen-P (OP) with increasing soil depth to 15 cm for three areas (P1, W1 and T1) improved by pioneer cropping in 1977. Arrows indicate the timing and amounts of fertiliser P applied.

In-situ mobilisation of phosphorus

Potential mobilisation of dissolved P (<0.45 μm) and particulate P due to rainfall impact in storm runoff from unimproved and improved areas was assessed by the DESPRAL test (Withers et al., 2007). Unimproved areas within RD4 and RD3 were compared with the improved areas P1, W1 and T1 (Table 4.2). Potential mobilisation of soil particles was variable (range 113-243 mg L^{-1}) with no clear pattern between unimproved or improved areas but with a significant difference (P<0.001) between fields T1 and RD4-unimproved and the other fields. These differences may relate to differences in organic matter (OM) content. Some data for OM content from different unimproved and improved areas suggest values vary from 20-90% based on loss on ignition.

Table 4.2 Concentrations of suspended solids (SS), total dissolved P (TDP), particulate P (PP), total P (TP) and P in SS (SS-P) potentially mobilised in field runoff as predicted by the DESPRAL test.

Field area	SS (mg L^{-1})	TDP (μg L^{-1})	PP (μg L^{-1})	TP (μg L^{-1})	SS-P (mg kg^{-1})
Improved					
P1	113	31	179	210	1576
W4	153	29	175	203	1139
T1	238	16	227	243	952
Unimproved					
RD3	130	25	78	102	597
RD4	243	28	105	133	432
P value	<0.001	NS	<0.01	<0.01	<0.01

NS, not significant

Concentrations of TDP, which includes both the reactive and unreactive fractions, were very similar across all areas and represented between 7 and 24% of TP concentrations. Concentrations of PP and TP were up to twofold greater (P <0.01), and the calculated P content of the SS (SS-P) was up to nearly four-fold greater (P <0.01), on improved areas compared to unimproved areas (Table 4.2). Values of SS-P were greatest on P1 which had received the highest amount of P during and subsequent to pasture improvement (Table 4.2).

4.4 Effect of pasture improvement on stream phosphorus

4.4.1 Catchment streamflow

Variations in streamflow broadly followed seasonal rainfall patterns but there were some notable differences between years. Annual rainfall in 1994/95 (909 mm) and 1996/97 (888 mm) was close to the long-term (1971-2000) average for Redesdale of 893 mm, but was ca. 200 mm lower in 1995/96 (682 mm) resulting in lower total streamflows in both catchments (Fig. 4.3a,b). The April to August period in 1995 was particularly dry resulting in lower flow rates in RD3 compared to RD4 in the first year (Fig. 4.3b). Conversely, very high rainfall in February 1997 resulted in higher flow rates in RD3 than in RD4 in the third year. Baseflow separation analysis confirmed a lower baseflow index in RD3 (ca. 0.65) than in RD4 (ca. 0.85) due to the surface drainage grips directing stormflow down the steeper slopes on RD3's northern boundary. The greater contribution of stormflow to total flow in RD3 was reflected in a consistently faster flow response causing higher peak flow rates in RD3 than in RD4 (Fig. 4.3c). Streamflow generally reached a maximum in February each year (average of 116 mm in RD3 and 84 mm in RD4) and a minimum in August (7 and 13 mm, respectively). Snowmelt also contributed to flow at various times, most notably during January, February and March 1996 (up to 30 cm of snow) and during January 1997 (up to 16 cm of snow) causing streamflow to sometimes exceed rainfall.

4.4.2 Stream phosphorus

To assess the effect of pasture improvement on P transfer, average daily stream P concentrations in RD3 and RD4 were compared. Summary data of orthophosphate P concentrations in the streams draining RD3 and RD4 before any extensive application of P fertilisers to RD4 are given by Webber and Wadsworth (1976). They cover a period of 16 months between 1973 and 1974 when samples were taken at approximately monthly intervals. Mean concentrations in RD3 and RD4 streams just before they join, and in RD4 upstream of the subsequently improved areas, were 13 (range 6-30 $\mu g\ L^{-1}$), 11 (range 5-32 $\mu g\ L^{-1}$), and 11 $\mu g\ L^{-1}$ (range 3-26 $\mu g\ L^{-1}$), respectively. With regard to dissolved P concentrations, the two catchment streams were therefore identical in P status. No data are available for DUP and PP concentrations pre-improvement. Instantaneous concentrations of DRP, TDP and TP during the 1994-1997 monitoring period ranged up to 0.68, 0.76 and 4.0 mg L^{-1}, respectively in RD3 and 0.13, 0.18 and 2.1 mg L^{-1}, respectively in RD4 (Table 4.3).

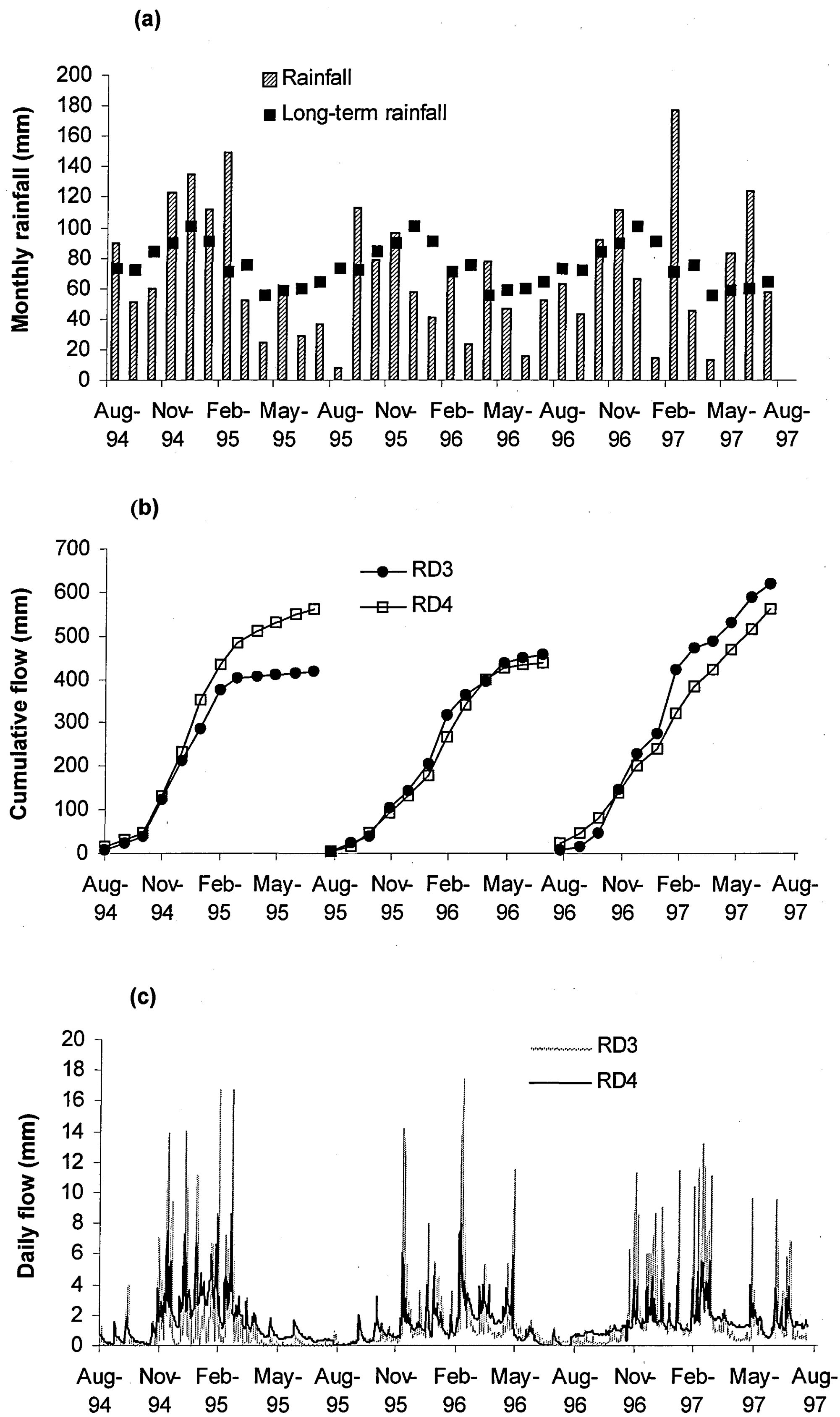


Figure 4.3 Rainfall and flow patterns in each catchment: (a) monthly rainfall in relation to the long-term (1971-2000) mean, (b) cumulative flows in each year and (c) variations in daily flow across the monitoring period.

Table 4.3 Distribution statistics for daily flow and concentrations of dissolved reactive P (DRP), total dissolved P (TDP), total P (TP) and suspended solids (SS) in streams draining RD3 (7% improved land) and RD4 (47% improved land).

	n	Range	Mean (s.e.)	25%	50%	75%
Flow (mm day^{-1})						
RD3	288	0.01-17.6	2.6 (0.2)	0.7	1.3	3.1
RD4	199	0.01-8.5	2.1 (0.1)	1.0	1.7	2.9
DRP (μg L^{-1})						
RD3	288	1-681	26 (5.1)	5	8	13
RD4	199	1-127	23 (1.5)	10	14	29
TDP (μg L^{-1})						
RD3	288	25-759	54 (5.5)	25	33	48
RD4	199	25-181	56 (2.4)	27	46	72
TP (μg L^{-1})						
RD3	288	50-4010	119 (18.8)	50	50	81
RD4	199	50-2110	200 (17.9)	56	110	217
SS (mg L^{-1})						
RD3	819	0.7-1021	93 (3.1)	40	71	116
RD4	628	0.5-859	95 (4.9)	16	55	124

Concentration data were variably skewed by high DRP, TDP and/or TP concentrations (Table 4.3), which in RD3 were up to an order of magnitude greater than annual average values and most likely due to direct faecal contamination by sheep grazing close to, or in, the stream. After log transformation, mean concentrations of DRP, TDP and TP were significantly ($P<0.001$) higher in RD4 than in RD3. Values were 16 vs 9 μg L^{-1} for DRP; 48 vs 38 μg L^{-1} for TDP and 129 vs 72 μg L^{-1} for TP. Although mean TDP (RD3 and RD4) and TP (RD3) values were below detection limits, they gave excellent agreement with single sample concentrations in RD3 of 30 and 69 μg L^{-1} taken on 30 November 2006 using the much more sensitive persulphate and colour P determination method. Similarly, TDP and TP concentrations upstream of improved areas within RD4 in November 2006 were 37 and 81 μg L^{-1}, respectively.

Mean daily concentrations of the different P forms DRP, DUP and PP were therefore calculated as 9, 29 and 34 μg L^{-1} in RD3 and 16, 32 and 81 μg L^{-1} in RD4. Hence, the majority of the significant increase in P concentrations in the 47% improved RD4 catchment was in the DRP and PP fractions, rather than in the DUP fraction despite this being the dominant dissolved P fraction in both streams. The proportional contribution of DRP to TP remained the same in both catchments (ca. 12%), whilst particulate P (PP) represented 50% of TP in RD3 and 63% of TP in RD4. Hence, whereas dissolved P forms were slightly more dominant in RD3, particulate P forms became more dominant in RD4 despite the latter having a higher contribution of baseflow. Although the number of baseflow samples collected was relatively low (11% of total samples in RD3 and 38% in RD4), P concentrations were significantly ($P < 0.05$) higher in RD4 than in RD3: + 5 μg DRP L^{-1}, + 8 μg TDP L^{-1} and + 53 μg TP L^{-1}. The apparent large contamination of baseflow with TP suggests some mobilisation of particulate material by livestock, and/or a higher algal biomass, in the RD4 stream. Corresponding increases (RD4 vs RD3) for days with a stormflow component were + 10 μg DRP L^{-1}, + 19 μg TDP L^{-1} and + 69 μg TP L^{-1}; $P < 0.001$. Larger differences in mean P concentrations between the two catchments were therefore obtained under stormflow than under baseflow.

Analysis of a number of storm events common to both RD3 and RD4 showed higher peak flow rates in RD3 than in RD4 but higher concentrations of DRP and PP in RD4 than in RD3. Typical example trends for a storm event during December 1996 are shown in Fig. 4.4. Flow peaked twice on 18 and 19 December 1996; the higher peak measured ca. 0.6 mm in RD3 but only 0.23 mm in RD4. In the largely unimproved RD3 catchment, DUP and PP concentration responses were similar for both peaks (maximum ca. 60 μg L^{-1}), whilst DRP concentrations increased up to only 25 μg L^{-1}. In the 47% improved RD4 catchment, a large increase in PP concentrations (maximum 270 μg L^{-1}) occurred with the first peak flow, with much lower concentrations (ca. 0.1 mg L^{-1}) associated with the second flow peak. This suggests that P-enriched soil (or particulate P derived from faecal matter) from RD4 became detached during the rising limb of the storm hydrograph, but that the supply became exhausted by the time of the second flow event. In contrast, the dissolved P fractions showed the reverse, with maximum concentrations (DUP ca. 0.14 mg L^{-1} and DRP ca. 0.1 mg L^{-1}) coinciding with the broader second peak (Fig. 4.4). This suggests that runoff water in the second flow event had greater contact time with the soil allowing greater opportunity for soluble P release by desorption and dissolution. These trends were absent in RD3 owing to the lack of P enrichment at the soil surface.

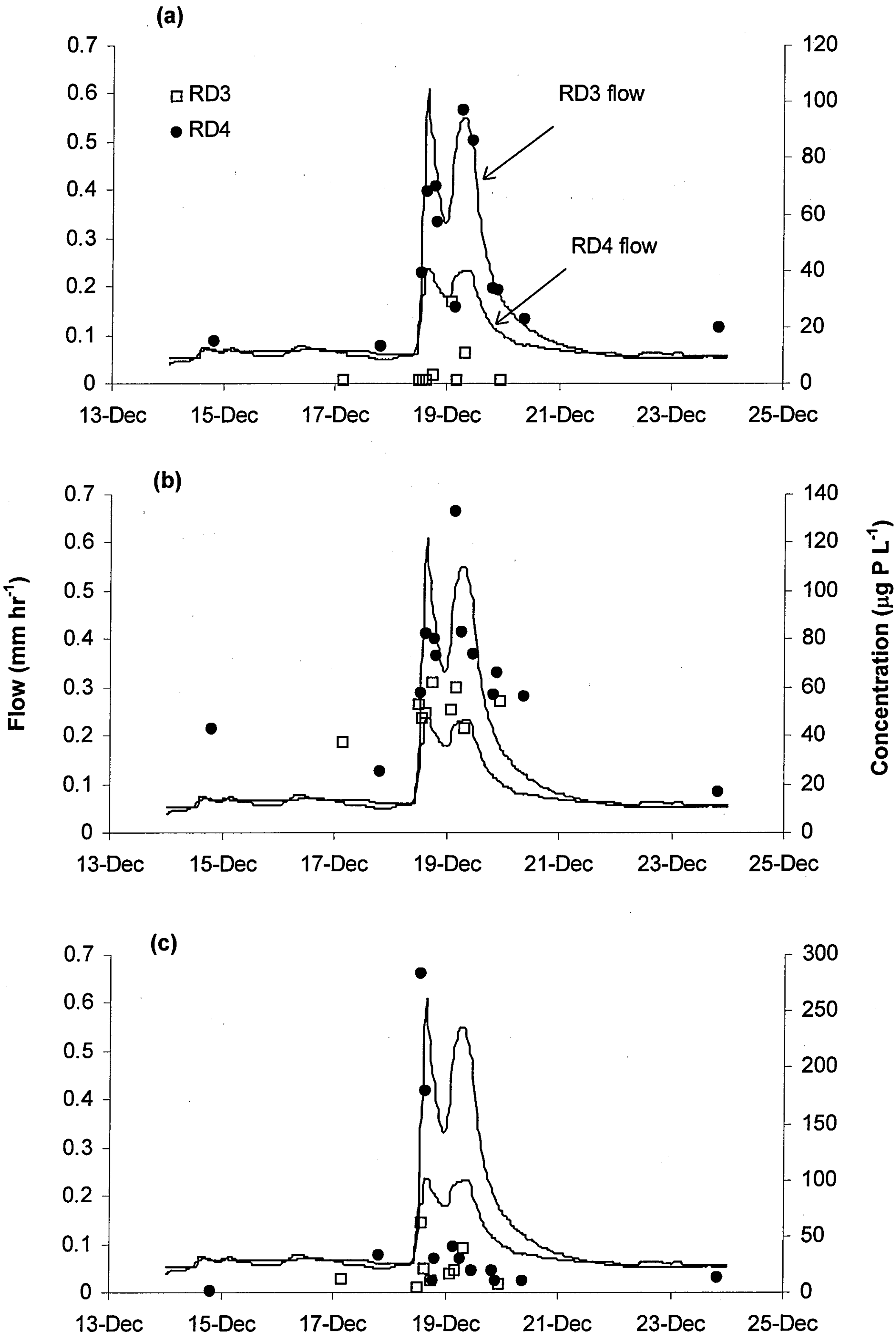


Figure 4.4 Flow and concentrations of (a) dissolved reactive P (DRP), (b) dissolved unreactive P (DUP) and (c) particulate P (PP) in RD3 and RD4 streams during a storm event in December 1996.

4.4.3 Annual phosphorus export

Annual stream loads of TP ranged between 0.31 – 0.52 (mean 0.39, s.e. 0.06) kg ha^{-1} in RD3 and between 0.65 – 0.95 (mean 0.79, s.e. 0.09) kg ha^{-1} in RD4. Smallest loads were obtained in the dry year of 1995/96 and as the BFI decreased (increased proportion of stormflow) the loads of all P forms increased. For example, stream load data in RD3 and RD4 for each month of 1996/97 (the year with the largest number of samples) are shown in Fig. 4.5. For both PP and DUP, there was a statistically significant difference ($P < 0.001$) between the two catchments in the intercept value but not the gradient. The intercepts for DUP and PP were 15 g ha^{-1} (s.e.d. 3.1 g ha^{-1}) and 37 g ha^{-1} (s.e.d. 6.0 g ha^{-1}), respectively greater in RD4 than in RD3 (Fig. 4.5). This suggests that the supply and rates of PP and DUP loss were similar (i.e. common transport mechanism) for both catchments but that the magnitude of the P loss was significantly higher for RD4 than for RD3 due to P enrichment. For DRP, the relationships between BFI and load for RD3 and RD4 were completely independent, with both a significantly ($P < 0.001$) different intercept and gradient (r^2 0.65). The relationship for RD3 was described by the equation, DRP = 0.027 kg ha^{-1} – 0.087*BFI, whilst that for RD4 was described by the equation DRP = 0.069 kg ha^{-1} – 0.065*BFI (Fig. 4.5). These two diverging regression lines for DRP reflect the greater flow response as BFI decreased diluting the limited rate of DRP supply in RD3 but not in RD4.

After accounting for variation in flow, the mean flow-weighted TP concentrations in each stream were much less variable between years, averaging ca. 78 $\mu g\ L^{-1}$ for RD3 and 153 $\mu g\ L^{-1}$ for RD4 (Table 4.4). DRP and PP concentrations were more than doubled (DRP, 10 to 21 $\mu g\ L^{-1}$; PP, 39 to 97 $\mu g\ L^{-1}$), while flow-weighted concentrations of DUP were only slightly increased (29 to 35 $\mu g\ L^{-1}$). Temporal analysis of mean monthly flow-weighted concentrations over the 3-year monitoring period suggested that the increase in dissolved P (DRP and DUP) in RD4 compared to RD3 occurred more during the winter months (November – March) than during summer months (Fig. 4.6). For example, average DRP concentrations increased by between 7 and 19 $\mu g\ L^{-1}$ in winter months and between 3 and 8 $\mu g\ L^{-1}$ in summer months. In contrast, the largest consistent increases in PP occurred in July, August and September. Differences in monthly flow-weighted concentrations were not related to whether the improved areas were either ungrazed or grazed by sheep (Fig. 4.6).

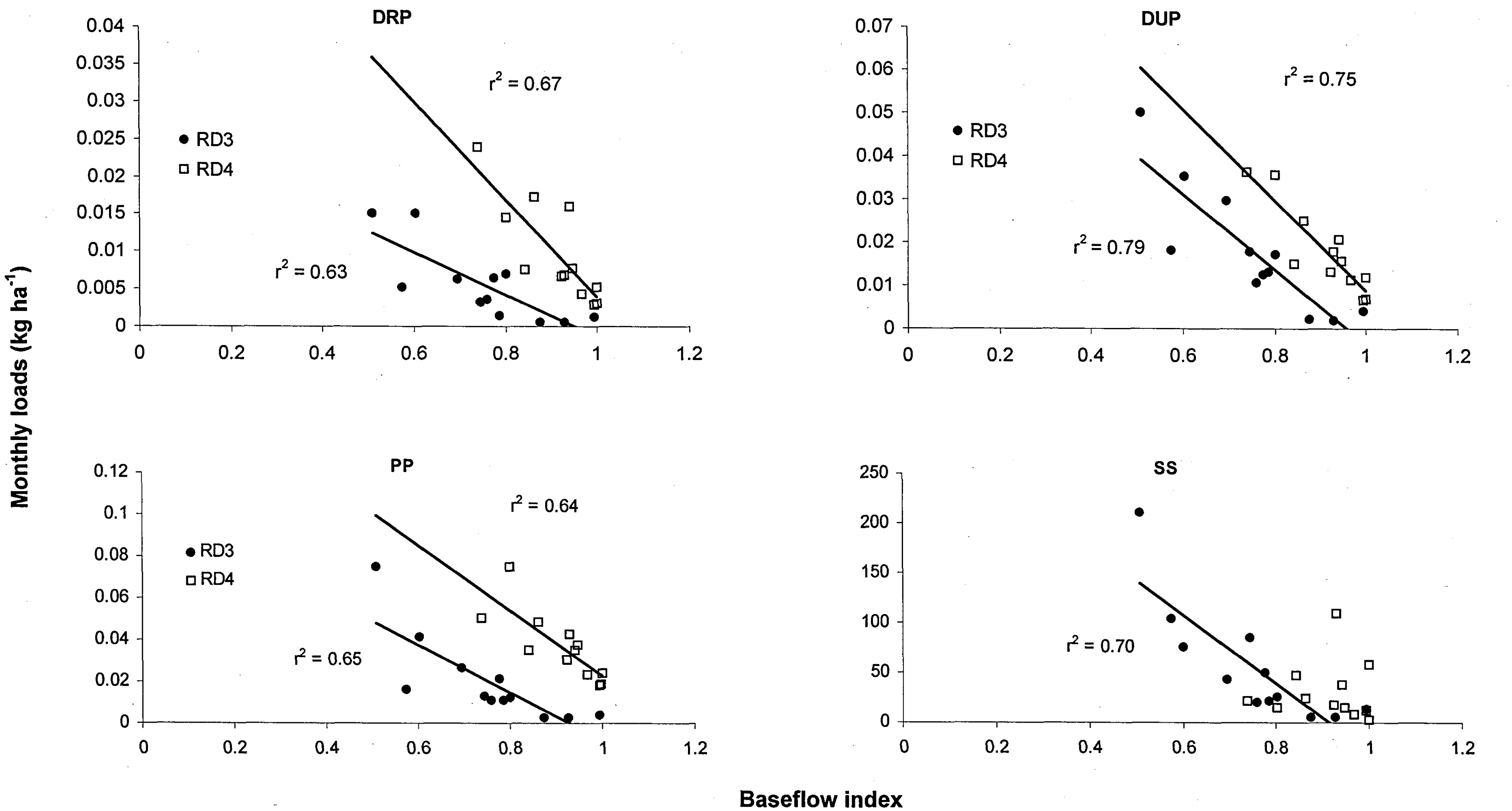


Figure 4.5 The effect of baseflow index on fluxes of dissolved reactive P (DRP), dissolved unreactive P (DUP), particulate P (PP) and suspended solids (SS) in RD3 and RD4 for each month during the 1996/97 monitoring year. Error bars represent the s.e.d. between the intercepts of RD3 and RD4 regression for DUP and PP.

Table 4.4 *Annual flow and flow-weighted concentrations* of dissolved *reactive* P (DRP), dissolved unreactive P (DUP), particulate P (PP), total P (TP) and suspended solids (SS) at each monitoring station.

Station	Monitoring period[a]	Flow[b] (mm)	MRP	DUP	PP	TP	SS (mg L^{-1})
				(µg L^{-1})			
RD3	1994/95	418	9	28	38	75	86
	1995/96	456	11	26	41	76	139
	1996/97	619	11	34	38	83	107
	Mean	498	10	29	39	78	110
RD4	1994/95	561	23	33	114	171	80
	1995/96	437	19	33	99	150	55
	1996/97	560	21	38	78	137	66
	Mean	519	21	35	97	153	67

[a]1 August - 31 July in each year. [b]Total over the monitoring period.

4.4.4 Stream suspended solids

Concentrations of SS ranged up to ca. 1 – 1.5 g L^{-1} (Table 4.3) with annual loads ranging between 0.36-0.66 t ha^{-1} in RD3 and 0.24 to 0.45 t ha^{-1} in RD4. Taking out the flow factor, annual flow-weighted SS concentrations averaged 110 mg L^{-1} in RD3 and 67 mg L^{-1} in RD4 (Table 4.4). Analysis of instantaneous daily concentration data after log transformation also showed a significantly ($P < 0.001$) higher SS concentration in RD3 (64 vs 43 mg L^{-1}), but with a strong contrast between concentrations measured under baseflow and those measured on days with a stormflow component. For storm flow samples, mean concentrations of SS in RD4 were only 18 mg L^{-1} compared to 72 mg L^{-1} in RD3, reflecting the greater mobilisation of particulate material in the faster peak flow rates in RD3. Concentrations of SS in baseflow in the two catchments were however very similar (ca. 50 mg L^{-1}), mainly because of relatively high SS values measured in RD4 during summer months (400-700 mg L^{-1}), which were most probably associated with algal growth. This strong contrast between baseflow and stormflow concentrations in RD4 was absent in RD3.

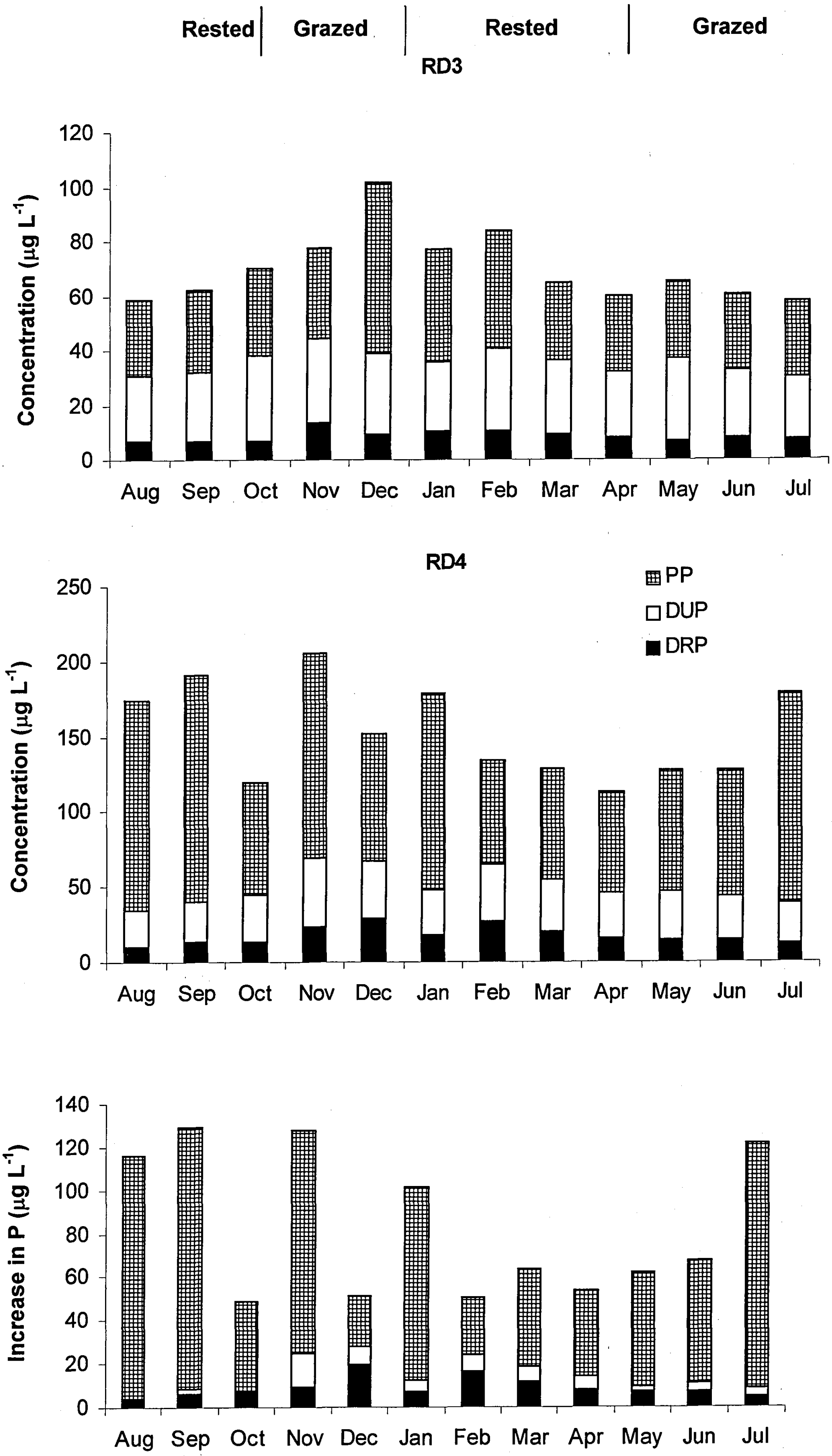


Figure 4.6 Temporal variation in the monthly flow-weighted concentrations of dissolved reactive P (DRP), dissolved unreactive P (DUP) and particulate P (PP) in (a) RD3 compared to (b) RD4 and (c) the resulting increase.

4.5 Discussion

Redesdale is typical of upland farming at lower altitudes where improvement of marginal agricultural land on the better soils can support an increase in sheep grazing intensity from 1 to 2.5 ewes ha^{-1} at different times of the year with annual N inputs of 60-80 kg ha^{-1}. Davies (1978) found that improving selected grazing areas could double the productivity of such marginal upland farms which according to Eadie (1985) produce over 80% of the total output from all hills and uplands. Profit margins in agriculture have declined in recent years and little pasture improvement is now undertaken. Recent changes in price support mechanisms under the EU Common Agricultural Policy is likely to further reduce livestock numbers in the uplands. However, even if fertiliser application rates and livestock numbers drop, areas that have been historically improved may still represent a long-term source of elevated nutrient concentrations and nutrient imbalances. Our study therefore provided a unique opportunity to investigate the impact of historic pasture improvement on P transfer at the catchment scale without interference from other anthropogenic P sources. Previous monitoring confirmed that RD3 and RD4 had identical P status before extensive application of P fertiliser.

Pasture improvement influences stream P fluxes by increasing P transfer from either the soil, fresh fertiliser applications and/or livestock excreta. However, no significant increase in P flux following the large single application of TSP (74 kg P ha^{-1}) to TW1 in May 1997 was detected in either RD3 or RD4, despite the fact that this fertiliser is highly water-soluble. This could be due to the lack of rain for 10 days following application and for RD4 a lack of hydrological connectivity; the TW1 reseed is located the farthest distance from the RD4 stream (Fig. 4.1). Previous work suggests that such direct P transfer is more likely where soluble fertilisers are applied in wet conditions (Roberts et al., 1989; Preedy et al., 2001). Direct P loss from excreta deposited by sheep either directly in the streams, or in feeding areas close to the streams, almost certainly contributed to the occasional high daily concentrations of dissolved and particulate P measured under stormflow conditions and sometimes during baseflow. However, there was no evidence to suggest that higher stocking rates in RD4 increased excretal P transfer, or sediment and P loss, since whether pastures were grazed or not had no apparent effect on P concentrations.

Although the two catchments differed slightly in their flow patterns, the results suggest that soil P accumulation was by far the most important source of increased P fluxes to the RD4 stream. Total P inputs using the Redesdale pioneer cropping technique (typically 150 kg P ha^{-1}) are greater than those (40-80 kg P ha^{-1}) more commonly used in alternative techniques for pasture improvement (Hornung, 1984; Newbould, 1985; Roberts et al., 1989). Soil available (Olsen) P levels might therefore be expected to increase relatively quickly, especially after large single P fertiliser applications. As noted by Haygarth et al. (1998b), soil P concentrations were much greater in the surface horizons (0-2.5 cm) raising concerns over increased P transfer in surface runoff that interacts with these surface layers. A positive link between soil available P and concentrations of DRP in storm runoff is now well established in plot studies (Maguire et al., 2005).

Increased DRP in surface runoff generated during storm events (Fig. 4.4) was shown to be the most likely cause of the doubling of stream annual flow-weighted DRP concentrations from 10 $\mu g\ L^{-1}$ in RD3 to 21 $\mu g\ L^{-1}$ in RD4. Exactly the same increase (+11 μg DRP L^{-1}) was measured when single samples were taken upstream and downstream of the improved areas within RD4 in November 2006. This magnitude of increase is similar to that obtained on New Zealand hill pastures when annual P fertiliser rates were increased from 11 to 64 kg P ha^{-1} (Lambert et al., 1985). Most of the DRP increase occurred during the winter months suggesting a storm event origin, although there was also an apparent small increase in RD4 baseflow DRP concentrations. The reason for this is not clear but may just relate to the rather low proportion of baseflow samples taken in RD3 and/or the technique used to separate the baseflow contribution. However, the largest proportional increase in P transfer was in particulate form, which in view of the slightly lower SS yields in RD4 must be associated with an increase in solids P content.

An approximation of the P content of SS can be obtained by dividing the annual PP load by the annual SS load. Annual SS yields at Redesdale (0.24 – 1.7 t ha^{-1}) are well within the range for small catchments (Lambert et al., 1985; Labadz, et al., 1991). This calculation estimated the mean SS-P concentration in RD4 (1405 mg kg^{-1}) to be four-fold higher than in RD3 (365 mg kg^{-1}). These data are in precise agreement with the results of the DESPRAL test which showed SS-P concentrations of between 952-1576 mg kg^{-1} on improved areas in RD4 and values of

432-597 mg kg^{-1} in unimproved areas (Table 4.2). Owens and Walling (2002) observed similar differences in solids P content between upland and the more intensively farmed lowland sections of the R. Swale. Compared to topsoil TP contents, the SS in the RD3 stream was slightly depleted in P (enrichment ratio of ca. 0.6), while the SS in the RD4 stream was slightly enriched in P (enrichment ratio of ca.1.5). In RD3, the low-P solids probably originated from the more deeply incised stream bank, or from the banks of the drainage grips transporting higher flow rates down the hillside. This is consistent with the larger SS concentrations in RD3 compared to RD4. In RD4, where hillsides were less steeply sloping, SS concentrations were less variable and selective erosion and transport of the P-enriched soil in surface runoff would be the dominant PP transport mechanism. The larger increases in SS and SS-P concentrations during summer in RD4 also suggest at least in part an in-stream organic (algal) source (Rigler, 1979).

Comparison of the concentrations of SS and P fractions mobilised by the DESPRAL test (Table 4.2) with those measured in stream flow (Table 4.4) may provide an indication of the likely average retention of P during transport from the field to the watercourse (Withers et al., 2007). Taking averaged DESPRAL data for the three improved areas and the two unimproved areas in Table 4.2, SS concentrations decreased by 40-60% and PP concentrations decreased by 50-58% while TDP concentrations apparently increased 1.5-2 fold. These reductions in SS and PP concentrations suggest significant retention of eroding soil particles either within-field or within the stream channel. The increase in TDP concentrations suggests either contamination from non-soil sources, such as sheep excreta or algal cell lysis, or a release of P from SS deposited within the stream channel, or both. Release of reactive P might be expected if the ambient DRP concentration in the water column is lower than the equilibrium P concentration of the bottom sediment (Jarvie et al., 2005). Since the apparent increase in TDP was greater for RD4 (25 vs 56 $\mu g\ L^{-1}$) than RD3 (27 vs 39 $\mu g\ L^{-1}$), sediment P release into the watercolumn maybe occurring.

The ecological significance of the two-fold increases in the average flow-weighted concentrations of DRP and PP in RD4 will depend on a number of factors (other nutrients, shade and flow). The increases in DRP occurred more during winter stormflow than during the biologically active summer months (Fig. 4.6). Low residence times and presence of high molecular weight organic matter can also be expected to have inhibitory effects on algal growth in upland environments (Biggs, 2000; Edwards et al., 2000). Increased DRP may suppress in-stream phosphatase activity in growth-limiting environments (Whitton et al., 2005), and will reduce the ability of upland streams to dilute nutrient richer waters downstream (Bowes et al., 2003). The role of P-rich solids in scouring streams is unclear but those of mineral origin may be bioavailable to aquatic organisms when deposited in oligotrophic waters downstream (Jarvie et al., 2005; Whitton et al., 2005). These potential adverse ecological effects suggest it is important to strike a balance between improved and unimproved areas in upland catchments. Water quality is currently assessed only on annual average concentrations of DRP (unfiltered), with threshold values of 20 and 40 μg DRP L^{-1} for good and high stream quality, respectively proposed for a type 2n stream such as at Redesdale (Duncan et al., 2006). The results presented here indicate that ca. 50% of the upland area at Redesdale could be improved and still meet an acceptable water quality standard.

4.6 Conclusions

There is a paucity of data on transfers of dissolved and particulate P in runoff from upland catchments where marginal rough grazing has been converted to grass/clover swards capable of supporting 2 or 3-fold higher sheep stocking densities. Such pasture improvement has potential to impair the quality of upland streams through erosion and eutrophication. Comparison of two adjacent catchments with variable amounts of improved land (7% vs 47%) has shown that pasture improvement with lime and P fertilisers doubled the transfers of dissolved inorganic P and particulate P, but not suspended solids, to their draining streams. This P enrichment caused a shift in P form towards a greater proportional contribution from the particulate fraction but not in the dissolved inorganic P fraction. Analysis of storm runoff dynamics suggested that the increased P transfer was most likely associated with soil P enrichment rather than to losses following the application of P fertilisers or due to increased stocking densities. Contrary to previous work, this study suggests it is important to strike a balance between improved and unimproved areas in upland catchments in order to avoid potential eutrophication impacts.

5

FARMING PRACTICES AND PHOSPHORUS LOSS IN THE LOWLANDS: ROSEMAUND

5.1 Introduction

Unlike upland streams, lowland rivers are influenced by a greater range and intensity of anthropogenic activities. These typically include higher population densities and associated wastewater and industrial discharges, increased runoff rates from large areas of impervious surfaces associated with urban infrastructures, and a greater intensity of agricultural activity on more manageable soils. This increased anthropogenic activity invariably leads to greater concentrations and loads of nutrients in lowland rivers (Muscutt and Withers, 1996; Bowes et al., 2003; Owens and Walling, 2002). Changes in the river flow velocities in the geomorphological transition between the uplands and lowlands also result in increased residence times which together with gradually increased nutrient loadings associated with downstream population centres lead to an increased risk of eutrophication (Edwards et al., 2000b). In the absence of upland areas, water quality in lowland rivers is heavily dependent on the diluting potential of headwater streams in rural areas dominated by forest and agriculture. The influence of farming practices on nutrient concentrations and loads in headwater lowland streams is therefore of high potential significance, not only in terms of site-specific eutrophication impacts but also in respect of helping to offset the ecological impacts of larger nutrient discharges downstream.

In recent years, the effects of agricultural intensification on nutrient loads to freshwaters and potential eutrophication has received much attention, especially with regard to P. In some countries, agriculture is now perceived as the major source of P inputs to freshwaters (e.g. in Denmark, Kronvang et al., 2005; and in Ireland, Lucey et al., 1999). Freed from the production constraints associated with the harsh climate and poor quality soils in upland areas, lowland areas encompass a wider range of soil types capable of producing higher yields of a variety of crops and supporting higher livestock numbers. The use of genetic engineering to produce higher yielding varieties and more productive livestock, improved agronomic practices allowing better control over pest and diseases and the introduction of field underdrainage systems that reduce surface wetness problems are just a few examples of the advances in land management that have led to intensification of lowland agriculture.

This intensification has led to dramatically increased use of the major nutrients (N, P and K), an increased dependence on monoculture and a range of environmental problems, including uneven distribution of nutrient surpluses at a range of scales, accumulation of surplus P in soils and increased soil erosion (see Chapter 2). Concentrations of P in rural agricultural streams therefore show considerable spatial and temporal variation with current annual export rates varying up to 6 kg P ha^{-1} within Europe (Kronvang et al., 2007). The form of P transported in runoff is very variable depending on the farming system and specific farming practices, but with a tendency towards dissolved P (DP) forms in runoff from grassland (Nelson et al., 1996; Hooda et al., 1999; Tunney et al., 2000), and particulate P (PP) from arable land (Sharpley and Smith, 1990; Catt et al., 1998; Uusitalo et al., 2000), although at the catchment scale, PP is often dominant in both catchment types (e.g. Douglas et al., 2007; Hodgkinson and Withers, 2007).

A large amount of research has been conducted worldwide over the last three decades to improve our understanding and knowledge of P loss in surface and sub-surface runoff from agricultural land (Sims et al., 1998; Haygarth and Jarvis, 1999; Dougherty et al., 2004). In general terms, P losses in land runoff are dependent on both the quantities of P present in, or on, the soil, and the extent to which water moving through, or over, the soil captures these sources. These sources include soil, crop residues, fertilisers and livestock faeces and land management practices (cultivation practice, nutrient inputs, crop and livestock management) and their interaction with landform and climate have a large influence on the P mobilised in runoff within fields and subsequent delivery to the watercourse. Additional sources of P in flowing waters include erosion of channel banks and resuspension of bottom sediments during storm events and increased organic matter (OM) generation under baseflow conditions due to biological activity and algal growth (Johnson et al., 1976; Laubel et al., 2003). Linking field management activities to stream P concentrations and loads in rural catchments is therefore complex and has not been widely investigated. Some data are available within grassland catchments (Lennox et al., 1997; Lazzarotto et al., 2005; Monaghan et al., 2007), but little data exist for intensive arable or mixed catchments, especially in the U.K.

The previous chapter investigated the impact of upland farming practices on dissolved and particulate P concentrations in headwater streams and found a strong link between surplus P accumulation in soil due to pasture improvement and increased P transfer in runoff. This chapter assesses the impact of farming practices (P input management, soil management and soil P status) on stream P concentrations and loads in a lowland mixed farming catchment at Rosemaund. Since P transfer in cultivated catchments is largely particulate in nature, it was expected that both soil P accumulation and cultivation practices would influence the amounts and timing of the PP transported in runoff at Rosemaund. Whilst much data shows large amounts of PP can be transported in runoff at field scale (Ryden et al, 1973; Catt et al., 1998; Chambers et al., 2000), there is less evidence of a farm management influence at the catchment scale due to retention of mobilised P within fields and streams, lack of connectivity between the field and the watercourse and dilution by groundwater or runoff from non-agricultural areas.

5.2 The Rosemaund catchment

The Rosemaund catchment (150 ha) is located in Herefordshire, England and includes crops of winter cereals and a range of break crops including potatoes, (62% of catchment area), permanent and ley grassland (33%) used for beef, sheep and deer grazing and small areas of woodland and farm buildings (5%). Slopes are gentle (usually <5°). A farm sewage treatment works (STW) servicing 65 people discharges secondary-treated overflow water directly into the stream. As well as the monitoring station at the catchment outlet (Belmont), a further nested sub-catchment monitoring station was located at Jubilee (31 ha) and a single field tile drain (6 ha) within Jubilee was monitored at Foxbridge (Fig. 5.1). The field drain receives drainage water from parts of both Foxbridge and Stoney fields. The Jubilee catchment has 61% of its area in arable cropping, whilst the Foxbridge catchment was continuously cropped to cereals. Stream flow at the three stations was monitored continuously over a three year-period (1997-2000) to examine the spatial and temporal variation in P concentrations in relation to detailed information on P inputs, cultivation practices and soil P concentrations. Annual inputs of P in fertilisers and recycled farmyard manure (FYM) are broadly in balance with crop P offtake but soil total and available (Olsen) P (OP) concentrations vary widely and are high (P Index >4, Ministry of Agriculture, Fisheries and Food, 2000) in fields where hops have been previously grown (Fig. 5.1). No hops were grown during the study period.

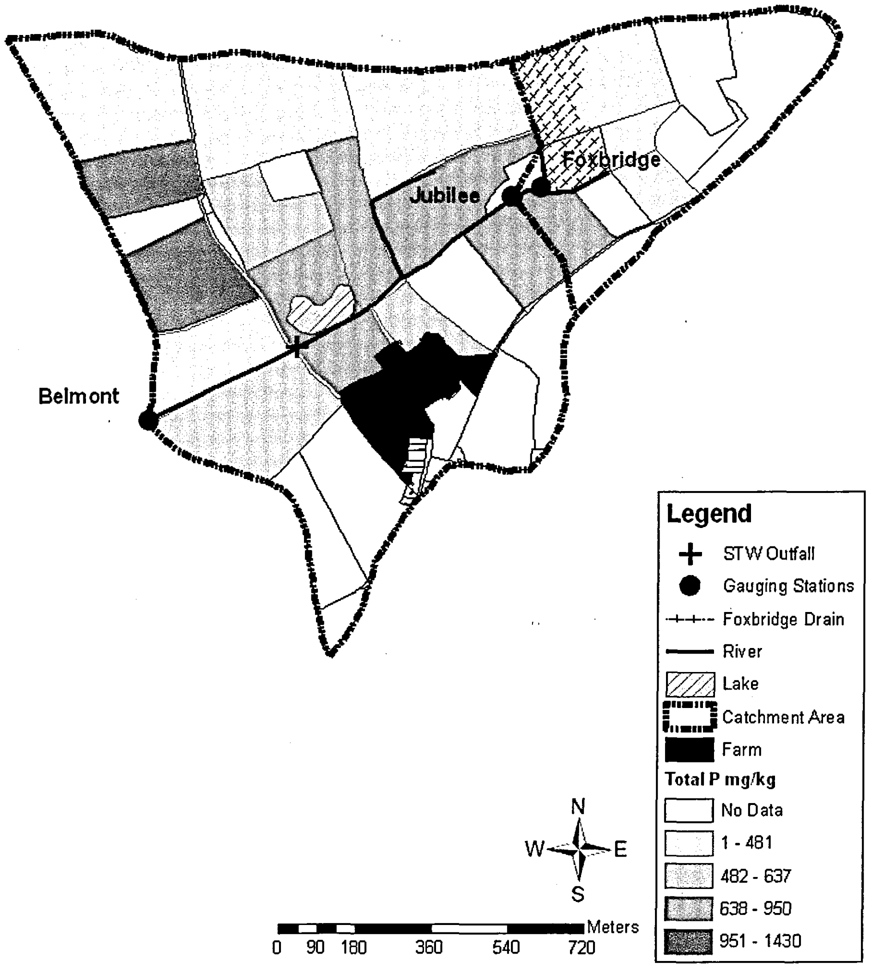


Figure 5.1 Location of monitoring stations and distribution of soil total P concentrations within the Rosemaund catchment.

The majority of fields have been underdrained to remove surface wetness and the soils are well-fissured with heavier-textured areas cracking during late summer and early autumn. The soil at Rosemaund is also particularly dispersive (Bromyard Association) and large concentrations of suspended particles and associated P are transported from the soil surface via the tile drains to the stream (Russell et al., 2001; Chapman et al., 2005). Shallow rill and sheet erosion is common with overland flow and sub-surface flow through tile drains contributing ca. 35% and 55%, respectively of the SS load reaching the stream (Russell et al., 2001; Walling et al., 2002).

The remaining SS in the stream (ca. 10% of total) is derived from the channel banks. Previous work has also highlighted that a relatively high proportion (20%) of OP in the Rosemaund soils is extractable in water (Withers et al., 2001) suggesting high concentrations of DRP might be expected in runoff water from some fields. Comprehensive details of the Rosemaund catchment, the soil types, land use history, P inputs and methodologies used to measure stream P concentrations and loads are provided in Chapter 3.

5.3 Soil phosphorus

Samples of topsoil (0-10 cm) from a range of arable, grassland and old hopyard fields were sampled in 1999 to determine soil P status. Concentrations of TP, OP and WEP ranged from 337-1430, 16-150 and 2-34 mg kg^{-1}, respectively with largest concentrations recorded in the old hopyard fields. On average, WEP was 20% of OP and OP was 8% of TP. The distribution of total P concentrations is shown in Fig. 5.1. The potential mobilisation of soil particles (>0.45 μm) and P forms due to rainfall impact was also assessed by the DESPRAL test (Withers et al., 2007). Field areas with slightly different soil types were split for sampling to assess any effect of the three soil types at Rosemaund; Bromyard, Middleton and Compton Associations. Russell et al. (2001) found that the heavier-textured Compton soils that occupy the river corridor contributed proportionally more SS than did the Middleton and Bromyard soils which tend to occupy the lower and upper field slopes, respectively. Summary distribution statistics are presented for all the field areas sampled together with mean values for the Stony/Brushes and Foxbridge/Longland fields that form the Jubilee catchment (Table 5.1).

The DESPRAL test suggests concentrations of SS mobilised in the field might range from 0.9-3.5 g L^{-1}. Bromyard soils tended to mobilize more SS than the heavier-textured Middleton and Comptom soils, although this was not always statistically significant ($P > 0.05$). However, within the Foxbridge/Longlands field, Middleton soils mobilised about half as much of the SS as Bromyard soils. Concentrations of TDP and PP ranged up to 0.5 and 2 mg L^{-1}, respectively and on average 78% of the TP mobilised by the test was in particulate form. Within the Jubilee fields, TDP and PP concentrations were relatively similar (ca. 152 and 809 μg L^{-1}, respectively), except for the Middleton soils in Foxbridge/Longlands which showed lower values (Table 5.1).

Table 5.1 Concentrations of suspended solids (SS), total dissolved P (TDP), particulate P (PP) and total P (TP) predicted by the DESPRAL test for all field areas (mean (s.e.) and range) and the two fields within the Jubilee catchment.

Field area	SS (mg L^{-1})	TDP (μg L^{-1})	PP (μg L^{-1})	TP (μg L^{-1})
All fields (n=36)	1910 (95) (877-3480)	290 (17.4) (125-501)	1020 (77) (491-2083)	1310 (88) (616-2565)
Stoney/Brushes				
Bromyard Association	2252	143	798	941
Middleton Association	1573	154	842	996
Compton Association	2100	159	669	828
Foxbridge/Longlands				
Bromyard Association	2722	151	926	1077
Middleton Association	1302	125	491	616

5.4 Stream phosphorus

5.4.1 Streamflow

Annual rainfall was 766, 718 and 715 mm in the 97/98, 98/99 and 99/00 monitoring periods, respectively and well above the long-term (30-yr) average of 660 mm. Monthly totals were quite variable with January, April and September being particularly wet in two out of the three years (Fig. 5.2a). For example, a particularly intense 70 mm storm event during April 1998 produced over 20 mm of flow at Foxbridge, and over 50 mm of flow at the Jubilee station, in one day, although rainfall exceeded 20 mm day^{-1} on only a few occasions (Fig. 5.2b). The spring/early summer of 1998 was particularly wet (96 mm in June) as was the autumn/winter of 1998/99 and the summer/autumn of 1999/00.

Streamflow usually peaked in January or April and average annual flows were 175 mm (153-200 mm), 366 mm (289-424 mm) and 268 mm (261-284 mm) for Foxbridge, Jubilee and Belmont (Fig. 5.2c), representing 24% (20-28%), 50% (40-59%) and 37% (36-37%) of annual rainfall, respectively. The flow at Jubilee was particularly large in the last year due to a relatively wet summer/autumn causing increased flow from groundwater springs and early flow through the land drains during September. In other years, flow through Foxbridge commenced only slowly in October or November with the majority occurring from December to April (Fig. 5.2c).

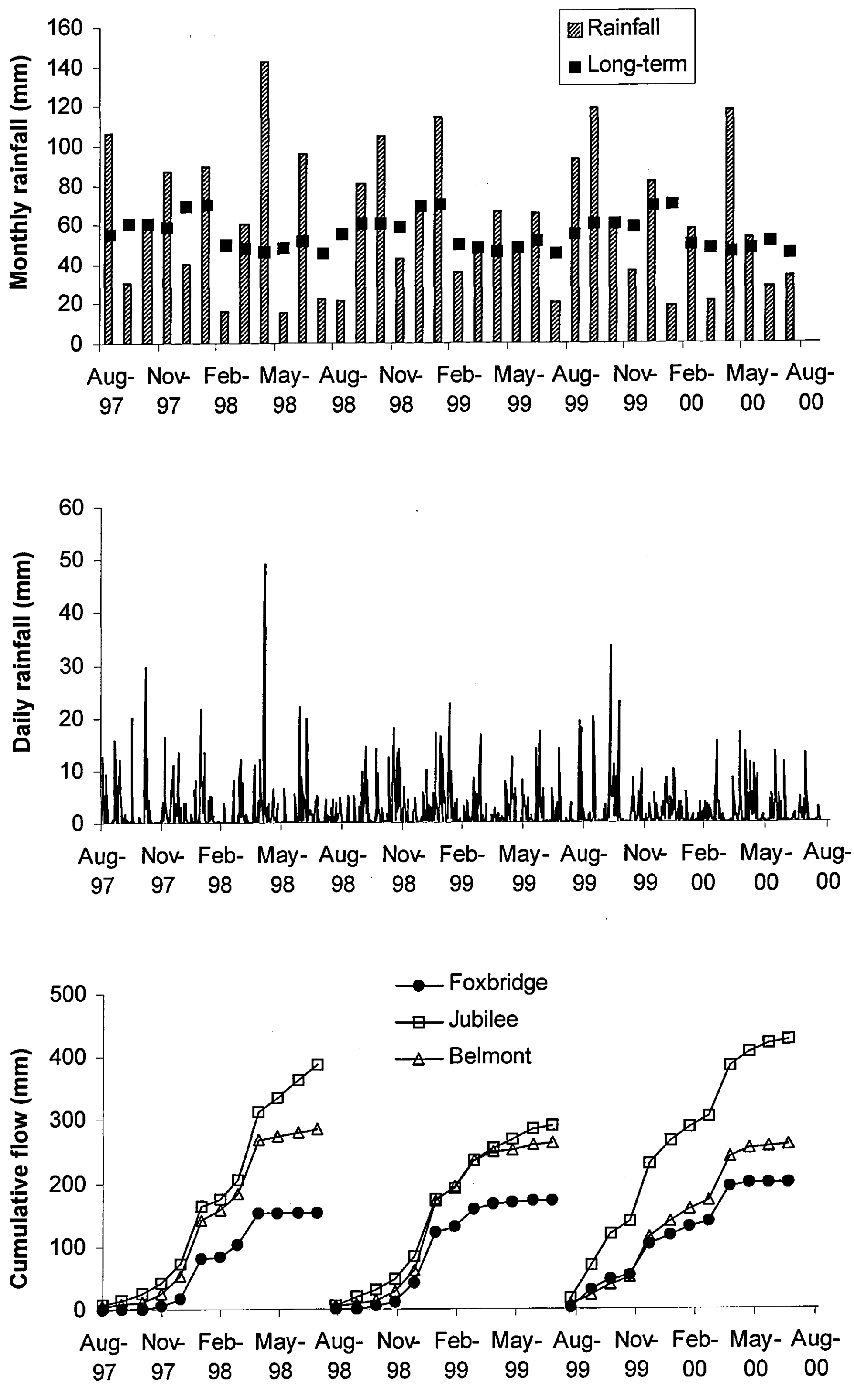


Figure 5.2 Rainfall and flow patterns in each catchment: (a) monthly rainfall in relation to the long-term (1971-2000) average, (b) variations in daily rainfall across the monitoring period and (c) cumulative flows in each year.

The volume of annual flow (m^3) at Foxbridge was ca. 1% of the flow at Jubilee which was ca. 30% of the flow at Belmont. There are 22 known drain outfalls within the whole catchment suggesting a drain flow contribution of approximately 22% assuming the flow at Foxbridge is representative of other drained areas. Baseflow was estimated to contribute between 68-71% of the flow at Jubilee and between 65-69% of the flow at Belmont according to the Dingman method (Dingman, 1994). This suggests an overland flow component of about 11-13%.

5.4.2 Stream phosphorus

Average daily concentrations of DRP, TDP and TP were very variable and not uniformly distributed, ranging up to 4 mg L^{-1} for dissolved fractions and 16 mg L^{-1} for TP (Table 5.2). Foxbridge and Jubilee stations showed very similar average values for all P fractions, although data for Foxbridge tended to be slightly more variable. Concentrations of DRP at Belmont were three-fold greater than at the other two stations, and represented a higher proportion (54%) of TP than at Jubilee (33%) or Foxbridge (38%). At all three stations, the highest P concentrations were always recorded under very low flows but both dissolved and total P concentrations also tended to increase with flow despite a great deal of scatter (data not shown).

Table 5.2 Distribution statistics for average daily concentrations of dissolved reactive P (DRP), total dissolved P (TDP), total P (TP) and suspended sediment (SS) at Foxbridge, Jubilee and Belmont stations 1997-2000.

	n	Range	Mean (s.e.)	Lower quartile	Median	Upper quartile
DRP ($\mu g\ L^{-1}$)						
Foxbridge	457	1-3730	181 (11)	59	128	241
Jubilee	473	2-2510	159 (8)	60	114	199
Belmont	428	4-1650	413 (13)	212	332	515
TDP ($\mu g\ L^{-1}$)						
Foxbridge	451	5-3800	228 (13)	75	160	290
Jubilee	459	5-2630	196 (10)	75	136	234
Belmont	427	4-1660	476 (14)	255	391	620
TP ($\mu g\ L^{-1}$)						
Foxbridge	451	20-5960	596 (38)	100	263	730
Jubilee	458	6-13800	598 (57)	110	236	612
Belmont	424	79-15900	877 (75)	320	532	907
SS (mg L^{-1})						
Foxbridge	170	2-8401	561 (74)	29	195	809
Jubilee	106	6-8385	872 (151)	76	369	882
Belmont	117	1-4215	319 (53)	51	140	306

Fewer samples were taken of baseflow than of stormflow, but baseflow concentrations of TDP at Jubilee ranged up to 0.6 mg L^{-1} and concentrations of TP ranged up to 3 mg L^{-1}. Concentrations of TP were highly skewed. Median concentrations DRP, DUP and PP were 96, 14 and 29 μg L^{-1}, respectively at Jubilee (n = 100) and 556, 80 and 77 μg L^{-1}, respectively at Belmont (n = 34). Spot concentrations of groundwater sampled during August 2007 under very low flow indicated a DRP concentration of <7 μg L^{-1}. Whilst the much greater DRP concentrations at Belmont might be expected due to the discharge from the farm STW, the concentrations of DRP at Jubilee were still relatively large (ca. 0.1 mg L^{-1}) for groundwater. Whilst DRP concentrations in baseflow tended to decrease with increasing flow at Belmont, they increased significantly (P <0.05) with flow at Jubilee suggesting a diffuse origin (Fig. 5.3).

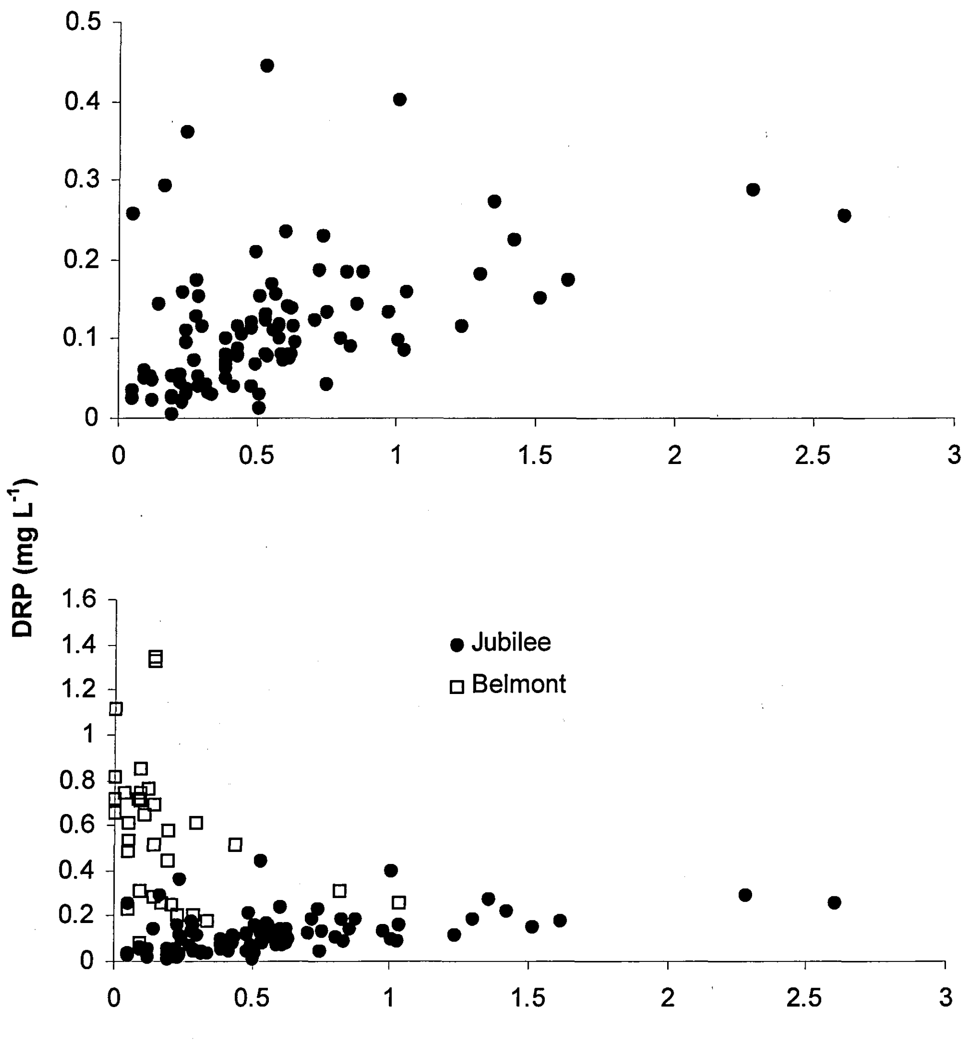


Figure 5.3 Effect of baseflow on concentrations of dissolved-reactive P (DRP) measured at Jubilee and in relation to those at Belmont. Note the different effects between the two stations.

Examples of the changes in P concentrations during two representative storm events at each monitoring station are shown in Figs. 5.4 and 5.5. In the first example (Storm 1), 22 mm of rain fell over 4 days from 22 December 1997 to 25 December 1997, with daily totals not exceeding 9 mm. In the second example (Storm 2), 28 mm of rain fell over 3 days from 2 April 2000 to 4 April 2000, with daily totals reaching 17 mm. Both storm events produced similar peak flow rates in the Foxbridge drain and stream monitoring points. For storm 1, TP concentrations peaked at 9.1 mg L^{-1} at Foxbridge, 3.1 mg L^{-1} at Jubilee and 1.8 mg L^{-1} at Belmont and coincided with peak flow at all three stations (Fig. 5.4). In contrast, peak TP concentrations were much smaller (up to 1.4 mg L^{-1}) for storm 2, did not decline greatly between stations and preceded peak flow (Fig. 5.5). Fluctuations in TP tended to be mirrored by fluctuations in the concentrations of SS confirming the predominance of particulate forms.

Dissolved P forms also rose with increasing flow for both storms and all stations but the response was damped within the range 0.4-1.0 mg L^{-1} in relation to TP response; for example contrast TDP response against the two scales of TP response in Figs. 5.4 and 5.5. Peak TDP concentrations did not always correspond with peak flow or with peak TP concentrations and in some cases decreased when TP concentrations rose; e.g. storm 1 at Foxbridge. A general pattern of fluctuating TP response but more stable and delayed TDP response to increasing flow was noted in a number of storm events.

5.4.3 Annual phosphorus export

Annual stream export of TP ranged between 1.7-2.1 (mean 1.94, s.e. 0.06) kg ha^{-1} at Foxbridge, 2.1-3.8 (mean 3.0, s.e. 0.09) kg ha^{-1} at Jubilee and between 2.5 - 2.9 (mean 2.7, s.e. 0.14) kg ha^{-1} at Belmont. Variation in annual export was strongly related to variation in annual flow and between 60-80% of the total annual load typically occurred in the two or three months of the year with very high rainfall (e.g. January and April). Hence flow-weighted concentrations of each of the P fractions are presented to allow comparison between stations and years (Table 5.3). Concentrations of DRP were significantly ($P < 0.05$) greater at Belmont than at Jubilee due to the influence of the effluent discharge from the farm STW. Concentrations of PP were 35% lower ($P < 0.05$) at Jubilee and Belmont than at Foxbridge due to dilution by the spring water feeding the stream. DUP concentrations were similarly low at all stations and never accounted for more than 10% of the TP load.

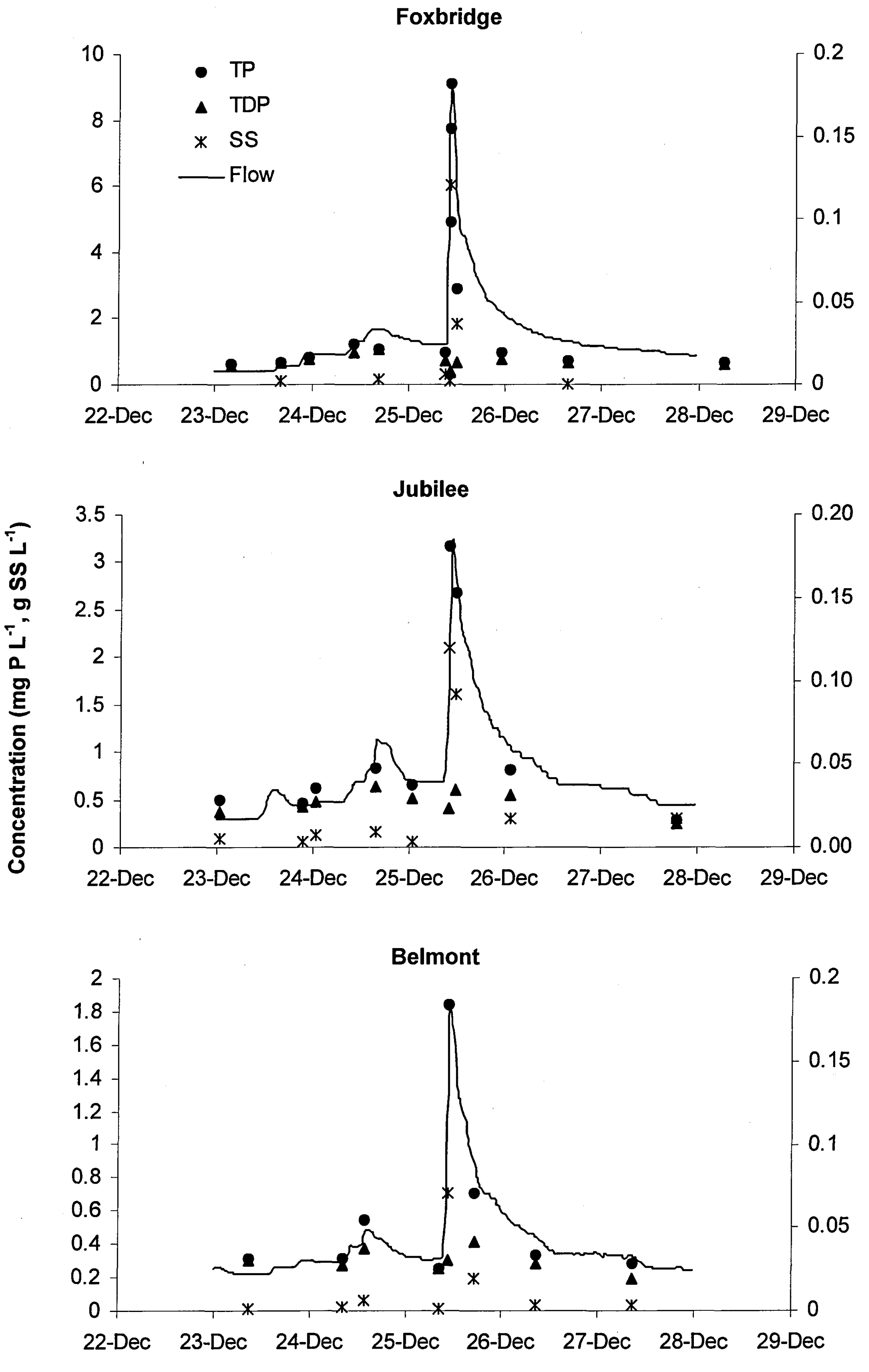


Figure 5.4 Flow and concentrations of total dissolved P (TDP, total P (TP) and suspended solids (SS) at Foxbridge, Jubilee and Belmont during a storm event in December 1997.

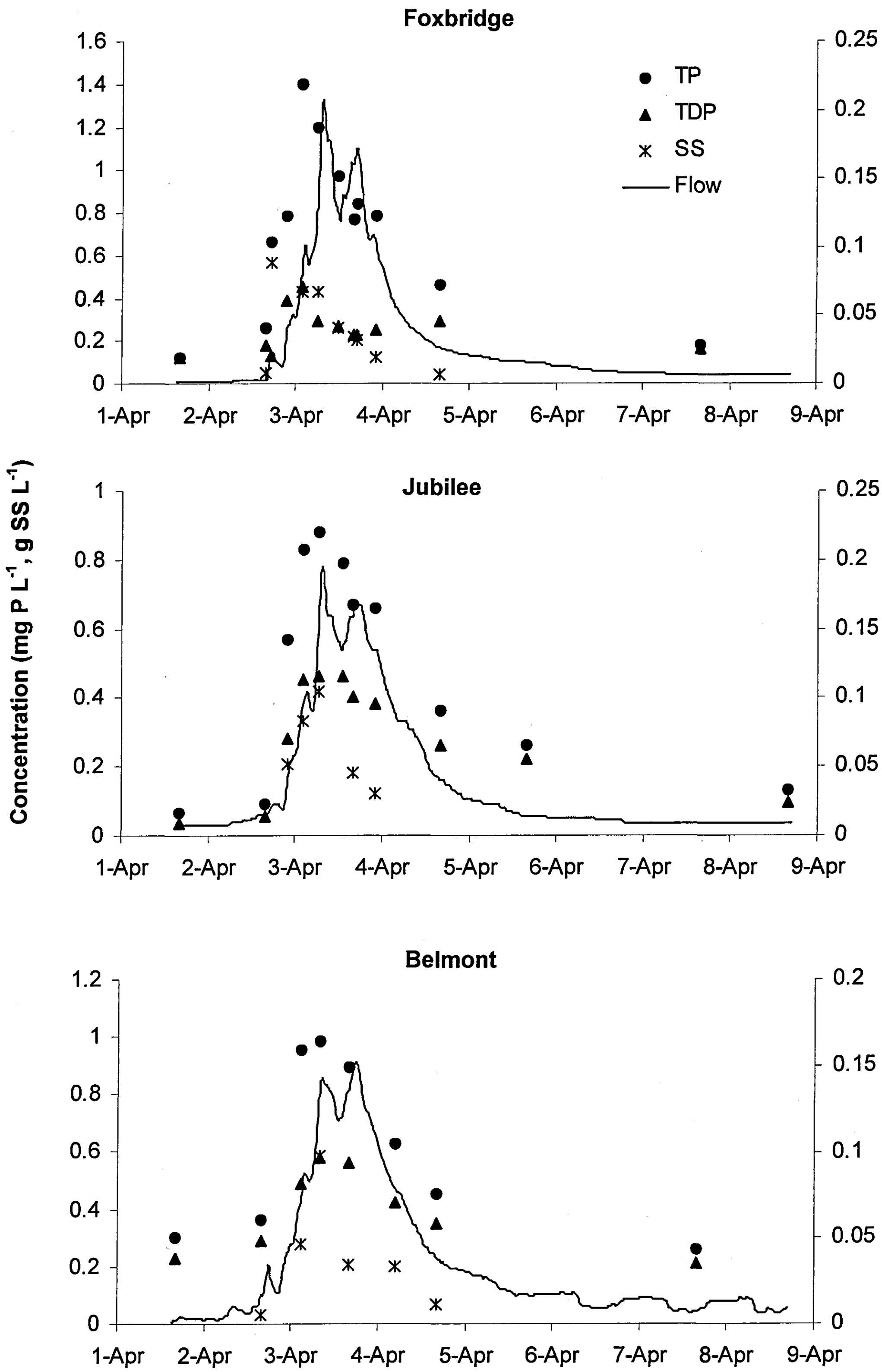


Figure 5.5 Flow and concentrations of total dissolved P (TDP, total P (TP) and suspended solids (SS) at Foxbridge, Jubilee and Belmont during a storm event in April 2000.

The particulate fraction dominated at Foxbridge (71%) and Jubilee (64%) and was in equal proportion with the dissolved fraction at Belmont (53%). At Foxbridge and Jubilee, DRP concentrations were notably lower in the third year which maybe an effect of the higher flows in 1999/00 exhausting the DRP supply. At Belmont, DRP concentrations were lower in the first year when PP concentrations were large. Concentrations of both DUP and PP (and hence TP) were larger in the first year at all stations (Table 5.3).

Table 5.3 Annual flow and flow-weighted concentrations of dissolved reactive P (DRP), dissolved unreactive P (DUP), particulate P (PP), total P (TP) and suspended solids (SS) at each monitoring station at Rosemaund.

Station	Year[a]	Flow[b] (mm)	DRP	DUP (μg L^{-1})	PP	TP	SS (mg L^{-1})
Foxbridge	1997/98	153	336	136	911	1383	715
	1998/99	173	286	52	669	1007	619
	1999/00	200	148	30	802	980	562
	Mean	175	257	73	794	1123	643
Jubilee	1997/98	336	243	108	646	997	803[c]
	1998/99	289	287	43	411	741	333
	1999/00	423	165	26	496	687	357
	Mean	366	232	59	518	808	498
Belmont	1997/98	284	328	110	598	1036	398
	1998/99	258	442	59	472	973	276
	1999/00	248	405	58	485	948	229
	Mean	268	392	76	518	986	301

[a]1 August - 31 July in each year. [b]Total over the monitoring period. [c]Values based on continuous turbidity data suggest a value of 539 mg L^{-1}.

Temporal variation in P export expressed as average flow-weighted concentrations for each month for each station are shown in Fig. 5.6. During the months when the Foxbridge drain was fully flowing (normally December-April), TP concentrations were often well over 1 mg L^{-1} and largely in particulate form. In February 2000, over 2 mg L^{-1} was recorded even though rainfall for that month was close to the long-term average. High TP concentrations were also recorded outside the normal drainage period, for example in June either as TDP (June 98) or PP (June 99) when small amounts of storm water were generated in response to rainfall. Relatively high TDP concentrations were recorded in September and October 1999 when FYM was applied to Foxbridge field although no similar increase was observed at Jubilee until the following January when DRP increased as flow increased (Fig. 5.6).

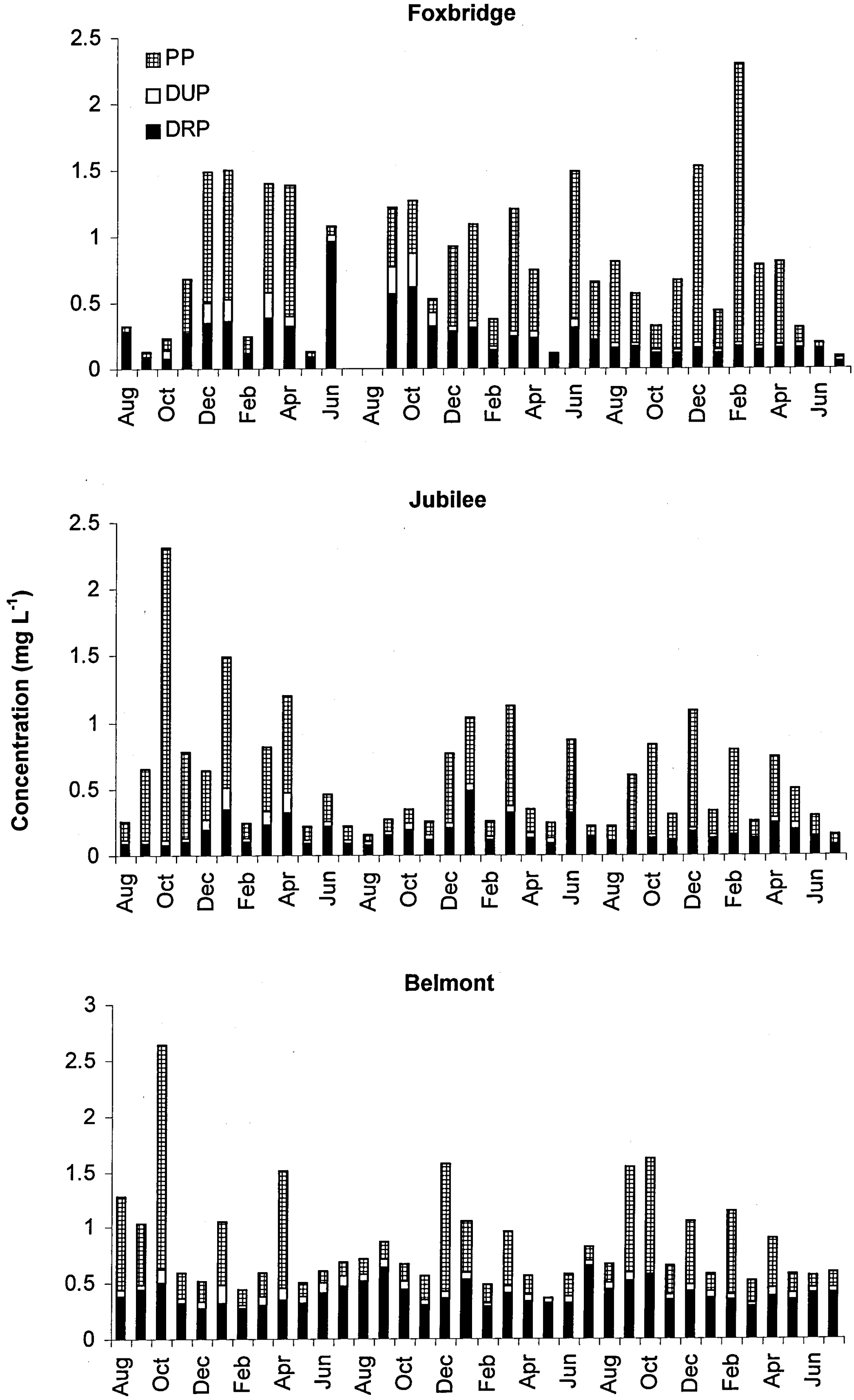


Figure 5.6 Monthly flow-weighted concentrations of dissolved-reactive P (DRP), dissolved unreactive P (DUP) and particulate P (PP) for the three monitoring stations in the Rosemaund catchment during 1997-2000.

At Jubilee and Belmont, flow-weighted TP concentrations were above 1 mg L^{-1} on fewer occasions, but very high PP concentrations (> 2 mg L^{-1}) were observed in October 1997 when much of the catchment was cultivated. During the main drainage period, concentrations tended to be greater in the wetter months (P <0.001), except for the DRP concentrations at Belmont which were consistently greater than at Jubilee due to the influence of the farm STW. The outfall from the farm STW into the stream was monitored on three occasions, 25 October 2006, 9 January 2007 and 15 June 2007. The average concentration of TP was 3.6 mg L^{-1} (s.e. 0.17) with ca. 94% in dissolved form, mostly DRP. The outfall pipe was usually drowned by stream flow during winter storm events but a flow rate of 0.2 l sec^{-1} was measured on one occasion during summer when stream levels were lower. Since the outfall takes only trickle overflow from the final sedimentation tanks, it is reasonable to assume that the flow rate is fairly constant throughout the year. When flow rates might be increased during storms the extra flow is likely to dilute the concentrations so the overall effect on P load will remain the same.

Hence, assuming this flow rate is typical of the remainder of the year, the total flow and total P load from the outfall can be calculated as 4.2 mm and 22.7 kg, respectively. Averaged over the whole catchment, the areal load is 0.15 kg ha^{-1}, which represents 6% of the TP export at Belmont. This load would increase the flow weighted TP concentration by ca. 0.06 mg L^{-1} which is less than half of the difference in the mean P concentrations between Belmont and Jubilee (0.18 mg L^{-1}). This suggests an additional P input from the farmstead was contributing to the flux at Belmont (see Chapter 7). Although these catchment P sources represented a relatively small proportion of the total P load, the continuous delivery (trickle) of concentrated effluent P caused major temporal fluctuations in the stream TDP concentrations at Belmont (Fig. 5.7). Concentrations showed a cyclical pattern with low concentrations during the winter and high concentrations (up to 1.5 mg L^{-1}) during the summer, typical of a stream receiving point sources.

5.4.4 Stream suspended solids

Instantaneous concentrations of SS were also very variable ranging up to 8 g L^{-1} and tended to be lowest at Belmont and highest at Jubilee (P<0.05), Table 5.1. As with TP concentrations, highest SS concentrations were measured at low flows and there was wide scatter in SS concentrations as flow rate increased. Estimated annual stream loads of SS ranged between 1.07-1.13 (mean 1.10, s.e. 0.02) t ha^{-1} at Foxbridge, 0.96-3.10 (mean 1.86, s.e. 0.64) t ha^{-1} at Jubilee and between 0.59-1.13 (mean 0.81, s.e. 0.16) t ha^{-1} at Belmont.

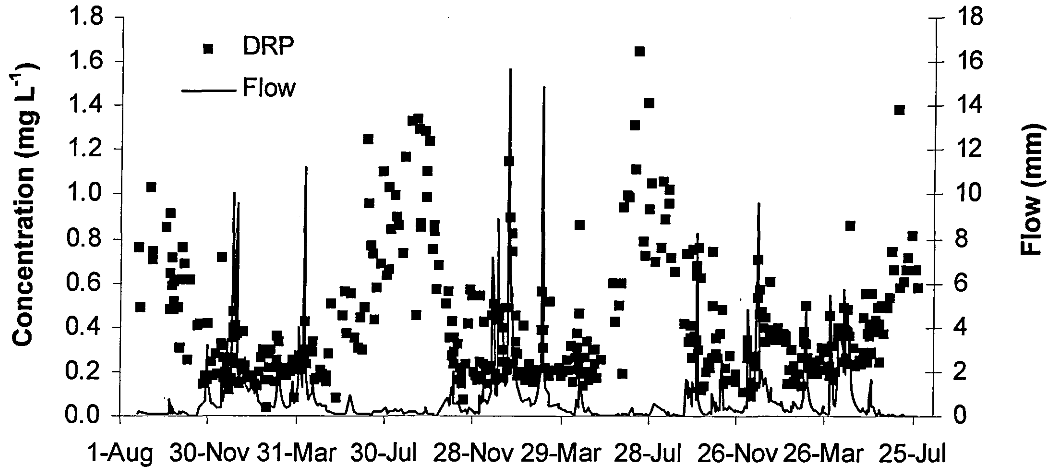


Figure 5.7 Changes in the concentration of dissolved reactive P (DRP) in the stream at Belmont during the monitoring period 1997-2000.

These estimates are however based on relatively few values and must be treated with caution, especially in the first year at Jubilee when there were very few measurements taken during high flows. Annual loads based on continuous turbidity measurements at Jubilee were much lower in the first year (3.10 v 1.81 t ha^{-1}) whilst comparable in the second year (0.96 v 0.81 t ha^{-1}). At Belmont, the differences were smaller; 1.13 v 0.80 t ha^{-1} in the first year and 0.72 v 0.84 t ha^{-1} in the second year. There were no turbidity data available for the third year. As with PP concentrations, flow-weighted SS concentrations tended to be larger in the first year at all stations, but particularly at Jubilee even when using turbidity data (Table 5.3).

There were very strong relationships between instantaneous PP concentrations and SS concentrations at all three stations with intercept values close to zero and regression gradients in the range 1-1.4 g kg^{-1} (Fig. 5.8). Comparison of mean SS-P concentrations (calculated as PP/SS) after log transformation indicated that SS-P concentrations decreased slightly ($P < 0.05$) from 1.2 g kg^{-1} in drain flow to 1.1 g kg^{-1} in the stream at Jubilee, and then increased sharply ($P < 0.001$) to 1.6 g kg^{-1} in the stream at Belmont.

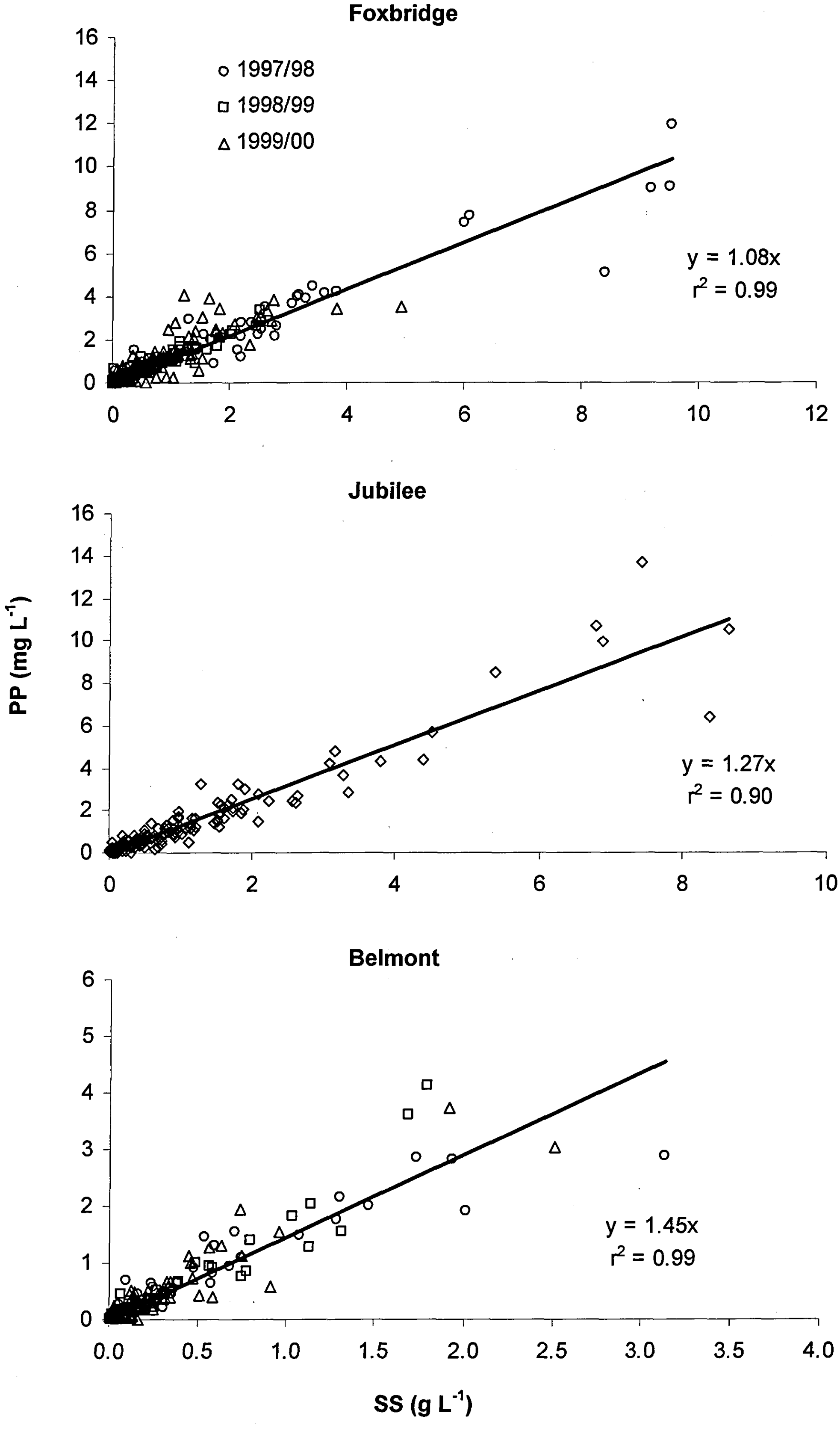


Figure 5.8 The relationship between concentrations of particulate P (PP) and suspended solids (SS) in drain flow (Foxbridge) and stream flow (Jubilee and Belmont) at Rosemaund.

5.5 Effects of farming practices

Temporal variability in dissolved P, particulate P and SS concentrations at the Foxbridge and Jubilee catchments was further explored in relation the precise timing of cultivations, fertiliser and manure applications during each monitoring year. Belmont data were influenced by inputs of P from the farm STW and therefore not used directly in this comparison, except where similarities between response at Jubilee and Belmont occurred. The Stony and Brushes fields (farmed as one field), and the Foxbridge and Longlands fields (also farmed as one field) form the Jubilee catchment, whist the Foxbridge drain receives water from both Foxbridge and Stoney fields (Fig. 5.1).

5.5.1 Cultivations

Winter cereals are usually ploughed, power-harrowed, rolled and drilled between the middle of September and the middle of October. However, actual cultivation dates varied between years depending on weather conditions. In the first year, most fields were cultivated/drilled in September but in the second year, cultivation was delayed until October and November due to a wet autumn. In the third year, some crops were drilled in early September but heavy rain during the remainder of September delayed cultivations until the middle of October or early November. As both Foxbridge/Longlands and Stoney/Brushes were cropped to cereals during the monitoring period, some impacts of cultivation on stream P concentrations might be observed.

In the first year, both fields in the Jubilee catchment were cultivated on the 26/27 September 1997. The first major storm after cultivation occurred on 8/9 October (>30 mm) with a further 16 mm falling on 11 October. Whilst stream flow was not greatly increased by these storm events, the concentrations of PP and SS at Jubilee increased dramatically up to 14 mg L^{-1} and 8 g L^{-1}, respectively (Fig. 5.9). Similar large increases in PP were also observed at the Belmont station even though flow was not greatly increased. At Foxbridge, there was no evidence of increased PP concentrations until a major storm event occurred on the 7/8 November (20 mm) when a peak concentration of 5 mg L^{-1} was measured in a small event (Fig. 5.10). This November peak was not observed at Jubilee. The patterns of high PP concentrations during small events at Jubilee contrasted strongly with the smaller increases in PP (generally < 2 mg L^{-1}) under larger flows during December and January when land drains were flowing continuously (Fig. 5.9).

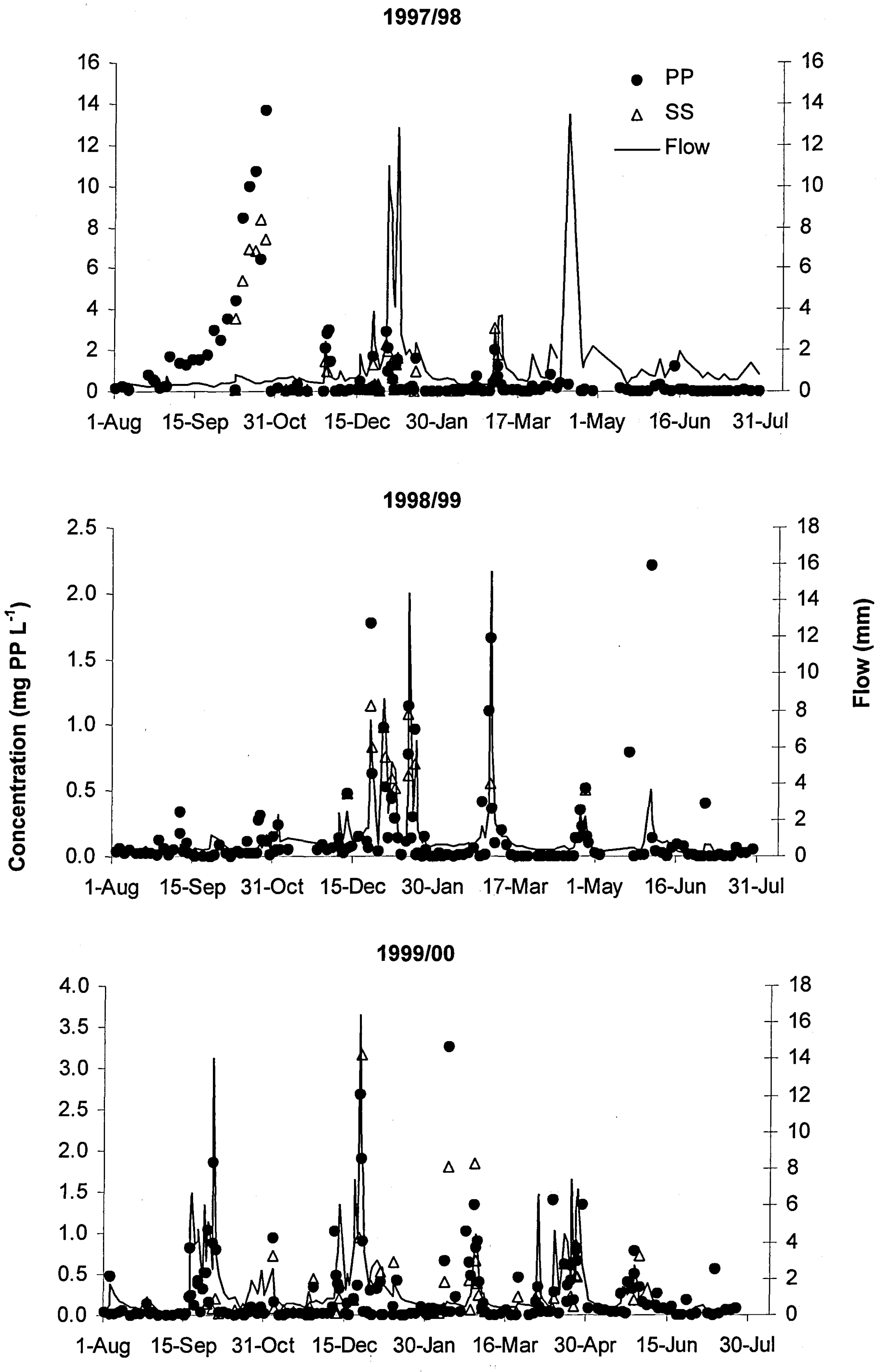


Figure 5.9 Temporal changes in the concentrations of particulate P (PP) and suspended solids (SS) in relation to flow at Jubilee in each year at Rosemaund.

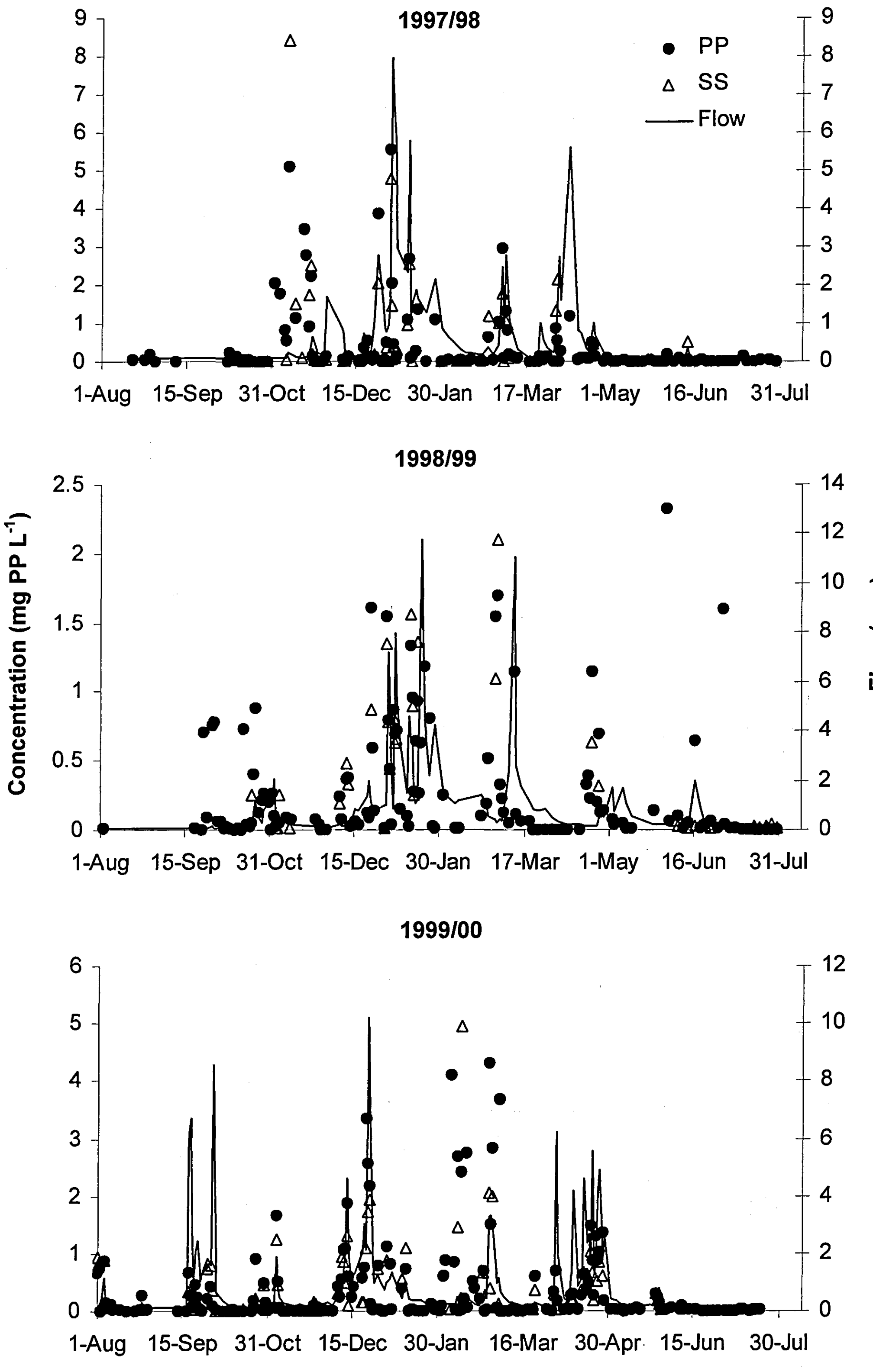


Figure 5.10 Temporal changes in the concentrations of particulate P (PP) and suspended solids (SS) in relation to flow at Foxbridge in each year at Rosemaund.

In the second year, Stoney/Brushes (the furthest of the two fields from the stream in Jubilee) was cultivated on 12 October and the Foxbridge-Longlands field on the 5 November 2008. Storm events after cultivation occurred on the 16 October (12 mm), 22-24 October (47 mm), 27 October (13 mm), 31 October (16 mm) and 2 November (10 mm), but thereafter the remainder of November was dry. Increased PP concentrations relative to flow were observed after cultivation at Foxbridge (up to 0.8 mg L^{-1}), but to a lesser extent at Jubilee (up to 0.4 mg L^{-1}) compared to the first year when both fields were cultivated (Figs 5.9 and 5.10). However, increased PP concentrations were also observed before cultivation.

In the third year, Stoney-Brushes was ploughed on 7 October and drilled on the 14 October 1999. Foxbridge-Longlands field was ploughed on 16 October but not harrowed until 9 November and drilled on 11 November 1999. Comparatively little rain fell during October and November except for low intensity storms on 21-24 October (21 mm) and 4-5 November (13 mm). There was a clear PP signal (0.9 mg L^{-1}) relative to flow on 5 November at Jubilee but otherwise PP concentrations varied very much in line with flow, with largest increases in flow occurring during September when rainfall was well above normal (Fig. 5.9). A very similar pattern was observed at Foxbridge, with increased PP concentrations (up to 1.7 mg L^{-1}) in small amounts of storm water occurring in early August (when 56 mm fell over 6 days) and on 25 October and 5 November. High PP concentrations (up to 4.5 mg L^{-1}) at low flow were also recorded at both Foxbridge and Jubilee (but not at Belmont) during February 2000 (Figs. 5.9 and 5.10), especially at Foxbridge with a number of peaks throughout the month despite no recorded cultivation or P input during February.

5.5.2 Fertiliser and manure applications

The timing of fertiliser and manure applications varied significantly between the three years allowing some comparison of the effects of these applications on stream P concentrations. Most arable fields in the catchment received P fertiliser during August and September 1997 and large amounts of FYM during September 1998. One field (Moorfield) received a large application of FYM in January 1999 prior to cropping with potatoes. In the third year, very few arable fields received any P in the autumn but P was applied to grassland fields during May and June 2000. Both arable crops and grassland typically received ca. 20 kg P ha^{-1} which is broadly equivalent to crop P offtake. Since P in runoff following fertiliser and manure applications can be expected to occur largely in dissolved form (Withers et al., 2001; Hart et al., 2004), temporal changes in TDP concentrations were evaluated in relation to the timing of P inputs.

Inorganic fertiliser (22 kg P ha^{-1}) was applied to Stoney-Brushes and Foxbridge-Longlands on 24 August 1997. The fertiliser was top-dressed onto stubble and was not incorporated until the fields were ploughed on 26 September 1997. After application, there were two days without rain and then 33 mm fell over the subsequent 5 days. On the first day of the storm, peak concentrations of 3.8 mg L^{-1} of TDP were recorded in a negligible amount of flow from the Foxbridge drain but no similar increase was observed at the Jubilee station on this date (Figs. 5.11 and 5.12). A concentration of 0.6 mg DRP L^{-1} under low flow was observed at Jubilee but this was two days before any fertiliser was applied. Smaller increases in DRP under low flow were occasionally observed at Foxbridge and Jubilee during subsequent storm events (e.g. 11 October and 7 November), but concentrations were generally maintained at ca 0.2 mg L^{-1} with occasional higher concentrations (up to 0.6 mg L^{-1}) during storm events. Some high TDP concentrations (up to 1 mg L^{-1}) relative to flow were observed later on in the year at both Foxbridge and Jubilee; for example in February, March and/or June.

In the second year, FYM (ca. 90 kg P ha^{-1}) was applied to both arable fields within the Jubilee catchment on 22 September 1998. This manure was not incorporated until 7 October on Stoney-Brushes and until 5 November on Foxbridge-Longlands. Significant rain fell 4 days after application on 26 September (16 mm) and sporadically during October and early November but no more than ca. 10-13 mm day^{-1}. These storm events did not increase flow greatly at Foxbridge but produced high concentrations of TDP (up to 1.4 mg L^{-1}) on all days when there was rain (Fig. 5.11). However, on only a few occasions did similar increased concentrations of TDP (up to 0.8 mg L^{-1}) occur at Jubilee. Thereafter, TDP concentrations at both stations increased as flow increased except for a sustained increase up to 0.2-0.3 mg L^{-1} from early June until the middle of July, after which concentrations declined again (Fig. 5.12).

In the third year, only the Foxbridge-Longlands field within Jubilee received P fertiliser (23 kg ha^{-1}) and not until the 1 June 2000. Very little rain fell after application except for an 8 mm storm on 9 June which produced a sharp rise in TDP concentrations up to 0.3-0.4 mg L^{-1} at both Foxbridge and Jubilee without increasing flow (Figs. 5.11 and 5.12).

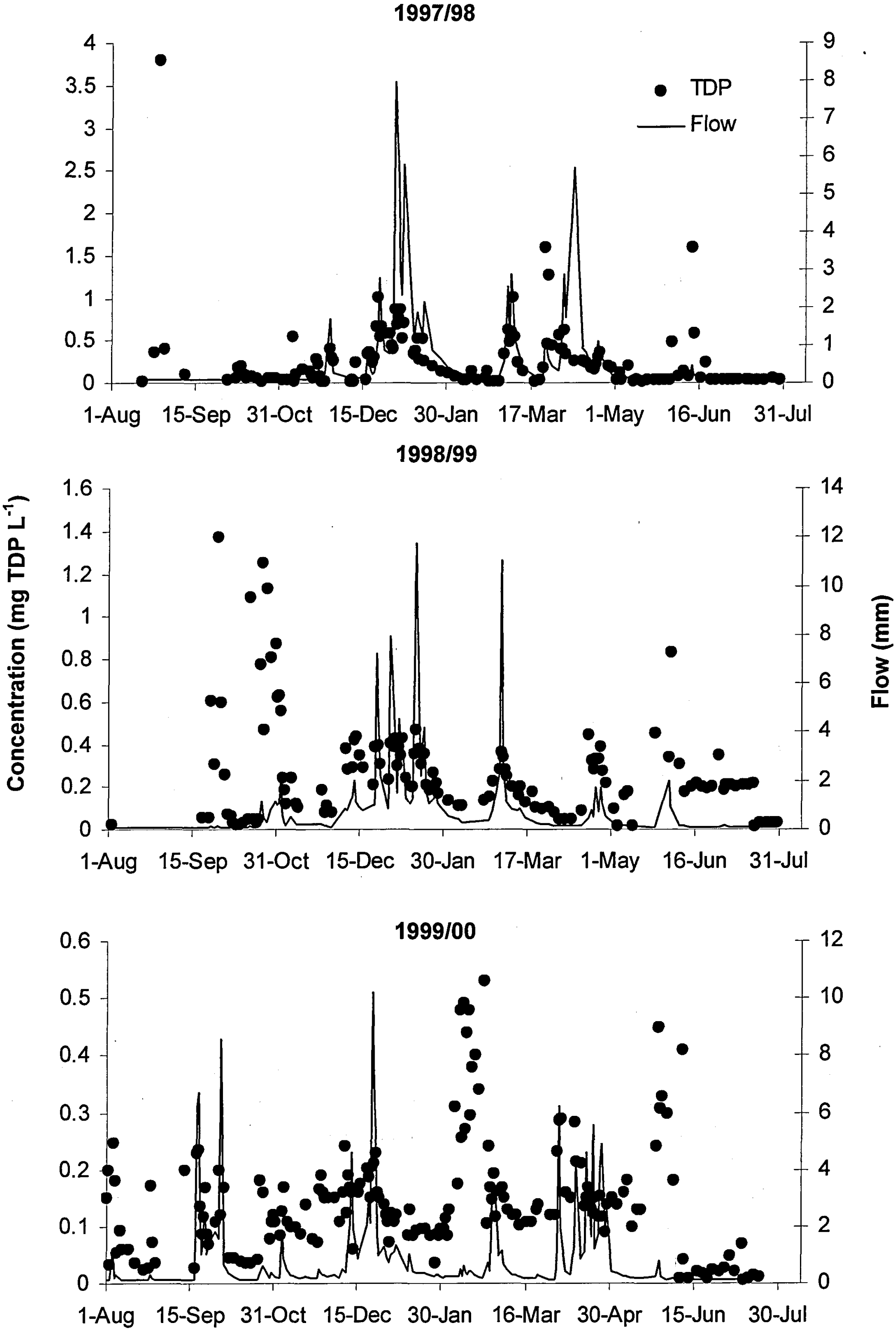


Figure 5.11 Temporal changes in the concentrations of total dissolved P (TDP) in relation to flow at Foxbridge in each year at Rosemaund.

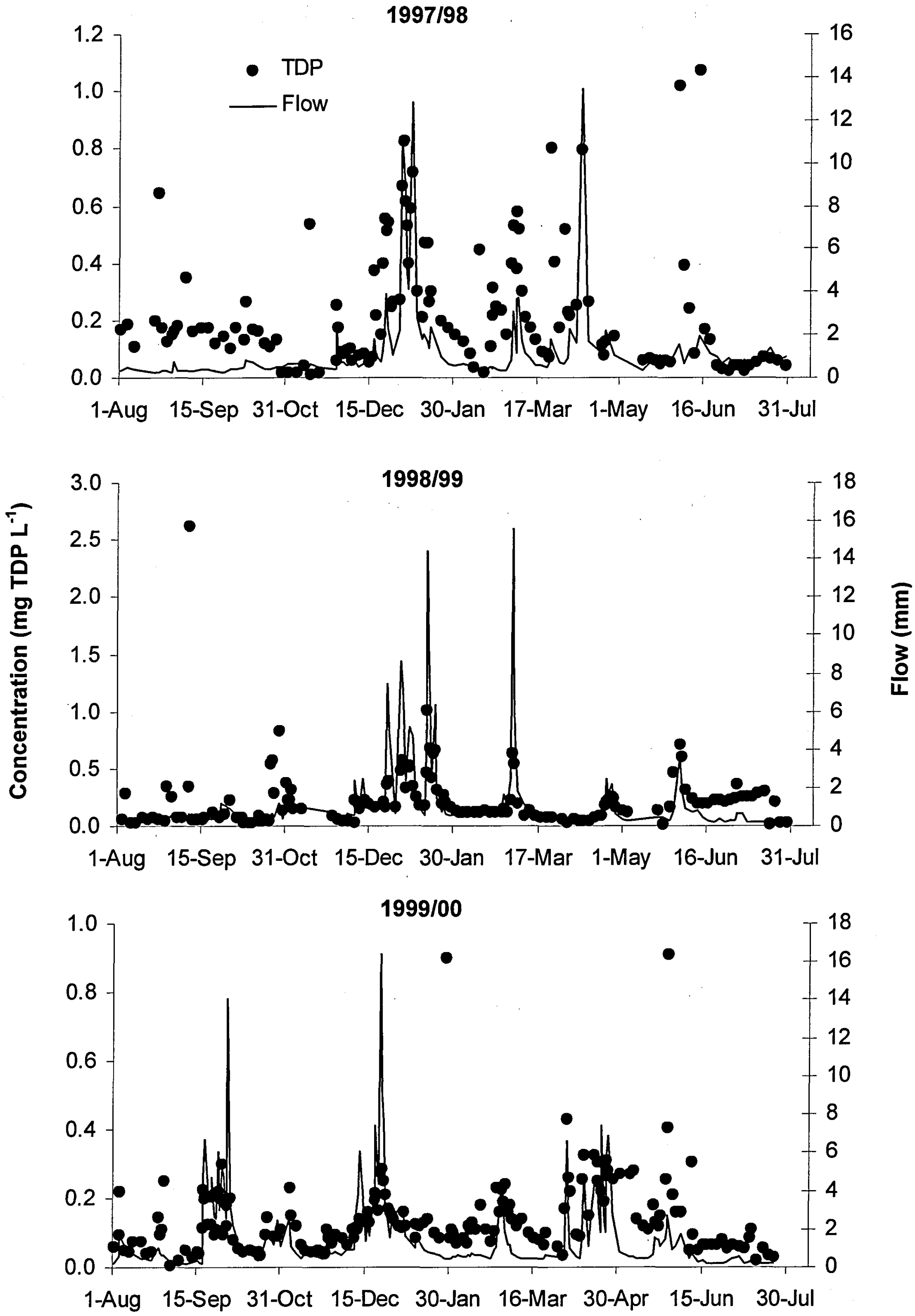


Figure 5.12 Temporal changes in the concentrations of total dissolved P (TDP) in relation to flow at Jubilee in each year at Rosemaund.

However, as in other years, high concentrations of TDP (ca. 0.5-0.6 mg L^{-1}) also occurred under relatively small increases in flow during February and May. There was a sustained increase in TDP during February 2000 at both stations, which coincided with the large increase in PP concentrations at Foxbridge and Jubilee, particularly at Foxbridge, suggesting a common source.

The FYM applied to Moorfield supplied approximately 100 kg P ha^{-1} and was applied over 4 days (11-14 January 1999) when the soil was wet. Concentrations of TDP rose to 9 mg P L^{-1} in a drain outfall from the field the day after the start of manuring even though very little rain fell (1.2 mm). More intense rain during 15-20 January 1999 maintained large TDP concentrations in the tile drain, and generated even greater concentrations (up to 17 mg P L^{-1}) in an adjacent field ditch. Maximum concentrations of total P in the tile drain and ditch were 11 and 21 mg P L^{-1}, respectively. The higher P concentrations in the ditch reflected additional P transfer in surface runoff from the field during the storm. The TDP concentrations at Belmont also increased above background after the manure application, but values were no greater than 1.7 mg P L^{-1}, a 10-fold dilution compared to the field values. TP concentrations at Belmont reached 3.5 mg P L^{-1}. The data have been published by Hodgkinson and Withers (2007) and are not illustrated further here.

5.6 Discussion

The monitoring period included three years with above average rainfall providing ample opportunity to detect any effects of soil cultivations and P input management on stream P concentrations and loads during storm events. Sub-surface flow through tile drains is the major stormflow pathway at Rosemuand with previous sediment-source tracing studies suggesting 55% of the catchment sediment yield arrives by this pathway following mobilisation at the soil surface (Russell et al., 2001; Walling et al., 2002; Chapman et al., 2005). About 92% of the catchment area is underdrained with 22 known drain outfalls. Assuming the Foxbridge drain is representative of other drains in the catchment, the main drainflow period (December to April) typically provided over 90% of the total annual flow. In one year, large flows were also observed during September and the December to April flow accounted for only 70% of annual flow.

Nevertheless some seasonality in flow is clearly evident at Foxbridge. Drainflow was very turbid during these main flow periods with SS and TP concentrations often over 0.5 g L^{-1} and 1 mg L^{-1}, respectively reflecting the highly dispersive nature of the Rosemaund soils. In one month (February 2000) flow-weighted TP concentrations reached over 2 mg L^{-1}.

Since there were no cultivations, or P inputs, within the Foxbridge catchment during the main drainage period, these large P concentrations are due to mobilisation of TDP and PP from the soil. The PP fraction was on average ca. 70% of TP and the P concentration of the SS transported was ca. 1000 mg kg^{-1}. The latter represents a P enrichment ratio (PER) of nearly 3 compared to the surface soil TP concentration in Foxbridge field of 340 mg kg^{-1}. Walling et al. (2002) considered that the SS transported through the Rosemaund drains was not totally derived from the surface soil but will have included some mobilisation of subsoil P. Hence the PER of surface transported P may even be slightly greater than 3. The DESPRAL laboratory test predicted SS and TP concentrations of 1.3–2.7 g L^{-1} and 0.6-1.1 mg L^{-1} for Foxbridge depending on soil type for particles <20 μm in size.

Whilst test TP concentrations are very similar to measured values, test SS concentrations are much greater suggesting significant retention (60-80%) of coarser soil particles with lower P content in transit from the field surface to the drain. Similar retention values for eroded soil in the fields at Rosemaund (i.e. gross.net erosion rates) were measured by Walling et al. (2002). The flow-weighted DRP concentration averaged 257 μg L^{-1}, which is well above the proposed riverine target of ca. 100 μg L^{-1} to limit eutrophication impacts (Duncan et al., 2006). The concentrations of TDP predicted by the DESPRAL test (ca. 150 μg L^{-1}) were much smaller than flow-weighted concentrations and more closely linked to spot median values. This difference maybe due to some release of DRP from the P-enriched transported particles during the equilibration time the samples remained in the automatic samplers before analysis. However, this does not account for the difference between mean flow-weighted concentrations and mean/median spot concentration values.

High flow-weighted TP concentrations (up to 1 mg L^{-1}), and sometimes high TDP concentrations (up to 0.9 mg L^{-1}), were also observed in small amounts of storm water from the Foxbridge drain. For example, during autumn (e.g. September and October 1998) and during summer months (e.g. June 1998 and 1999, and August 1997). These small flows will largely be preferential in nature due to the susceptibility of the underdrained Rosemaund soils to crack during summer and before soils have wetted up to field capacity. The source of the P mobilised in autumn 1998 is undoubtedly the large amount of FYM that was applied in September. Elevated spot concentrations of DRP and TDP were measured directly after application and during October, as has been observed in many studies (Withers et al., 2001; Owens and Shipitalo, 2005). There was also a large rise in DRP concentrations at Jubilee during the following January when flow rates increased under the high rainfall in that month suggesting some residual P effect of the FYM applied.

The greater TDP concentrations measured in June 1998 are probably due to a combination of mineralisation effects stimulated by the unusually high rainfall in that month (96 mm) falling on dry soil (e.g. Pote et al., 1999; McDowell and Trudgill, 2000) and P release from senescing cereal leaves (Sharpley, 1981). Since these concentrations were observed at both Foxbridge and Jubilee it is unlikely that the elevated values at Jubilee were due to release of DRP from stream bottom sediments. In a field trial (Holbach plots) at Rosemaund, Withers et al. (2001) recorded very high TDP concentrations in surface runoff during spring and summer. Normally such high concentrations would never reach the stream but a combination of sufficient rainfall, cracked soils and underdrainage may have been sufficient for P delivery. Indeed, it was surprising how often drainflow with high P concentrations was measured entering the stream during the biologically active spring and summer period. These large P concentrations under low flow maybe more ecologically relevant than the much larger P loads arriving during winter (Edwards et al., 2000b; Jarvie et al., 2006).

The large TDP concentrations measured under low flow at Foxbridge were not always measured at Jubilee due to the dilution by groundwater which made up the bulk of the flow. Similarly annual flow-weighted concentrations of PP were lower at Jubilee and Belmont than at Foxbridge, probably due to retention within the stream system. Conversely, high PP concentrations (up to 14 mg L^{-1}) were measured under low flow at Jubilee but not at Foxbridge; for example in autumn 1997. Similarly large PP concentrations were also noted at Belmont during that autumn after the majority of arable fields in the catchment had been cultivated for sowing of winter cereals. Since this PP signal was not observed at Foxbridge, it suggests that the PP was transported to the stream in overland flow. The sediment-sourcing work of Russell et al. (2001) found that 35% of the SS delivered to the Rosemaund stream was via surface runoff, particularly from winter cereal and old hopyard fields. Overland flow will also have contributed to the very high TDP and TP concentrations measured in the Moorfield ditch following the FYM application during January 1999. Large TDP and PP concentrations were also observed during February 2000 but the source of this P is unknown.

Storm transfers of PP during the main drainage period also showed a large degree of temporal variation; for example in contrasting the storms in December and April in Figs. 5.4 and 5.5. In December, large TP concentrations coincided with peak flow while in April, much lower TP concentrations preceded peak flow suggesting a shortage of entrained soil particles (Pacini and Gachter, 1999). This difference probably relates to a greater degree of soil protection provided by growing cereal crops in April as compared to December. The much smaller but consistent range in DRP concentrations during the two storm events reflects a continuous supply of soil P release, with the often observed lag in peak DRP concentrations relative to flow due to P adsorption onto detached soil particles. Hence a lag DRP peak was observed when SS concentrations were large in the December storm, whereas there was no lag for the April storm. Similar storm variability was reported by Lazzarotto et al. (2005).

Some exhaustion of the soil P pool does appear to have occurred in the third year of the study when flow-weighted DRP concentrations were consistently lower at both Foxbridge and Jubilee. No fertiliser was applied during autumn in the third year. These contrasting patterns in temporal variability of P concentrations relative to flow between Foxbridge and Jubilee were useful in separating out sources of P at Rosemaund. The farming operations (cultivations and fertiliser applications/FYM) were mainly carried out at a time of year when drainflow volumes were small and therefore their main effect was on stream concentrations and not loads, although some residual effects were observed. The majority of the P load was transported during high drainflows when soil P was the main source.

The Rosemaund stream is heavily influenced by groundwater and flows all year round. Baseflow analysis suggested a groundwater contribution of ca. 70% with average DRP concentrations of 0.1 and 0.6 mg L^{-1} at Jubilee and Belmont, respectively, whilst < 7 μg L^{-1} was measured in summer groundwater. This suggests considerable enrichment of baseflow from a non-point source at Jubilee, which tended to increase with flow. Lazzarotto et al. (2005) measured similarly large groundwater concentrations in a Swiss grassland catchment and suggested P release from stream sediments may be involved. This apparent enrichment may be due to the insensitivity of the Dingman baseflow separation analysis; for example by including P in slow seepage from tile drains or another sub-surface flow source as storm events subside. However, the P enrichment may also be due to DRP release following re-mobilisation of stream bottom sediments during the larger baseflows. Release of DRP from disturbed bottom sediments/gravels or sediments trapped in macrophytes is an important source of P for internal recycling (Koski-Vahala and Hartikainen, 2001; Gainswin et al., 2006).

If one assumes a DRP (or TP) concentration of 0.1 mg L^{-1} for baseflow at the Jubilee station, it can be calculated that stormflow contributed ca. 90% of the measured TP load of 3 kg ha^{-1} and 70% of the DRP load of 0.8 kg ha^{-1}. These loads correspond to average flow-weighted concentrations of DRP and TP in stormflow of 0.5 and 2.4 mg L^{-1}, respectively. If the discharge from the Foxbridge drain with flow-weighted concentrations of DRP and TP of 0.26 and 1.1 mg L^{-1} are typical of other drains in the catchment, the disparity between drain outflow and streamflow concentrations during storms suggests some additional P sources were mobilised, especially particulate-associated P. Similar calculations for Belmont assuming an average DRP and TP concentration in baseflow of 0.6 mg L^{-1} suggest stormflow contributed no DRP and 60% of the TP load, with a flow-weighted TP concentrations of 1.8 mg L^{-1}. Since stormflow will have contributed some DRP based on the Jubilee estimates, these approximate calculations suggest some net retention of DRP downstream of the farm STW. This is also suggested by the significantly larger P concentrations in the SS at Belmont compared to Foxbridge and Jubilee.

The stream at Belmont had much greater flow-weighted DRP concentrations than at Jubilee and was clearly influenced by the discharge from the farm STW and possibly contaminated runoff from the farmstead. The calculated discharge of TP from the farm STW (22.7 kg) represents only 6% of the average TP export measured at Belmont. The corresponding contribution to TDP export is 11%. The farm STW serves ca. 65 people (although only half of these are actual residents) which gives a per capita P loading of 0.35 kg TP or 0.65 kg TP based on only the resident population. For the same catchment, Johnes and Hodgkinson (1998) estimated a contribution of 0.38 kg ha^{-1} for a population of 35 people during an export coefficient modelling exercise.

5.7 Conclusions

The spatial and temporal variation in SS and P concentration in the flow from a field tile drain and at two downstream locations was investigated in relation to detailed knowledge of the timing of farm practices on the mixed farm at Rosemaund over a three-year period. Key land use and P input factors influencing the concentrations measured were:

1. the presence of extensive underdrains in the catchment which provided a rapid delivery route for mobilised P even under low flow conditions
2. autumn cultivations which provided a source of loose soil for entrainment in surface and sub-surface runoff and increased PP concentrations during major storm events
3. fertiliser and manure applications which not only maintained the supply of desorbable soil P but also provided a source of immediate and residual DRP supply to runoff water

Large concentrations of TDP and PP were observed under both low and high stormflow volumes, including the ecologically sensitive spring and summer periods. Large DRP concentrations delivered during spring and summer through tile drains maybe of direct ecological relevance. However, cultivation and P input effects on P concentrations were most apparent outside of the main drainage period and therefore did not contribute significantly to P export, which was largely governed by flow volumes and the detachment and release of soil P. The results highlight the disparity between concentration and load effects of farming practices at the catchment level and a greater understanding of the ecological significance of variably timed P inputs to headwater streams is needed to develop effective catchment management. Such understanding needs to take account of potential sources of P within rural catchments other than those investigated in this chapter.

6

THE EFFECT OF SOIL PHOSPHORUS ON PHOSPHORUS IN SUSPENDED SOLIDS IN RUNOFF: HOLBACH PLOTS

6.1 Introduction

Sources of P in runoff from arable and pasture land include the fresh application of fertilisers and manures to the land surface, and the accumulated residues of surplus P in the soil arising from previous P applications (Sharpley and Withers, 1994; McDowell et al., 2005). Unlike fresh P applications, which are often applied only once or twice a year, soil P is a more ubiquitous source that becomes active every time it rains. As reviewed in chapter 2, differences in farm type, land management and the polarisation of farming systems have created large local variation in surplus P inputs, and in concentrations of soil P accumulation depending on the depth over which surpluses have been incorporated into the soil (Karlen et al., 1991; Edwards and Withers, 1998).

Soil P concentrations therefore range from deficiency levels in extensive farming systems to excessively high values where farming practices have intensified, particularly on granivore farms, horticultural enterprises and arable land with potatoes and sugar beet in the rotation (Pierzynski and Logan, 1993; Domburg et al., 1998; Skinner and Todd, 1998). Within-field variability in soil P is also very high due to non-uniformity in inputs (e.g. grazing animals (Fisher et al., 1998), or manure spreading accuracy (Smith et al., 1998)), or due to differences in crop yields and P offtake (Sylvester-Bradley et al., 1999). Where soil P concentrations are high and these areas coincide with hydrological pathways to the watercourse, there is an increased risk of P transfer in storm runoff with potential to cause eutrophication.

Phosphorus is transported in runoff largely in association with soil colloids, particles and aggregates. For example, Pionki and Kunishi (1992) report that 70-80% of P in streams draining two catchments of varying size in Pennsylvania, USA was associated with suspended solids (SS) even though stormflow only accounted for 20% of the total flow in the stream. This pattern of P export is very similar to that in the Rosemaund catchment (Chapter 5). Soil P accumulation increases both dissolved reactive P (DRP) and particulate P (PP) concentrations in runoff (Heckrath et al., 1995; Sharpley, 1995). Large DRP concentrations in runoff occur as a result of P desorption from P sorbing surfaces, dissolution of unstable P precipitates in cultivated soils receiving lime, and transfer of colloids (Froelich, 1988; Pierzynksy et al., 2005a).

Large PP concentrations reflect the selective transport of P-enriched fine soil particles detached at the soil surface and hence are a function of both the amount of SS transported and the P content of the SS (Sharpley, 1980; Sharpley, 1985b; Quinton et al., 2001). The range of total (TP) concentrations in runoff from soils can therefore be quite variable; for example Quinton et al. (2003) measured a range of 0.4-5.5 mg TP L^{-1} for 26 European soils under standardised simulated rainfall. The dynamic interaction between dissolved and particulate phases during transport and originating from variable source areas within catchments adds further complexity to the P transfer process (Sharpley et al., 1981).

In principle, soil P accumulation must therefore be monitored and controlled to minimise potential eutrophication impacts. Most research has focused on the DRP fraction in runoff and relating runoff DRP to various indices of soil P release (e.g. Sharpley et al., 1996; Hooda et al., 2000; Maguire et al., 2005), particularly soil test P (STP) which is a measure of the labile store of available P in soils routinely used for predicting fertiliser needs. This has resulted in the development of models to predict DRP concentrations in runoff based on STP (Vadas et al., 2005) and recommendations to limit STP levels on farms (Sharpley et al., 2003). However, there has been relatively little research conducted on the impacts of soil P on PP concentrations in runoff. This is surprising because the particulate fraction is often the dominant form of P transported in land runoff, and plays an important role in nutrient cycling and biotic response within the water column (Syers et al., 1973; Klotz, 1985; Evans et al., 2004; Stutter et al., 2007). Some work has been conducted on the bioavailable P (BAP) portion of runoff which will include a desorbable P component (e.g. Sharpley and Smith, 1993; Uusitalo et al., 2001) but not the whole PP fraction. The research focus on DRP rather than PP probably reflects the greater bioavailability of dissolved P fractions to aquatic organisms (Hatch et al., 1999; Reynolds and

Davies, 2001), although measures to control P loss from agricultural land are targeted at both dissolved and particulate forms (Withers and Jarvis, 1998; Sims and Kleinman, 2005).

Suspended solids originating in land runoff are much finer textured than the soil from which they are derived due to the preferential mobilisation of clay and silt-sized aggregates during rainsplash and their subsequent aggregation during transport. For example, Walling et al. (2000) report median particle size diameters of only ca. 12 μm (no particles > 63 μm) for SS in a number of UK rivers and similar median values have been reported by Wallbrink et al. (2003) for Australian rivers (< 10 μm). The particle size of SS in outflow from tile drains in fields has been shown to be even finer; 2-4 μm (Laubel et al. 1999; Russell et al., 2001). These fine particles usually have a greater P content than the bulk soil due to the selective adsorption of added P onto the large surfaces areas associated with primary clay particles and associate microbial debris (Williams and Saunders, 1956; Syers et al., 1969; He et al., 1995). The degree of enrichment by clay and P is defined by an enrichment ratio (CER and PER) which is the ratio of clay/P in the SS to that in the soil, (Ryden et al., 1973).

The CER will depend on the distribution of clay particles within dispersed soil aggregates of varying size due to storm intensity and is usually lower in clayey soils than in sandy soils (Foster et al., 1985). The PER depends on the distribution of P within the particles transported and can range up to 5 in field and catchment studies (Pionki and Kunishi, 1992; Sharpley, 1985b), although values of 1-3 are much more common. While PER values are often quoted there is very little information on how PER changes with increasing soil P content. Sharpley and Smith (1990) concluded that PER was more influenced by the amounts of SS transported than the soil P content. However, their study included sites with a very large range in SS concentrations and it is not clear whether this relationship holds true for sites with much small SS concentrations that are typical of rainsplash detachment and sheet erosion from soils. The almost continuous loss of very fine soil particles and aggregates by sheet erosion during storm events is of considerable environmental significance (Stone and English, 1993; Owens et al., 2005).

Whether SS in runoff acts as a P source, or a P sink, within the stream channel depends not only on P sorption properties (e.g. P content, particle size distribution (PSD), organic matter (OM) content and mineralogy) and SRP concentration gradients operating within the water column, but also on a number of other interrelated biotic (e.g. microbial and algal activity) and abiotic factors (pH, redox conditions, calcite), (House, 2003; Stutter et al., 2007). However, the P sorption properties of SS are of fundamental importance in understanding their potential role in stream P dynamics and is usually characterised by the equilibrium P concentration (EPC_0), (Taylor and Kunishi, 1972; Froelich, 1988; House et al., 1995). For example, Scalenghe et al. (2007) found that the amounts of water-extractable P (WEP) in clay fractions of different EU soils were less than those in silt and sand fractions from the same soils, due to their greater P buffering capacity. However, the effect of soil P level on the P sorption properties and EPC_0 of suspended particles in runoff has not been widely investigated since the seminal work of Sharpley (1980). This represents an important gap in our ability to understand the ecological relevance of PP transfer in land runoff and identify appropriate control measures. If soil P does not have a significant influence on PP concentrations and potential P release to receiving waters, this may have important consequences for developing programmes of measures in catchments to reduce agriculture's contribution to eutrophication.

The two previous chapters have demonstrated that soil P is a major source of PP transfer in runoff water within an upland and lowland environment. This chapter further explores this linkage by investigating more closely the effect of increasing soil P concentration on the P content and sorption properties of SS in overland flow from field plots at Rosemaund. As highlighted in Chapter 5, the arable soils at Rosemaund are both P-rich due to a legacy of hop production and highly dispersive, causing large concentrations of SS and P to enter the stream in surface and sub-surface runoff. This site therefore provided a unique opportunity to study particulate P enrichment and the potential consequences for eutrophication associated with their P release properties. Runoff water from hillslopes in this catchment might be expected to have a greater DRP concentration than the receiving stream (e.g compare Foxbridge and Jubilee monitoring stations in Chapter 5) which might trigger a release of P from the particulate P component in runoff on entering the stream. It was envisaged that the effects of soil P accumulation on the PP in runoff would be less marked than for DRP because of the selectivity of finer P-enriched particles during the detachment process. In particular, it is not clear whether an increase in soil P fertility (for example from 20 mg L^{-1} to 60 mg L^{-1}) at Rosemaund would significantly influence PP concentrations or the EPC_0 of the transported SS.

6.2 The Holbach plots

The experiment was conducted on some field (Holbach) plots at Rosemaund that had been previously established to study the impact of various fresh P additions on P export in overland flow at the field scale (Withers et al., 2001). Variable amounts of fertiliser (250-4000 kg P ha^{-1}) were added to these existing plots in 1998 to establish a wide range in soil total P (TP), Olsen-P (OP) and WEP concentrations. A different soil TP and OP concentration was established on each of the fifteen plots such that there was no replication. Surface runoff from the plots was collected for a total of 22 storm events over the three winter monitoring periods (Periods 1-3) for determination of SS and P concentrations. The plots were not cropped and no further P was applied. In monitoring period 3, samples of SS from each plot were collected for determination of their particle size characteristics and P sorption properties relative to those in the soil. More comprehensive details of the treatments and the methodologies used to collect runoff and measure SS, P concentrations and P sorption properties were provided in Chapter 3.

6.3 Soil phosphorus

Balance calculations based on soil analysis results in April 2000 indicated that all of the applied P fertiliser was recovered as increased soil TP within the top 20-25 cm, and that ca. 30% and 13% of the added P was bound in OP and (WEP) forms, respectively. These soil extractable P accumulation rates are identical to those measured in the previous experiment after consecutive applications of smaller amounts of fertiliser P over a longer time period (Withers et al., 2001). Whilst soil TP concentrations remained constant, OP and WEP concentrations decreased slightly (e.g. OP by 6%) during the 4-year experimental period probably due to P fixation processes in the absence of fresh fertiliser inputs. Averaged over the 4 years, the range in soil P concentrations achieved across the plots was 378-1033 mg kg^{-1} for TP, 19-194 mg kg^{-1} for OP and 3-75 (mg kg^{-1}) for WEP (Table 6.1). Concentrations of $CaCl_2$-extractable P at the start of the experiment ranged from 0.5-20.3 mg kg^{-1} and with a tendency to increase once OP concentrations exceeded ca. 35 mg kg^{-1}. According to Hesketh and Brookes (2000), this change-point represents the concentration of OP at which accelerated P release to runoff occurs at this site. Concentrations of oxalate-extractable P represented between 60 and 95% of total P suggesting that Fe and Al sesquioxides were the major sorption sites for the applied P.

Table 6.1 Phosphorus concentrations and sorption characteristics of the Holbach soils.

Plot	Soil P			Langmuir parameters[1]				
	TP	OP	WEP	Q_{max}	k	EPC_0	P_{sat}	PSI
		(mg kg^{-1})		(mg kg^{-1})		(mg L^{-1})	(%)	
1	408	24	5.1	270 (33.6)	0.33 (0.05)	0.12	8.9	5.1
2	378	19	3.2	288 (16.7)	0.34 (0.07)	0.08	6.3	5.5
3	842	125	40.0	269 (55.6)	0.09 (0.01)	3.02	50.9	2.5
4	840	146	56.6	293 (76.6)	0.09 (0.01)	4.32	52.6	2.2
5	1033	194	75.4	289 (4.1)	0.02 (0.03)	10.15	74.4	0.7
6	713	112	42.0	254 (64.4)	0.11 (0.02)	3.88	52.4	2.0
7	496	49	14.0	311 (18.7)	0.17 (0.03)	0.47	16.4	4.4
8	459	41	11.4	283 (11.0)	0.22 (0.04)	0.31	14.5	4.3
9	984	171	52.2	265 (3.9)	0.05 (0.05)	8.15	71.7	0.9
10	552	88	24.9	278 (16.8)	0.11 (0.02)	1.03	28.1	3.8
11	908	149	60.0	239 (3.6)	0.05 (0.01)	6.72	70.7	1.2
12	383	29	5.4	291 (39.3)	0.27 (0.05)	0.16	9.6	5.0
13	428	36	8.7	287 (35.3)	0.21 (0.03)	0.35	14.3	4.3
14	393	25	4.8	302 (16.1)	0.27 (0.05)	0.15	8.9	5.0
15	455	38	9.6	336 (18.1)	0.18 (0.04)	0.35	11.6	4.5

[1]Determined by double Langmuir function except for plots 5, 9 and 11 which were better fitted by a single Langmuir function. s.e. values given in brackets. k values were determined by single Langmuir over the range 0-15 mg L^{-1}.

The maximum P sorption capacity (Q_{max}) as fitted by the Langmuir function ranged from 239 to 336 mg kg^{-1} (mean 284 mg kg^{-1}, s.e. 6.1) but was not related to soil P level. The soil P saturation percentage (P_{sat}, calculated as OP/Q_{max}) ranged from 6-74% and there was an order of magnitude variation in the EPC_0, 0.1 –10 mg L^{-1}, Table 6.1. Values of P_{sat} were well predicted by the single point P sorption index (PSI, P_{sat} = 81 – 14.4 PSI, r^2 0.98, P <0.001), and highly correlated to the degree of P saturation (DPS) as measured by oxalate with no α correction (DPS = 0.46 P_{sat} + 10, r^2 0.97, P <0.001). The close correlation between P_{sat} and DPS suggested that native sorbed P was adequately described by OP. The fertiliser additions tended to slightly decrease soil pH (range 6.6-7.0) and increase total Ca (range 4.2-5.1 g kg^{-1}) and oxalate–extractable Al (range 0.7-1.2 g kg^{-1}), but the changes were small. The range in oxalate-extractable Fe was 1.3-2.0 g kg^{-1} and not related to the P additions.

In-situ mobilisation of phosphorus

Dispersion of the soil using sodium chloride showed that the P content of both the fine silt (2-20 μm) and clay (<2μm) fractions significantly increased as soil P levels increased, but that the coarse silt fraction (20-63 μm) was not P enriched (Fig. 6.1a). The degree of P enrichment of the <2 μm fraction was considerably greater than that of the <20 μm fraction. Potential mobilisation of soil particles (>0.45 μm) and P forms due to rainfall impact in storm runoff from selected plots (Plots 2, 4, 5, 6, 7, 8, 10, 13 and 15) was assessed by the DESPRAL test (Withers et al., 2007). Concentrations of SS dispersed were uniformly high (2.3-2.8 mg L^{-1}) and showed no relationship with STP level. Concentrations of TDP and PP were related to STP (Fig. 6.1b), when the data for plot 5 were excluded. Addition of P therefore increased both the TDP (<0.45 μm) fraction and the PP (>0.45 μm) fractions.

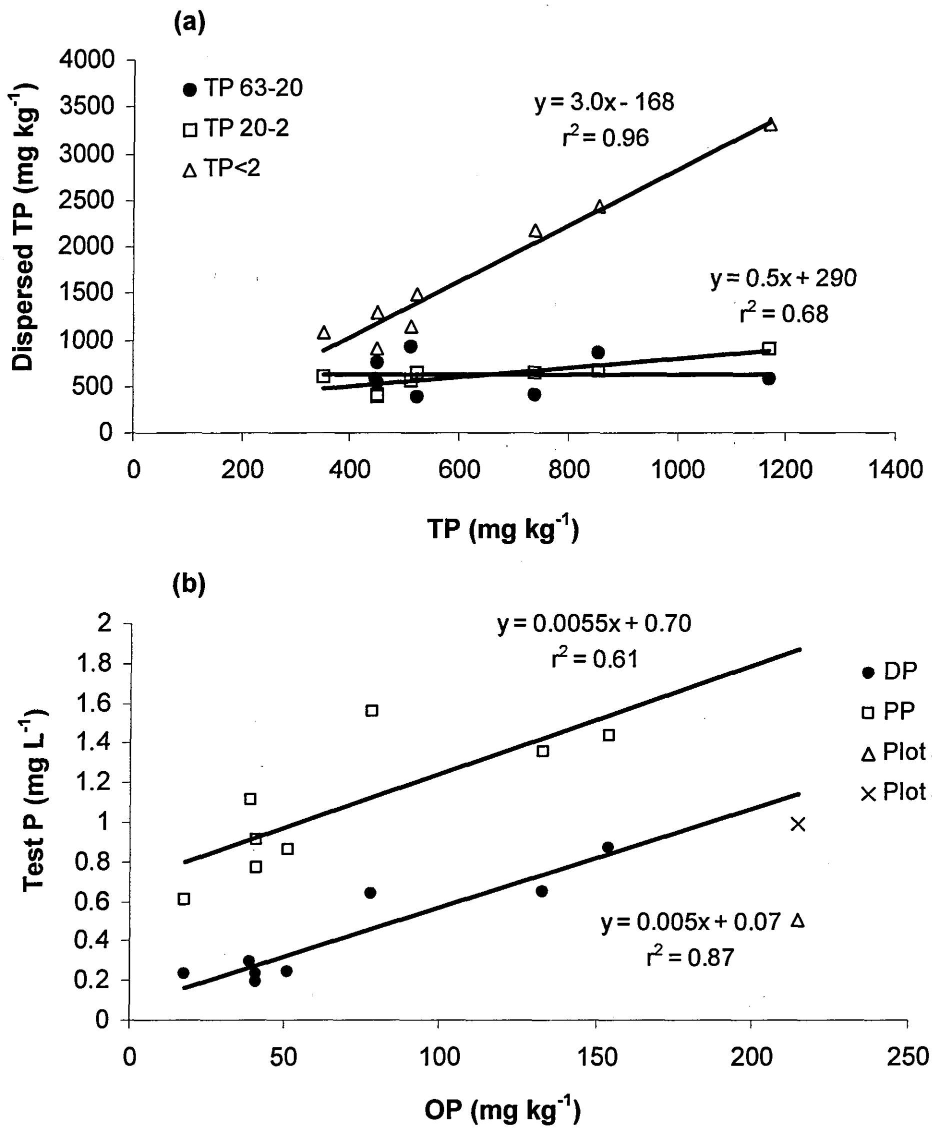


Figure 6.1 Effects of soil P concentration on dispersed P, (a) within soil particle size fractions dispersed by sodium chloride (Data supplied by E. Barberis, Italy) as a function of soil total P (TP) and (b) concentrations of total dissolved P (TDP) and particulate P (PP) dispersed by the DESPRAL test as a function of Olsen-P (OP). Data for plot 5 were treated as outliers.

6.4 Suspended solids and phosphorus in surface runoff

6.4.1 Surface runoff

The storm events monitored resulted in variable amounts of overland flow ranging up to 3 mm per event (Fig. 6.2). Most runoff was generated when the soil was left uncultivated during the first monitoring period (average of 1.7 mm plot^{-1}), representing between 3 and 11% of incoming rainfall. In the second and third monitoring periods after the soil was cultivated, the average amounts of overland flow fell by nearly an order of a magnitude to ca. 0.1-0.3 mm plot^{-1}, and representing only 1% of rainfall. This low runoff rate after cultivation is similar to that found in other experimental work at Rosemaund (Smith et al., 2001), and also in the previous study at this field site (Withers et al., 2001).

Within a single storm event, individual plots showed up to ten-fold variation in runoff volume when the soil was cultivated, depending on the micro-topography of the soil surface within any one year. This variation was notably absent in the first monitoring period when the undisturbed and therefore more consolidated surface generated more uniform amounts of overland flow. Cumulative runoff over the whole experimental period averaged 11.7 mm (s.e. 1.02, range 4.4 - 17.4 mm).

6.4.2 Suspended solids

No rill erosion was observed on any plot and rainsplash detachment was the dominant process of soil particle entrainment. Concentrations of SS in runoff varied from 0.1 – 5 g L^{-1}, with mean and median values of 0.6 g L^{-1} (s.e. 0.03 g L^{-1}) and 0.5 g L^{-1}, respectively. In the first monitoring period when runoff volumes were larger, concentrations of SS fell within a much narrower range (0.3 – 1.2 g L^{-1}), and only after cultivation, when plot runoff volumes were frequently below ca. 0.5 mm (<15 L plot^{-1}), were values outside this range (Fig. 6.3). The lower range in SS concentrations over a wide range in runoff volumes in the first year may be at least partly due to the filtering effect of the weed cover that had built up on the plots. After cultivation, variation in SS concentrations between events was related to storm intensity and antecedent soil moisture. For example, in the third monitoring year, a mean SS concentration of 0.3 g L^{-1} was obtained for event 14 under low intensity rainfall, 1.3 g SS L^{-1} for event 19 under high intensity rainfall and 2.2 g L^{-1} for event 20 under low intensity rainfall which followed the day after event 19.

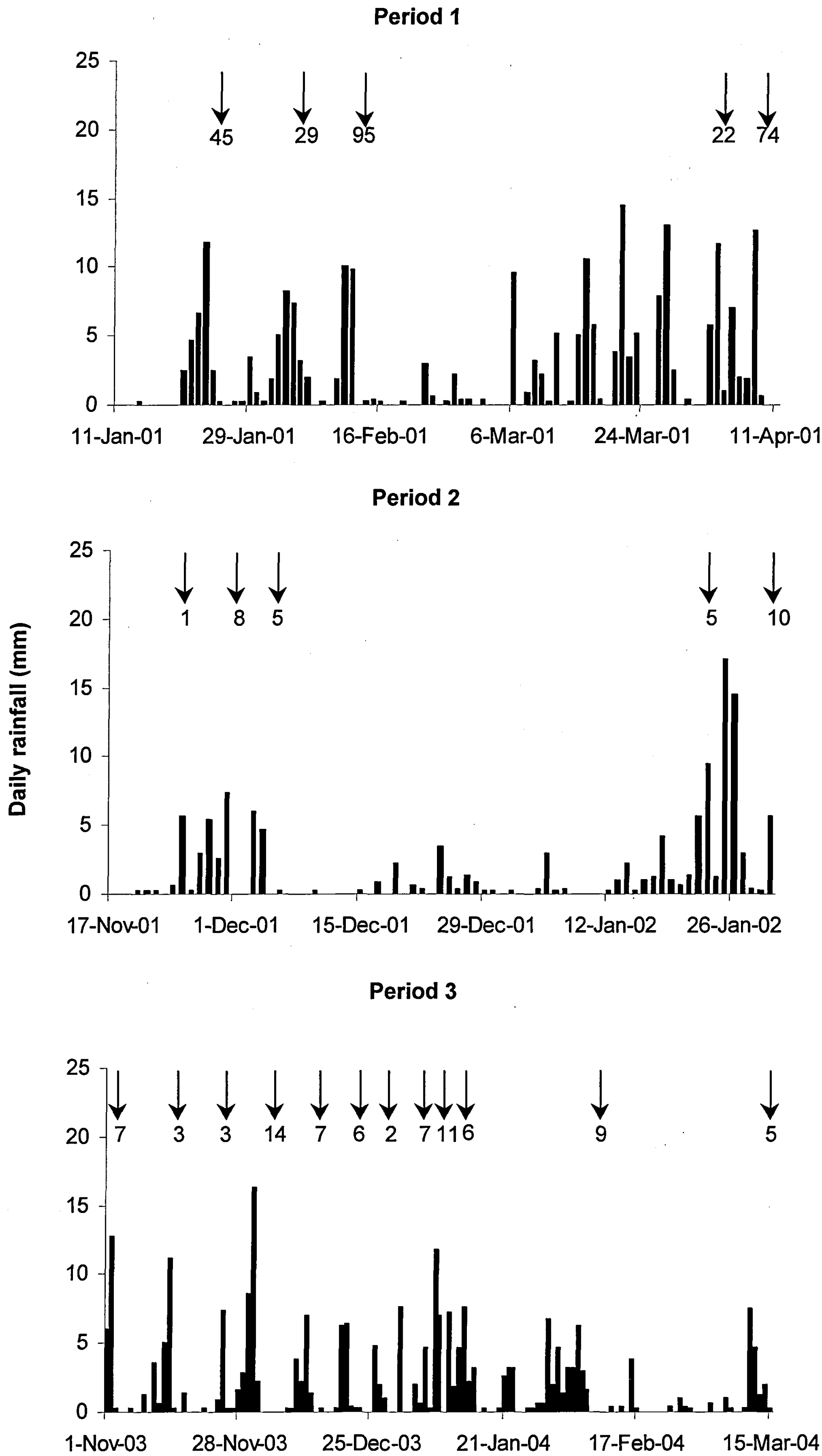


Figure 6.2 Monitored runoff events (indicated by down arrows) in relation to daily rainfall during each monitoring period. Mean runoff volumes (L plot^{-1}) collected for each event are also shown.

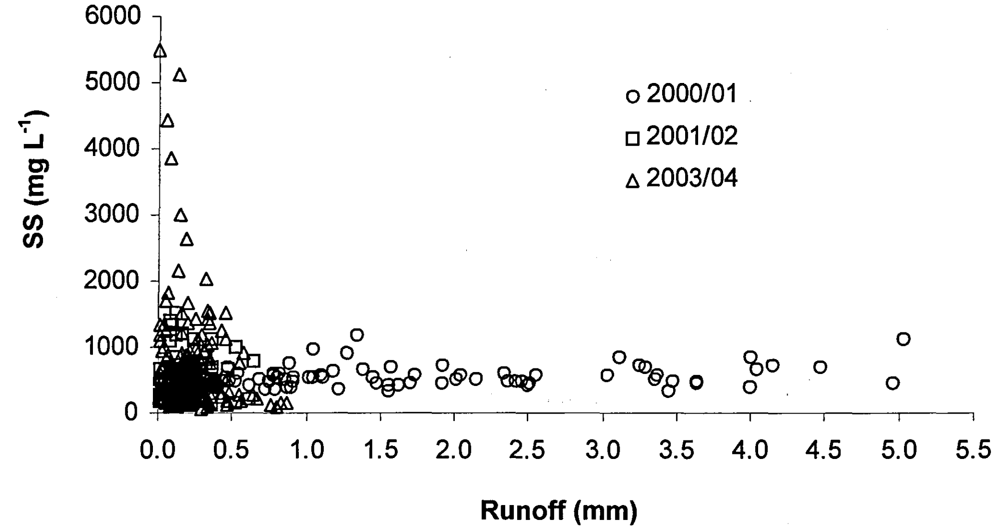


Figure 6.3 Concentrations of suspended sediment (SS) as a function of runoff volume.

Cumulative export of SS over the three monitoring periods ranged from 29–112 g ha^{-1} (Table 6.2), with greater export in the first monitoring period when runoff volumes were much greater, except on plot 3 which always generated relatively low runoff. Flow-weighted SS concentrations averaged 0.6 g L^{-1} in the first two monitoring periods and 0.7 g L^{-1} in 2003/04. There was no effect of soil P level on export, or on flow-weighted concentrations, of SS.

Table 6.2 Cumulative runoff and associated exports of suspended sediment (SS), dissolved-reactive P (DRP), dissolved-unreactive P (DUP), particulate P (PP) and total P (TP) from each plot at Rosemaund.

Plot	Runoff (mm)	SS (g ha^{-1})	P export (mg ha^{-1})			
			DRP	DUP	PP	TP
1	10.3	57	25	6	70	101
2	14.5	67	38	7	126	172
3	4.4	29	78	9	62	148
4	11.4	76	110	7	125	241
5	10.3	76	219	22	130	371
6	15.3	81	104	11	148	263
7	6.0	31	16	4	42	62
8	10.9	51	35	6	90	130
9	7.6	46	80	10	90	180
10	15.0	85	80	11	134	224
11	16.4	93	153	8	173	334
12	16.3	96	36	6	131	172
13	11.1	75	36	5	101	141
14	17.0	112	32	5	150	187
15	8.7	60	26	5	85	116

The SS collected in runoff from storm events 11-18, 19 and 21 in the third monitoring period contained no sand (>63μm). The SS from events 11-18 and event 21 showed very similar particle size distribution (PSD) across all plots, with a median particle size of 6 μm and comprising 19% clay and 81% silt. Compared to the original soil, the sediment from these two storms were enriched in the 0.45-2, 2-6 and 6-20 μm sizes and depleted in the 20-63 μm fraction (Fig. 6.4). Sediment from storm 19 showed a different PSD; the median particle size was significantly smaller (4 μm) and the clay content much greater (29%). This sediment was more enriched with particles of size 0.45-2 and 2-6 μm than the sediment from the other storms. There was no enrichment in the 6-20 μm size range and there were no particles >20μm (Fig. 6.4). There was no apparent enrichment in the <0.45 μm fraction (i.e. colloidal material) in any of the storm events.

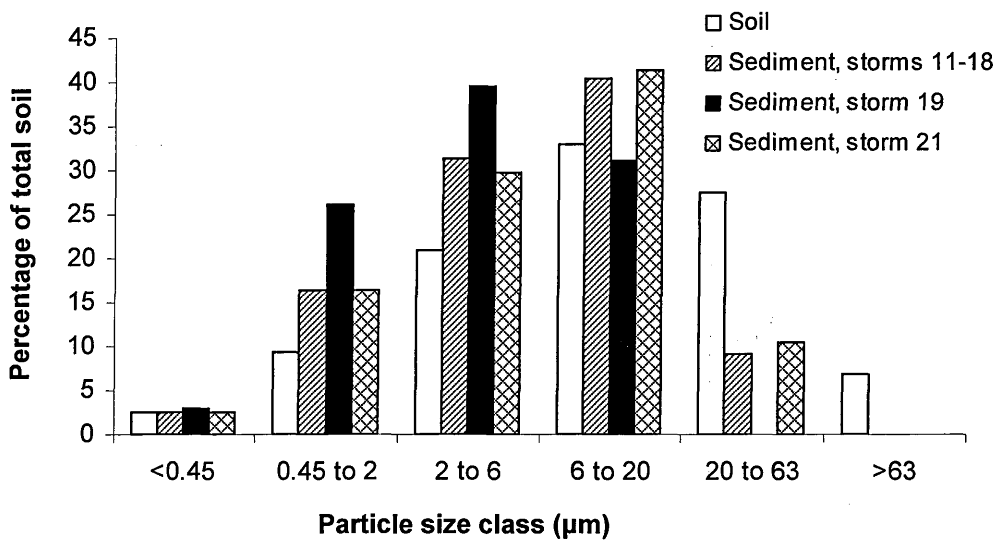


Figure 6.4 Particle size groupings of turbid runoff from storms 11-18, 19 and 21 in relation to the original soil.

For event 19, 0.34 mm of runoff was generated after 19 mm of rain fell in two days including a period of 5 hours when 2-4 mm of rain fell every hour. Event 21 generated a similar amount of runoff (0.27 mm) as event 19 but as a result of a prolonged period (48 mm in 28 days) of light rain where daily rainfall never exceeded 7mm. This contrasting pattern of intensity of precipitation resulted in differences in both SS entrainment (1.3 g L^{-1} for storm 19 vs 0.8 g L^{-1} for storm 21) and in the particle size transported (finer particles). Combining all storm sample particle size data available, there was a statistically significant (P <0.001) positive relationship between the concentration of SS in the runoff and the amount of clay enrichment (Fig. 6.5). The relationship suggests that CER would increase by 1 for every g of SS per litre of runoff. As more soil particles became entrained in the runoff, more clay particles became preferentially transported and average median particle size decreased.

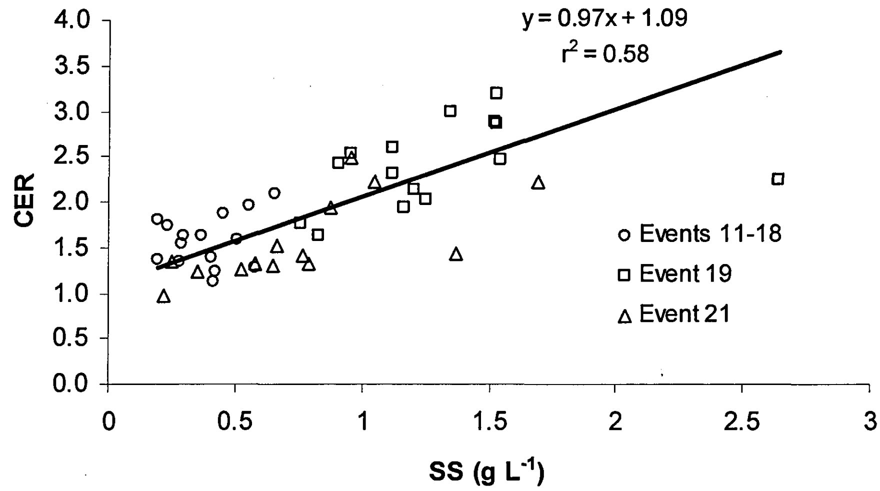
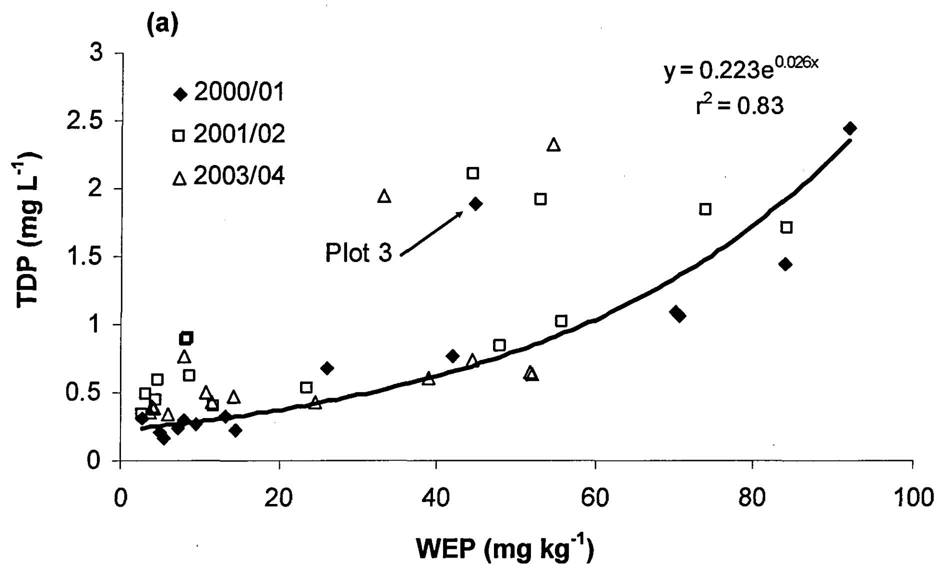


Figure 6.5 Relationship between the concentration of suspended sediment (SS) in runoff from storm events 11-18, 19 and 21 and the clay enrichment ratio (CER) of the sediment.

6.4.3 Phosphorus

Concentrations of DRP, TDP and TP in runoff ranged up to 4.1, 4.5 and 7 mg L^{-1}, respectively depending on soil P level; data for flow-weighted TDP and PP concentrations in relation to soil WEP and TP concentrations are shown in Fig. 6.6. The concentrations of dissolved P after cultivation (2001/02 and 2002/03) were often higher than in the first monitoring period due to the low runoff volumes measured. These differences were observed at both low and high soil P concentrations but were more consistent at low soil P and where runoff volumes were least. Concentrations of PP were also much more variable under low runoff volumes (Fig. 6.6b). Trends in runoff P concentrations due to differences in soil P fertility are therefore best represented by values obtained in 2000/01 when runoff volumes (with the exception of plot 3) were more uniform.



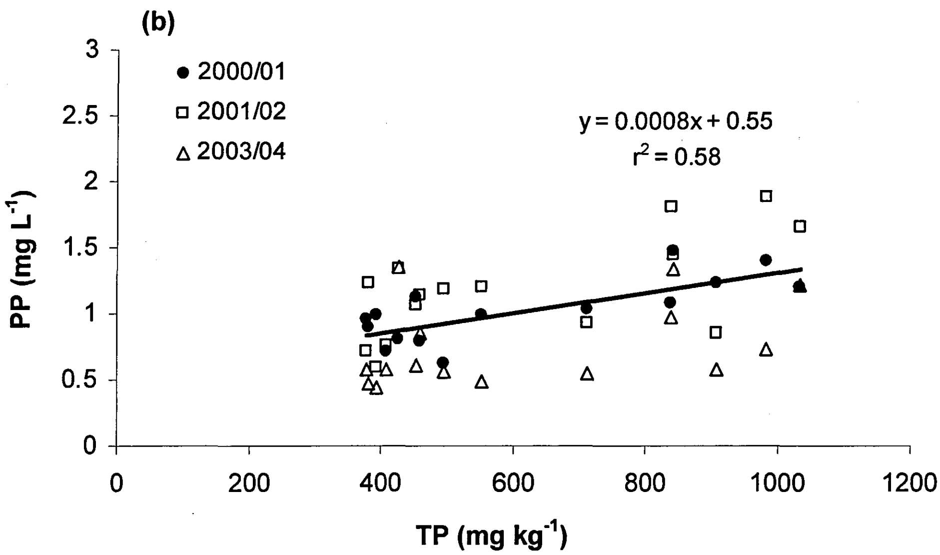


Figure 6.6 Relationship between (a) total dissolved P (TDP) and soil water-extractable P (WEP) and (b) particulate P (PP) and soil total P for the Holbach plots. The regression line shown represents the first year's data when runoff volumes were more uniformly large.

In 2000/01, runoff TDP concentrations were very similar at low soil WEP and OP concentrations (ca. <20 and <50 mg kg^{-1}, respectively), and tended to increase exponentially only at higher soil P concentrations (Fig. 6.6a). The majority (80-95%) of the measured TDP was molybdate-reactive. In contrast to TDP, runoff PP concentrations increased linearly rather than non-linearly (Fig 6.6b). As the soil P level increased, the proportion of TP in runoff which was in dissolved form increased and at soil WEP concentrations of ca. 50 mg kg^{-1} (ca. 140 mg kg^{-1}OP), the runoff P switched from being mostly (>50%) in particulate form to being mostly in dissolved form.

6.4.4 Phosphorus in suspended solids

Concentrations of TP (841-1499 mg kg^{-1}), OP (63-171 mg kg^{-1}) and WEP (6-43 mg kg^{-1}) in the SS collected for storms 11-18 were uniformly high across all plots compared to the large initial range in soil P concentrations (Table 6.3), and to which they were unrelated. Average values were 1120, 118 and 24 mg kg^{-1}, respectively. PER values were consequently greatest on plots with initially lower soil P and declined as soil P increased (Fig. 6.7a). For storms 11-18, PER values ranged from 1.2 - 3 for TP, 0.7 – 6.6 for OP and 0.5 – 6.1 for WEP. TP enrichment of the SS collected from storms 19 and 21 tended to be slightly lower than those for storms 11-18 (Fig. 6.8a), whilst very similar OP and WP enrichment of SS was obtained for storms 19 and 21 where there was enough sediment for analysis (Fig. 6.7b). There was no relationship between sediment CER and PER due to the dominant effect of soil TP on sediment PER.

Table 6.3 Concentrations of TP, OP and WEP, and P sorption characteristics, of suspended sediment collected from each plot during storm events 11-18 in 2003/04.

Plot	TP	OP	WEP	Langmuir parameters[1] Q_{max}	k	EPC_0	P_{sat}	PSI
		(mg kg^{-1})		(mg kg^{-1})		(mg L^{-1})	(%)	
1	1220	146	19.5	433 (75.2)	0.07 (0.04)	2.54	33.7	3.3
2	971	114	6.0	469 (60.1)	0.09 (0.03)	0.72	24.3	5.2
3	1400	165	25.6	396 (9.2)	0.12 (0.02)	2.93	41.7	4.2
4	1380	171	26.8	422 (24.1)	0.10 (0.01)	1.68	40.5	4.7
5	1320	157	43.1	528 (7.5)	0.06 (0.01)	4.57	29.7	3.8
6	1040	136	32.1	455 (131.9)	0.06 (0.04)	5.47	29.9	2.6
7	981	114	27.7	420 (137.8)	0.12 (0.00)	1.96	27.1	5.0
8	1210	138	26.5	435 (22.4)	0.09 (0.02)	1.94	31.7	4.8
9	1200	122	11.6	496 (54.7)	0.125 (0.02)	0.85	24.6	5.9
10	944	82	22.2	470 (9.1)	0.09 (0.00)	1.76	17.4	5.8
11	1290	100	22.7	398 (13.6)	0.12 (0.01)	1.58	25.1	5.7
12	841	63	21.4	416 (103.5)	0.11 (0.02)	1.01	15.1	5.7
13	920	85	24.8	404 (92.4)	0.08 (0.02)	1.93	21.0	4.9
14	968	78	22.4	415 (36.0)	0.07 (0.02)	2.40	18.8	4.4
15	1110	104	31.1	461 (102.7)	0.07 (0.03)	3.00	22.6	3.9

[1]Determined by double Langmuir function except for plots 3, 5, 10 and 11 which were better fitted by a single Langmuir function. s.e. values given in brackets. k values were determined by single Langmuir over the range 0-15 mg L^{-1}.

The P sorption capacities of the SS collected from each plot during storms 11-18 ranged from 398 – 528 (mean 441, s.e. 9.8) and were consistently greater than those of the soils by ca. 57% reflecting their clay enrichment (Table 6.3). Similarly, values of P_{sat} (15 - 42%), EPC_0 (0.7 – 5.5 mg L^{-1}) and PSI (2.6 – 5.9) fell within relatively narrow ranges compared to the original soils, and were unrelated to the initial soil P concentrations.

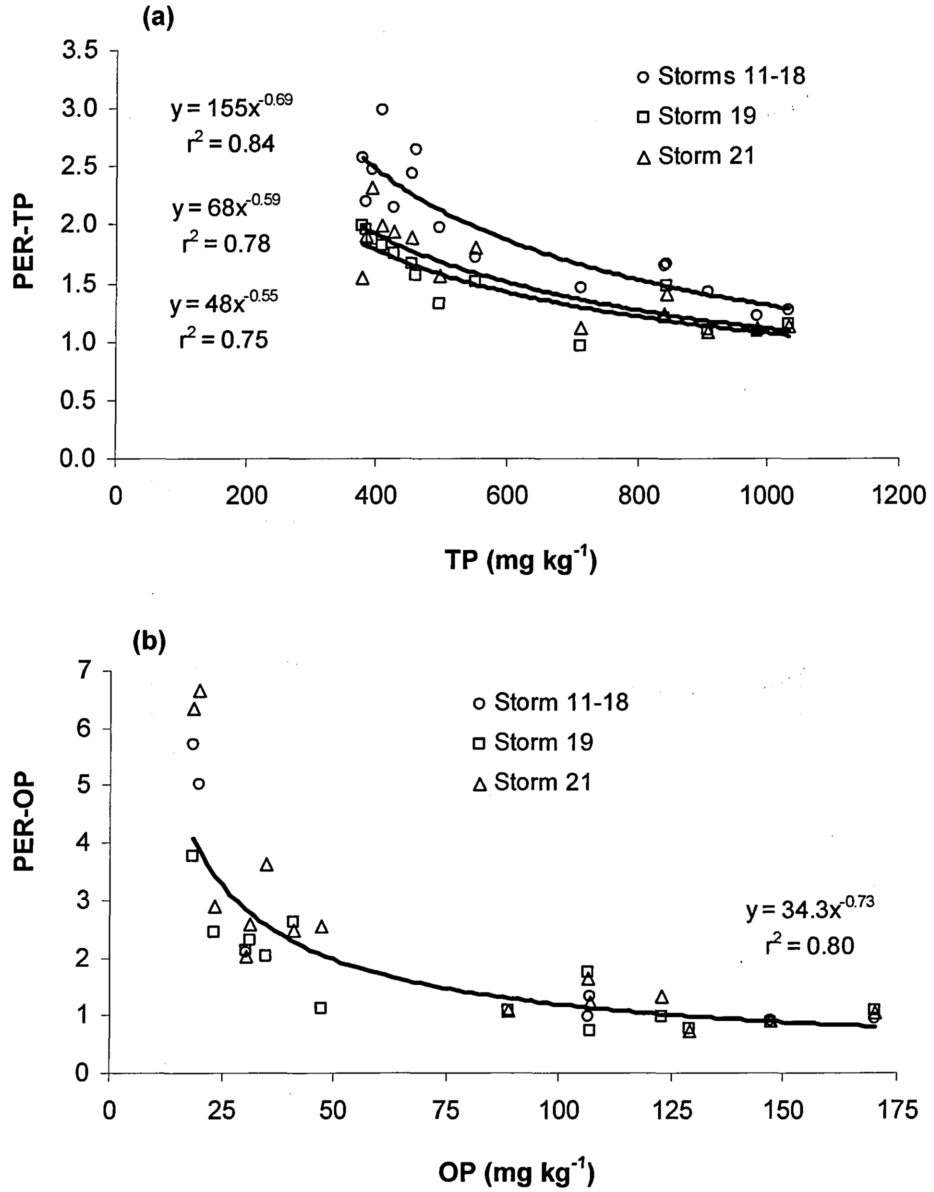


Figure 6.7 Relationship between (a) enrichment ratio of TP in SS (PER-TP) and soil total P, and (b) enrichment ratio of Olsen-P (PER-OP) and soil Olsen-P for storms 11-18, 19 and 21 in 2003/04.

At low soil P concentrations, sediment P_{sat} and EPC_o tended to increase, and sediment PSI tended to decrease, compared to the original soil. At high soil P concentrations, the reverse occurred; changes in sediment EPC_0 concentrations with increasing OP concentrations in the original soil are shown in Fig. 6.8a. The OP concentration in the soil at which SS changed from being a P sink (soil EPC_0 > sediment EPC_0) to a P source was ca. 110 mg kg^{-1} as shown in Fig. 6.8b. This OP concentration is equivalent to ca. 40% saturation of the soil P sorption capacity of the soil and ca. 25% P saturation of the P sorption capacity of the SS.

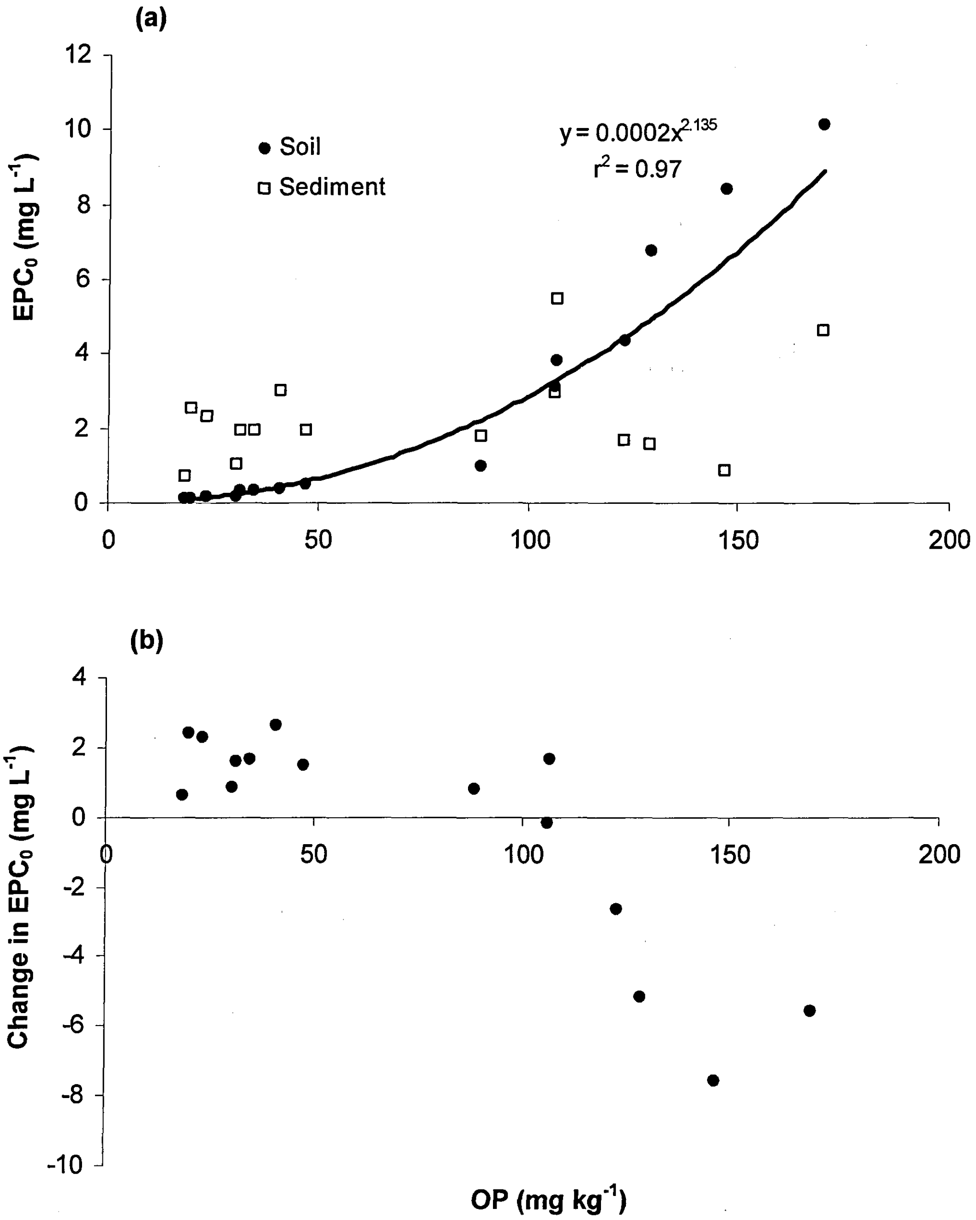


Figure 6.8 Relationship between (a) the equilibrium P concentration at net zero sorption (EPC_0) in sediment and soil and (b) the change in EPC_0 (sediment minus soil) as a function of the original Olsen-P concentrations in the soil.

6.5 Discussion

The excessive accumulation of surplus P in soils is a major source of P in land runoff and it is important to manage soil P concentrations such that they are adequate to provide agronomic benefits but without causing deterioration in water quality through eutrophication impacts. The initial particle size dispersion and DESPRAL analysis of the soil indicated that soil P accumulation had increased both the dissolved and the particulate fractions but to varying degrees. In the runoff, soil P had raised DRP concentrations at a greater rate than PP concentrations, although DRP concentrations did not start to increase until OP exceeded 50 mg kg^{-1}. The lack of any effect of soil P level on DRP concentrations below 50 mg OP kg^{-1} was also apparent in the results of the DESPRAL laboratory test. This apparent change-point correlates to the 25% P saturation above which Langmuir adsorption theory predicts P binding energy decreases (Holford et al., 1974; 1997). Change-points have been identified for a number of soils using $CaCl_2$ as a surrogate for runoff DRP concentrations (Hesketh and Brookes, 1997). For the Holbach soils, the $CaCl_2$ change-point was found to be ca. 35 mg kg^{-1}, a value which is slightly different to the value based on DRP in runoff. However, even at the lowest soil OP concentration tested (19 mg kg^{-1}) and for all soils with <50 mg kg^{-1}, DRP concentrations in plot runoff and in the DESPRAL test were above 0.2 mg L^{-1}. These concentration values are identical to those measured in field drain flow from Foxbridge over a three-year period (Chapter 5), and are more than double the proposed river target of 0.1 mg L^{-1} required to achieve good ecological status under the Water Framework Directive (Duncan et al., 2006).

However, a separate smaller study found that DRP concentrations in rapid runoff generated from these soils under a much greater intensity of simulated rainfall increased linearly with increasing OP with no indication of a change-point (Fig. 6.9). Since runoff would have remained in the collecting tank for the duration of the storm event, DRP concentrations would have equilibrated with transported SS rather than reflect initial soil P status. The SS collected from plots with low OP had greater EPC_0 concentrations than the soil and therefore would maintain larger DRP concentrations in solution. This effect would be absent at higher soil OP concentrations where differences between the EPC_0 of SS and soil were nil or negative (Fig. 6.8). This equilibration effect would have also been operating in runoff samples collected from the wider Rosemaund catchment (Chapter 5). The importance of soil contact time is also apparent in the difference in DRP concentrations between Period 1 (high runoff volumes) and Periods 2 and 3 (low runoff volumes) for plots with low soil OP. At higher soil OP concentrations, these differences were largely absent. These data suggest that change-point theory may be explained by equilibration of runoff P with the soil during transport, as suggested by Koopmans et al. (2000). As the DESPRAL test involves a sedimentation period of only 4 m 40 s, this is clearly sufficient time for equilibration processes to occur.

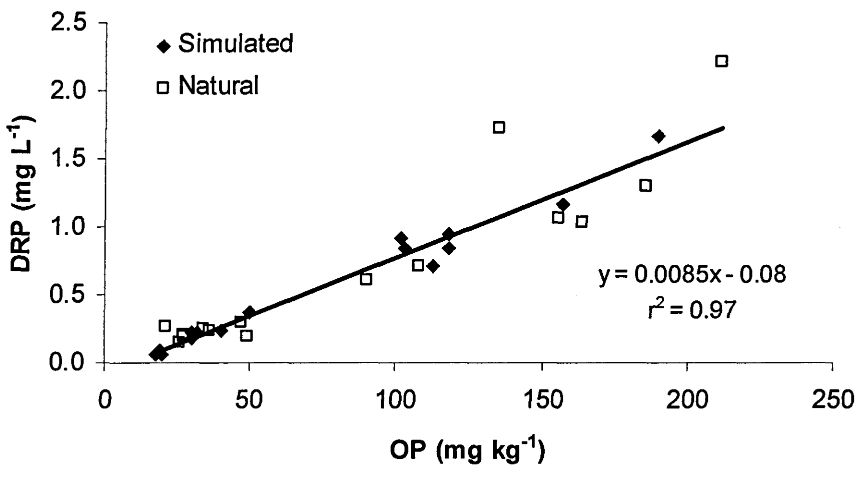


Figure 6.9 Concentrations of dissolved-reactive P (DRP) collected in runoff under either natural rainfall in 2000/01 or simulated rainfall as a function of soil Olsen-P (OP) across the Holbach plots. The regression line shown is for the simulated runoff data. The OP concentration relating to a runoff DRP concentration of 0.1 mg L^{-1} is 22 mg kg^{-1}.

Particulate P concentrations in runoff increased much more slowly than DRP as soil P increased, as was also observed in the fine silt fraction of the soil (Fig. 6.1). The SS in the runoff collected from storm events in the third monitoring period had a spectrum of particle sizes with median values in the range 3-6 μm and mode values below 20 μm. This particle size range is very similar to that measured in SS in UK rivers (Walling et al., 2000), to the median particle size of SS in drainflow from the Foxbridge drain (7 μm, Russell et al., 2001) and is consistent with the preferential transport of fine silt-sized particles in rainsplash detachment and sheet erosion (Hairsine and Rose, 1991; Wan and El Swaify, 1998). This selectivity is most probably due to particles settling out during transport across the 15 m length of plot rather than the rainsplash detachment phase (Foster et al., 1985). This is supported by the results of the DESPRAL test which showed a large difference between in-situ mobilisation of SS (ca. 2.6 mg L^{-1}) and the average flow-weighted concentrations of SS (0.6 mg L^{-1}) in runoff from the field plots, which was identical to the flow-weighted SS concentrations measured in the Foxbridge drain (Chapter 5). Contrary to the commonly observed transport of coarser soil particles with lower P content as SS concentration increase (e.g. Sharpley and Smith, 1990), there was a positive relationship between CER and SS in this study (Fig. 6.5. This suggests that the entrainment of soil particles on these plots was supply limited rather than transport limited as has been observed in other catchment studies, as reviewed by (Walling and Moorehead, 1989).

The narrower range in TP, OP and WEP concentrations in the SS compared to the soil also reflects the preferential transport of very fine particles in runoff. Fine soil particles would be much more enriched with P compared to coarser particles in runoff from plots with low soil P than they would be in runoff from plots that were more P saturated (Maguire et al., 1998). This would have the effect of averaging out the TP content in the SS collected over a range of initial soil P contents. The resulting PER values were therefore greatest in soils with low P contents declining as soil P increased. The average P content of the SS collected across all the storms in the third monitoring period was 0.1%. This value is identical to that measured in the Foxbridge drain within the Rosemaund catchment (Chapter 5) and has also been found in a number of agricultural field and catchment studies (Johnston et al., 1976; Probst, 1985; Pionki and Kunishi, 1992; Uusitalo et al., 2001). Owens and Walling (2002) found river SS-P concentrations of up to 0.2 % in rural catchments but there maybe additional more concentrated P sources other than runoff from arable and pasture land in rural catchments (Edwards and Withers, 2008). The expected increase in SS-P concentration associated with an increase in soil OP from 20 mg kg^{-1} (Index 2, Ministry of Agriculture, Fisheries and Food, 2000) to 60 mg kg^{-1} (P Index 4) at this site would be only 161 mg kg^{-1} or 10%.

The degree of clay and silt enrichment in the SS compared to the soil resulted in a 57% increase in P sorption capacity and variable effects on EPC_0 concentrations depending on the degree of P saturation of the SS, as found by Sharpley (1980). On plots with low soil P concentrations, the selective entrainment of finer P-enriched soil particles increased EPC_0 concentrations in the SS because the SS was more P-saturated (15-34%) than the soil (8-15%). However, on plots with high soil P concentrations, the SS was less P saturated (25-40%) than the soil (55-76%). This may be due to P sorption onto coarser soil particles, or precipitation of P, in heavily-enriched soils (e.g. Pierzynski et al., 1990) and/or some release of P to the more dilute rainwater during the storm event. It is likely that relationship between P_{sat} of the SS and soil will vary from site to site and with experimental conditions (Ryden et al., 1973; Scalenghe et al., 2007). The switch from entrained particles being a P source or a P sink was ca. 110 mg kg^{-1} of Olsen-extractable P. This level of OP is much higher than the level of OP above which DRP concentrations in the runoff apparently increased (50 mg kg^{-1}). The expected increase in % P saturation of the transported SS associated with an increase in soil OP from 20 mg kg^{-1} (P Index 2) to 60 mg kg^{-1} (P Index 4) would be negligible (ca. 2%) and would not have influenced the

EPC_0 concentration. These data suggest that SS in runoff from both low and high P soils have the potential capacity to release P on entering streams depending on the stream dissolved P concentration and the EPC_0 of the SS. The range in EPC_0 concentrations in the SS (0.7-5.5 mg L^{-1}) measured in this study are considerably greater than the flow-weighted concentrations of DRP (0.15-0.45 mg L^{-1}) in the Rosemaund stream (Chapter 5), suggesting P release would occur when SS entered the stream.

This range in EPC_0 concentrations in the SS collected from the Holbach plots during storm events is very much greater than that measured in natural river systems. For example Jarvie et al. (2005) measured concentrations up to only 0.2 mg L^{-1} in a range of moist river bed sediments from catchments on Old Red Sandstone lithology in the Wye river basin under base flow conditions, and similar ranges are reported by House and Denison (2000). Pionke and Kunishi (2002) measured an EPC_0 of 1 mg L^{-1} in storm SS with a P content of 0.1% in a Pennsylvania watershed but the SS was dried. Hence, while it is well known that river bed sediments undergo various transformations that alter their P sorption characteristics (House et al., 1995), these large differences in EPC_0 could relate to (a) release of P as SS enters the watercourse and becomes bottom sediment, (b) the effects of drying and oxidation of the samples and/or (c) differences in the solution:soil ratio between the laboratory and the river (Ryden et al., 1973). Drying of anoxic bed sediments can both increase and decrease P sorption depending on the relative transformations and ageing of iron (Fe^{2+}) and Fe^{3+} compounds (Twinch, 1987; Baldwin et al., 2002). To provide enough sediment to conduct P adsorption isotherms, runoff from consecutive storms was allowed to accumulate in the collecting dustbins over quite a long period (ca. 10 weeks). However, the SS from storms 19 and 21 were not stored but still showed similar enrichment in WEP (a surrogate for EPC_O), and so the influence of reducing conditions was not considered to have had a major influence. Clearly further work is required to determine whether the effects observed in this study are reproducible in SS kept moist and under a wider range in solution:soil ratios.

6.6 Conclusions

Little information exists on the impact of increasing soil P level on the P content and sorption properties of SS in land runoff despite the predominance of particulates in transporting P in land runoff and the importance of sediment P exchange in regulating DRP concentrations in the water column. In this study, the average particle size and P content of SS collected in overland flow from 15m plots was identical to those measured in the Foxbridge field drain at Rosemaund (Chapter 5), and similar to those of solids suspended in rivers in other agricultural regions. The Rosemaund soil is particularly dispersive and particles transported in surface and sub-surface flow largely originate from the surface soil as a result of rainsplash detachment and subsequent transport in sheet runoff. Under these conditions, enrichment of runoff with clay particles increased as SS concentrations increased, and enrichment of runoff with TP, OP or WEP was negatively related to the soil P content rather than to the amounts of SS transported.

Although entrained particles were always enriched in P, their capacity to desorb P to runoff both increased and decreased compared to the soil from which they were derived. Eroding particles therefore remain a potential source of DRP in the receiving water even on soils of low P fertility. Increasing STP from average to high soil P fertility (e.g. from 20 to 60 mg kg^{-1}) had relatively little influence on this desorption capacity but greatly increased the concentrations of DRP in runoff. This was consistent with dispersion analysis of the soils which suggested that added P was largely associated with the <0.45 μm fraction rather than the >0.45 μm fraction. A switch from PP dominance to TDP dominance in runoff water occurred at soil OP concentrations of ca. 140 mg kg^{-1}, well above that required for optimum crop production. Concentrations of DRP in runoff were always greater than the current target (ca. 0.1 mg L^{-1}) proposed for limiting eutrophication even at the lowest concentration tested (19 mg kg^{-1}), and it is unlikely that this demanding water quality target can be achieved at this site through nutrient management and soil P depletion.

7

CHARACTERISATION OF PHOSPHORUS SOURCES IN THREE RURAL CATCHMENTS: WYE

7.1 Introduction

Phosphorus (P) in catchment runoff is not directly toxic to aquatic biota but nevertheless causes a range of environmental, social and economic problems at the regional level that require holistic, catchment-based control strategies that address the sources and not just the symptoms. Nutrient criteria with regulatory applications have been established to define what constitutes water quality impairment (U.S. Environmental Protection Agency, 2002; Duncan et al., 2006) and cost-effective targeting of measures to reduce concentrations and loads of P in catchments is required to achieve these criteria. Within Europe, there is a statutory requirement for programmes of measures to be implemented in catchments by 2012 so there is some urgency. Correct apportionment of P sources, knowledge of their distribution and understanding of their mode of delivery to the watercourse is therefore an essential part of catchment management and targeting mitigation options where they will be most effective. Targeting all sub-catchment areas equally has been shown to be neither cost-effective (Schleich et al., 1996), nor likely to reduce pollutant discharge (Jokela et al., 2004; Granlund et al., 2005). For example, Pionke et al. (1997) found that 90% of the diffuse P export from a Pennsylvania catchment was derived from only 10% of the catchment area.

Sources of P entering surface waters have traditionally been grouped into point and diffuse (or non-point), (Novotny and Olem, 1994). This is a convenient grouping that allows diffuse sources to be quantified as the difference between a measured total P load at a catchment outlet minus the sum of the larger point sources. The latter are usually flowing continuously from single points, require some form of consent, or permit, and are therefore routinely monitored. Some definitions of point sources also include runoff from farmyards or animal feedlots (e.g. Carpenter et al., 1998), but these are often not monitored with the same precision as municipal and industrial discharges. Diffuse sources have a more rural origin and originate from a number of different areas within the upstream catchment and are more episodic in nature and therefore temporally and spatially more variable. Carpenter et al. (1998) included agricultural runoff from cultivated and pasture land, urban runoff, discharges from small sewage treatment works (STW), leakages from septic tanks and atmospheric deposition over a water surface within the diffuse group. In reality, the large variety of different P sources entering surface waters have different hydrological and compositional characteristics which often makes their simple grouping difficult (Edwards and Withers, 2008).

As a consequence, diffuse sources of P derived from anthropogenic activities are invariably lumped under 'agriculture' in source apportionment studies (Johnes, 1996; European Environment Agency, 2005; White and Hammond, 2007). Much research has increased our understanding of the amounts, forms, timing and processes of P transfer in runoff from agricultural land and the implications for control strategies (Withers et al., 2000; Kronvang et al., 2005). In agricultural areas, rates of P transfer from land have increased compared to those from more pristine environments as a direct result of the increased amount and accessibility of source material at the land surface. As outlined in Chapter 2, this extra source material reflects the greater quantities of P applied to agricultural land and the accumulation of surplus P in the soil. Additional, and more indirect, effects of farming on P transfer are associated with changes in crop and livestock production methods that have accelerated runoff and erosion rates (Evans, 2006), increased the water-solubility of P in livestock manures (Kleinman et al., 2005) and increased hydrological connectivity between the field and the waterbody (Sims et al.,1998).

Until recently, less attention has been paid to sources of P other than farmed land in agricultural catchments, such as farmyard and road/track runoff and septic tank discharges which are rarely monitored but contribute P under both low and high flow conditions more continuously during the year (Hively et al., 2005; Armscheidt et al., 2007; Edwards et al., 2007). These 'other' sources have hydrological and chemical properties which are intermediate between point and diffuse and are potentially more ecologically damaging than runoff from farmed land (Edwards and Withers, 2008). For example, using high frequency sampling, Jordan et al. (2005a) showed that whilst storm runoff from grassland was the main source of large episodic P exports in a rural catchment in Ireland, ambient P concentrations in the stream between events were being sustained by much smaller chronic inputs of P which were storm independent. Whilst the effects of major point source discharges on stream P dynamics has been well demonstrated in large catchments (Demars and Harper, 2002; Jarvie et al., 2003), there is less information on the abundance and importance of 'intermediate' sources of P in smaller rural catchments. Just targeting cultivated and pasture land when these other 'intermediate' sources may be ecologically more relevant will not be pragmatic, or achieve the desired water quality goals, and may cause unnecessary alienation, cost and social upheaval.

The previous two chapters investigated the influence of farming practices on stream P concentrations in streams and field runoff within the Rosemaund catchment with highly dispersive soils under mixed farming. In this chapter, the range in P concentrations measured at Rosemaund are placed into context with those associated with other 'intermediate' sources that contribute P to rivers in three slightly larger (7-10 km^2) rural catchments in Herefordshire on similar lithology. These headwater catchments are dominated by agricultural land use and have no major (>10,000 per capita population) municipal or industrial discharges but nevertheless have some urbanisation associated with rural (village) communities. It was hypothesised that this urbanisation might extend the range of sources contributing P to the headwater streams during storm events. To provide an indication of potential ecological response, storm runoff and stream samples were additionally analysed for dissolved nitrogen (N) forms and boron (B), the latter being used as a possible marker of domestic wastewater inputs (Dyer and Caprara, 1997). Some fertilisers also contain B but these were not used in the catchments.

7.2 Wye catchments

Streams and a range of storm runoff sites in the three catchments (Whitchurch, Dinedor and Kivernoll) were monitored over a two-year period (2005/06 and 2006/07), Fig. 7.1. All three catchments drain into first-order tributaries of the Wye river and are located on Devonian Old Red Sandstone lithology, but varied in the type and intensity of farming system (Table 7.1) One catchment (Kivernoll) also contained a village STW serving ca. 400 people. In each catchment, co-operative farmers were interviewed to collect information on land use and fertiliser and manure P inputs. Animal numbers were not high and differences in farming intensity between the catchments was predominantly in the proportion of cultivated land, the extent of field underdrainage and in P inputs and average soil P fertility.

Table 7.1 Selected characteristics of the three Wye catchments.

	Whitchurch	Dinedor	Kivernoll
Area (km^2)	6.5	8.7	9.9
Major soil associations[1]	Eardiston	Bromyard Eardiston	Bromyard Eardiston
Farming intensity	Low	Medium	High
Land use			
Arable[2]	23	53	68
Grassland[3]	60	23	11
Woodland	9	19	13
Other[4]	8	5	7
Animal numbers[5]			
Dairy/beef/calves	1.2	0.9	0.3
Sheep/lambs	3.8	2.5	1.1
Pigs	0.13	0.11	0.01
Poultry	62	127	170
Other[6]	0.09	0.03	0.01
No. of farms	13	10	9
Fertiliser/manure P (kg ha^{-1} yr^{-1})	<5-15	11-75	26-92
Average soil P Index[7]	1	1	3
Drain abundance	Low	Medium	High

[1]Ragg et al. (1984). [2]Includes ley-arable rotations. [3]Includes set-aside and in Kivernoll 4% orchard. [4]Urban areas and farmyards. [5]Numbers per ha of farmed land based on MAGPIE census data (Lord and Anthony, 2000). [6]Horses, goats and deer. [7]Ministry of Agriculture, Fisheries and Food (2000).

The Whitchurch catchment (6.5 km^2) has silt loam soils (Eardiston Association), steeply sloping riparian pasture grazed by cattle and sheep in the centre of the catchment and ley-arable crops receiving farmyard manure (FYM) on perimeter plateau land. Very little P fertiliser is used and very few field drains (Table 7.1). There are no major point source inputs but there are a large number of farmyards and the Whitchurch stream receives a runoff from steeply sloping roads. The Dinedor catchment (8.7 km^2) includes a mixture of beef and sheep farming on permanent and ley grassland in the west of the catchment with arable land growing cereals, oilseed rape, beans and potatoes in the east of the catchment. One farm in the catchment has pigs.

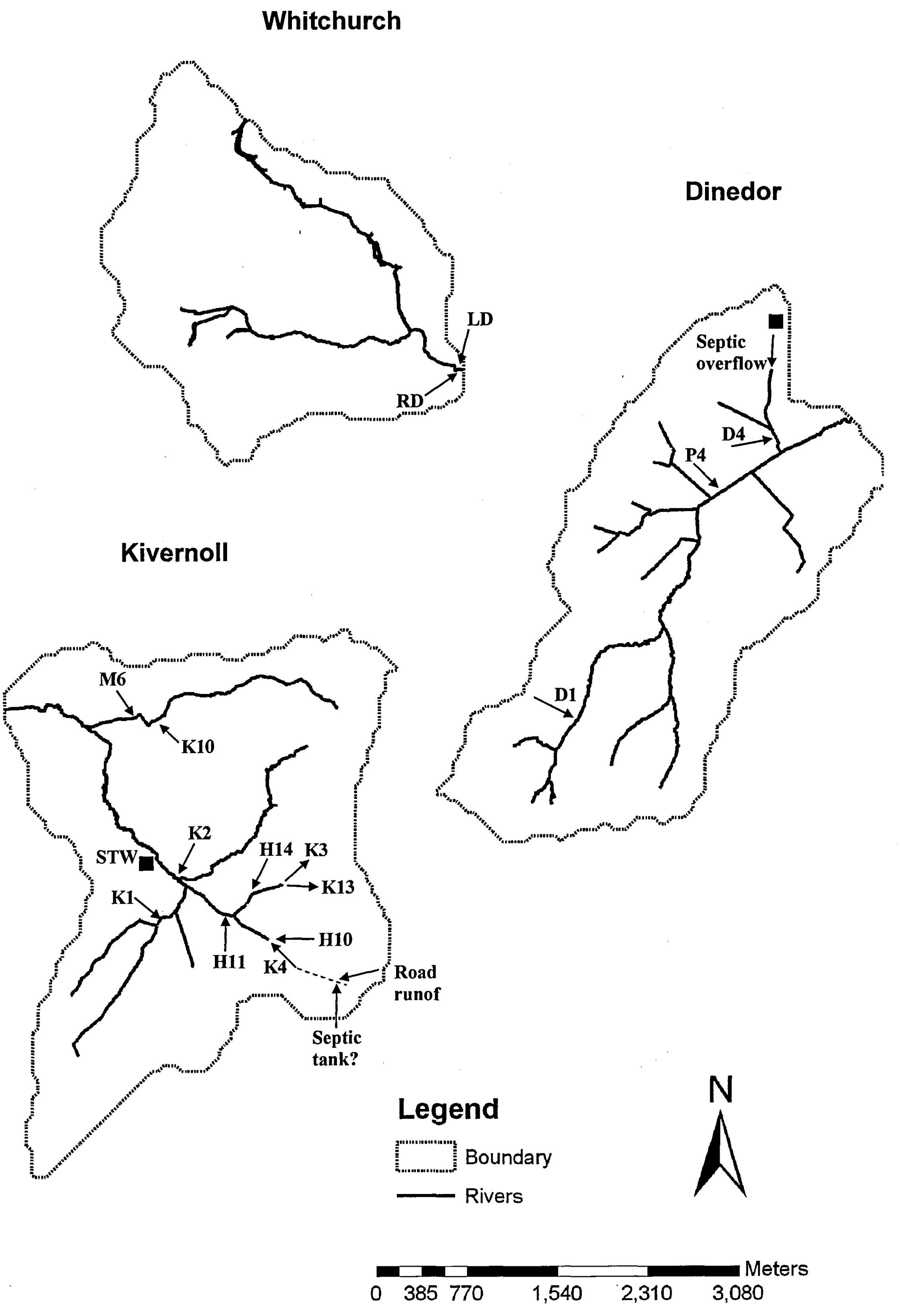


Figure 7.1 Location and distribution of storm runoff sampling points and other discharges within the three Wye catchments. The streams were monitored at their catchment outlets. Details of the sampling codes are given in Table 7.2.

Fertiliser and manure use is greater in the Dinedor catchment than in the Whitchurch catchment but very variable resulting in a greater range of Olsen-P (OP) concentrations but still rather low average soil fertility (P Index 1), (Table 7.1). The heavier-textured, silty clay loam soils (Bromyard Association) that predominate in the lower lying parts of catchment are underdrained. The overflow from a village hall septic tank was thought to enter a field ditch in the lower part of the catchment (Fig. 7.1), but otherwise there are no major wastewater discharges. Farmed land within the Kivernoll catchment (9.9 km^2) is almost totally under arable cultivation with intensive winter cereal, oilseed rape, sugar beet and potato production regularly receiving poultry manure and P fertiliser. Average soil P fertility is much greater than in the Whitchurch or Dinedor catchments and field drains are abundant (Table 7.1).

Soil samples (0-10 cm) were taken from 19 fields representative of soil type and soil P fertility to characterise their chemical characteristics, soil P release properties and potential mobilisation of SS and P during rainfall using the DESPRAL test (Withers et al., 2007). Instantaneous samples of streamwater were taken manually at each catchment outlet on a weekly basis for determination of suspended solids (SS) and a range of nutrients and trace elements of which only the N forms (ammonium-N (NH_4-N); nitrate-N (NO_3-N) and total dissolved N (TDN)), P forms (soluble reactive P (DRP); total dissolved P (TDP) and total P (TP)) and B are reported here. Additional stream samples were taken using an automated sampler at regular intervals during 8-10 storm events over the two-year monitoring period but these samples were analysed for only TP and SS. Storm runoff water representative of surface and sub-surface runoff from farmed land and different 'intermediate' sources entering the streams was collected from various locations within each catchment when the opportunity arose and similarly analysed. The locations of the storm runoff sites in each catchment are shown in Fig. 7.1.

Sites were selected on the basis of visual evidence of regular runoff and sampled once during each storm event. The sites sampled were field surface runoff (1 site), field drain outfall (8 sites), minor roads (4 sites), farmyards (1 site) and a field ditch receiving septic tank discharge (1 site). Additional samples were also collected from two farmyard sources (yard and sheep shed runoff) at Rosemaund. A total of 118 runoff samples were collected over the two years with additional data on flow rates at some sites in the second year, where this was feasible. A summary of the individual locations sampled and their main features is given in Table 7.2.

Table 7.2 Numbers and types of samples of each runoff type collected in each catchment.

Catchment	Runoff type	Location code	No. of samples	Main features
Whitchurch	Road	LD	4	Left road drain entering stream
		RD	8	Right road drain entering stream
Dinedor	Field surface	D1	13	Overland flow across grass field
	Field drain	P4	7	Set-aside field, P index 1[1]
	Filed ditch	D4	13	Contaminated by septic tank overflow
Kivernoll	Field drain	K1	4	Arable field
	Field drain	K10	3	Arable field receiving overflow from lake
	Field drain	H14	10	Arable field next to farm, P index 3[1]
	Field drain	H10	2	Arable field, P index 4
	Field drain	H11	3	Arable field, P index 4
	Field drain	M6	8	Arable field, P index 3
	Field drain	K13	2	Drain from hillside spring
	Farmyard	K3	13	Yard runoff from a poultry farm
	Road	K2	7	Highway runoff entering stream
	Road	K4	10	Highway runoff mixed with ditch runoff
Rosemaund	Farmyard	R1	5	Runoff from arable farmyard
	Farmyard	R2	6	Runoff from sheep shed

[1]The P Index system adopted by Ministry of Agriculture, Fisheries and Food (2000). Index 1, 10-15; Index 3, 26-45; Index 4, 46-70 mg OP L^{-1}.

In a detailed study over the 2006/07 winter, runoff samples from three separate tile drain outfalls were taken manually every 1-2 hr during four or five storm events. The tile drains represented flow from fields with variable soil P status but similar soil type; two of the fields were in the Kivernoll catchment (H14 and M6) and one field was in the Dinedor catchment (P4). Manual samples were taken during storm events on a similar basis from two road drains (right drain, RD and left drain, LD) discharging directly into the Whitchurch stream. Comprehensive details of the three catchments, the soil types, land use history, P inputs and methodologies used to measure stream P concentrations and storm runoff concentrations were provided in Chapter 3.

7.3 Soil analysis

Soil analysis results from fields representative of soil type and P inputs are given in Table 7.3. With the exception of field GT2 which was quite acid, soil pH was close to the optimum for arable crops (pH 6.5) and grassland (pH 6.0), (Ministry of Agriculture, Fisheries and Food, 2000). Organic matter (OM) concentrations ranged up to 76 g kg^{-1} although the majority of fields contained < 30 g kg^{-1}.

Table 7.3 Selected physical and chemical characteristics of the catchment soils and potential mobilisation of suspended solids (SS) and P forms determined by the DESPRAL test.

Catchment	Field	Land use	General characteristics					Soil P characteristics				Mobilised by DESPRAL test				
			pH	OM	TCa	TFe	TAl	TP	OP	WEP	DPS	SS	DP	PP	TP	SS-P
Whitchurch	GT2	Grass	5.3	29	0.09	21.3	16.8	328	9	2.9	10.9	200	113	47	160	235
	H2	Grass	6.8	40	0.25	22.8	19.5	601	18	8.4	17.7	175	50	270	320	1543
	H1	Grass	7.1	24	0.29	24.3	19.7	660	32	15.4	23.4	617	213	1554	1767	2519
Dinedor	P4	Grass	6.0	27	0.25	28.0	25.6	441	13	2.4	12.0	933	70	550	620	589
	P6	Grass	5.7	25	n.d.	n.d.	n.d.	458	12	1.6	n.d.	810	44	499	543	616
	R3	Arable	6.9	24	0.17	27.4	23.7	387	13	4.1	10.1	653	91	269	360	412
	L9	Arable	6.5	24	n.d.	n.d.	n.d.	430	16	2.9	n.d.	865	132	295	427	341
	L12	Arable	6.8	24	0.28	32.2	28.1	446	23	3.5	11.5	892	79	581	660	651
	L31	Grass	6.5	76	n.d.	n.d.	n.d.	1030	35	6.1	n.d.	303	99	361	460	1191
	L16	Arable	6.4	27	n.d.	n.d.	n.d.	413	8	2.0	n.d.	988	95	558	653	565
	L15	Arable	5.9	27	n.d.	n.d.	n.d.	411	21	1.7	n.d.	1485	76	617	693	415
Kivernoll	D3	Arable	7.1	25	0.98	29.2	28.5	692	20	2.6	19.0	758	152	661	813	872
	M4	Arable	6.3	46	0.26	29.4	24.8	496	12	2.3	12.5	225	47	126	173	560
	M6	Arable	6.8	30	0.29	33.6	30.6	586	34	10.7	21.4	1333	185	685	870	514
	H8	Arable	6.8	25	n.d.	n.d.	n.d.	678	40	9.7	n.d.	1378	148	2852	3000	2070
	D5	Arable	7.1	29	n.d.	n.d.	n.d.	582	29	5.9	n.d.	1150	135	845	980	735
	H10	Arable	6.9	25	0.35	23.4	23.9	802	52	14.9	23.5	1105	278	2022	2300	1830
	D4	Arable	7.1	24	n.d.	n.d.	n.d.	634	44	15.4	n.d.	1057	173	994	1167	940
	H14	Arable	6.7	33	0.28	29.4	26.2	576	31	6.4	16.9	543	128	265	393	488

n.d. – not determined.

Soil total Fe and Al concentrations were very similar across all three catchments (20-30 g kg^{-1}), soil TP, OP and WEP ranged from 328-1035, 9-52 and 1.6-15.4 mg kg^{-1}, respectively, while the degree of P saturation (DPS) ranged from 10-24%. DPS values were greater in the Whitchurch catchment (coarser-textured soils) and in the Kivernoll catchment (greater soil P fertility) than in the Dinedor catchment and strongly related to WEP.

The amounts of SS mobilised by the DESPRAL test varied from 175-1485 mg L^{-1}, but a large proportion of the cultivated soils had values above 500 mg L^{-1} (Fig. 7.2). Grassland fields with greater levels of OM (> 30 g kg^{-1}) dispersed less SS. Of the cultivated fields, the more P fertile soils in the Kivernoll catchment tended to show larger SS concentrations. Test concentrations of TDP tended to increase as OP increased and were significantly ($P < 0.01$) larger in Kivernoll fields (156 µg L^{-1}) than in Dinedor fields (86 µg L^{-1}). Hence, the DESPRAL test predicted that very variable concentrations of TP (160 – 3000 mg L^{-1}) would be mobilised in runoff water.

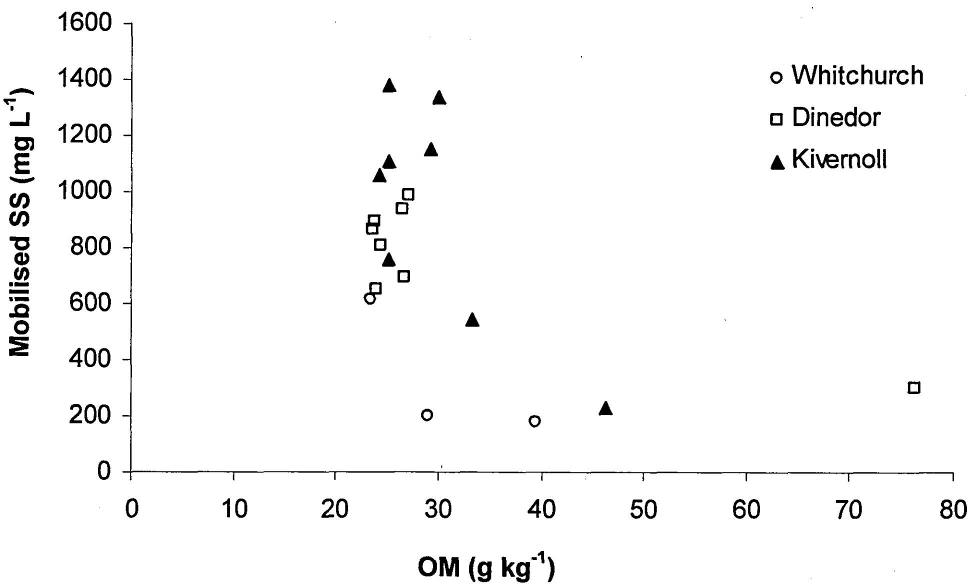


Figure 7.2 Variation in the mobilisation of SS during rainfall as predicted by the DESPRAL test as a function of soil organic matter (OM) in the three catchments.

7.4 Stream concentrations

Average and range concentrations of P forms in weekly stream samples were in the order Whitchurch < Dinedor < Kivernoll, with particularly large DRP concentrations measured at Kivernoll due to the influence of the village STW (Table 7.4).

Table 7.4 Distribution statistics for weekly flow, concentrations of P forms, suspended solids (SS) and their P content, N forms, TDN:TDP ratio and boron (B) over a two-year period. Standard error values are given in parenthesis.

Catchment	Statistic	Flow ($L\ sec^{-1}$)	DRP	DUP ($\mu g\ L^{-1}$)	PP	TDP (%)	SS[1] ($mg\ L^{-1}$)	SS-P ($g\ kg^{-1}$)	NH_4-N	NO_3-N ($mg\ L^{-1}$)	TDN:TDP ratio	B ($\mu g\ L^{-1}$)
Whitchurch	Range	12-95	4-303	0-87	0-247	18-100	2-525	0.2-20.6	0.002-0.46	6.0-9.4	25-579	21-34
(n = 103)	Mean (s.e.)	43 (1.9)	29 (3.3)	13 (1.5)	27 (3.4)	65 (1.7)	19 (5.3)	2.0 (0.22)	0.03 (0.01)	7.9 (0.05)	259 (9.2)	25 (0.2)
	Median	42	25	8	18	64	10	1.6	0.02	7.9	249	25
Dinedor	Range	1-578	4-431	0-184	2-973	32-97	2-323	0.1-18.9	0.002-0.81	8.7-27.1	29-353	18-41
(n = 102)	Mean (s.e.)	84 (8.9)	95 (7.6)	26 (3.2)	61 (12)	76 (1.4)	25 (4.8)	2.5 (0.22)	0.06 (0.01)	12.3 (0.24)	153 (6.3)	23 (0.5)
	Median	73	71	14	22	79	10	2.2	0.03	11.8	150	21
Kivernoll	Range	1-775	50-1264	0-223	0-2220	13-100	2-306	0.7-23.4	0.002-0.92	6.1-18.9	9-134	24-126
(n = 103)	Mean (s.e.)	58 (12.7)	382 (28)	27 (3.3)	94 (25)	86 (1.5)	22 (4.8)	4.8 (0.39)	0.06 (0.01)	11.7 (0.19)	50 (3.1)	33 (1.1)
	Median	33	246	16	29	91	6	4.0	0.02	11.6	45	31

[1]Storm event sampling showed a range of 8-2183, 8-7814 and 4-1983 mg SS L^{-1} at Whitchurch, Dinedor and Kivernoll, respectively. Corresponding range concentrations of TP were 10-2880, 56-1830 and 97-1870 $\mu g\ L^{-1}$, respectively.

Mean TP concentrations increased from 69 $\mu g\ L^{-1}$ (range up to 0.5 mg L^{-1}) at Whitchurch, to 182 $\mu g\ L^{-1}$ (range up to 1.5 mg L^{-1}) at Dinedor and to 503 $\mu g\ L^{-1}$ (range up to 2.6 mg L^{-1}) at Kivernoll. Average TDP concentrations represented between 65% of TP at Whitchurch and 86% of TP at Kivernoll. The proportion of TDP in dissolved-unreactive (DUP) form was greatest at Whitchurch (20% of TDP) and lowest at Kivernoll (5%). In contrast to TDP, site differences in PP concentrations were smaller due to similarly low average SS concentrations (19-25 mg L^{-1}). Hence any difference in PP concentrations was due to variation in the P content of the SS, which was lowest at Whitchurch (1.6 g kg^{-1}) and particularly large at Kivernoll (4.8 g kg^{-1}). Large seasonal variation in DRP concentrations was apparent at Kivernoll (much larger in summer and autumn) but not in the other two catchments. Concentrations of DUP were always lower in winter and concentrations of PP were greatest in spring at Whitchurch and Dinedor and in autumn at Kivernoll.

Weekly samples will not capture the larger storm events and will therefore underestimate the true range in concentrations mobilised in runoff and streamflow. Hence the storm event data (SS and TP only) collected for the three catchment streams showed a much greater range in SS concentrations, especially at Dinedor where flow rates were greater; the storm event data are provided as a footnote to Table 7.4. Whilst weekly samples showed SS concentrations did not reach above 0.5 g L^{-1}, the storm sampling showed concentrations up to 2 g L^{-1} at Whitchurch and Kivernoll and up to 8 g L^{-1} at Dinedor. Increased mobilisation of SS greatly increased the range in TP concentrations measured at Whitchurch, but only slightly increased the range at Dinedor, whilst the range in TP concentrations at Kivernoll was actually greater in the weekly samples than in the storm event samples. This pattern suggests that diffuse P sources are supplemented by intermediate or point sources in the Dinedor and Kivernoll catchments.

In all streams, NO_3-N was the dominant form of N with larger average concentrations measured in the Dinedor and Kivernoll catchments (ca. 12 mg N L^{-1}) than in the Whitchurch catchment (8 mg N L^{-1}). The average ratio of TDN:TDP ranged from a value of 259 in the Whitchurch catchment to 50 in the Kivernoll catchment (Table 7.4). Average and range concentrations of B were very similar at Whitchurch and Dinedor but increased at Kivernoll due to detergent in the wastewater effluent discharged from the STW.

7.5 Storm runoff concentrations

7.5.1 Field surface runoff

Surface runoff was collected from one site in the upper part of the Dinedor catchment. This runoff originated in an upslope arable field, became mixed with road runoff during transit and eventually flowed down a steeply sloping riparian grassland field before entering the stream. Concentrations of TP varied from 360-1950 µg L^{-1} (mean 968, s.e. 135, median 900 µg L^{-1}) with an average 66% in PP form (Table 7.5). Runoff TDP concentrations ranged from 28-567 µg L^{-1} with a large average value (mean 298, s.e. 34, median 325 µg L^{-1}) relative to suggested riverine targets of 100 µg L^{-1} (Duncan et al., 2006). The majority of the TDP was in dissolved-reactive form (70% of TDP). Runoff PP concentrations were strongly dependent on SS concentrations which varied from 160-705 mg L^{-1}, with an average calculated SS-P content of 1462 mg kg^{-1} (Table 7.5). This is a very similar value to that obtained in the Whitchurch stream. The dominant form of N was nitrate (mean 4.4 mg N L^{-1}) and TDN:TDP ratios averaged 105 (s.e. 25, median 65). Boron concentrations averaged 33 µg L^{-1}, but with range values up to 59 µg L^{-1}, suggesting some detergent contamination on some sampling occasions.

7.5.2 Runoff through field drains

Eight field drains discharging directly into the stream were monitored but at five of the seven Kivernoll sites the sample numbers were very low (2-4 samples). Concentrations of TP varied from 19-3600 µg L^{-1} (mean 710, s.e. 139, median 346 µg L^{-1}) with an average 53% in PP form. Runoff TDP concentrations varied from 15-621 µg L^{-1} with mean and median values of 162 (s.e. 20) and 108 µg L^{-1}, respectively, of which ca. 70% was in dissolved-reactive form. Concentrations of SS ranged up to 412 mg L^{-1} and the average SS-P content was 2343 mg kg^{-1}, a value which is greater than was found in surface runoff. Concentrations of TDN were generally large and dominated by NO_3-N with values up to 30 mg N L^{-1} (mean 13.4, median 11.4 mg L^{-1}). Ammonium-N concentrations were heavily skewed by two very large concentrations (6-9 mg N L^{-1}) at one site (K13) but were otherwise low (Table 7.5). The larger TDN concentrations relative to TDP concentrations in drainflow produced a very wide range of TDN:TDP ratios (11-2107), but again the data were heavily skewed with mean and median values of 228 and 90, respectively. Boron concentrations also showed considerable variation with concentrations up to 93 µg L^{-1}.

Table 7.5 Range, mean and median concentrations of P forms, SS and SS-P in storm runoff for each P source group within the Wye catchments.

Source group	Statistic	DRP	DUP	PP	TDP	SS	SS-P	NH_4-N	NO_3-N	TDN:TDP	B
		-----------------	μg L^{-1}	--------------	% of TP	mg L^{-1}	g kg^{-1}				
Field surface	range	7-412	21-155	267-1615	17-54	160-705	1.0-20.1	0.02-0.8	0.5-13.8	27-319	7-59
(n = 13)	mean (s.e.)	207 (30)	91 (11.8)	670 (127)	34 (4.1)	340 (54)	1.5 (0.1)	0.2 (0.06)	4.4 (0.9)	105 (25)	33 (4.2)
	median	217	84	470	28	305	1.5	0.1	4.9	65	35
Field drains	range	4-487	1-282	3-3276	4-91	1-412	0.8-10.0	<0.01-9.2	1-31	11-2107	15-93
(n = 39)	mean (s.e.)	125 (22)	55 (9.6)	530 (131)	47 (4.4)	108 (24)	2.3 (0.4)	0.5 (0.27)	13.4 (1.5)	228 (63)	31 (2.7)
	median	65	35	228	46	68	1.8	0.04	11.4	90	24
Roads	range	4-1330	17-517	40-15425	4-92	15-7859	0.3-5.1	<0.01-0.8	0.1-40	3-170	14-69
(n = 29)	mean (s.e.)	381 (70)	115 (18)	1534 (525)	36 (5.2)	857 (332)	2.2. (0.2)	0.2 (0.04)	7.1 (1.4)	36 (7.5)	34 (2.2)
	median	202	105	841	24	425	1.9	0.1	7.1	25	29
Farmyards	range	61-5680	5-1470	6-12860	10-98	13-1051	1.0-25.2	0.02-3.5	0.5-34	1-136	1-184
(n = 24)	mean (s.e.)	1108 (292)	183 (60)	1419 (567)	55 (5.4)	366 (81)	5.3 (1.6)	0.6 (0.16)	6.5 (1.8)	18 (6.2)	41 (8.2)
	median	535	116	666	50	330	2.5	0.3	2.5	9	35
Septic tank	range	311-1388	2-372	72-2090	46-90	24-204	2.0-5.9	0.06-2.2	4.0-8.2	6-30	37-79
ditch	mean (s.e.)	775 (92)	93 (26.7)	511 (149)	68 (3.2)	89 (18.7)	3.6 (0.4)	0.9 (0.20)	6.5 (0.4)	14 (2.1)	53 (3.2)
(n = 13)	median	772	63	398	67	77	3.2	0.9	6.6	13	49

Despite the large variation, there were some significant ($P<0.05$) differences between these drain sites. Two sites had large mean DRP concentrations (231 and 294 $\mu g\ L^{-1}$) relative to the overall mean, three sites had relatively large PP concentrations (>1.5 g L^{-1}). One of the latter sites which was thought to be a hillside spring was also heavily contaminated with NH_4-N (6-9 mg L^{-1}), although DRP concentrations were low. Low sample numbers at some sites restricted statistical comparison to three drain sites which were sampled more frequently; drains in fields P4 in the Dinedor catchment and fields M6 and H14 in the Kivernoll catchment. The fields varied in their topsoil OP concentration; field P4 (13 mg kg^{-1}), field M6 (34 mg kg^{-1}) and field H14 (38 mg kg^{-1}). Field P4 had been in set-aside and had received no N or P inputs prior to sampling. Fields M6 and H14 were both cropped to wheat and had received poultry manure in August 2006 at 5 t ha^{-1} supplying an estimated 77 kg P ha^{-1}. Field H14, which was directly next to the farm buildings, additionally received dirty water from a poultry house every 6 weeks at a rate of 20,000 L ha^{-1}. No dirty water applications were made during the January 2007 sampling period but the field did receive some on 20 December 2006.

The median, inter-quartile range and 95% confidence intervals for DRP, NO_3-N and B concentrations measured at these three sites are shown in Fig. 7.3. These three nutrients provided the greatest contrast between the sites. Median DRP concentrations were very much greater in the H14 drain, and lower in P4 than in M6. Considerably greater NO_3-N was measured in the two Kivernoll drains, especially M6, than in the Dinedor drain, which had a very low mean value (1 mg L^{-1}). Boron concentrations were also elevated in the H14 field compared to the other two sites. Drain runoff at these three field sites was also monitored more frequently (1-2 hour intervals) during 4-5 individual storm events. The largest concentrations of DRP, PP and TP were always obtained in the H14 drain but differences in DUP and PP concentrations were more variable and generally smaller (Fig. 7.4). The M6 drain tended to have greater DRP concentrations than P4 drain during the more intensive storm events, but lower DUP and PP concentrations, with the result that TDP and TP concentrations in the drains from these two fields were quite similar. Overall the largest differences in P concentrations between these three sites were in DRP (Fig. 7.4).

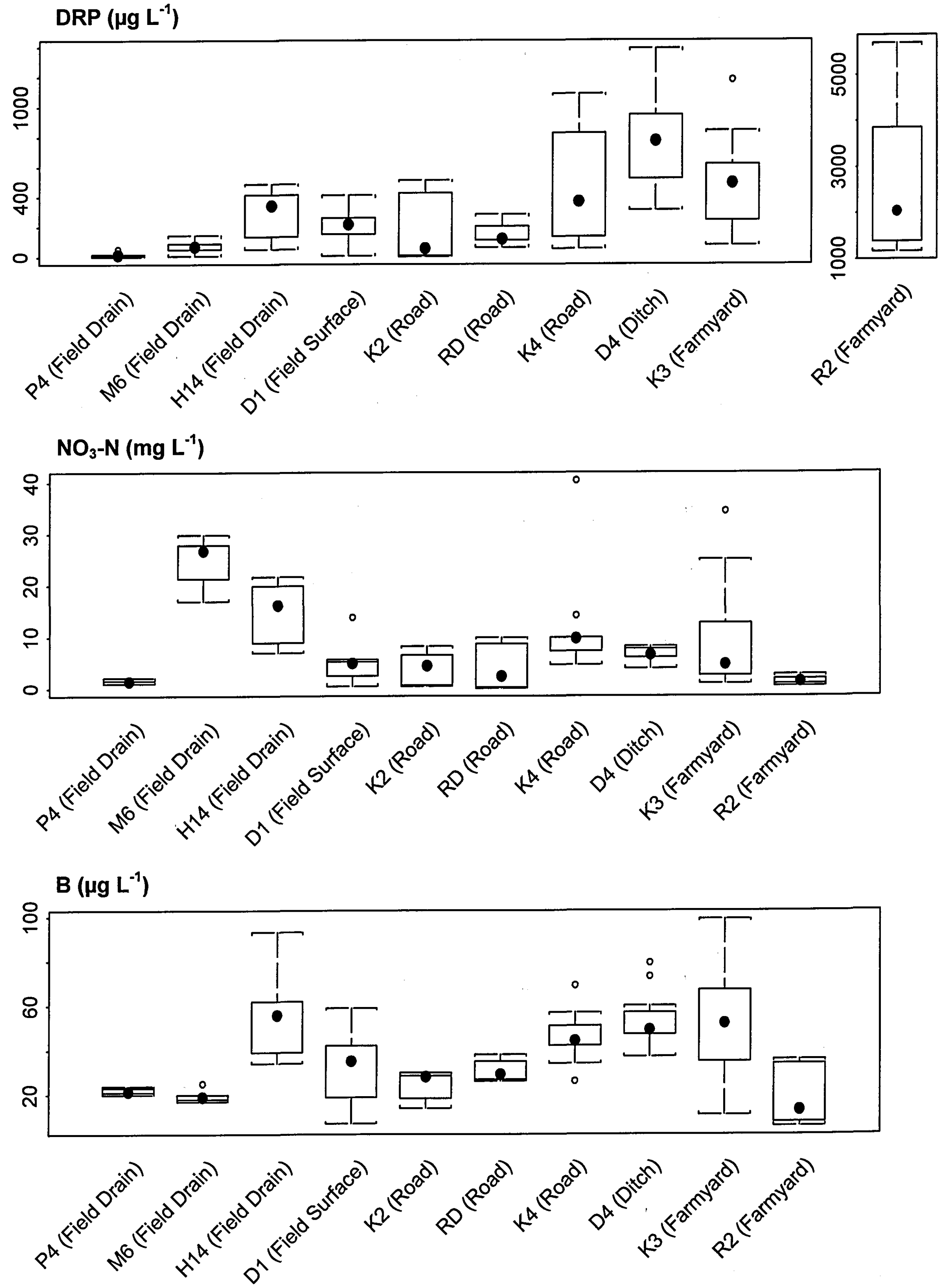


Figure 7.3 Box and whisker plots of storm runoff data from selected sites (>5 samples) showing the median, lower and upper quartiles, confidence intervals and outliers.

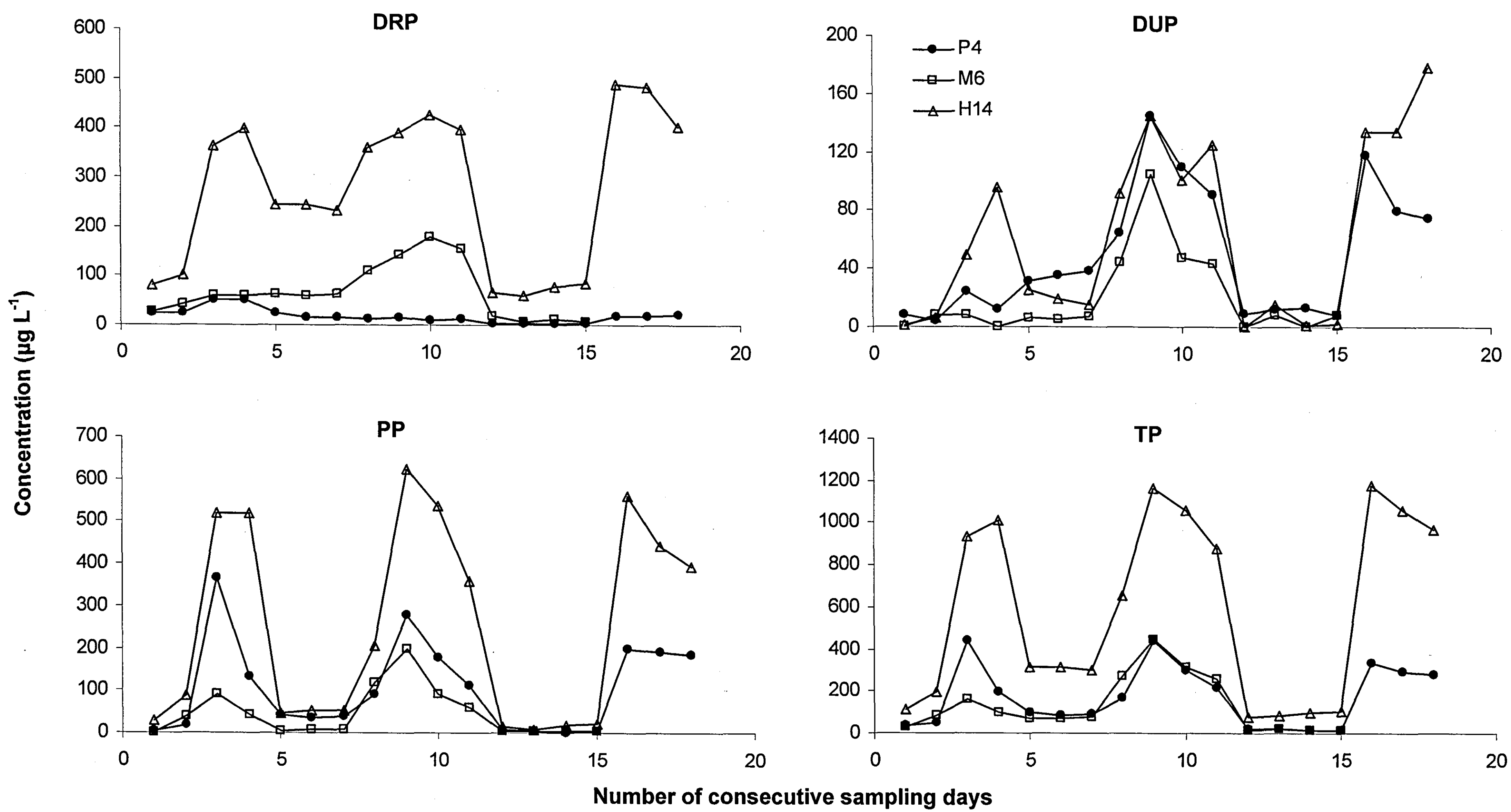


Figure 7.4 Changes in the concentration of dissolved reactive P (DRP), dissolved unreactive P (DUP), particulate P (PP) and total P (TP) in storm flow through three field drains during consecutive sampling days. P4 is in the Dinedor catchment and M6 and H14 are in the Kivernoll catchment.

Although the P content of the SS in drainflow was slightly lower in P4 than in the other two drains, this was not statistically significant (P >0.05). Hence, despite differences in soil OP and TP concentrations between the three fields (Table 7.3), all PP/SS values from all drains during the storm sampling campaign fell on a single gradient representing an average SS-P content of 1.9 g kg^{-1} (Fig. 7.5). Differences in PP concentrations between the three drains were therefore due to small differences in the amounts of SS transported.

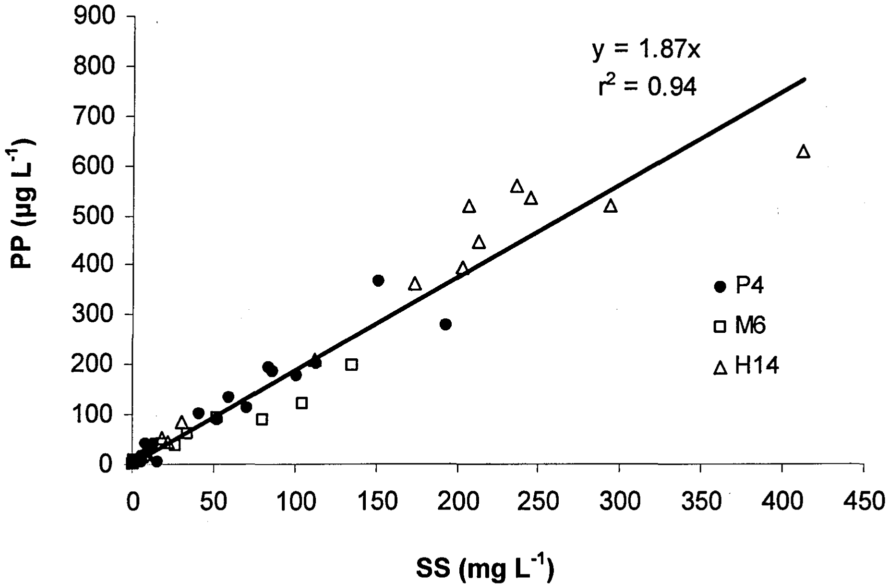


Figure 7.5 The relationship between suspended solids (SS) and particulate P (PP) in runoff through drains from three contrasting fields in the Dinedor (P4) and Kivernoll (M6 and H14) catchments.

7.5.3 Road runoff

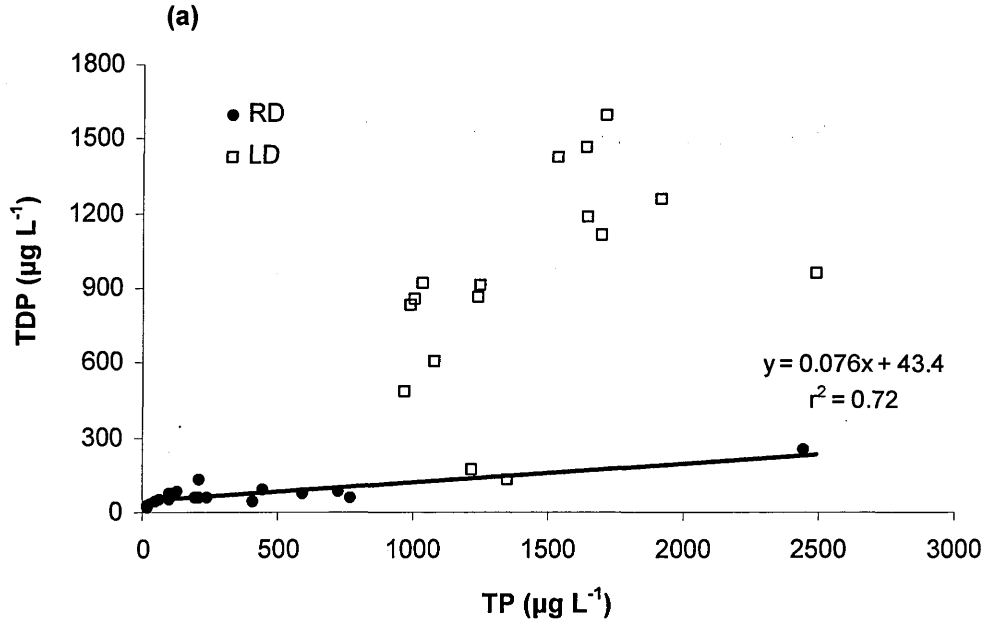
Road runoff was sampled at 4 locations: two sites were large road drains entering either side of the Whitchurch stream by the catchment monitoring station (Plate 7.1). The left drain (LD) received yard and roof runoff from a nearby house but otherwise was no different to the right drain (RD). The other two sites were in the Kivernoll catchment; one site was highway runoff entering the stream (K2) whilst at the other site (K4), highway runoff was combined with field ditch runoff before being discharged down a field drainage pipe into the headwater stream (Fig. 7.1). The field ditch was thought to receive septic tank discharges from nearby domestic dwellings.

This P source group showed a much wider range in TP concentrations (0.1-16 mg L^{-1}, mean 2.0, s.e. 0.5, median 1.4 mg L^{-1}) than field surface and drainflow sources (Table 7.5). This wide range was apparent in all the forms of P and in SS concentrations, with mean values of 381, 115 and 1534 µg L^{-1} for DRP, DUP and PP, respectively and 857 mg L^{-1} for SS. Corresponding median values were 202, 105 and 841 µg P L^{-1} and 425 mg SS L^{-1}. The majority of the P transported was therefore in PP form (average 76%). The range in SS concentrations was particularly wide and this runoff type was sometimes extremely turbid. Nitrate-N and ammonium-N concentrations ranged up to 40 and 0.8 mg N L^{-1}, respectively with average values of 7.1 and 0.2 mg N L^{-1}. The larger P concentrations in road runoff tended to produce much lower TDN:TDP ratios (3-170, mean 36, s.e. 7.5, median 25) than for field sources while B concentrations were elevated up to 69 µg L^{-1} on some occasions at some sites (Table 7.5).

(a) **(b)**

Plate 7.1 Road runoff entering the Whitchurch stream via two large road drains; (a) the right drain and (b) the road passes by a dwelling before entering the left drain.

Site differences within this P source group were also very clear. Of the two Whitchurch road drains, the runoff entering the LD contained significantly greater DRP (mean 0.96 mg L^{-1}) and NH_4-N (mean 0.6 mg N L^{-1}) concentrations than runoff entering the RD (means of 0.19 and 0.16 mg L^{-1}), respectively. Whilst the runoff from RD was dominated by PP (77% of TP), the LD was dominated by TDP (71% of TP). These road drains were monitored more intensively over 1-2 hourly intervals over successive storm events during January in the same way as for the field drains. The dominance of TDP in the LD runoff was very consistent across all storms (Fig. 7.6a). The majority of this increased TDP in the LD runoff was dissolved-reactive (mean 88% of TDP). Concentrations of SS in the LD runoff were also greater, and had a more variable SS-P content, compared to the SS transported in the RD, which had a very constant SS-P content of ca. 0.2% P (Fig. 7.6b). At low SS concentrations, the PP in the LD runoff tended to be greater than that in the RD runoff, whilst at high SS concentrations, PP tended to be lower. The LD runoff contained similar concentrations of B to the RD runoff.



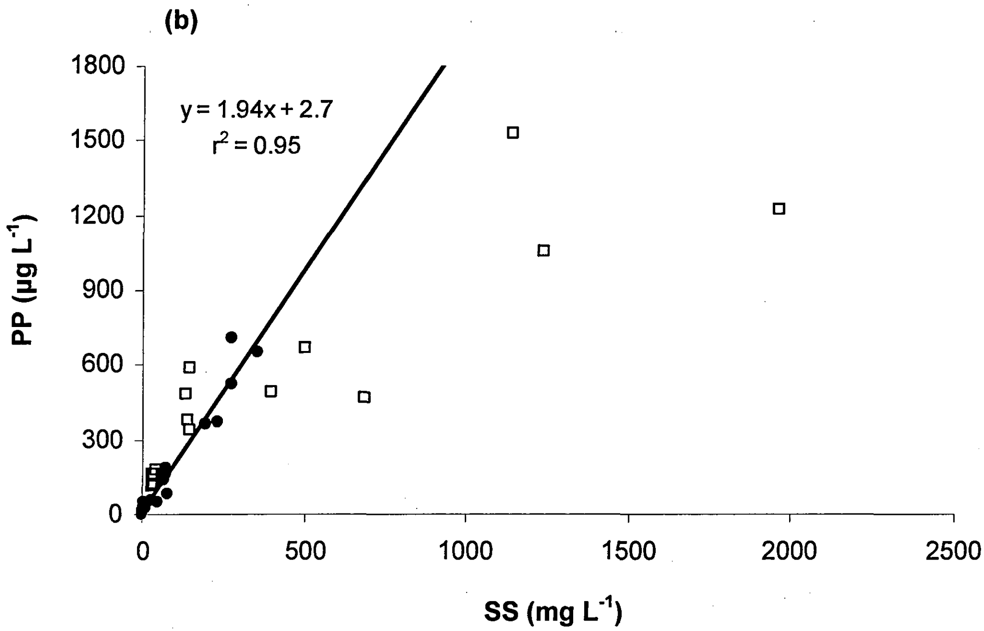


Figure 7.6 Relationships between (a) total dissolved P (TDP) and total P (TP) and (b) particulate P (PP) and suspended solids (SS) in road runoff from two sources entering the Whitchurch stream. The regression line is for samples from the RD drain only.

There was also a difference in runoff chemistry between the two Kivernoll road runoff sites (Fig. 7.3). A larger mean TDP concentration (77% DRP) at the K4 site (605 μg L^{-1}) was associated with a significantly elevated mean B concentration (46 μg L^{-1}) compared to runoff at K2, where mean TDP and B concentrations were 243 (17% DRP) and 30 μg L^{-1}, respectively. There was also a strong linear relationship between SRP and B (r^2 0.7) at K4 suggesting a common wastewater source. Mean NO_3-N (but not NH_4-N) was also much greater at K4 (12.1 mg L^{-1}) than at K2 (4.3 mg L^{-1}). However, mean concentrations of SS, and the P content of the SS, were very similar between these two sites. Indeed, SS-P concentrations in road runoff were very similar to those in drainflow at Kivernoll with a mean value of 2 g kg^{-1} (Fig. 7.7). This suggests the main source of SS on roads is from the surrounding fields. This SS-P concentration is not significantly different to that measured in the RD drain at Whitchurch.

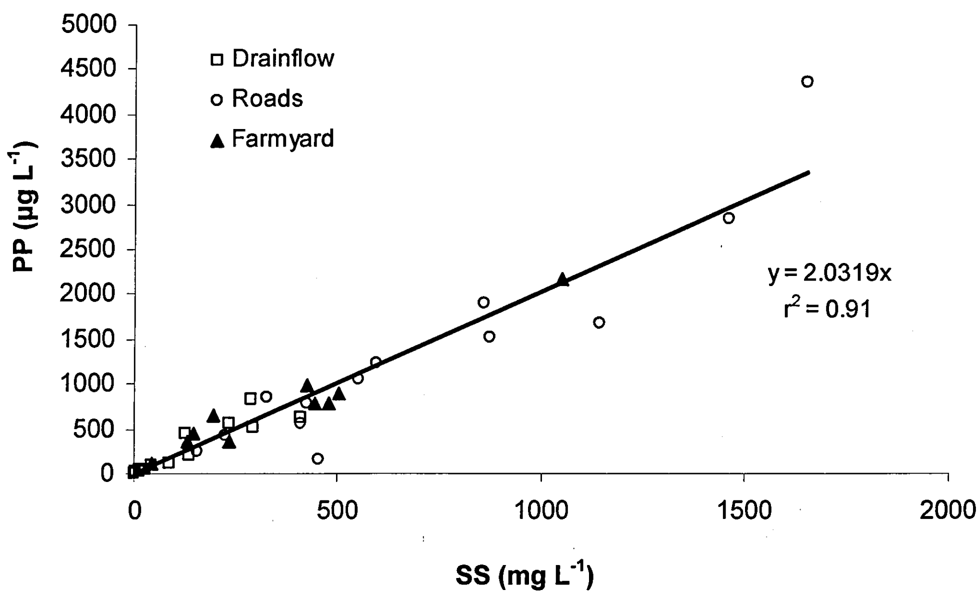


Figure 7.7 The relationship between suspended solids (SS) and particulate P (PP) for various runoff P sources in the Kivernoll catchment. A single regression line is appropriate for all runoff types/sources.

7.5.4 Farmyards

Farmyard runoff was sampled from 3 locations; two sites at Rosemaund and one site in the Kivernoll catchment (Fig. 7.1). One site at Rosemaund was an open yard area with livestock access. The other two sites were piped discharges, one from a sheep shed to a soak-away (Rosemaund) and one from a poultry farmyard to a stream (Kivernoll). Compared to other source groups, these samples had very large mean and range concentrations of dissolved P forms (DRP and DUP), TP and the largest range in PP and SS-P concentrations (Table 7.5). For example, concentrations of TDP varied from 70-5980 µg L^{-1} (mean 1291, s.e. 326, median 677 µg L^{-1}) with an average 80% in dissolved-reactive form. However, DUP concentrations also ranged up to 1470 µg L^{-1} accounting for up to 72% of TDP on some occasions. Concentrations of TP ranged from 76-14,500 µg L^{-1} and average values were the greatest of all the P source groups (mean 2711, s.e. 729, median 1428 µg L^{-1}). Mean and median SS-P concentrations were 5.3 and 2.5 g kg^{-1}, respectively and closer to the values recorded in runoff contaminated by septic tanks than in field or road runoff. Farmyard runoff was also quite turbid but not unusually so (up to 1 g SS L^{-1}). Ammonium-N and nitrate-N concentrations ranged up to 3.5 and 34 mg N L^{-1}, respectively and TDN:TDP ratios were consequently lower than field and road P sources (mean 18, s.e. 6.2, median 9). Boron concentrations were also elevated on some occasions with values up to 184 µg L^{-1}.

Within this source group, the largest mean concentrations of DRP, DUP and PP were recorded in the runoff from the sheep shed (R2), with values of 2.7, 0.5 and 3.8 mg L^{-1}, respectively (Fig. 7.3). Average concentrations of P in the SS in this runoff were over 12 g kg^{-1}. Ammonium-N (mean 1.3 mg L^{-1}) and B (mean 51 mg L^{-1}) were also greater at this site, whilst highest NO_3-N (9.2 mg L^{-1}) was recorded at the Kivernoll site, K3. Mean concentrations of DRP at R1 and K3 were also large (0.9 and 0.5 mg L^{-1}, respectively) relative to other source groups. Mean concentrations of SS were similar (300-500 mg L^{-1}) across the three sites. However, unlike R1 and R2, where SS-P concentrations were very large (6.6 and 12.5 g kg^{-1}, respectively), those at K3 were very similar to the SS-P concentrations calculated for field and road runoff in the Kivernoll catchment (Fig. 7.7). At K3, there was also a strong linear relationship between DRP and B (r^2 0.7) as was found at K4. Relatively large DRP and B concentrations were also measured in the yard runoff at Rosemaund and they were strongly correlated again suggesting a common source.

7.5.5 Ditch receiving septic tank effluent

A field ditch thought to be regularly receiving storm overflow from a village hall septic tank within the Dinedor catchment showed very large concentrations of DRP (but not DUP) with an average and median value of 0.8 mg L^{-1} compared to all other source groups except farmyards (Fig. 7.3). TDP concentrations ranged between 0.3-1.8 mg L^{-1} and was on average 68% of TP, which ranged between 0.4 and 3.9 mg L^{-1} (mean 1379, s.e. 255, median 1232 µg L^{-1}). Concentrations of SS were generally quite low (<204 mg L^{-1}) but the P content of the SS averaged 3.9 g kg^{-1} and contrasted strongly with the SS-P concentration measured in surface runoff and drainflow within this catchment (Fig. 7.8). Whilst NH_4-N concentrations were quite large relative to other source groups (mean 0.9 mg L^{-1}), NO_3-N concentrations were not elevated with a mean value of 6.5 mg N L^{-1} and a range up to 8 mg N L^{-1} (Table 7.5). TDN:TDP ratios were therefore comparatively low (mean 14, s.e. 2.1, median 13). Concentrations of B were well above background with a mean value of 53 µg L^{-1}, but a correlation with SRP concentrations was obtained for only some of the samples suggesting an additional source of SRP other than septic tank effluent at some sampling dates.

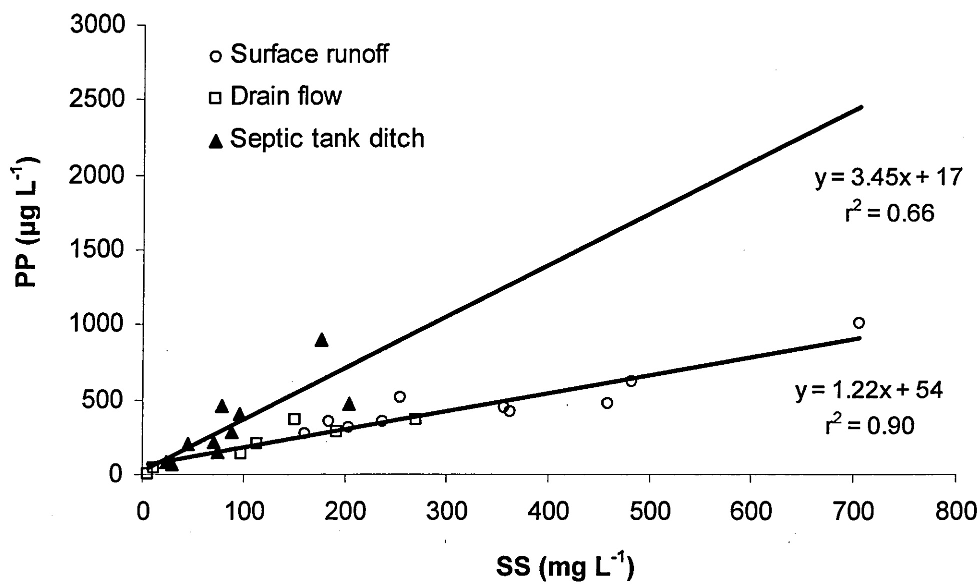


Figure 7.8 The relationship between suspended solids (SS) and particulate P (PP) in storm surface runoff and drainflow compared to a field ditch receiving septic tank effluent in the Dinedor catchment.

7.5.6 Sewage treatment works

Data on total daily flow and monthly concentrations of total reactive P (TRP, dissolved-reactive P determined on an unfiltered sampled) and TP discharging from the STW in the Kivernoll catchment for the period November 2005 to October 2006 were used to calculate an annual load to the Kivernoll stream of 256 kg TP, of which 93% was in dissolved-reactive form. As the STW services 400 people, this is equivalent to a per capita P loading of 0.64 kg. Discharge P concentrations over the year typically varied from 2- 8 mg L^{-1} with an average flow-weighted TP concentration of 5 mg L^{-1}. Highest concentrations were normally obtained during the summer months when flow rates were lower.

7.6 Discussion

These catchments are fairly typical of rural communities in Herefordshire but of a sufficiently small size to suggest that 'agriculture' would be the main source of P entering streams during storm events. The weekly stream sampling showed large differences in P export between the three catchments, particularly the DRP fraction. At Kivernoll the location of the village STW was clearly influencing the concentration of DRP, especially during low flow summer periods. However, it is likely that differences in stream P chemistry were at least partly due to not only differences in agricultural intensification but also in the distribution of other sources termed 'intermediate' by Edwards and Withers (2008). At Dinedor, where there was no STW, yet stream DRP concentrations were significantly greater than at Whitchurch. Stream concentrations upstream of the STW within Kivernoll (data not reported here) also showed considerable enrichment with DRP.

In the Whitchurch catchment, farming intensity was low, there was very little P fertiliser used and overland flow was the dominant pathway of P transfer. There were a large number of farms but these were located on the perimeter of the catchment and were generally not directly connected to the stream. In the Dinedor catchment, P inputs in fertiliser and manure were greater, soil P concentrations were more variable, there was a greater proportion of cultivated land and a larger number of underdrained fields feeding into a denser stream network on heavier-textured soils. In addition there was an overflow from at least one village septic tank. At Kivernoll, large amounts of P in fertiliser and poultry manure were regularly applied to underdrained cultivated fields, and there was evidence of farmyard runoff and septic tank overflows discharging directly into the stream at more than one location. In some instances, there was a multitude of drains from different sources discharging into the same headwater stream (Plate 7.2). Road runoff appeared to be an important source of P in all catchments.

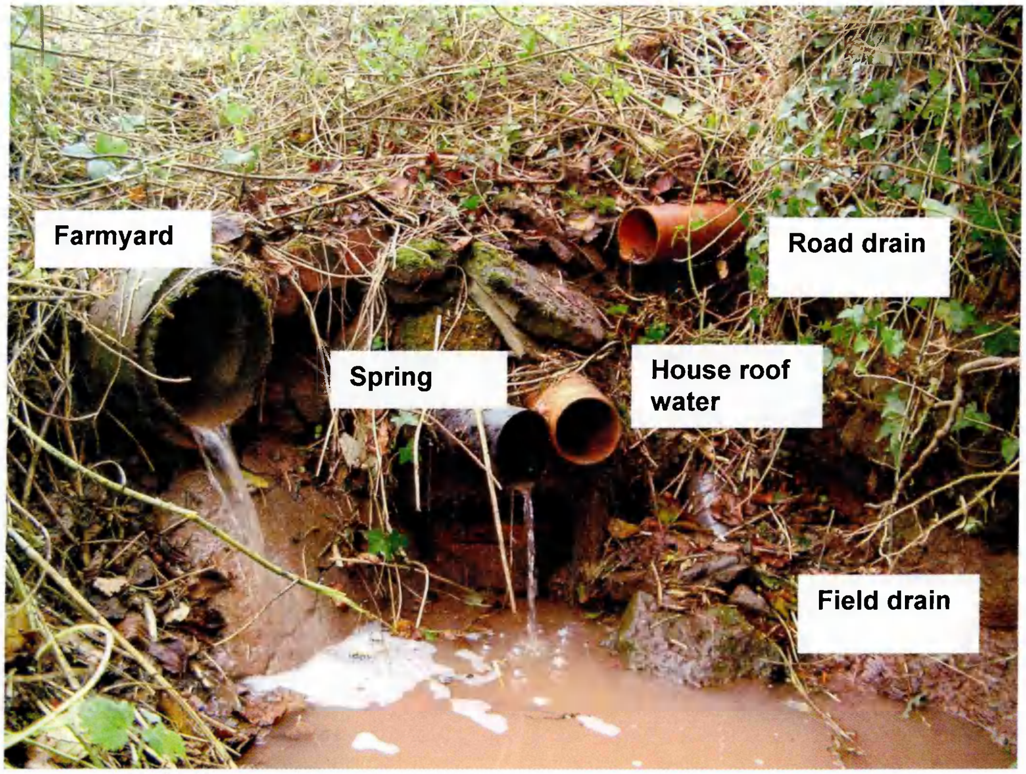


Plate 7.2 Piped discharges of runoff from multiple sources feeding into a headwater stream in the Kivernoll catchment.

Within these three rural catchments, concentrations of P in storm runoff from the different source groups sampled varied by up to two orders of magnitude between sites and sampling dates. This large variability is due to differences in rainfall intensity and flow rates on different types of surface, the chemical characteristics of the source group and the effects of a variety of controlling factors relating to anthropogenic and management activities (Duncan, 2005; Edwards et al., 2007). For example, the composition of farmyard runoff will depend on the type of farm (arable v livestock), the type of livestock and their period of housing, frequency of washing down, farm building design and the routing of water (Edwards et al., 2007). Concentrations in road runoff will depend on the type of surface, the number of vehicles passing per day (e.g. SS) and surrounding land use (e.g. whether urban open, residential or rural agricultural land), (Mitchell, 2001; Kayhanian et al., 2007). Concentrations in field runoff are dependent on antecedent soil moisture, soil P status, recent applications of fertilisers and manures and type and timing of soil cultivations (e.g. Schelde et al., 2006). The relationship between source availability and flow speed and routing will govern the concentration sampled at any one time. Although some measurements of spot flow were taken at the time of sampling, there was no clear effect of flow on concentrations due to the variable amounts of source P material available for mobilisation between sites and sampling dates.

Although sample numbers were low in this survey, some notable differences between the P source groups were clearly evident, particularly at sites where more than 5 samples were taken. In general, P concentrations were in the order: field drains < field surface runoff < roads < farmyards = septic tank ditch (e.g. Table 7.5), but there were some notable exceptions (Fig. 7.3). Lower concentrations of all P forms in field drains relative to overland flow probably reflects the greater opportunity for adsorption of DRP and filtering of PP as water percolates through the soil as shown by Haygarth et al. (1998b), and resulting in a greater proportion of P in dissolved form (close to 50%) compared to overland flow (ca. 20-25%). Hence differences in PP concentrations were due to differences in the amounts of SS transported rather than the P content of the SS, which was very similar between sub-surface and surface runoff and between sites, at ca. 0.2%.

The lack of any difference in runoff SS-P concentrations between the P4, M6 and H14 drains despite the large difference in field soil P status is consistent with the results of the Holbach field trial at Rosemaund, although the actual values were lower at Rosemaund (0.1%) compared with the values here (0.2%). This is probably due to differences in the particle size of the SS transported. However, one field drain (H14) also showed evidence of more significant contamination with significantly elevated DRP, NH_4-N and B concentrations, which reflected the close proximity of the field to the farmyard and possibly to the use as a sacrifice area for dilute poultry yard washings. However, the strong correlation between DRP and B suggests a common detergent source as has been found in larger catchment studies (Jarvie et al., 2003; Neal et al., 2005). This suggests that single field drains may be interconnected with other drains and deliver storm runoff from multiple sources and not just the drainage water from the field.

Road runoff tended to show larger concentrations of both dissolved and particulate P forms than runoff from fields, although the P enrichment of the SS transported was very similar between these source groups at Kivernoll. The much greater range in PP concentrations in road runoff is due to the rapid flow response to rainfall falling on an impervious surface mobilising SS present on the road. The origin of this SS is undoubtedly topsoil previously carried out of the field on tractor tyres and subsequently deposited on the road and carried in runoff during storm events. However, a greater range in DRP concentrations in road runoff was also noted at more than one site. Within the Kivernoll catchment, road runoff was mixed with a ditch receiving septic tank overflow before entering a collection pit which was piped to the stream (site K4). The strong relationship between DRP and B suggests this was the main source of the SRP rather than soil P release. This discharge was particularly important because it constituted the headwater stream in the Kivernoll catchment. It is interesting that the level of DRP contamination by the septic tank discharge was not sufficient to raise SS-P concentrations probably due to dilution by the larger SS concentrations associated with the road runoff component.

In the Whitchurch catchment, the runoff entering the left road drain was very enriched with DRP compared to the right road drain which may be due to the close proximity of the road to a house where roof and yard water was contributing to the runoff before entering the stream. Edwards et al. (2007) observed high DRP concentrations in roof runoff which they associated with bird droppings. However, a couple of samples collected from a Kivernoll site (K13) which was thought to be from a spring contained very little DRP but exceptionally high concentrations of NH_4-N. It is possible that DRP enrichment may partly reflect the difficulty of filtering highly turbid samples in the field causing very fine colloidal particles to pass through the filter. However, the DRP enrichment at the Whitchurch site was also combined with a large average NH_4-N concentration suggesting some faecal contamination from the nearby houseyard or roof possibly from farmyards further upslope. Since roads universally pass by domestic dwellings it is perhaps not surprising that they are prone to contamination from a variety of P sources of domestic origin. For example Mitchell (2001) found road runoff from residential areas tends to be higher than from main highways. Road runoff must therefore be considered a more concentrated source of P than field runoff.

As expected, farmyards were particularly concentrated in P due to the presence of livestock excreta and potential detergents used for washing down (Edwards and Hooda, 2007; Edwards et al., 2007). The majority of this P was highly bioavailable. The largest DRP concentrations were measured in runoff from the sheep shed at Rosemaund where the amount of faecal material present at the surface was much greater. Mobilisation of faecal material was high due to the rapid surface runoff that is generated on impervious surfaces causing an increase in all P forms measured. Preedy et al. (2001) observed mobilisation of both dissolved and particulate forms in overland flow when manure was applied to a wet clayey soil with limited permeability. Some samples from all three farmyard sources contained elevated concentrations of B. As pointed out by Edwards et al. (2007), farmyards combine a number of multiple sources including farm buildings, adjacent livestock collecting areas, access tracks and overflows from domestic wastewater systems. Indeed, farmyard runoff was as concentrated as the field ditch which regularly received septic tank overflow, which was stimulating abundant growth of macrophytes (Plate 7.3).

Plate 7.3 Macrophytes flourish in a field ditch regularly receiving overflow from a village septic tank (autumn 2007).

Contrary to the diluted septic tank contamination in the road drain at Kivernoll, the amount of septic tank overflow entering the field ditch in the Dinedor catchment was sufficient to greatly increase SS-P concentrations relative to those observed in surface and sub-surface field runoff. A similar effect of STW effluent on SS-P content was observed in the weekly Kivernoll stream samples and SS and stream bottom sediments are clearly able to attenuate septic tank derived P (Zanini et al., 1998). The calculated P export from the Kivernoll village STW (0.64 kg P ha^{-1}) is twice as large as that calculated for the farm STW at Rosemaund (Chapter 5). The Kivernoll STW is quite a bit larger (400 v 65 people) and the difference may reflect greater detergent use in a village as opposed to a farm with a few visitors. Values of P export coefficients for domestic wastewater used in modelling range from 0.38 kg P ha^{-1} (Johnes, 1996) to 0.9 kg ha^{-1} (Smith et al., 2005).

Runoff from farmyards and septic tanks are not only more concentrated in P than runoff from farmed fields but also contain a much greater proportion of the P in dissolved and therefore bioavailable forms. Some road runoff was also contained a high proportion of dissolved P. Furthermore, these 'intermediate' sources of P are delivered to the streams more continuously than field runoff and all through the year. This is because the discharge is not always dependent on storm flow and hence semi-continuous (septic tanks), or because storm flow always generates runoff from impervious surfaces which have a renewable supply of source material (e.g. soil, dust, faeces, bird droppings etc). These discharges will be continuing through the ecologically sensitive spring and summer periods, in contrast to discharges from cultivated land and grassland which largely arrive in the water during winter storms and when biological activity is low.

The ratio of TDN:TDP concentrations measured in these intermediate sources was always greater than the optimum (elemental) ratio of 7:1 required by algae (Redfield, 1958), despite the large P concentrations measured. This is largely due to the considerable leaching of nitrate from these catchments, especially in the drains at Kivernoll where poultry manure was regularly applied. This was most apparent in the large difference in NO_3-N concentrations between the set-aside field which had received no N fertiliser (P4, 1 mg L^{-1}) and the two arable fields receiving poultry manure (>11 mg L^{-1}). Clearly nitrate remains an issue for water quality in this region.

7.6 Conclusions

Accurate identification of the major sources contributing P to streams in catchments suffering from eutrophication is essential if improvements in water quality are to be achieved. The results of this preliminary survey of a range of potential P sources clearly show this is a very difficult task even in small rural catchments with limited urbanisation. Not only were there a number of runoff entry points into streams associated with roads, but there was evidence of direct contamination of runoff water with septic tank overflows and farmyard runoff. In many cases there were a number of drain discharges whose origin was uncertain or where individual drains had a number of combined runoff sources. Road runoff was often more concentrated in dissolved P than field runoff due to the greater opportunity to become contaminated by domestic wastewater, detergents and/or roof water.

It cannot therefore be assumed that cultivated and pasture land are the only sources of P in small rural catchments with no major point sources. Current attempts to improve catchment management, such as the governments Catchment Sensitive Farming Initiative, must recognise the greater ecological importance of sources associated with urbanisation of rural areas rather than targeting just agricultural practices. On the basis of this survey, only small amounts of urbanisation are required in rural catchments to have a profound influence on stream P chemistry and are probably more important than agricultural sources. Given the complex origins and routing of multiple P sources found in these small rural catchments, it is difficult to see how catchment models can ever be anything else than screening tools at the larger scale.

8

THE EFFECT OF SOIL MANAGEMENT ON RUNOFF PHOSPHORUS: AVON

8.1 Introduction

As reviewed in Chapter 2, and demonstrated by the results of the field and catchment studies at Redesdale and Rosemaund (Chapters 4 - 7), much of the agriculturally-derived P entering water bodies is associated with fine soil particles enriched with P previously added in fertilisers and manures. In reviewing the results from sediment source tracing in a number of UK catchments, Walling (2005) found that 60-96% of the SS measured in rivers during storm events was derived from surface land sources as opposed to river channel banks or sub-surface sources. The largest surface soil contributions were in intensively cultivated lowland catchments, where combinations of vulnerable soils, frequency, timing and method of cultivation, presence of compacted tramlines and lack of crop cover during storm events often cause significant erosion and off-site impacts (Davidson and Harrison, 1995; Chambers et al., 2000; Evans and Boardman, 2003)

Land use management therefore has a large influence on runoff generation and the proportion of the catchment area contributing soil particles and associated P to the watercourse. Cultivated riparian fields are most at risk since they have direct hydrological connectivity with the watercourse. However as pointed out by Edwards and Withers (2007), fields some distance from the watercourse have become more hydrologically-connected in recent decades through features such as artificial land drainage systems, tramlines introduced to improve spreading accuracy of agrochemicals, and tracks/roads. Consequently, measures to control SS and P loss need to be targeted at contributing fields that have both a high potential for P mobilisation and rapid hydrological connectivity to the watercourse (Heathwaite et al., 2000). In-field 'source' control is therefore an important part of the integrated approach to catchment management required for diffuse pollution control.

A number of improved soil and crop management practices have been identified to help reduce SS and P mobilisation at its source (Withers and Jarvis, 1998; Sims and Kleinman, 2005), often at relatively low cost. These include providing adequate crop cover during critical periods, avoidance of compaction by timely cultivations, incorporating organic matter to improve structural stability, changing cultivation method and/or cultivation and tramline direction on sloping fields, or a change in land use in eroding areas (Ministry of Agriculture, Fisheries and Food, 1999; Chambers et al., 2000; Environment Agency, 2001; Evans and Boardman, 2003). Most research has been directed at cultivation methods (reduced cultivation or conservation tillage) and cultivation direction in relation to field slope. Reduced cultivation includes direct drilling (no cultivation prior to drilling), shallow (<10 cm) tillage without soil inversion and deep (>10 cm) tillage without inversion (Davies and Finney, 2002). In theory, reduced cultivation techniques offer a number of benefits including lower energy (cultivation) costs, decreased susceptibility to soil structural degradation, carbon sequestration and a richer biological community in the soil (Holland, 2004). Many studies have reported large reductions in runoff and erosion when reduced cultivations or conservation tillage techniques have been adopted (Carter, 1998), with studies by Jordan et al. (2000), Chambers et al. (2000) and Quinton and Catt (2004) of relevance to the U.K.

In practice, however, reduced cultivation has led to reduced yield and increased runoff in some instances, and has been variably effective for erosion and P control because the technique must be adapted to the local soil and climatic conditions, thus requiring a higher degree of management skill. Particular problems in the UK have been encountered with build-up of grass weeds, poor crop establishment due to increased pest damage and/or severe topsoil compaction where the technique has been adopted on structurally unstable soils (Davies and Finney, 2002). A number of studies have also highlighted an increase in the more biologically active 'dissolved' fraction of P in land runoff under conservation tillage due to the preferential accumulation of available P at the surface of non-inverted soils (Karlen et al., 1991; Carter, 1998). In one study, the increase in dissolved P increased P export even though the transport of soil particulate-P was reduced (Gaynor and Findlay, 1995). Increased leaching of other more mobile nutrients has also been recorded (Carter, 1998).

There is less quantitative information on the effects of cultivation timing on SS and P loss risk, or on the interaction between cultivation timing and cultivation methods, despite numerous field and catchment observations showing surface roughness and lack of crop cover are major factors influencing the incidence of erosion risk (Chambers et al., 2000; Evans and Boardman, 2003). Martin (1999) found that late cultivation led to the greatest rates of runoff and erosion in a comparative study of cultivation and cropping techniques to control 'muddy flooding' in Northern France. Early drilling to quickly establish a crop cover is therefore a potentially useful management option to help reduce erosion risk and more information is needed on the effectiveness of this option in relation to other aspects of cultivation practice. Although one of the more pragmatic technical advances in recent decades from an agronomic perspective, tramlines have also been implicated as a cause of increased surface runoff, soil erosion and transport of diffuse pollutants to watercourses (Robinson and Naghizadeh 1992; Chambers et al., 2000; Basher and Ross, 2001).

Tramlines are semi-permanent wheelways for farm machinery to travel down during spraying and fertilising operations without causing wheeling damage to the rest of the field, sometimes referred to as 'controlled trafficking'. They were first introduced in the UK in the early 1970's, are usually standardised at either 12 m or 24 m and have become a ubiquitous feature of arable farming. However, tramlines not only reduce infiltration rates due to the soil compaction created beneath the tractor wheel, but also act as a channel for any runoff that is initiated due to the indentation of the soil, especially after multiple tractor passes (Fullen and Reed 1987). Rill erosion is often seen in tramlines that run up-and-down the hillslope due to the critical shear stress of water flow created by this channelling effect and subsequently encroaching on the surrounding soil (Basher and Ross 2001). In surveys of erosion in lowland England and Wales, compacted wheelings and tramlines were considered by farmers and researchers to be the major cause of erosion in arable fields. In the most recent survey between 1989 and 1994, the presence of tramlines and wheelings was the major causal factor in 34% of 146 surveyed fields where erosion occurred (Chambers *et al.* 2000). However, little quantitative information exists on the relative importance of tramlines relative to other soil and crop management practices and what aspects of tramline management might be important in directing measures to minimise SS and P loss.

The objective of this study was to evaluate the effects of cultivation method, cultivation timing and the presence of tramlines on the amount of infiltration excess overland flow, and associated transfer of SS and P, from farmer demonstration areas in a priority catchment suffering the effects of siltation and eutrophication. It was hypothesized that cultivation timing is at least as important as cultivation method as a management factor influencing P transfer and that tramlines running up-and-down slope will exacerbate runoff and entrained SS and P due to soil compaction and channelling effects.

8.2 The Avon catchment

The Hampshire Avon is a groundwater-dominated catchment (1706 km^2) that spans the counties of Wiltshire and Hampshire in Southern England. It has been designated as a Special Area of Conservation (SACs) under the EC Habitats Directive, and a pilot area for the development of Eutrophication Control Action Plans by the EA (Environment Agency, 2002). Siltation and eutrophication are considered to have increased as a result of agricultural intensification in recent decades. In particular, the upper part of the catchment above Salisbury has been suffering from 'Chalk Stream Malaise' with a loss of key macrophytes and decline in salmonid and coarse fish species and invertebrates (Huggins, 1998). A local 'LANDCARE' initiative was begun in 1999 to raise awareness of the declining water quality and encourage more environmentally sustainable land management practices. Small demonstration sites were established on key soil types in the catchment to provide a focus for farmer discussions and show how soil and crop management could be modified to reduce the risk of SS and P loss. To supplement this farm advisory activity, field plots at three of the demonstration sites were monitored over two successive winters (2002/03 and 2003/04) to quantify the effect of selected best management practices (BMPs) on overland flow and the concentrations and loads of SS and P in the runoff.

The field demonstration areas were established on the three main lithologies that dominate the catchment area: Upper Chalk (Wilton), Upper Greensand (Pewsey) and Kimmeridge clay (East Knoyle), Fig. 8.1. Calcareous Chalk soils are stable, easily worked, silty textured, variable in depth, very free draining and used for continuous arable cropping. However, they are often cultivated up and down steep slopes, where runoff might concentrate in compacted tramlines.

Greensand soils are deep and also well suited to intensive arable farming, but contain a high proportion of very fine sand that restricts rapid infiltration and causes the soil surface to seal over (cap) easily during storms, accelerating the risk of overland flow and erosion on hillslopes, especially where tramlines run up-and-down slope. Kimmeridge Clay soils are inherently poorly drained but have tile drainage systems to improve infiltration rates in the flatter landscapes dominated by grass-based livestock farming. In recent years, silage maize has become a popular dairy cow feed but the introduction of this arable crop into the area has increased the risk of soil wash contaminated with manure running off these clay fields during heavy storms.

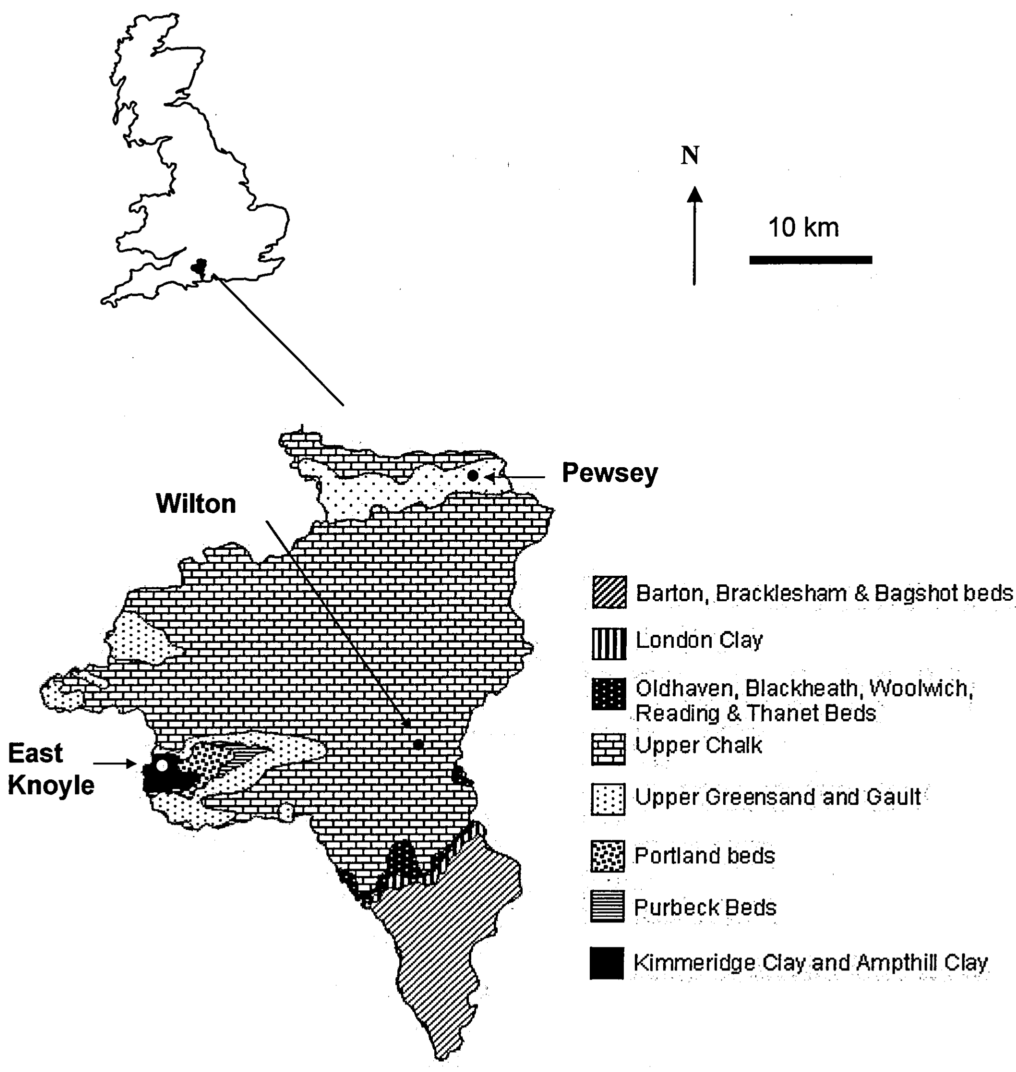


Figure 8.1 Location of the demonstration sites (Pewsey, Wilton and East Knoyle) in relation to the geology of the Hampshire Avon catchment in England.

The demonstration areas were cultivated and drilled either early (E) or late (L), and were either traditionally cultivated (TC) or reduced cultivated (RC), providing four treatment combinations:

E-TC, E-RC, L-TC and L-RC. Some treatment combinations and/or runoff plots were not established due to weather or field restrictions (Table 8.1). Early drilling was usually at the end of September, but late drilling varied from late October to early January depending on the weather. Traditional cultivation included ploughing to 20-25 cm (with or without a press), and either tine harrowing or power harrowing before drilling. Reduced cultivation was with either heavy discs (Pewsey and Wilton) or a heavy harrow (East Knoyle) to 5-8 cm.

Table 8.1 Treatments compared at each demonstration site and the dates monitoring started.

Year/treatment	Pewsey	East Knoyle	Wilton
2002/03			
E-TC	√ (4 Dec)	x	√ (4 Dec)
E-RC	√ (4 Dec)	x	√ (4 Dec)
L-TC	√ (17 Jan)	x	x
L-RC	√ (17 Jan)	x	x
no tramlines	√ (17 Jan)	x	x
2003/04			
E-TC	√ (22 Oct)	√ (22 Oct)	√ (22 Oct)
E-RC	√ (22 Oct)	√ (22 Oct)	√ (22 Oct)
L-TC[1]	√ (23 Dec)	√ (7 Jan)	√ (7 Jan)
L-RC	√ (23 Dec)	√ (7 Jan)	√ (7 Jan)
no tramlines	√ (22 Oct)	x	x

[1]At East Knoyle, the L-TC treatment was replaced with an E-RC 'headland' treatment.

On each demonstration plot, three hydrologically-isolated runoff plots (15m long by 2m wide) were installed as soon as practicable after drilling to monitor the mobilisation of SS and P in surface runoff generated by successive storm events. Typically 9-10 storm events were monitored on the early drilling treatments, and 5 events were monitored on the late drilling treatments. Each plot contained a tramline which was established soon after early and/or late drilling and before the runoff plots were installed. To compare the effects of the tramline, three additional runoff plots without a tramline were installed on selected cultivation treatments at Pewsey. At this site, three additional runoff plots were also established on an off-plot area where the direction of drilling was across slope rather than up-and-down slope. More comprehensive details of the demonstration plots in the Avon catchment, the treatments compared and the methodologies used to collect runoff and measure SS and P concentrations and loads are provided in Chapter 3.

8.3 Rainfall, runoff and erosion

Rainfall differed markedly between the two monitoring years, providing a range of conditions for runoff sampling (Fig. 8.2). Prolonged heavy rain (ca. 320 mm) during October and November 2002 delayed establishment of the runoff plots until early December and prevented drilling of the late treatments at Wilton and East Knoyle. At East Knoyle, the soil remained so wet that attempts to establish the plots had to be abandoned. At Pewsey, the heavy rain caused surface sealing of the soil on the E-TC treatment and the uncultivated plots designated for the late drilled treatments showed extensive rill erosion downslope.

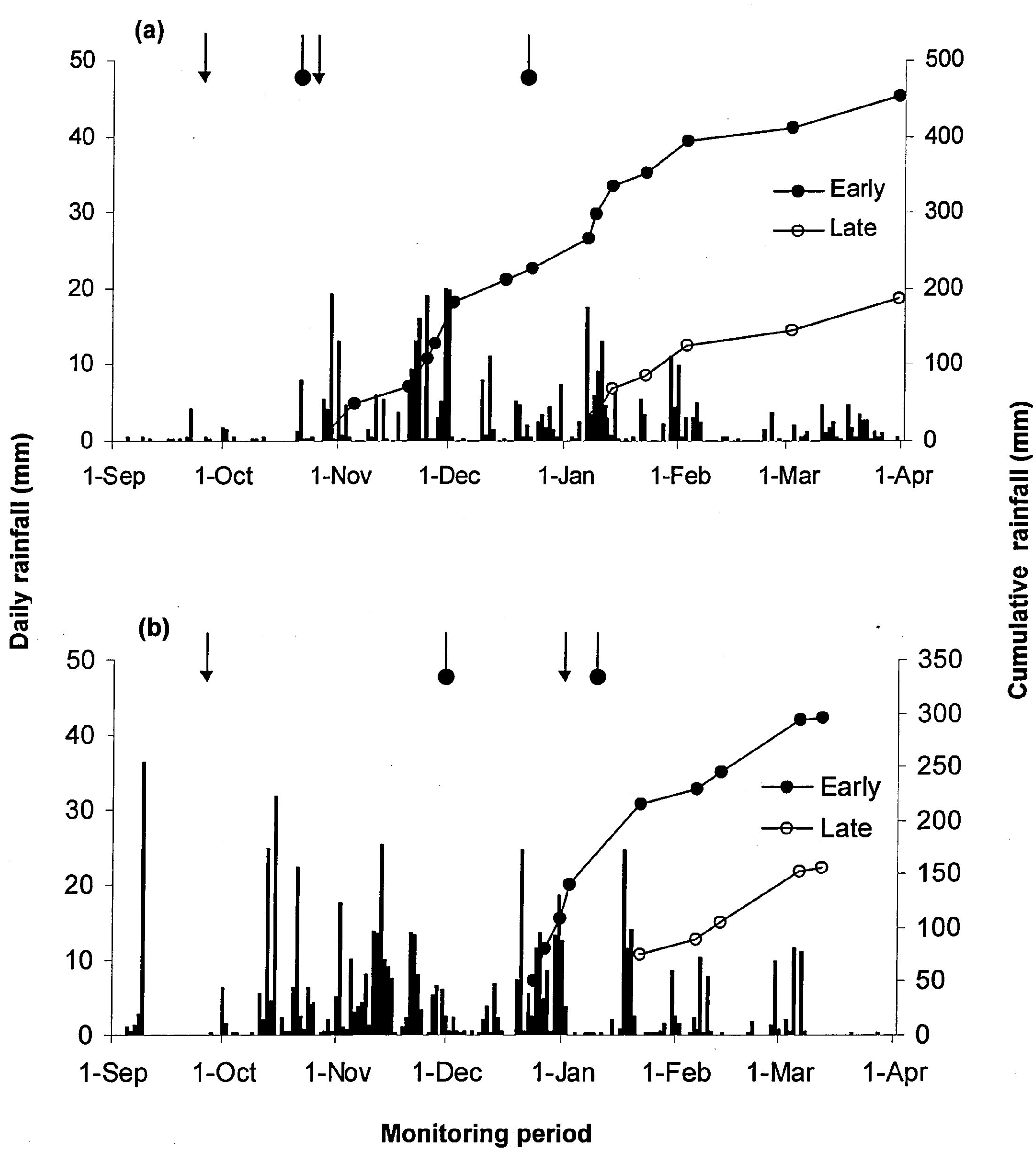


Figure 8.2 Daily and cumulative rainfall in (a) 2002/03 and (b) 2003/04. Arrows denote dates of treatment establishment (——▸) and plot establishment (——•).

In contrast, the E-RC plot showed little surface sealing due to the presence of straw residues left from the previous cereal crop. Much less precipitation occurred after the late-drilled treatments were established at Pewsey in early January 2003 (Fig. 8.2). Despite the delayed establishment of runoff plots, cumulative precipitation on the early drilled treatments was still ca. 300 mm while ca. 150 mm of rain fell during the late-drilled monitoring period. This pattern of rainfall contrasted strongly with that in the second year when very little rain fell directly after establishment of the early-drilled treatments, but heavier and more persistent rain fell during November 2003 following the establishment of the late-drilled treatments (Fig. 8.2). Consequently, runoff collectors were not installed on the late-drilled treatments until late December/early January. Cumulative precipitation over the monitoring period of the early-drilled and late-drilled treatments in 2003/04 was ca. 450 and 200 mm, respectively. At Pewsey, the heavy November rain falling onto the recently cultivated and bare soil again caused extensive surface sealing, and eventually rill erosion down the tramlines on the late-drilled treatments. Gushing of runoff down tramlines was also observed on the late-drilled treatments at the more steeply sloping Wilton site, while large amounts of runoff were collected from the headland treatment at East Knoyle which was drilled up-and-down the slope. These effects were not observed on the early-drilled treatments in 2003/04 at any of the sites.

8.4 Soil phosphorus

Potential mobilisation of soil particles (>0.45 μm) and P forms due to rainfall impact in storm runoff from each of the demonstration areas was assessed by the DESPRAL test (Withers et al., 2007). At each field site, traditionally-cultivated plots (E-TC) were compared with reduced-cultivated plots (E-RC) for the early-drilling treatment in 2003 (Table 8.2). Concentrations of SS varied from 843-1670 mg L^{-1} with significantly higher values at the East Knoyle site compared to the other two sites. Concentrations of PP and TP were largest at East Knoyle and lowest at the chalkland site (Wilton). The Wilton soil also showed significantly lower TDP concentrations than the other two sites reflecting its much lower level of soil P fertility (OP, 10 mg kg^{-1}) compared to Pewsey (OP, 33 mg kg^{-1}) and East Knoyle (OP, 67 mg kg^{-1}). Compared to site differences, there were no significant effects of cultivation type on SS or PP concentrations but TDP concentrations were always greater on reduced-cultivated plots than on ploughed plots (Table 8.2). However, TDP only accounted for between 6 and 16% of TP..

Table 8.2 Concentrations of suspended solids (SS), dissolved P (TDP), particulate P (PP) and total P (TP) predicted by the DESPRAL test.

Field area	SS (mg L^{-1})	TDP (μg L^{-1})	PP (μg L^{-1})	TP (μg L^{-1})
Pewsey				
E-TC	960	178	1105	1283
E-RC	843	220	964	1184
East Knoyle				
E-TC	1163	194	1534	1728
E-RC	1670	256	1701	1957
Wilton				
E-TC	963	37	746	783
E-RC	990	61	767	828
P value				
site	0.001	<0.001	<0.001	<0.001
treatment	NS	<0.001	NS	NS

NS, not significant, E-TC, early-drilled, traditionally-cultivated, E-RC, early-drilled, reduced-cultivated.

8.5 Cultivation effects

8.5.1 Runoff volume

Treatment effects on runoff volume were site specific and variable. At Pewsey, twice as much runoff was collected from E-TC than from E-RC in 2002/03 (Table 8.3). This difference was related to the degree of surface sealing which was evident with ploughing but not with reduced cultivation where cereal straw residues protected the surface. Tramlines were also deeper and more incised in ploughed soil than with reduced cultivation causing greater channelling of runoff (see section 8.6). Greater runoff from E-TC was most apparent when rainfall was high during the early part of the monitoring period. Hence, there was no difference in runoff between E-TC and E-RC when less rain fell after the late-drilled treatments were established at Pewsey in 2002/03 (data not shown).

With less intense rainfall after crop establishment, runoff was not different between E-TC and E-RC at Pewsey in 2003/04 (Table 8.3). Using a water balance model (Bailey and Spackman, 1996), a soil moisture deficit of 31 mm was estimated in the surface 20 cm root zone at drilling in 2003/04 compared to a deficit of 14 mm in 2002/03. Tramlines were not deeply incised on either cultivation type and did not channel runoff to the same degree as in the first year (see section 8.6). However, runoff volume from the late-drilled treatments at Pewsey, where surface sealing occurred under the November rain, was 5-fold greater than from the early-drilled treatments (Fig. 8.3). At Wilton and East Knoyle, runoff volume was not affected by cultivation method in either 2002/03 or 2003/04 (Table 8.3). Significantly greater runoff was obtained in 2003/04 on the headland at East Knoyle where the direction of drilling was up-and-down the slope (Fig. 8.3). A trend for greater runoff volume in late versus early-drilled treatments was observed at Wilton in 2003/04 (Fig. 8.3).

Table 8.3 Effect of early-drilled treatments (E-TC and E-RC) on runoff volume, and on the loss and flow-weighted concentration of suspended solids (SS), total P (TP) and total dissolved P (TDP) in 2002/03 and 2003/04.

Year	Pewsey		East Knoyle		Wilton	
Measurement	E-TC	E-RC	E-TC	E-RC	E-TC	E-RC
2002/03						
Runoff (mm)	20	10**	-	-	3	3
SS loss (kg ha^{-1})	476	90**	-	-	27	6**
SS concentration (g l^{-1})	2.4	0.9**	-	-	1.0	0.2***
TP loss (g ha^{-1})	387	105***	-	-	31	14***
TDP loss (g ha^{-1})	22	12	-	-	6	5
TP concentration (mg l^{-1})	1.9	1.1***	-	-	1.1	0.5**
TDP concentration (mg l^{-1})	0.1	0.1	-	-	0.2	0.2
SS-P concentration (mg kg^{-1})	809	1016			961	1411*
2003/04						
Runoff (mm)	7	6	3	5	4	5
SS loss (kg ha^{-1})	103	89	58	87	84	61
SS concentration (g l^{-1})	1.6	1.4	1.5	1.7	1.9	1.2
TP loss (g ha^{-1})	74	68	71	107	80	67
TDP loss (g ha^{-1})	10	11	18	26	9	7
TP concentration (mg l^{-1})	1.1	1.0	1.7	2.1	1.8	1.3
TDP concentration (mg l^{-1})	0.1	0.2	0.4	0.5	0.2	0.1
SS-P concentration (mg kg^{-1})	611	624	941	957	852	1001

E, early-drilled; TC, traditional cultivations; RC, reduced cultivations. *, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively.

8.5.2 Suspended solids

Suspended solids in runoff during individual storm events varied from <0.1 - 8 g L^{-1}. However, a significant relationship between flow and SS concentration was observed only for L-RC at Wilton, when water gushing down the tramlines mobilised additional soil particles. At Pewsey in both years, and at Wilton in 2002/03, flow-weighted SS concentrations were significantly (P <0.05) lower under reduced cultivation than under traditional cultivation (Tables 8.3 and 8.4).

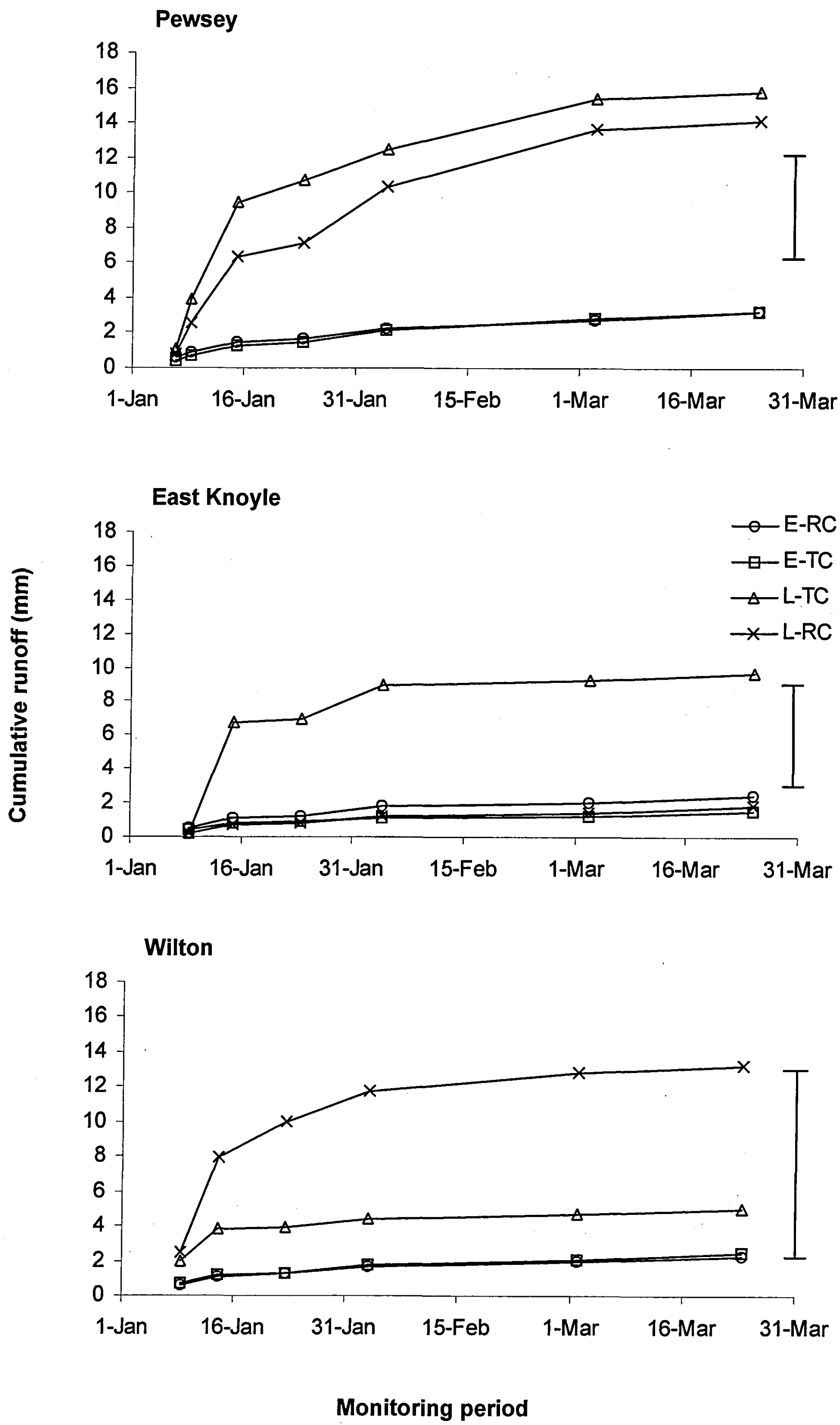


Figure 8.3 Cumulative runoff from successive storm events during the monitoring period following the establishment of the late-drilled treatments in 2003/04. Error bars denote least significant difference (P <0.05). (At East Knoyle, the L-TC treatment was replaced by a headland area).

Table 8.4 Effects of cultivation method and timing on the load and flow-weighted concentration of suspended solids (SS), total P (TP), total dissolved P (TDP) and the P concentration in the SS (SS-P) after the late-drilled treatments were established in 2003/04.

Site	SS		TP		TDP		SS-P
Treatment[1]	load (kg ha^{-1})	Concentration (g l^{-1})	load (g ha^{-1})	concentration (mg l^{-1})	load (g ha^{-1})	concentration (mg l^{-1})	concentration (mg kg^{-1})
Pewsey							
E-TC	77	2.3	48	1.5	3	0.08	600
E-RC	57	1.8	38	1.1	4	0.12	553
L-TC	650	4.3	548	3.6	31	0.18	796
L-RC	184	1.3	183	1.3	23	0.16	888
Significance	***	***	***	***	**	**	**
LSD	129	1.1	107	0.9	17	0.06	132
East Knoyle							
E-TC	50	2.5	53	2.5	5	0.32	866
E-RC	77	2.7	82	2.9	9	0.39	866
Headland	70	0.7	148	1.4	19	0.22	1643
L-RC	32	1.6	30	1.6	5	0.28	760
Significance	NS	**	NS	***	*	*	**
LSD		1.4		0.9	9	0.10	433
Wilton							
E-TC	63	2.8	51	2.3	3	0.12	774
E-RC	43	1.8	39	1.6	2	0.07	896
L-TC	150	3.0	97	2.0	12	0.06	732
L-RC	787	4.6	583	3.4	3	0.10	677
Significance	NS	NS	NS	NS	NS	NS	NS

E, early-drilled; L, late-drilled; TC, traditional cultivations; RC, reduced cultivations. *, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively. NS, not significant; l.s.d., least significant difference.

At Pewsey in 2003/04, SS concentrations were greater when ploughed soil was drilled late than when drilled early (Table 8.4). This greater SS mobilisation with L-TC was evident as rill erosion in the tramlines. In contrast, at East Knoyle, greater runoff on the headland diluted SS concentrations (Table 8.4). Samples of solids collected from storm events on the 5 November and 24 November 2003 at each site were analysed for their particle size distribution (Table 8.5). During both storms, the SS from Wilton contained a higher proportion of clay, had lower median and mode particle sizes and contained greater amounts of P than the SS collected from East Knoyle and Pewsey. Pewsey SS contained significant proportions of sand-sized particles, and showed more variability between cultivation treatments (E-TC vs E-RC on 5 November storm). Clay enrichment ratio (CER, defined as the ratio of the percentage clay in the SS to that in the original soil) varied from 1.1 at East Knoyle up to 2 at Pewsey (Table 8.5).

Table 8.5 Particle size, total P content and P enrichment ratio (PER) of suspended solids collected from each site during storm events on the 5 and 24 November 2003. Organic matter was not removed prior to particle dispersion and counting. Samples are the mean of three replicates unless otherwise stated.

Storm date	Particle size							
Treatment Site	<2 μm	2-63 μm %	>63 μm	Median μm	Mode μm	Total P mg kg^{-1}	CER	PER
5/11/03								
E-TC								
Pewsey	12	76	12	14.8	52.1	748	1.97	1.56
East Knoyle	15	85	0	6.7	17.4	n.d.	1.71	n.d.
Wilton	31	69	0	3.6	4.5	1657	1.61	1.66
5/11/03								
E-RC								
Pewsey	7	53	40	39.1	96.5	545	1.19	1.82
East Knoyle	16	84	0	6.0	15.2	1310	1.86	1.16
Wilton	27	73	0	4.3	10.6	1763	1.44	1.76
24/11/03								
E-TC								
Pewsey	7	54	39	42.2	93.6	483[1]	1.13	1.45
East Knoyle	14	86	0	7.5	18.4	1240[1]	1.67	1.10
Wilton	25	75	0	5.0	8.5	2413[1]	1.29	2.41

n.d. – not determined. [1]Single samples only.

Mean cumulative SS load in runoff was affected by cultivation method at Pewsey in both years, and at Wilton in the first year (Tables 8.3 and 8.4). The larger load of SS from E-TC than from E-RC in 2002/03, and from L-TC than from L-RC in 2003/04, at Pewsey reflected both greater runoff volume from the sealed soil surface and increased flow-weighted concentration in runoff. The lower SS load from E-RC at Wilton in 2002/03 was due solely to decreased particle detachment rather than to any decrease in runoff (Table 8.3). An effect of cultivation timing was obtained only at Pewsey in 2003/04, where late drilling increased SS load compared to early drilling for both methods of cultivation (Table 8.4).

8.5.3 Phosphorus

Cumulative TP load in runoff, and flow-weighted TP concentration, followed the same general pattern as for SS, since the majority (60-97%) of TP was in particulate (> 0.45 μm) form at all three sites. Concentrations typically varied from 0.1 to 6 mg TP L^{-1} for individual storm events, with maximum TP load up to 0.6 kg P ha^{-1} at Pewsey and Wilton, and up to 0.15 kg P ha^{-1} at East Knoyle (Tables 8.3 and 8.4). Significantly greater TP loss occurred with E-TC than with E-RC at Pewsey and at Wilton in 2002/03, and with L-TC than with L-RC at Pewsey in 2003/04. At Pewsey in 2003/04, greater TP loss occurred when soil was drilled late than when drilled early (Table 8.4). As with SS, flow-weighted TP concentrations were consistently lower when the soil was not inverted, except at Wilton where water gushing down the compacted tramline generated additional particulate P with L-RC (Tables 8.3 and 8.4).

As expected there was a strong positive relationship between SS and TP (and PP) concentrations at all sites (r^2 = 0.9). In 2002/03, the concentration of P in the SS (SS-P, calculated as PP/SS) tended to be lower on the ploughed plots (larger SS losses) than on reduced cultivated treatments at both Wilton and Pewsey, although this was only statistically significant at Wilton (Table 8.3). In 2003/04, SS-P concentrations were significantly greater on late-drilled treatments than on early-drilled treatments at Pewsey, while at East Knoyle, they were noticeably high where flow-weighted SS concentrations were very low on the headland treatment (Table 8.4). When SS concentrations were broadly similar between the sites, SS-P concentrations were significantly greater at East Knoyle and Wilton than at Pewsey. For example, on the early-drilled treatments in 2003/04, average concentrations were 618, 949 and 926 mg kg^{-1} for Pewsey, East Knoyle and Wilton respectively ($P < 0.001$) representing average P enrichment ratios (PER) of 1.9, 0.84 and 0.93. Similar site differences were obtained within individual storms (Table 8.5). These site differences disappeared when treatment effects on SS and PP concentrations became significant (e.g. the late-drilled treatments).

Dissolved P concentrations in runoff were generally not affected by cultivation method or timing except at Pewsey and East Knoyle in 2003/04. At Pewsey, TDP concentration was increased with increasing runoff volume, while at East Knoyle the reverse was obtained (Table 8.4). The molybdate reactive form was dominant (>95% of TDP), and flow-weighted average DRP concentrations over the respective monitoring periods were notably greater at East Knoyle (ca. 0.22-0.50 mg L^{-1}) compared to Pewsey (0.08-0.18 mg L^{-1}) and Wilton (0.06-0.21 mg L^{-1}). Hence, flow-weighted concentrations of TDP were very similar at Wilton and at Pewsey despite the large difference in soil P fertility.

8.5.4 Soil and crop effects

At Pewsey and Wilton, reduced cultivation resulted in a more consolidated topsoil structure than when the soil was ploughed (Table 8.6). Not all differences were significant at the 5% level, but some were significant at the 10% level. The consistent soil cultivation effect was obtained at both drilling dates, although the magnitude of the difference was generally greater with later drilling. Differences in soil strength were most pronounced between 5 and 25 cm depth (Figs. 8.4 and 8.5). At Pewsey, there was evidence of a plough pan below 25 cm depth. In contrast, there was no difference in soil physical quality among treatments at East Knoyle, except for better soil structure and slightly increased air capacity with E-RC compared to other treatments. In May 2003 (data not shown), soil shear strength was greater with E-RC than with E-TC, but bulk density air capacity was not affected by treatment. Infiltration was not affected by cultivation treatments, but was lower at Pewsey compared to the other two sites (Table 8.6). Similar data were obtained in 2002/03. At Pewsey, there was virtually no infiltration of water at some measuring points due to surface sealing.

Surface cover

Surface cover was a function of crop cover and previous crop residues. Residues left by the previous crop helped to provide a greater degree of surface cover where the soil was not inverted. Visual differences in the degree of surface cover were most pronounced at Wilton in 2002/03 and at Pewsey in both years. At East Knoyle, the previous crop was maize (*Zea mays* L.) providing a residue of coarse stems spaced widely apart. Percentage of bare ground is shown in Table 8.5. Surface cover was 20-30% on early-drilled ploughed areas and 15% on late-drilled ploughed areas.

Table 8.6 Effects of cultivation method and timing on soil physical quality and on the percentage of bare ground in 2003/04.

Site / Treatment	Dry bulk density (g cm^{-3})		Air capacity (%)		Structure Score[1]	Soil strength[2]		Infiltration Rate[3] (mm)	Bare ground (%)	
	Topsoil	Subsoil	Topsoil	Subsoil		Vertical (N m^{-2})	Horizontal (kPa)		23 Jan	9 Apr
Pewsey										
E-TC	1.15	1.08^{a}	14.3^{bc}	18.1^{b}	7.6^{c}	35	37	19.1	78	41
E-RC	1.19	1.17^{a}	8.8^{a}	14.8^{b}	4.7^{a}	70	59	12.3	36	23
L-TC	1.21	1.14^{a}	11.5^{ab}	18.4^{b}	6.7^{b}	49	39	6.6	85	40
L-RC	1.23	1.32^{b}	7.9^{a}	10.2^{a}	5.2^{a}	90	80	3.6	60	47
Significance	NS	**	***	**	***	***	***	NS	***	***
LSD						8.1	6.2		6.0	4.5
East Knoyle										
E-TC	0.86	0.84	18.8^{a}	17.9	5.2^{a}	46	64	62.2	71	8
E-RC	0.82	0.81	21.6^{b}	18.8	6.9^{b}	49	64	55.4	62	8
Headland	0.83	0.93	17.8^{a}	13.3	4.6^{a}	50	69	79.2	66	14
L-RC	0.88	0.85	18.3^{a}	14.4	4.7^{a}	47	57	47.7	86	38
Significance	NS	NS	*	NS	***	NS	NS	NS	**	***
LSD									10.9	6.0
Wilton										
E-TC	0.88^{a}	0.94^{ab}	27.3^{b}	25.3^{b}	6.9^{b}	29	41	76.0	79	44
E-RC	1.01^{b}	1.04^{ab}	20.9^{a}	23.1^{ab}	5.4^{a}	35	54	67.6	71	31
L-TC	0.95^{ab}	0.97^{a}	27.1^{b}	20.7^{ab}	6.2^{b}	34	49	102.0	85	52
L-RC	1.05^{b}	1.08^{b}	19.8^{a}	17.3^{a}	5.2^{a}	73	72	58.5	82	50
Significance[5]	*	*	*	*	***	***	*	NS	***	***
LSD						10.5	18.4		5.3	7.4

E, early-drilled; L, late-drilled; TC, traditional cultivations; RC, reduced cultivations. *, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively. Values with different letters are significant from one another. NS, not significant; l.s.d., least significant difference from Genstat analysis. [1]Score out of ten, 1 = poor, 10 = good. [2]Mean values over a depth of 30 cms. [3]Mean values over 5 hrs.

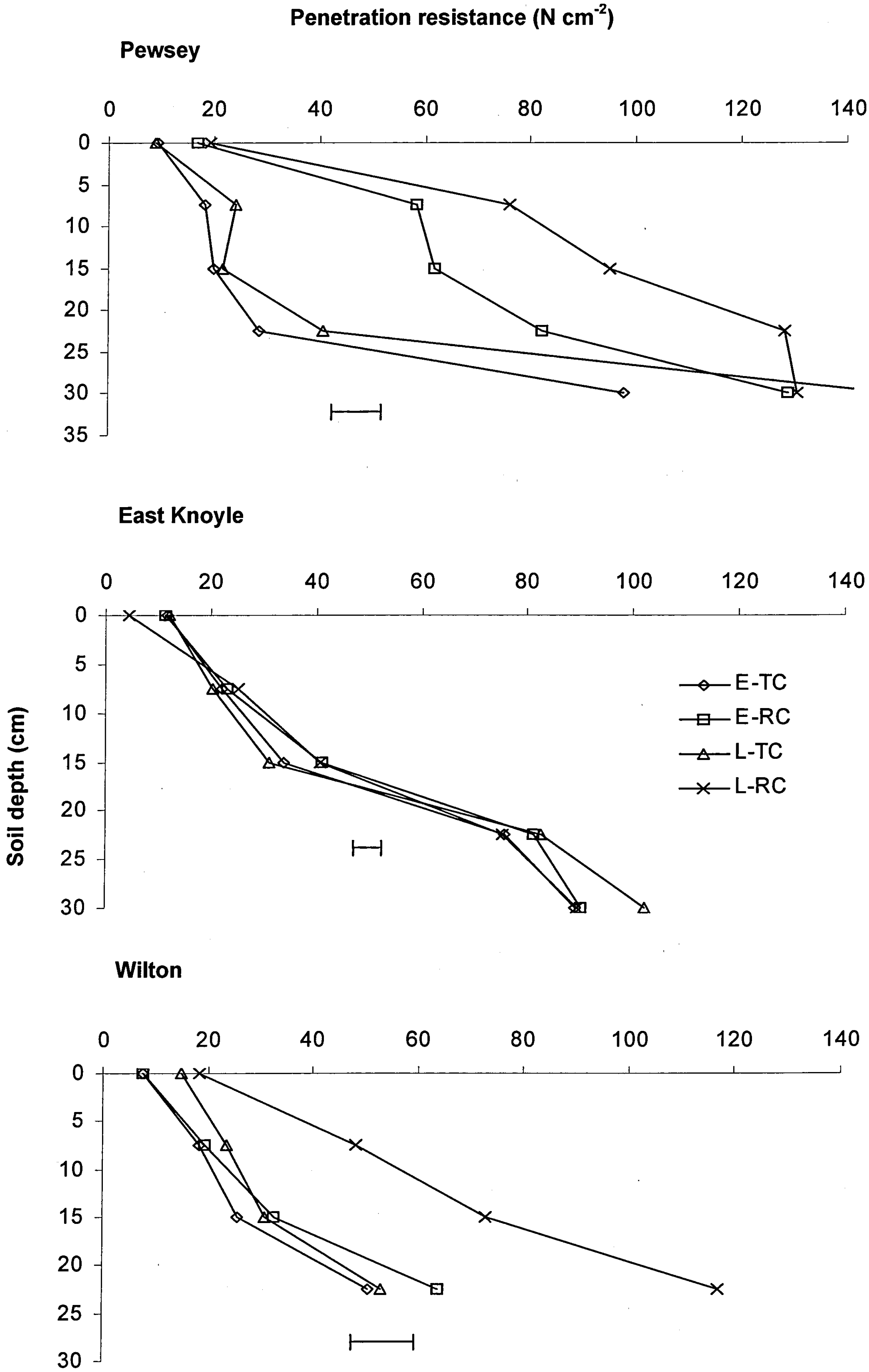


Figure 8.4 Effect of cultivation method and timing on soil penetration resistance to 30 cm in January 2004. Error bars denote least significant difference (P <0.05). (At East Knoyle, the L-TC treatment was replaced by a headland area).

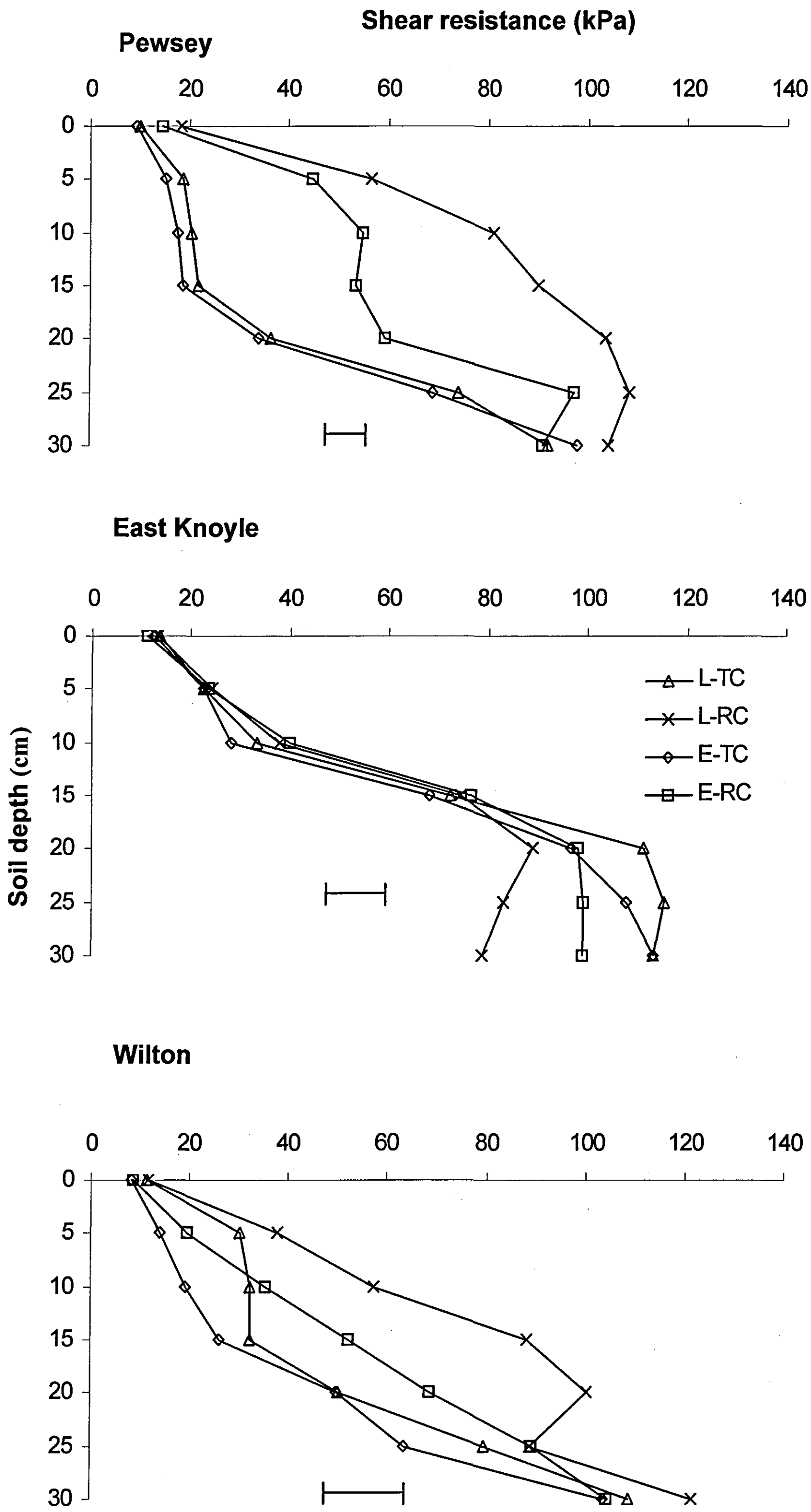


Figure 8.5 Effect of cultivation method and timing on soil shear strength to 30 cm in January 2004. Error bars denote least significant difference (P <0.05). (At East Knoyle, the L-TC treatment was replaced by a headland area).

At Pewsey, reduced cultivation produced 25-40% greater surface cover than ploughing. At Wilton and East Knoyle, reduced cultivation produced 8-9% greater surface cover than poughing. had only 8-9% greater crop cover than the E-TC plot. By April, differences in surface cover disappeared. Crop growth was noticeably more abundant at East Knoyle than at the other two sites, perhaps reflecting the better soil structure (Table 8.5).

8.6 Effects of tramlines - Pewsey

8.6.1 Runoff volume

Combining all the data for the early-drilled treatments in both years at Pewsey, there was a significant relationship between rainfall and runoff for plots with and without a tramline (Fig. 8.6). The gradient of the relationship was significantly (P <0.001) shallower for plots without a tramline, resulting in an increasing divergence in runoff rates between tramlined and non-tramlined areas as rainfall increased. For plots with a tramline, 1.9% of incoming rainfall was measured in runoff, whilst the corresponding figure for plots without a tramline was 1.3%. Tramlines therefore increased runoff on average by 46% when crops were drilled early.

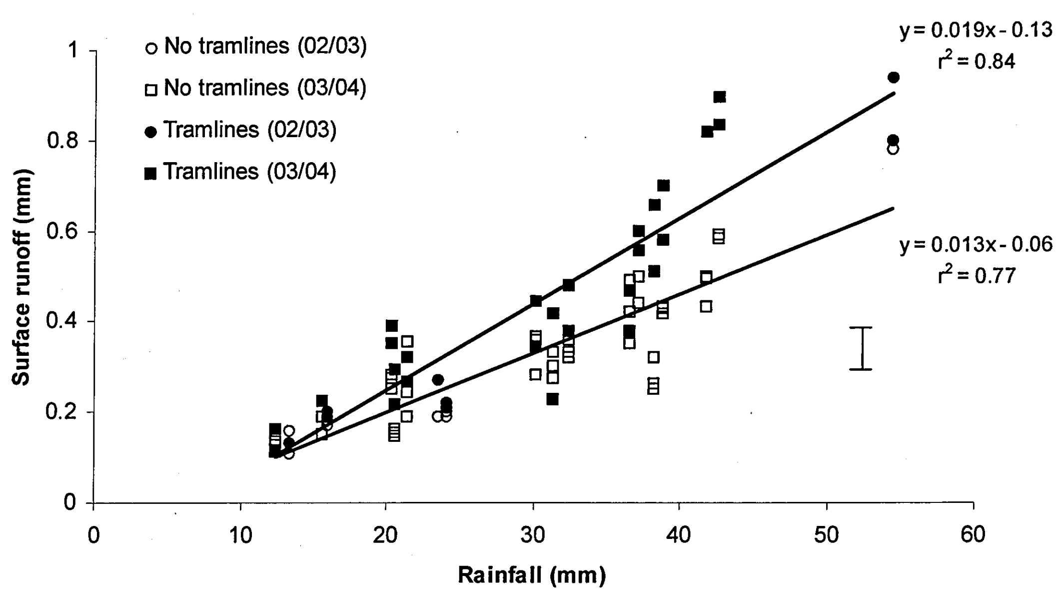


Figure 8.6 Rainfall-runoff relationships for plots on the early-drilling treatments with and without tramlines, and including plots with tramlines running across the field slope. Data points represent individual replicate plots in each year. The error bar represents the least significant difference.

Runoff volumes measured on the late-drilled treatments were much greater and more variable than on the early-drilled treatments reflecting the greater sensitivity of the capped soil surface to rainfall intensity. For the L-TC treatment, runoff averaged ca. 7% of rainfall (range up to 13%) from plots without a tramline and ca. 8% (range up to 18%) from plots with a tramline. Significant tramline effects on runoff were therefore absent where the soil became capped with considerable amounts of runoff clearly travelling down inter-tramline areas as well as down tramlines. Differences in runoff between the early and late drilling treatments in 2003/04 were therefore much greater than the differences in runoff due to the presence of tramlines.

8.6.2 First monitoring year (2002/03)

Soon after the early drilling treatments were established in the first year, runoff was being initiated in the tramlines sooner than from non-tramline areas, and those on the ploughed soil, which had tyre lugs pointing in a downward ∨ direction, appeared deeper and more incised than on the E-RC treatment which had tyre lugs pointing in an upward ∧ direction (Fig. 8.7).

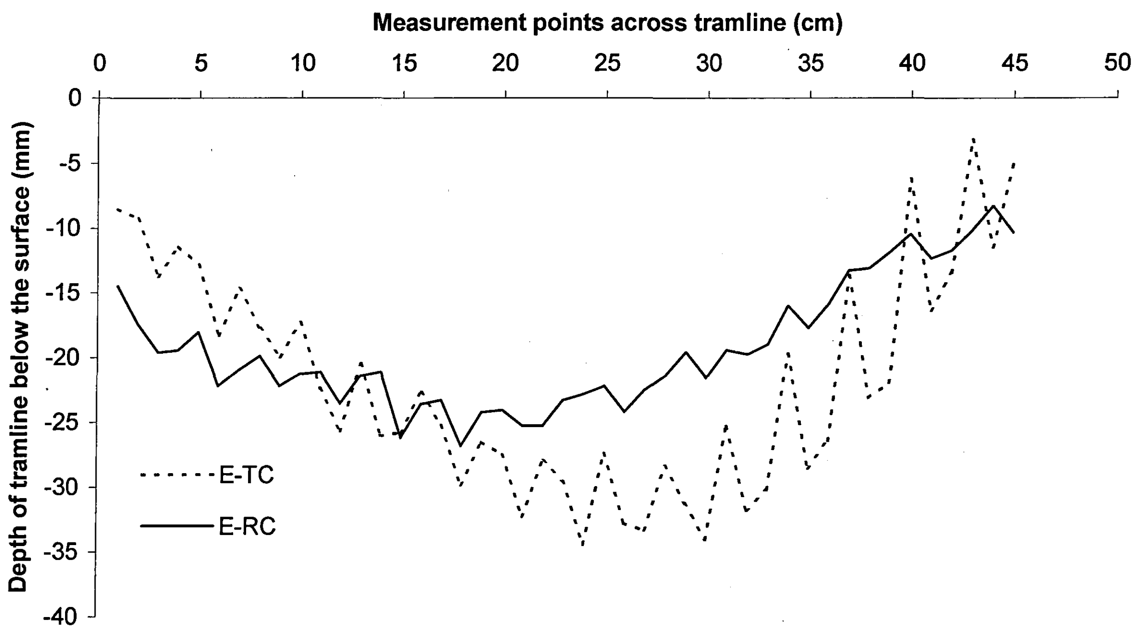


Figure 8.7 Tramline cross-sectional profiles for the traditionally cultivated (E-TC) and reduced cultivated (E-RC) treatments in June 2003.

Maximum tramline depth was 10 mm greater on the E-TC treatment than on the E-RC treatment, and with a more irregular surface as a result of the depressions left by the tractor tyres and/or the effects of the runoff water. After the plots with no tramlines were established in January 2003, the cumulative amounts of runoff collected from the plots on the E-TC treatment over the 5 monitored storm events were 65% greater (+1 mm) on the plots containing a tramline (Table 8.7). Much of this small but significant (P <0.01) difference occurred in the first heavy storm event, although runoff volumes were always greater where a tramline was present. This small amount of additional runoff caused considerably greater mobilisation of soil particles as evidenced by large significant increases in the flow-weighted concentrations of both SS and TP (Table 8.7). However, the extra runoff was too small to generate any rill erosion down the tramlines and did not increase concentrations of dissolved P (TDP), which consequently represented a smaller proportion of the total P lost. Plot exports of SS and TP from the E-TC treatment (70 kg ha^{-1} and 58 g ha^{-1}, respectively) were 4-5 fold greater where tramlines were present than where they were not present, reflecting both increased runoff and increased concentrations of SS and TP in the runoff (Table 8.7). The absence of tramlines had reduced SS and TP export by ca. 75% under the compacted soil surface conditions caused by the heavy rain that fell after the early drilling treatments were established on the ploughed soil.

Table 8.7 Effects of tramlines on the cumulative mean volumes of cumulative runoff, and associated loads and flow-weighted concentrations of suspended solids (SS), total P (TP) and dissolved P (TDP) in 2002/03.

Treatment[a]	Runoff[b] (mm)	Loads			Flow-weighted concentrations		
		SS (kg ha^{-1})	TP (g ha^{-1})	TDP (g ha^{-1})	SS (g L^{-1})	TP (mg L^{-1})	TDP (% of TP)
E-TC	2.4	70	58	2	2.9	2.4	4
E-TC-NT	1.4	14	15	2	0.9	0.9	12
P value	**	**	**	NS	**	**	*
l.s.d	0.4	23	18		0.9	0.8	6
E-RC	1.9	19	19	3	1.0	1.0	17
E-RC-NT	2.6	23	30	4	0.9	1.1	14
P value	NS	NS	NS	NS	NS	NS	NS

[a]Early-drilled (E), traditional cultivations (TC), reduced cultivations (RC), no tramline (NT).
[b]Cumulative runoff over the monitoring period 17 Jan. – 12 Mar. 2003.
*, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively.
NS – not significant (P >0.05), l.s.d – least significant difference

In contrast, there was no statistically significant (P >0.05) effect of tramlines on runoff, or on the concentrations and exports of SS and TP from the plots that had been reduced cultivated (Table 8.8). However, it was observed that the runoff volumes from one replicate plot on the E-RC treatment which contained a tramline that became partially crop covered was 50% less than for the other two replicates where the tramlines were bare (Fig. 8.8).

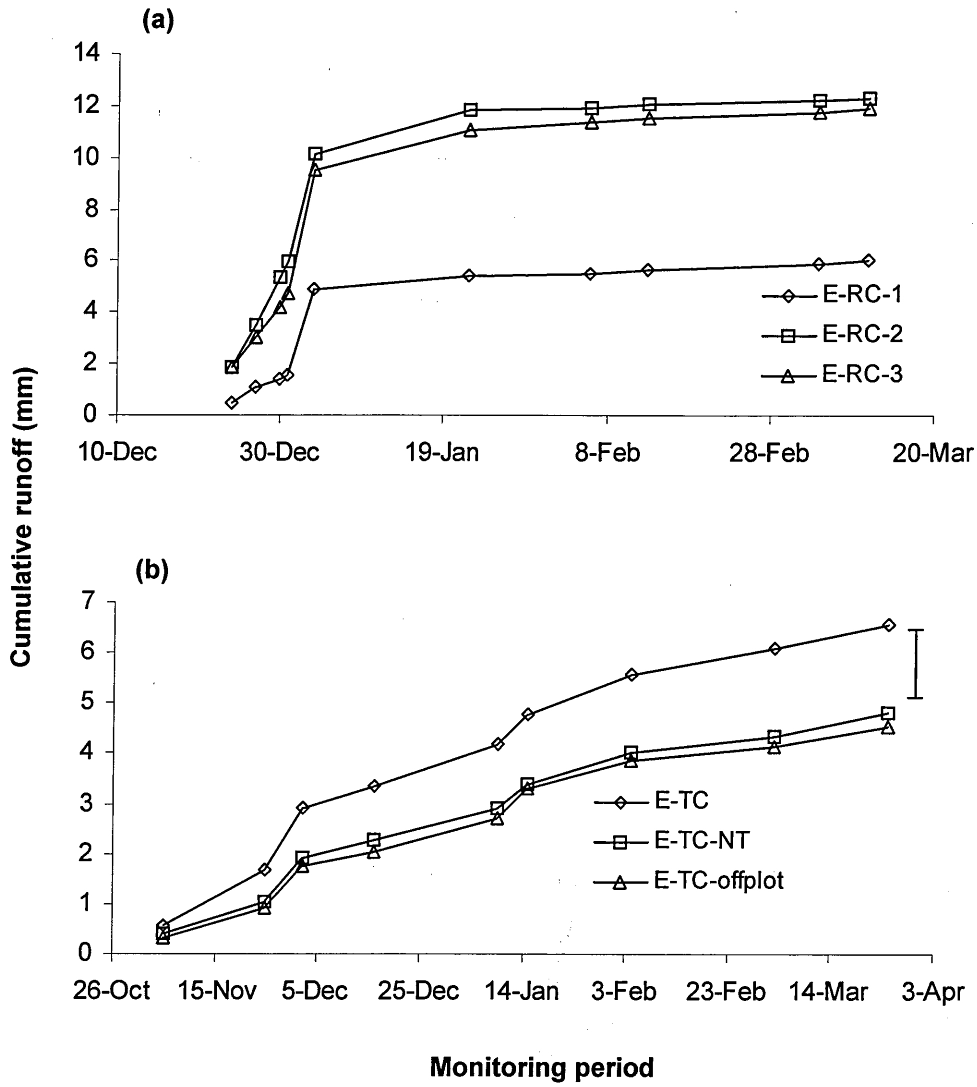


Figure 8.8 Pattern of cumulative runoff from the early-drilled traditionally cultivated treatment (a) during the 2002/03 monitoring period (E-RC-1 is the replicate plot with partial crop cover as shown in Figure 2b) and (b) either with a tramline (E-TC), without a tramline (E-TC-NT) or with tramlines running across slope (E-TC-offplot). Error bars represent the least significant difference.

Table 8.8 Effects of tramlines on the mean volumes of cumulative runoff, and associated loads and flow-weighted concentrations of suspended solids (SS), total P (TP) and dissolved P (TDP) in 2003/04.

Treatment[a]	Runoff[b]	Loads			Flow-weighted concentrations		
	(mm)	SS ($kg\ ha^{-1}$)	TP ($g\ ha^{-1}$)	TDP ($g\ ha^{-1}$)	SS ($g\ L^{-1}$)	TP ($mg\ L^{-1}$)	TDP (% of TP)
E-TC	6.6	103	74	10	1.6	1.1	13
E-TC-NT	4.8	94	55	10	2.0	1.1	18
E-TC-Offplot	4.5	97	69	11	1.9	1.4	15
P value	*	NS	NS	NS	NS	NS	**
l.s.d.	1.5						3
E-RC	6.2	89	68	11	1.4	1.0	17
E-RC-NT	4.6	75	47	7	1.6	1.0	15
P value	NS	NS	NS	NS	NS	NS	NS
L-TC	16	650	548	31	4.3	3.6	5
L-TC-NT	14	270	210	11	1.9	1.5	5
P value	NS	**	**	NS	*	*	NS
l.s.d.		221	195		1.7	1.5	

[a]Early-drilled (E), late-drilled (L), traditional cultivations (TC), reduced cultivations (RC), no tramline (NT).
[b]Cumulative runoff over the monitoring periods; E treatments 22 Oct. 2003 – 25 Mar. 2004; L treatments 23 Dec. 2003 – 25 Mar. 2004.
*, ** and *** denotes significance at the 5%, 1% and 0.1% level, respectively. NS – not significant (P >0.05), l.s.d – least significant difference

The degree of crop cover on the tramline is illustrated in Plate 8.1, and the effect on runoff was consistent over the extended monitoring period covered by the main study. Cumulative exports of SS and TP were also much smaller from the plot that contained the partially covered tramline. For example, cumulative exports of SS were 64 v 95 and 112 kg ha^{-1}, and cumulative exports of TP were 69 v 114 and 131 g ha^{-1} for the three replicate plots respectively, where the first plot was partially crop covered. These differences in SS and P export were solely due to differences in runoff, and SS, TP and TDP concentrations were quite similar across all the three plots.

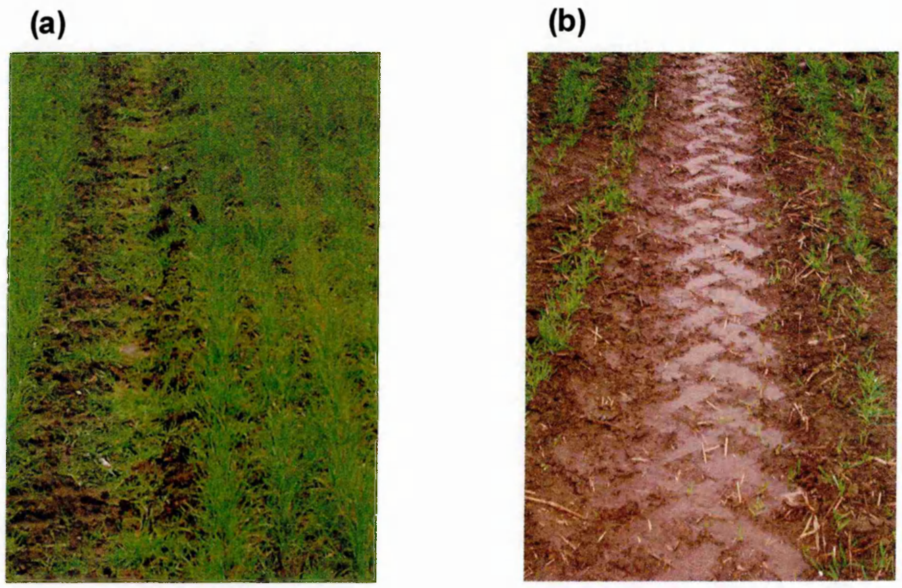


Plate 8.1 Runoff plots showing (a) partial crop cover in the tramline on the E-RC treatment (December 2002) and (c) runoff generation before non-tramlined areas (November 2003). Tyre lugs were positioned in a downward v encouraging flow.

8.6.3 Second monitoring year (2003/04)

In the second year, the effects of tramlines on runoff, SS and TP loss from the early-drilled treatments were compared over a longer period, and the cumulative volumes of runoff were greater (ca. 5 mm, Table 8.8). Also in this year, a comparison was made with an E-TC treated area outside the demonstration areas where the tramline direction was running across the slope (offplot) rather than up-and-down the slope. As in the first year, runoff was initiated in the tramline (Plate 8.1). However, in contrast to the first year, the soil was not greatly indented when the tramlines were established despite almost identical drilling dates in the two years (24 and 28 September, respectively). Two or three passes of the tractor wheel were required simply to establish a visible tramline both on the E-TC and the E-RC treatments reflecting a dry and stable soil structure, as confirmed by estimates of SMDs (31 cm in 2003/04 vs 14 cm in 2002/03).

On the E-TC treatment, there was significantly (P <0.05) less runoff where there was no tramline or where the tramline and drilling direction was across the field slope (Fig. 8.8). Despite these ca. 30% lower runoff volumes, flow-weighted concentrations of SS and TP tended to increase relative to those in runoff from E-TC plots with a tramline (Table 8.8), although this trend was significant at the 10% level only. Hence, there was no significant net effect of tramlines on exports of SS and P from the early-drilled ploughed plots. Flow-weighted concentrations of TDP were significantly (P <0.05) lower on the E-TC plots with a tramline running up-and-down the slope, but this was not sufficient to influence TDP export because runoff volumes were greater (Table 8.8).Similar reductions in runoff (ca. 2mm) were obtained on the E-RC treatment where tramlines were absent but this was not significant even at the 10% level. Neither was there any significant effect of tramlines on the concentrations, or cumulative exports of SS, TP and TDP on the reduced cultivated treatment (Table 8.8).

After establishment of the L-TC treatment, much larger volumes of runoff were generated due to capping of the soil surface irrespective of whether a tramline was present or not. Runoff volume was greater on the tramlined areas (+ 1.7 mm), but not significantly so, indicating that there was substantial runoff occurring on the non-tramline area within the plot. Despite these small runoff differences, there was considerably more entrainment of soil particles where tramlines were present with two-fold greater flow-weighted concentrations of SS, TP and TDP (Table 8.7). Cumulative loads of SS, TP and TDP from plots with tramlines (650 kg ha^{-1}, 548 g ha^{-1} and 31 g ha^{-1}, respectively) were consequently ca. 2.5 times greater than from plots without tramlines. During the monitoring period the tramlines on the late-drilled treatments became progressively more incised as rills developed during consecutive storm events.

8.6.3 Soil compaction in tramlines

Soil shear strength and penetration resistance measurements taken in 2003/04 were almost always greater in the tramlines than in non-tramlined areas but the differences were generally only statistically significant (*P* <0.05) on the traditionally cultivated treatments (Fig. 8.9).

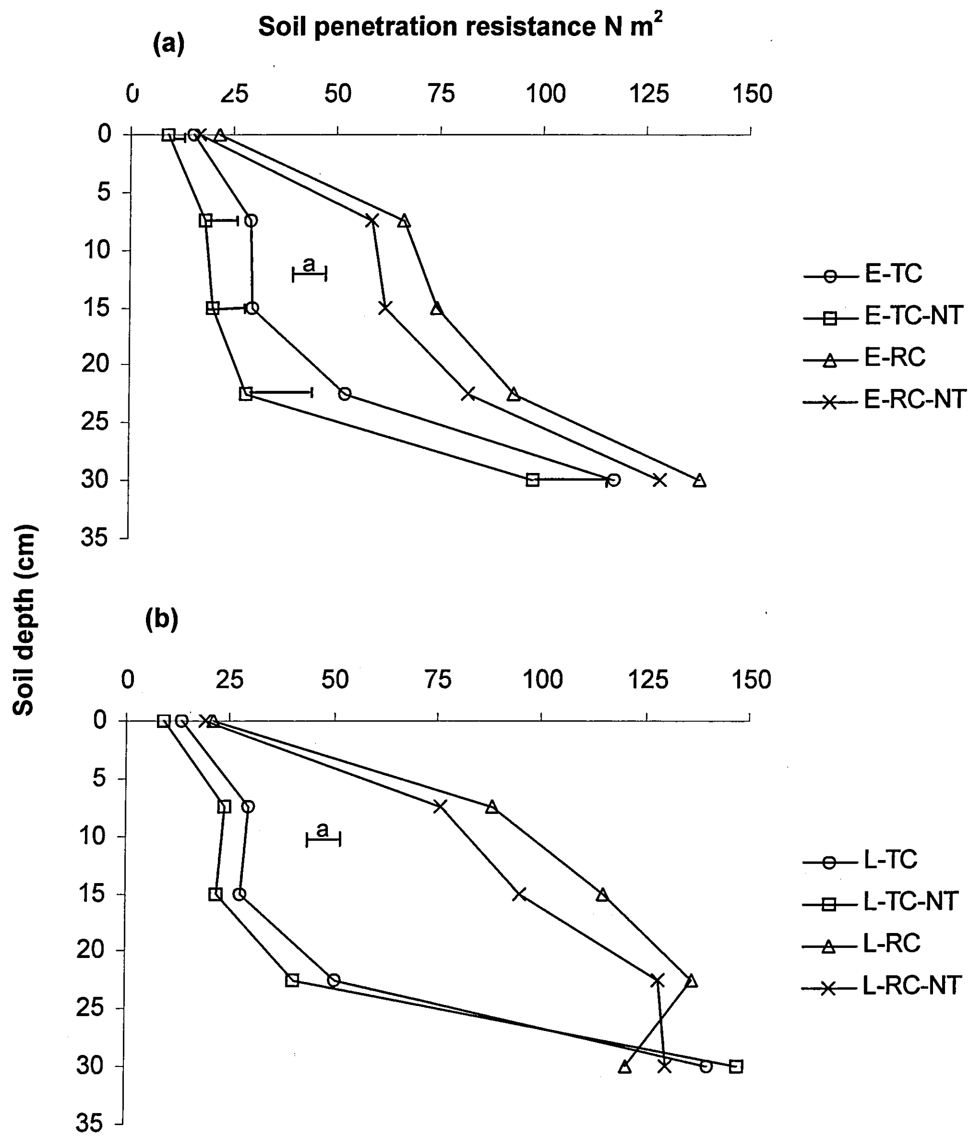


Figure 8.9 Effects of tramlines on soil penetration resistance with increasing soil depth down to 30 cm on (a) early-drilled treatments and (b) late-drilled treatments in January 2004. Error bars denoted by 'a' represent the least significant difference (*P* <0.05) between cultivation treatments meaned across soil depth. Error bars for significant differences between tramline and tramlined areas (where present) are given at each soil depth.

Penetrometer readings suggested that the soil under the tramlines on the E-TC treatment was significantly more compacted throughout the top 30 cm, but differences in soil penetration resistance in tramlines on the E-RC treatment were not sufficiently large to be statistically significant (Fig. 8.9). Similarly, significant differences in soil shear strength between tramlined and non-tramlined areas were obtained only at the soil surface (0-10 cm) of traditionally cultivated treatments (E-TC and L-TC), whilst the increases on the tramlines of the E-RC treatment tended to be smaller and not significant. The differences between tramlined and non-tramlined areas were also small in relation to the much larger and highly significant (P <0.001) differences in both shear strength and penetration resistance between the two cultivation treatments (Fig. 8.9). The reduced cultivated plots were considerably more consolidated at 5 cm depth, and below, than the traditionally cultivated plots. Hence differences in measured values under the tramlines of the E-RC treatment would have to be proportionally larger to show a significant effect.

8.7 Discussion

8.7.1 Cultivation effects

Cultivation method and timing effects on runoff and subsequent entrainment of soil particles and associated P were site specific and variable between years. Although the effects of precipitation during the critical months of October and November as observed by Boardman (2001) were not always captured, there was still sufficient precipitation to allow comparison of treatment effects on SS and P mobilisation in surface runoff in both years. At Pewsey, the inherent vulnerability of the fine sandy soil to surface sealing when there was little surface cover was the main factor causing increased runoff and erosion. In the first year, surface sealing occurred when heavy rain fell within two weeks of drilling, and there was little the farmer could have done to prevent this using traditional ploughing. However, in 2003/04, it did not rain appreciably until after the late-drilled treatments were established, and under these circumstances, the beneficial effects of early drilling were considerable. Late cultivation increased runoff 5-fold and increased SS and P mobilisation by an order of magnitude.

These data provide new quantitative evidence of the adverse effects of surface sealing and late drilling on runoff, and the resulting impacts on the mobilisation of SS and P. The results show that early-drilling as a management technique to reduce the risk of soil and P loss may not always be successful on this vulnerable Greensand soil in years with wet autumns. Similarly the adverse consequences of late-drilling on runoff, SS and P loss risk may only occur in some years depending on rainfall patterns. However early drilling can clearly be an extremely effective management technique for reducing runoff and erosion and has other benefits for crop yield. Its adoption in combination with other erosion control measures such as a targeted changes in land use is therefore recommended as part of catchment management (Evans and Boardman, 2003).

The inherent vulnerability of the Pewsey soil to structural degradation from raindrop impact was generally absent at the other two sites due to the more stable soil structure resulting from their higher clay, organic matter and/or calcium carbonate contents. For example, the heavy rain that fell after the establishment of the early-drilled treatments in December 2002/03 did not increase runoff rates on the steeply sloping chalkland site at Wilton, even though this soil has a high silt content. Any surface sealing that occurred on the Chalk soil tended to disintegrate between storm events. In the second year, late cultivation did increase runoff rates by causing soil compaction on the L-RC treatment, especially in the tramline, which acted as an efficient conduit for the runoff that was generated. Similar observations of tramline compaction generating runoff and erosion were recorded by Robinson and Boardman (1988) in the South Downs. Hence, the key factors influencing runoff rates at Wilton were timeliness of cultivation and the presence of tramlines running up and down the slope, which concentrated runoff.

The comparatively low runoff volumes and associated loads of SS and P measured at East Knoyle were surprising in view of the heavy texture of the soil. Recent subsoiling would have helped to maintain good infiltration to tile drains, but drilling of the crop across the slope may also have helped to reduce runoff, SS and P losses. Significantly greater runoff occurred on the headland where the soil was cultivated up and down the slope. Since soil physical measurements did not indicate that the headland was any more compacted than the other treatments, direction of drilling was probably the main factor influencing runoff rates despite the slight slope. Greater runoff actually reduced SS mobilisation on the headland, reflecting the more stable soil structure associated with this well-fissured clay soil with long-term grass in rotation. Greater runoff has not always led to increased erosion, because of the lack of loose soil to become entrained (Robinson and Naghizadeh, 1992; Martin, 1999; Chambers et al., 2000).

The consistent beneficial effects of non-inversion cultivation methods in reducing runoff, and/or SS and P mobilisation under heavy rain at Pewsey and Wilton support earlier work as reviewed by Carter (1998). These effects are due to the greater crop cover associated with reduced tillage and were obtained despite evidence from a range of soil physical measurements that the reduced-cultivated soils were structurally more consolidated. In comparing drilling and ploughing techniques on the South Downs chalk soils, Robinson and Boardman (1988) found that soil erosion was not related to differences in soil consolidation or soil moisture. Limited soil moisture measurements (data not shown) in our study indicated that reduced cultivated soils initially contained more moisture than ploughed soils. Greater bulk density and lower total porosity in reduced-cultivated soils can be offset by a greater continuity of medium sized pores and a firmer surface for trafficking (Hill et al., 1985). This was clearly apparent in the deeper cross-sectional profiles of the tramlines on the ploughed soil compared to the reduced cultivated soil at Pewsey in the first year. Hence the beneficial environmental effects of reduced cultivation were due to better crop cover and improved resilience to trafficking after drilling. The lack of any effect of reduced cultivation on the heavier clay soil at East Knoyle soil, despite its suitability to this cultivation technique, may have been related to the lack of difference in surface cover. Maize is a widely spaced crop that leaves single coarse stems and therefore did not provide a good crop residue cover.

Contrary to previous work, there was no indication of higher dissolved P concentration under reduced cultivation. This probably reflected the short-term nature of this study and the lack of P fertilisation during the study periods. Since runoff would have remained in the tank for the duration of the storm event, dissolved P concentration may have equilibrated with P sorbing surfaces on the SS rather than on the soil in situ. Dissolved P would have probably decreased when finer clay particles with large P sorption capacity were preferentially transported.

8.7.2 Tramline effects

The soil at Pewsey is vulnerable to erosion and fans of deposited soil are frequently seen at the bottom of tramlines where the hillslopes level out into the riparian zone, or which connect to roads and tracks (see Plate 3.3). The observations and measurements presented support earlier studies showing that tramlines running up-and-down the slope can increase volumes of overland flow and/or the entrainment of soil particles and associated P transfer (Basher and Ross 2001). In this study, runoff volumes were increased by only small amounts (1-2 mm) where tramlines were present but this was sufficient to increase SS and TP export up to 5-fold and 4-fold larger, respectively. Earlier initiation of runoff, the lack of any crop cover to protect the soil and the channelling effect created by the depth and pattern of indentation left by the lugs of the tractor wheel were key contributory factors. These factors were also identified by Fullen and Reed (1979) in studies on erosion-prone sandy soils in the West Midlands.

However, as noted by Robinson and Naghizadeh (1992) in a similar experiment on chalk soils, there was considerable variation in the impact of tramlines depending on the extent to which these key factors were represented. Most obvious in this study was the significant difference in tramline effects between the traditionally cultivated and reduced cultivated treatments in the first year, and the lack of such differences between cultivation type in the second year. In the first year it was noted that the tramlines were shallower and smoother on the reduced cultivated treatment, where the topsoil structure was much more consolidated than on the ploughed treatment. A looser ploughed soil has a greater vulnerability to the force exerted by the tractor tyre and hence the lugs of the tractor tyres caused a deeper, more irregular and slightly more compacted surface than on the more consolidated reduced cultivated treatment. The greater depression left under ploughed soil may also have been accentuated by wet weather that occurred soon after drilling in autumn 2002.

The greater cross-sectional area of the tramline on the ploughed area in the first year therefore acted as an effective channel for increased mobilisation of SS and P as runoff was initiated. Robinson and Naghizadeh (1992) also found higher rates of SS mobilisation from tramlines on ploughed as compared to shallow cultivated soil, and suggested this occurred when the tyre treads or lugs broke up the soil rather than simply compressed the soil. However, contrary to the findings here, these authors found that runoff was less on ploughed plots where the soil in the tramlines had broken up, and suggested that this was due to ponding of runoff water in the depressions left by the tractor treads allowing greater time for infiltration. This difference may be related to the direction of the tyre treads which pointed in a downward $\vee$ shape in this study, and also to the tendency for the lug depressions to fill up with silt detached by splash erosion during rainfall. The depressions caused by the tyre lugs were therefore directing water down the tramline and once in the tramline the water could not escape to either side. The markedly increased soil strength on the reduced cultivated treatments did not cause increased runoff. This suggests that the channelling effect of the tramlines and the greater susceptibility to aggregate breakdown on the looser ploughed soil were much more important contributory factors than soil compaction in increasing runoff generation.

In the second year the ploughed soil was not deeply incised by the tramline, which was not visibly different to the tramline channel left in the reduced cultivated plot. Indeed, in contrast to the first year, the farmer had to make 2 or 3 tractor passes to establish a tramline on both cultivation treatments after early drilling when soil conditions were much drier. Hence, although there was slightly more runoff generated on the tramlined areas (+35% on both early-drilled treatments), flow-weighted SS and P concentrations tended to be lower than on non-tramlined areas. Dilution of SS concentrations with increasing runoff in tramlines was also noted by Chambers et al. (2000) in their erosion surveys, and by Robinson and Naghizadeh (1992) where the soil surface of the tramline was not greatly disturbed and there was less loose soil material available for entrainment. Enhanced stability of the soil surface in a dry autumn will also have helped to maintain infiltration rates and reduce any channelling effect. In contrast, on the late-drilled treatments which became severely capped after heavy rain, much larger amounts of soil and P (2.5 fold greater) were mobilised from the tramlines despite small differences in runoff, reflecting the lack of crop cover in the tramlines and high runoff velocities. These data strongly suggest that the timing of tramline establishment in relation to antecedent soil moisture conditions can have an important influence on the risk of increased runoff and erosion from these soils.

A number of management techniques to reduce the risk of increased runoff and erosion along tramlines have been suggested (Ministry of Agriculture, Fisheries and Food, 1999; Chambers et al. 2000). These include: delaying tramline establishment until >25% crop cover has been established, establishing tramlines across, rather than up-and-down, the hillslope, and drawing a tine behind the tractor wheel to disrupt any compaction caused by the tractor wheel. Although not specifically tested here, the observations on some replicate plots on the E-RC treatment in

2002/03 suggest that providing some crop cover was effective in reducing runoff volumes. Tramline establishment normally involves not drilling in the tramline and reversing this so that the crop is sown, and the tramline established by a tractor pass, is one possible management option. The lack of any accompanying reduction in SS and P concentrations when the tramline has some crop cover may just reflect the more stable nature of the tramline surface under reduced cultivation, as indicated by the tramline profiles. Under a ploughed soil, crop cover in the tramline would be expected to have had an effect on soil entrainment as well as runoff. The offplot plots with drill lines and tramlines running across the field slope produced the same amounts of runoff as the plots with drill lines running up-and-down the slope but without a tramline.

However, these management options are not always practical for the farmer who needs to apply a herbicide to control weeds in early autumn, or on slopes where concerns over safety take priority. In some fields, ponding of water in tramlines running perpendicular to the slope can reach sufficient proportions to cause breakthrough down slope. Nevertheless they will be suitable for adoption at many sites with simple rather than complex slopes. The use of tines behind the tractor wheels was not tested here but was found by Basher and Ross (2005) to be very effective in reducing runoff and erosion on a clay soil in intensive horticultural areas where crops are grown in raised beds. However, Chambers et al. (2000) found that this technique increased erosion on sandy soils because the tine smeared the soil at the depth of operation and the loosened soil became more vulnerable to entrainment by the runoff. Individual field management therefore appears to be a significant factor in whether some options are effective or not.

The data presented here also suggest that an additional important measure to reduce soil and P mobilisation is to avoid establishing tramlines on loose (puffy) seedbeds or in wet conditions that will cause deep channelling and irregular tramline profiles. Ensuring tractor lugs are pointing in an upward $\wedge$ rather than a downward $\vee$ is not a practical option for the farmer who needs to travel in both directions across the field. No one technique is likely to be suitable to all site conditions, and in-field controls may need to be integrated with other control practices which seek to prevent SS and P-laden runoff entering waterbodies directly (Withers and Jarvis 1998). However these results support earlier studies that more sensitive management of tramlines is required on soils vulnerable to runoff and erosion in areas suffering from diffuse pollution, and that such management would be effective in reducing the risk of flooding and diffuse pollution.

8.7.3 Reducing phosphorus transfer

By comparing the effects on TP export obtained by the different BMPs tested at Pewsey in 2003/04 relative to the maximum TP loss obtained on the late-drilled traditionally cultivated treatment (0.55 kg ha^{-1}), it is possible to draw some tentative conclusions as to which of the BMPs is likely to be more effective (Fig. 8.10). Whilst omission of tramlines and adopting reduced tillage individually cut TP loss by over 50%, early drilling was by far the more effective measure reducing TP loss by over 90%.

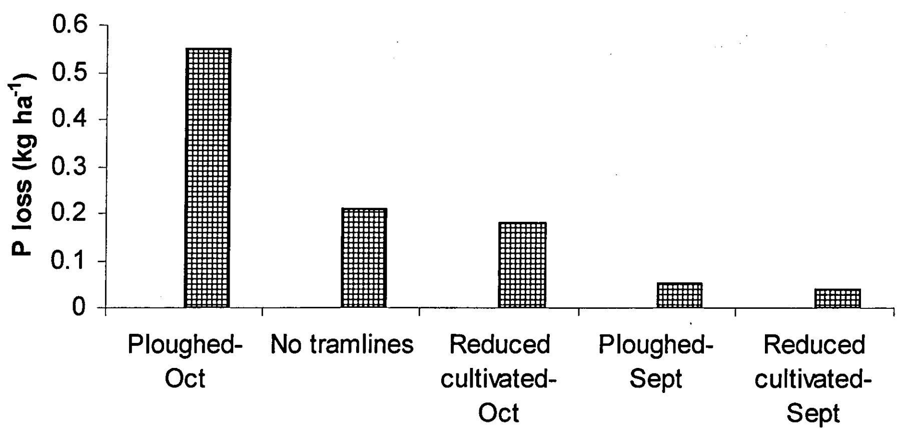


Figure 8.10 Comparison of the relative effectiveness of selected BMPs on the loss of TP in runoff from field plots at Pewsey in 2003/04.

When the crop was drilled early, further adoption of additional measures (reduced-cultivation plus no tramlines) did not make so much of an impact on the TP load (from 0.5 down to 0.2 kg TP ha^{-1}) or the flow-weighted concentration (from 1.4 to 0.9 g L^{-1}). However, since early-drilling will not always be successful in all years, it seems pragmatic to adopt such combinations of measures.

Concentrations of SS mobilised in rainsplash as predicted by the DESPRAL test were slightly lower than those obtained in runoff from the field plots; e.g. for the early-drilled treatments Pewsey: 0.8-1.0 vs 0.9-2.4; East Knoyle: 1.1-1.7 vs 1.5-1.7; Wilton: 1.0 vs 0.2-1.8 mg L^{-1}. The largest differences were on the more steeply sloping sites at Pewsey and Wilton where flow energy may have mobilised more soil particles over the 15 m length of plot during the heavier storms. However, predicted TP concentrations were either slightly lower, or similar, to actual values ; e.g. Pewsey: 1.2-1.3 vs 1.0-1.9; East Knoyle: 1.7-2.0 vs 1.7-2.1; Wilton: 0.8 vs 0.5-1.8 mg L^{-1}. This is also consistent with actual mobilisation of coarser soil particles with lower P content during some storm events.

A comparison of the flow-weighted concentration obtained in the early-drilled traditionally cultivated plots at Pewsey with those obtained in the two Avon tributaries representative of the greensand lithology (East and West Avon) and over the same time period was undertaken by analysis of the relationship between load and flow. This relationship allows potential diffuse sources to be examined in large catchments by assuming that point sources do not have a variable flow component but represent the intercept on the y axis. This comparison suggests there is significant retention of mobilised P on route from the field to the watercourse (Fig. 8.11).

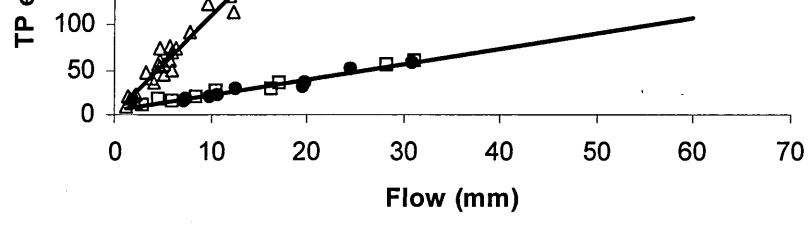


Figure 8.11 Load-flow relationships for P export at the field (Pewsey) and catchment scale (East and West Avon). The difference between the two scales suggest significant retention of P on route from the field to the watercourse.

For each mm of flow, the TP loss measured at the catchment outlet was only 16% of the TP loss measured at the field scale. This suggests a retention of 84% of the TP mobilised at the field scale over the length of the monitoring period. Hence, in relation to these much larger differences in flow-weighted concentrations of TP observed between the field and the catchment scale, the differences in TP concentrations predicted by the DESPRAL test at Pewsey and those obtained in runoff from the field plots were relatively small. This suggests that the DESPRAL test might be used as a means of quantifying the amount of P retention within catchments by comparing the results of the test with measured data after accounting for point sources (Withers et al., 2007).

8.8 Conclusions

Early drilling of crops to quickly establish a crop cover and timeliness of cultivations to avoid compaction were key management techniques to help reduce the potential risk of runoff, erosion and P loss. The greater susceptibility to runoff and erosion when crops were sown late can be minimised by adoption of reduced cultivation. Reduced cultivated soils provided greater crop residue cover, were less susceptible to surface sealing and provided a firmer surface for tractor traffic due to their more consolidated and uniform soil structure. Tramlines were important vectors of runoff causing increased mobilisation of SS and P and need to be managed carefully. The greatest impact of tramlines was obtained where they caused significant indentation of the soil surface after traditional inversion cultivation or where erosion rills developed in the tramline. Smallest losses were obtained when tramlines were established on more consolidated reduced-cultivated soil, or under dry soil conditions, when any increase in runoff generation was compensated for by a decrease in SS entrainment. Establishment of partial crop cover in the tramline, and establishing tramlines across the slope rather than up and down the slope, were also effective in reducing runoff and/or entrainment of SS and P. The data support the adoption of these improved soil management practices in combination on farms in the catchment area.

9

SYNTHESIS, CONCLUSIONS AND FURTHER RESEARCH

9.1 Synthesis

The farming community within the UK is under increasing government pressure to reduce P emissions from agricultural land to water because of the risk of eutrophication, which is becoming a major environmental issue for the developed world. The focus on P is because of the central ròle this nutrient plays in biological productivity and the current emphasis on the farming community relates both to the established link between agricultural intensification and increased P loss in land runoff and the need to control both point and diffuse sources of P in catchments to meet ecological water quality targets. This thesis queries whether this pressure is justified in view of the limted evidence of links between field scale farming activities and ambient P concentrations in draining streams or rivers, the disparity in timing between agricultural P transfers and ecological response and the presence of other sources of P in rural catchments which might be ecologically relevant. Whilst there is much research data confirming a link between soil P, P inputs in fertilisers and manures and soil management on P transfer in runoff from field plots, there is less, or conflicting, evidence of a catchment scale response. This is due to the confounding effects of P storage within terrestrial and aquatic environments, the modifying influence of in-river processes on P cycling, forms and availability and the greater contribution from point sources as catchments increase in size.

9.1.1 Historic v current land management

To overcome these confounding influences, and to help bridge a research gap between field scale activities and catchment response, the effects of farming practices on P transfer in small headwater catchments (<2 km^2) were investigated within the context of upland and lowland farming. In terms of effects on water quality, farming practices can be separated into the impacts of historic land management (i.e. soil P accumulation and soil test P (STP) concentrations) and current land management (i.e. annual inputs of fertilisers and manures, soil cultivations, stocking densities). The comparison of two adjacent upland catchments with variable amounts of improved pasture land at Redesdale showed a direct link between soil P accumulation and increased stream P concentrations in a 3-year study. There was no evidence of a link between current land management (increased sheep stocking densities and fertiliser application on improved land) and stream P concentrations, probably because sheep stocking rates were not sufficiently high over a wide enough area of the catchment and/or a lack of hydrological connectivity (overland flow) between the field and the watercourse at the time of the year the farming actions were carried out.

In the lowland mixed farming catchment at Rosemaund, the hydrological connectivity between fields and the stream is greatly accentuated by the ubiquitous presence of intensive underdrainage systems that were installed when grant aid was available. Previous fingerprinting work in this catchment had confirmed that mobilisation of P at the surface was rapidly transported down cracks and fissures to the field drains and into the stream. The soil at Rosemaund is also highly dispersive and runoff and streams become very turbid during heavy storms. Under these conditions, and in three study years with above average rainfall, both soil cultivations and fertiliser/manure applications greatly increased the concentrations of particulate P (cultivations) and dissolved P (fertilisers and manures) either in a monitored field drain (Foxbridge) or in the stream (Jubilee). However, in most instances, these increased P concentrations were observed in only small amounts of stormwater and outside of the main land drainage period (December – April). Only when large amounts of manure were applied to the fields in the second year was there any evidence of carryover into the main drainage period. This disparity in the timing of farming activity (i.e. autumn cultivations and fertiliser applications prior to sowing), the main period of drainflow (i.e. winter) and the main period of biological activity (spring and summer) is therefore a key factor governing the potential contribution of agricultural activities to eutrophication risk.

9.1.2 Impacts of soil phosphorus

At both Redesdale and Rosemaund, the main source of P for transport during hydrological events was the soil, and the majority of the P transported was in particulate P (PP) form rather than dissolved reactive P (DRP) or dissolved unreactive P (DUP), as has been observed in many catchment studies at this small scale (Chapter 2). Soil P status reflects the impacts of historic land management and soil P release to runoff water is governed both by the particle size and the P sorption characteristics of the particles being transported together with any desorption of DRP/DUP from the soil *in-situ*. Monitoring of the Holbach field plots under natural rainfall at Rosemaund provided an opportunity to investigate the impact of soil P status on P enrichment of runoff water and in particular the PP fraction. Soil P had a smaller influence on the particulate P fraction than the dissolved P fraction in the runoff due to the preferential adsorption of previously added P onto the finer soil particles with larger surface areas for P sorption. When soil P fertility was optimum of crop production (Olsen-P (OP) of ca. 20 mg L^{-1}, Ministry of Agriculture, Fisheries and Food, 2000), concentrations of DRP in runoff were also more than double those proposed to limit eutrophication impacts in flowing waters (0.1 mg L^{-1}, Duncan et al., 2006) due to DRP exchange with the transported SS. Concentrations increased further when soil OP rose above 50 mg L^{-1}. Similar effects were noted at the Avon demonstration sites; for example, DRP concentrations in plot runoff at Wilton were frequently above 0.1 mg L^{-1} despite the soil being P deficient.

The results at Rosemaund suggest that the P content of the SS in field runoff will be almost as great on a soil with average soil P fertility (the lowest soil OP concentration tested on the plots was 19 mg kg^{-1}, P Index 2) as on a soil with a much greater soil P fertility (e.g. P index 4+). The average SS-P content across all the plots at Rosemaund was ca. 1000 mg kg^{-1} and varied from 768 mg kg^{-1} on the plot with the lowest soil P to 1277 mg kg^{-1} on the plot with the largest soil P (P Index 8). This average value is identical to that observed in the drainflow and in the stream at Rosemaund. The estimated SS-P concentration in runoff from improved pasture at Redesdale (P Index 2) was similar to that observed in the stream (ca. 1400 mg kg^{-1}) and close to the values recorded at Rosemaund. Similarly, the calculated PP concentrations for the field plot runoff collected at the three farmer demonstration sites in the Avon catchment (Pewsey, East Knoyle and Wilton) were ca. 1000 mg kg^{-1} except at Pewsey where values were lower due to the presence of sand particles in the runoff. The P fertility of these soils ranged from deficient (P Index 0) at Wilton to high (P index 4) at East Knoyle.

These data suggest a relative constancy in PP concentrations entering streams in sheet runoff from agricultural land, which has been observed in a number of catchment studies (Johnson et al., 1976, Probst, 1985; Owens and Walling, 2002; Nemery et al., 2005). This reflects the selective partitioning of soil primary particles and aggregates during the erosion and runoff process. The particle size distribution of the particles transported therefore has a much greater influence on PP content than does soil P status and the main impact of soil P accumulation on runoff P at Rosemaund was therefore on the 'dissolved P' fraction. The particle size distribution of SS in runoff collected at the plot scale (d_{50} <10 μm) was identical to that observed in drainflow and in the streams at Rosemaund and within the rivers in the larger Avon catchment. This suggests that any deposition of coarser particles during transport had already occurred within the 15 m plot length and any further retention of SS during transport would be dependent on the presence of flow-retarding features in the landscape such as hedges or ponds.

The PP fraction may nevertheless play a role in eutrophication where there is a release of P on entering the water column due to a concentration gradient between the equilibrium P concentration (EPC_o) of the SS transported and the ambient P concentration of the stream, or where in-stream processes modify the P sorption characteristics of the SS during transport downstream. As observed in previous studies, SS in runoff generally has a greater P sorption capacity (Q_{max}) than the original soil due to the selective enrichment of fine clay particles (Sharpley, 1980, Scalenghe et al., 2007). In this study, the increase in Q_{max} was the same as that found by Sharpley (1980), 56%. However, the % saturation of the P sorption capacity of runoff SS was always greater than that of the soil until the OP content of the soil exceeded about 110 mg kg^{-1}. The results from the rapid runoff experiment showed that this increase in EPC_0 at low OP could not be due to adsorption of DRP released from the soil *in-situ* (only 0.1 mg L^{-1} at OP of 20 mg kg^{-1}), and must therefore reflect the percent P saturation of the fine particles transported. Consequently, both EPC_o and water-extractable P (WEP) concentrations in the SS were still larger on the plots with the lowest soil P fertility (ca. 2 mg L^{-1}) than the ambient concentrations of DRP in the Rosemaund stream (0.25 mg L^{-1}) suggesting considerable potential for P release.

Contrary to the observations by Jarvie et al. (2005) and others, who found river bed sediments were a P sink rather than a P source, the Rosemaund plot data suggest SS in hillslope runoff is likely to be a P source on entering the water column. These differences may relate to the transformations that river bed sediments undergo during deposition and colonisation by microbial and algal communities (House et al., 1995) but may also reflect that river bed sediments may already have released their P store during transport and prior to deposition. The concentrations of EPC_0 measured in river bed sediments are an order of magnitude lower than those measured in the Rosemaund runoff and other hillslope-derived SS (e.g. Pionki and Kunishi, 1992). However, it is not clear to what extent the large EPC_0 concentrations measured in plot runoff are an artefact of the sample collection and drying process and further work is required to test these relationships under moist conditions that are more representative of *in-situ* soil, storm runoff solution to soil ratios and stream environments.

9.1.3 Ecological relevance

Soil P transfers in runoff from the farmed land at Rosemaund are therefore of potential ecological significance but they arrive in the stream during winter storm events when biological activity is low. The temporal variation in P transfer is therefore dictated by the distribution of hydrologically effective rainfall over the year both in terms of amount and intensity. Transfers of P during the ecologically sensitive spring and summer periods were generally absent at Rosemaund, except for those years when drains continued to flow during April and to some extent in May due to high spring rainfall. Some high P concentrations were observed in small amounts of stormwater from the Foxbridge drain during summer when the Rosemaund soils crack readily, although this was not related to any farm application or operation. Direct contamination of water by dirty water applications percolating down cracked soils in summer has been observed by others (Williams and Nicholson, 1995), but this was not the source of the P at Rosemaund. Runoff from cultivated and pasture land may therefore not be as ecologically relevant as other potential sources of P in rural catchments which are delivered to the stream more continuously, such as discharges from sewage treatment works (STW), discharges associated with septic tanks and runoff generated from impervious surfaces associated with urbanisation. The dominating effect of even a small farm STW at Rosemaund on the temporal variation in P concentrations during the year was clearly demonstrated by the characteristically high DRP concentrations measured at Belmont during summer.

To examine the potential significance of rural P sources other than farmed land, the range in P concentrations and forms in a wide variety of storm runoff samples collected from various sites in three catchments and their associated streams were investigated. The scale of study increased from <2 km^2 at Redesdale and Rosemaund to 7-10 km^2 at Whitchurch, Dinedor and

Kivernoll. Although the streams were still headwaters, the catchments included a greater number of domestic dwellings, farms and roads associated with village communities. The variability in concentrations between sites and sampling dates for all sources was considerable and the relative abundance of P sources associated with farmyards, roads and septic tank discharges directly entering the streams was surprising in these catchments. In many cases it was not clear where the multitude of drain discharges originated from, whilst in other cases it became clear that discharges were associated with a combination of sources rather than a single source. Septic tank discharges were detected in at least two of the sites and detergent contamination was suspected in discharges from some farmyards and roads. Runoff from impervious surfaces was more concentrated in P (both DRP and PP) than surface and sub-surface runoff from farmed land and would be generated all year round whenever it rained, including the ecologically sensitive summer period. On the basis of these observations, the urbanisation of these small rural catchments was clearly a major source of P inputs to their streams.

Stream monitoring data for the different catchments studied in this thesis also showed a clear switch from a domination by particulate P forms at Redesdale and Rosemaund to domination by dissolved P forms at Whitchurch, Dinedor and Kivernoll. This apparent switch maybe partly due to the differences in sampling frequency adopted; at Redesdale and Rosemaund the sampling was storm orientated with fewer baseflow samples taken, whilst sampling in the other three catchments was on a weekly basis supplemented by occasional storm event sampling. The latter sampling programme will tend to underestimate the PP contribution from farmed land. However, this difference will also be due to the greater number of P concentrated sources entering the stream in the larger catchments as discussed above. A dominance of DRP forms associated with a greater distribution of point source inputs in large catchments has been widely observed elsewhere.

9.1.4 Implications for catchment management

The results of the data evaluation undertaken in this thesis have important implications for catchment appraisal and management under UK climatic conditions. In terms of agricultural P transfers, the majority of hydrologically effective rainfall is distributed over the winter months and clearly any field operations which coincide with a rapid hydrological connectivity with the water course must be avoided. These operations include fertiliser and manure applications to wet or frozen soils over winter, harvesting of root and forage crops in riparian fields leaving bare compacted soils in early winter, late drilling of winter cereal crops and high stocking densities over winter. As shown in this study, cultivations and fertiliser/manure applications during autumn may cause high P concentrations in a small amount of runoff water but hydrological connectivity is more limited and stream ecological response is likely to be low. Applications of P in late spring and summer are similarly less likely to reach the watercourse, although if applied in liquid form to cracked soils may cause leakage of large P concentrations during an ecologically sensitive period. These 'incidental' P transfers can therefore be largely overcome by careful timing of field operations in combination with additional storage facilities for livestock manures where this is required.

The importance of soil P as the main source of P transfer within the Redesdale and Rosemaund catchments suggests mitigation of agricultural P emissions should be directed at both long-term nutrient management to reduce the risk of dissolved P (<0.45 μm) transfer and soil management to reduce susceptibility to soil dispersion and particulate P transfer. At Redesdale, DRP concentrations were still below the 0.1 mg L^{-1} threshold for good ecological quality even though nearly 50% of the catchment had been improved, whilst DRP concentrations in field runoff at Rosemaund were still above this threshold at the lowest soil OP concentrations tested and required for optimum crop growth. This suggests that nutrient management goals must be site and catchment specific and will vary depending on soil and PP release characteristics. At Rosemaund, it is unlikely that concentrations of DRP lower than 0.1 mg L^{-1} can be achieved without long-term depletion of soil P reserves or addition of immobilising agents to increase the soil P sorption capacity.

The lack of sensitivity of PP transfer to increasing soil P fertility at Rosemaund also suggests that control of PP in land runoff should focus on reducing SS transfer rather than the P content of the SS. Reducing SS transfer requires measures to minimise the inherent susceptibility of soil particles to rainsplash detachment/dispersion and reduce the opportunity for rainsplash interaction with the soil by providing crop cover. The results of the DESPRAL analysis in the Wye catchments (Whitchurch, Dinedor and Kivernoll) hint at the importance of organic matter

(OM) in soil structural stability (e.g. Greenland et al., 1975). When OM concentrations fell below 30 g kg^{-1}, the vulnerability to particle dispersion increased dramatically, although there were relatively few grassland fields included in this analysis and the impact of changes in bulk density was not investigated. However, this threshold is very similar to the critical level of OM discussed by Loveland and Webb (2003) which they argue is based on anecdotal evidence. The greater susceptibility of the Kivernoll soils to dispersion may also be related to their greater soil P fertility based on regular application of poultry manure, as found by Celi et al. (1999).

The demonstration plots established by the Environment Agency (EA) provided the opportunity to examine some cultivation and crop management options that have not been previously investigated in combination; namely early sowing in combination with either ploughing, or reduced cultivation, and removal of tramlines. All the options tested provided some degree of control although providing a crop cover as quickly as possible by early drilling was the most effective option. Since early drilling will not always be effective due to the distribution of autumn rainfall, the results suggest a combination of measures is required. All of the options tested were of relatively low cost and could be adopted on most farms, although the removal of tramlines raises significant practical difficulties relating to weed, pest and disease control that reduce its acceptability to farmers. What is clear is that the effects of mitigation options are likely to be very site specific depending on the particular combination of environmental factors and land management factors operating.

The comparison of field runoff P concentrations with other potential sources of P in catchment runoff or in direct discharges to the stream raises a number of points for catchment management. Firstly, it must not be assumed that cultivated and pasture land are the only sources of P in small rural catchments with no major point sources, as is commonly assumed in source apportionment budgets (e.g. Johnes, 1996). In many cases, septic tank effluent and runoff from farmyards and roads are more concentrated P sources, they have a higher proportion of their TP content in a dissolved and therefore highly bioavailable form and they are delivered to the watercourse more continuously than runoff from farmed land. Hence, the second point is that these sources with compositional and hydrological properties intermediate between point and non-point sources (Edwards and Withers, 2008) are therefore potentially more ecologically relevant than non-point source runoff from cultivated and pasture land and a greater priority in terms of control. Thirdly, the routing of these multiple P sources to the watercourse is complex which makes the effective targeting of control measures very difficult without local knowledge.

This strongly suggests that catchment management must incorporate a wider collective social responsibility for the multiple sources of P that enter rural streams. Targeting just the agricultural sector for causing downstream eutrophication when other sources in rural headwaters may be more important will not achieve the desired goals and may cause alienation and social upheaval. Actions are urgently needed to control the 'intermediate' sources which may require soft 'end-of-pipe' engineering solutions rather than tackling sources or a combination of both. Strategically placed wetlands have been introduced to tackle highway runoff but personal observations within the Avon catchment alone suggest these are not maintained and largely neglected such that their retention capacity soon becomes very limited and they ultimately become a source rather than a sink for downstream eutrophication.

Septic tanks would appear to be a particular problem. Whilst it is recognised that any risk of P loss via discharge from septic tanks can be very effectively controlled by installation design and adequate maintenance, regulation guidelines are often circumvented for expediency and/or tanks are not emptied regularly enough to ensure their continued effectiveness (Butler and Payne, 1995; Gold and Sims, 2001). They also have a limited lifespan yet replacements are seldom installed and many systems are now probably too old to remain effective. With an expanding population, solutions to these issues need to be identified and be supported by investment into the infrastructure of our rural communities. As pointed out by Malmqvist and Rundle (2000), urban expansion is probably the greatest threat to our rivers and this thesis also suggests that this urbanisation is probably more important in terms of water quality than the way the land is farmed.

9.2 Conclusions

1. Investigation of the spatial and temporal variation in P concentrations and loads in streams draining an upland (Redesdale) and a lowland (Rosemaund) agricultural

catchment showed that historic land management (i.e. soil P accumulation and frequency of cultivation affecting OM levels) was a more important factor controlling P transfer than current land management (i.e. annual inputs of fertilisers and manures and cultivation) due to the disparity in the timing of annual farm operations in relation to the main periods of maximum runoff and ecological sensitivity.

2. The majority of the P transfer in both catchments was in particulate form due to the mobilisation and transfer of P-enriched soil particles during winter storm events. Since current standards for good ecological status in flowing waters are currently based on concentrations of reactive P rather than total P, up to 50% of the catchment land area at Redesdale could be agriculturally improved and still meet an acceptable water quality standard. In contrast at Rosemaund, it is unlikely that the ecological threshold can be achieved under current best management without increasing the soils inherent P sorption capacity.

3. Increasing soil P fertility had a greater impact on dissolved (<0.45 µm) P concentrations in hillslope runoff than particulate (>0.45 µm) P concentrations. The lack of sensitivity of the PP fraction to increasing soil P reflected the selective adsorption of added P onto very fine soil particles (<10 µm) and their subsequent mobilisation and sorting during rainsplash and runoff. The particle size of the soil particles mobilised and transported in runoff therefore have a greater impact on PP concentrations than the amount of P present in the soil and mitigation actions to reduce PP transfer in field runoff should be focused on minimising SS transfer rather then the P content of the SS.

4. The P content of soil particles transported in sheet runoff from hillslope plots (15 m long) of varying soil P status were relatively constant (ca. 0.1%) and almost identical to that observed in their draining streams at catchment scales of <2 km^2. The particle size of the transported SS at this plot scale was also identical to that in SS entering streams. The DESPRAL laboratory test which quantifies the P in clay and fine silt particles and aggregates (<20 µm) provided a reasonable estimate of the potential mobilisation of P in hillslope runoff but overestimated the mobilisation of SS due to particle deposition along the plots. The P release properties of the SS in field plot runoff suggested that entrained particles would release P on entry into the stream at Rosemaund.

5. The results of the DESPRAL test suggested that raising soil OM concentrations above 30 g kg^{-1} would be an effective measure to reduce the inherent susceptibility of agricultural

soils to particle dispersion during storm events. Evaluation of three best management practices to reduce SS and P transfer in fields with different soil types in the Hampshire Avon catchment showed that the effectiveness of these options are likely to be site specific depending on weather, inherent soil susceptibility to structural degradation and management. Early drilling, timeliness of cultivation to avoid soil compaction, better tramline management and reduced cultivation techniques would all help reduce agriculture's impact on water quality in the catchment area and reliance on one option may not be effective in all years.

6. Surface and sub-surface runoff from agricultural land is not the only source of P entering streams in small rural catchments with some degree of urbanisation. Farmyard runoff, road runoff, and overflow discharges from septic tanks were all more concentrated sources of P than runoff from farmed land. The more continuous delivery of P in runoff from impervious surfaces and/or in direct piped discharge of wastewater to the stream during the ecologically sensitive summer period suggests these sources need to be urgently controlled in order to achieve the water quality goals required by the EU Water Framework Directive by 2015. Only a small amount of urbanisation would therefore appear to be needed to pose a serious threat to eutrophication, and catchment management strategies to control P transfer should include collective actions by the general public in addition to the farming community.

9.3 Suggestions for further research

1. The finding that PP in field runoff is relatively insensitive to soil P accumulation at the field scale needs to tested on other soil types and should include more deficient and low P soils. There was limited opportunity to evaluate how soil P concentrations affect the potential bioavailability of runoff PP and this aspect also requires further investigation on a wider range of soil types.
2. Phosphorus sorption studies suggested that SS in field runoff would release P on entering the watercourse but these data were probably influenced by the method of collection and subsequent drying. There is an important distinction between soil scientists

who tend to work on dried samples and hydrochemists who prefer to work under more natural conditions. Further work is required to bridge this gap.

3. Suspended solids play an essential role in regulating P availability within the aquatic environment and further work is required to understand the transformations of PP that occur during transport from the field to the watercourse including downstream transport, and the main controlling factors affecting these transformations. This research should help understand better the role of PP in stream ecology and to what extent PP inputs from agricultural land will continue to cause eutrophication after catchment control measures have been implemented.
4. The survey of potential P sources in rural catchments was only preliminary but suggested that the abundance and relative significance of 'intermediate' sources needs to be investigated in a wider range of catchments. Development work is urgently needed on measures to control these source P inputs.
5. Consideration should be given to establishing demonstration catchments where the combined targeting of agricultural, intermediate and point source control options can be evaluated and the lessons learnt transferred to other catchments. These catchments should include both upland and lowland environments.

10

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Appendix 1

METHODOLOGY FOR QUANTIFYING THE UK PHOSPHORUS CYCLE

The imports, exports and internal recycling of P within UK agriculture in 1993 were estimated to the nearest 1000 t using national census statistics, surveys of feed, fertiliser, livestock manure and sewage sludge use on farms, production levels of different agricultural crops and commodities, and a knowledge of their average P content. This approach is similar to that adopted in 1973 by Centre for Agricultural Strategy (1978), but excludes domestic and industrial P use, and any estimate of national P loss from agriculture to water. The year 1993 was chosen on the basis of general availability of data from a number of different sources. Imports and exports are defined as transfers which enter or leave the agricultural P cycle, as distinct from inputs and outputs which are recycled transfers. The term 'arable' encompases cereals, break crops, fodder crops and horticultural crops. Temporal trends in surplus P were examined over the period 1935-2005 as the difference between fertiliser imports plus manure inputs and the total P removal in grass and arable products, excluding baled straw which is largely returned in farmyard manure.

Imports of feeds and inorganic calcium phosphates for animal consumption were obtained from Ministry of Agriculture, Fisheries and Food (MAFF), Statistics, Branch C (Ministry of Agriculture, Fisheries and Food, pers. com.). The types and amounts of these imports were cross-referenced with data on the usage of raw materials by UK feed compounders (Feed Facts Quarterly, 1995). Concentrations of total P in each feed ingredient were calculated separately based on values given in Ministry of Agriculture, Fisheries and Food (1990). Data on manufactured fertiliser sales were provided by the Fertiliser Manufacturers Association (2001), which agreed well with data from the British Survey of Fertiliser Practice on inorganic P fertiliser use in 1993 (Burnhill et al. ,1997). An estimate of P in sewage sludge applied to land was based on a detailed survey of sludge disposal in 1990/91 (Department of the Environment, 1991), which indicated that c. 0.5 million ha of farmland receives sludge each year of which 44% is anaerobically digested. Average P concentrations in digested and undigested sewage sludge were taken as 16 and 12 kg t^{-1}, respectively (Ministry of Agriculture, Fisheries and Food, 2000). Annual P inputs from the atmosphere were estimated as 0.3 kg ha^{-1}. This average agrees with more recent figures of 0.5 kg ha^{-1} for lowland areas (Withers et al., 1999) and 0.1-0.2 kg ha^{-1} for upland areas (Gibson et al., 1995).

Exports of P in crop and livestock products for human consumption were calculated according to June census statistics on annual production (Ministry of Agriculture, Fisheries and Food, 2001a), and taking typical values for the P content of crops (Ministry of Agriculture, Fisheries and Food 1985; 1990), milk and meat (Agricultural Research Council 1980; 1981). Phosphorus contained in wool (20 t) and in eggs (1400 t) were too small to be itemised separately. Oilseed and linseed crops grown on set-aside land were assumed to be for industrial use only. Estimates of the amounts of P in cereal grain used by the milling and malting industry assumed that 20% of P in wheat and oat grain is removed in the flour and that 10% of the P in barley grain is removed in the malt. These proportions were calculated from the P content of whole grain and of the grain offal which is used in feedstuffs (Ministry of Agriculture, Fisheries and Food, 1990). Similarly, rapeseed oil was assumed to have a P content of 2.5 kg t^{-1} based on the P content of whole seed and rapemeal.

Export of P in liquid milk assumed an average concentration of 0.9 kg P m^{-3} (Withers et al., 1999) and that 50% of skimmed milk production was fed to animals. Export of P in meat was estimated by converting deadweight production to liveweight assuming killing out percentages of 55% for ruminants and 75% for non-ruminants (Centre for Agricultural Strategy, 1978), and taking average P contents in ruminant and non-ruminant liveweight of 8 and 6 kg P t^{-1}, respectively (Agricultural Research Council, 1980; 1981; Tunney, 1990). It was assumed that 80% of body P is contained in the bones (Agricultural Research Council, 1980) and that c. 10% of bone P is returned to agriculture. The majority of bone P and offal is therefore assumed to be used in either manufacturing industry, feeds for domestic pets or sent for refuse. All horticultural produce was assumed to be for human consumption.

Amounts of P recycled in excreta from housed and grazing livestock were calculated according to the numbers of livestock at the MAFF June Census, the amounts of excreta produced each day by different classes of livestock, the number of days housed and taking typical values of the P content of fresh manure (Ministry of Agriculture, Fisheries and Food, 2000, 2001a). These calculations indicated that cattle and calves, sheep and lambs, total pigs and total fowl excrete an average 9.30, 0.99, 2.76 and 0.27 kg P per head, respectively each year. The amounts of P removed from the soil in home-grown crops were calculated from census data on average crop yields and typical values of their P content (Ministry of Agriculture, Fisheries and Food, 1985; 1990; 2001a). The removal of P in cut and grazed grass from grassland < 5 years (temporary grass), > 5 years (permanent grass) and from rough grazing assumed utilized yields of 8, 5 and 2 t DM ha^{-1}, respectively, and herbage P contents of 3.5, 3.0 and 1.5 kg t^{-1}, respectively. These values are similar to those used by Centre for Agricultural Strategy (1978) but assume 10% greater output in 1993 compared to 1973. Offtake of P in cereal crops assumed that straw yields were 65% of grain yield (Withers, 1993), that only barley and oats straw was removed from the field and took account of the proportion of grain used for home-saved seed. Offtake of P in other arable and horticultural crops assumed that all straw was incorporated and that 20% of sugar beet tops were grazed by livestock. The fate of cereal straw is considered an oversimplification but straw contains very little P (Withers, 1993).

The total amount of P in home-grown feedstuffs (arable crops and grassland) recycled to livestock was calculated as the difference between (a) the exports of P in milk and meat plus the P excreted by livestock and (b) the imports of P in purchased feedstuffs and minerals. This calculation assumed that there is no significant net gain or loss of body P in UK livestock on an annual basis and suggested total P amounts fed to UK livestock of 259,000 t in 1993. A check on this figure can be obtained by reference to the amounts of purchased feeding stuffs and minerals used in the UK. These were estimated at 16.8 million t in 1993, of which 11.5 million ts were compound concentrates, 4.6 million t in straight concentrates and 0.7 million t equivalent in low-energy feeds (Ministry of Agriculture, Fisheries and Food, 2001b). Taking their typical P contents as 7, 3 and 1 kg t^{-1} respectively, and assuming that dairy cows receive 2.5 kg of supplementary mineral P as dicalcium phosphate per day (UKASTA, pers comm), total P fed in purchased feedstuff is 95,000 t. According to MAFF census data, about 20% of this figure is additionally fed from on-farm cereal supplies and, including P in cut and grazed herbage (139,000 t), total P ingested by livestock in 1993 was estimated as 253,000 t. Similarly, after taking account of UK imported P, and the P exported in crop products for human consumption, P offtake in arable crops and grassland was calculated as 222,000 t. This compares with a value of 224,000 t calculated from the removal of P from individual crops. All values are therefore within 5% of each other.

These estimates are by necessity only approximate because of the limitations imposed by the MAFF agricultural census data, the uncertainty over the precise fate of crop and livestock products and the inaccuracies associated with taking typical P contents of different commodities, especially livestock manures whose P content can vary widely. For example, the running total of animal numbers during the year may be different to those counted on the one census day in June. On a national basis, where figures are reported to the nearest 1000 t, these uncertainties are likely to be small and the data is considered to give an adequate indication of the national P cycling and surplus in 1993.