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**NATIONAL UNIVERSITY OF IRELAND, GALWAY**

**THE ERODIBILITY AND PHOSPHORUS LOSS  
POTENTIAL OF A SELECTION OF IRISH  
TILLAGE SOILS**

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Thesis submitted in fulfilment of the requirements for the degree of Doctor of  
Philosophy.

August 2012

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*For my parents*

“Tell me and I'll forget; show me and I may remember; involve me and I'll understand.”

Chinese proverb

## ABSTRACT

Arable cropping, due to its intensive nature, can leave soil with reduced ground cover and impaired soil structure, making it vulnerable to erosion under heavy rainfall. Runoff containing suspended sediment (SS) and nutrients, particularly phosphorus (P), from agricultural fields is considered to be one of the main causes of water quality impairment. To date, in Ireland, no study has investigated erosion and associated P loss from tillage soils when subjected to high intensity rainfall, even though international research indicates significantly higher P export coefficients from this land use than from grassland. As a result, only agronomic nutrient advice is available and this has been adopted into current legislation. Research was therefore necessary to assess the potential P losses arising from complying with the legislation. This objective was addressed in the first part of the study using two simulated rainfall experiments. A related objective involving the development of a screening methodology to identify tillage fields with erosion risk and soil quality problems was addressed in the second part.

The aim of the first part of the study was to quantify the amount of dissolved reactive phosphorus (DRP), total phosphorus (TP), particulate phosphorus (PP) and SS released in runoff from five tillage soils with varying soil test P (STP) when subjected to a rainfall intensity of  $30 \text{ mm hr}^{-1}$  applied in three successive events. Soil physical and chemical parameters, slope, and time interval between storm events were ranked in order of importance for the prediction of P and SS releases, and a runoff dissolved phosphorus risk indicator (RDPRI) was developed to identify the STP for Irish tillage soils above which there may be a potential threat to surface water quality. The effect of soil type on the flow weighted mean concentration (FWMC) of DRP ( $p = 0.013$ ) depended on both slope and time between rainfall events. The effect of soil type depended only on surface slope for the FWMCs of SS ( $p = 0.044$ ), TP ( $p = 0.014$ ) and PP ( $p = 0.022$ ) in surface runoff. Increasing the overland flow rate over the soil surface in the presence of rainfall had the effect of increasing the concentrations of SS, PP and TP (but not DRP) in surface runoff ( $p < 0.05$ ) across all soils. An increase in extractable soil P resulted in an increase in concentrations of DRP in surface runoff ( $p < 0.05$ ) across all soils. Of the five methods used to extract soil P in these experiments, water extractable P (WEP) was identified as having the greatest potential to be used as an indicator of the risk of P movement from soil into runoff water. However, despite its apparent advantage over Morgan's

Phosphorus ( $P_m$ ) in determining environmental risk, it would appear to be impractical and costly to run two soil P tests side by side given that  $P_m$  gives a good approximation for both agronomic and environmental purposes.

Combining the results of both experiments (rainfall only and rainfall and overland flow) indicated that if the current agronomic optimum  $P_m$  range for tillage soils of 6.1 - 10 mg  $L^{-1}$  is maintained by tillage farmers through adherence to recommended P application rates, then the risk of runoff with DRP concentrations in excess of the level at which eutrophication is likely to occur (0.03 mg molybdate reactive phosphorus (MRP)  $L^{-1}$ ) should be minimal.

Ireland has a valuable resource in terms of its land and soil quality, and promoting sustainable soil management is one of the areas of action included in Food Harvest 2020, the national strategy for the development of the agri-food sector. Agricultural activities that can negatively impact on soil quality must be tackled if Ireland is to meet the ambitious growth targets set out in this vision. Therefore, in the second part of the study, the five soils above, and a sixth soil, were assessed in their natural field conditions to determine their erosion risk and soil quality status. At each study site, detailed soil classification results and visual soil assessments were used in conjunction with observed erosion levels to select the most appropriate indicators for assessing erosion risk and soil quality status. Parameters selected include: texture, slope, erosion features, structure, ponding, potential rooting depth, soil organic matter (SOM), average annual rainfall and current land use. Assessment of these indicators at each study site using the user friendly grading system developed here, made it possible to correctly identify the sites where the erosion risk was high. This work showed that adoption of an erosion and soil quality screening method by tillage farmers and advisory specialists in Ireland will enable the quantification of the extent of erosion risk and soil quality degradation on farms. This will allow remediation measures to be prioritised for the most vulnerable sites, which is likely to result in cost and resource savings for farmers and advisory services.

## **DECLARATION**

This dissertation is the result of my own work, except where explicit reference is made to the work of others, and has not been submitted for another qualification to this or any other university.

John Regan



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## ABBREVIATIONS

AAR	Average annual rainfall
ACP	Agricultural catchments programme
AgSt	Aggregate stability
AIC	Aikake information criterion
Al	Aluminium
Al <sub>ox</sub>	Ammonium oxalate-oxalic acid extractable aluminium
Be	Beryllium
C	Carbon
Ca	Calcium
CaCO <sub>3</sub>	Calcium carbonate
CEC	Cation exchange capacity
CORINE	CO-oRdination of INformation on the Environment
CO <sub>2</sub>	Carbon dioxide
CSA	Critical source area
Cs	Caesium
CSO	Central Statistics Office
DAFF	Department of Agriculture, Fisheries and Food
Defra	Department for Environment, Food and Rural Affairs
DEHLG	Department of the Environment, Heritage and Local Government
DRP	Dissolved reactive phosphorus
EEA	European Environment Agency
EPA	Environmental Protection Agency
EQS	Environmental quality standards
Fe	Iron
Fe <sub>ox</sub>	Ammonium oxalate-oxalic acid extractable iron
FWMC	Flow-weighted mean concentration
GIS	Geographical information systems
GLMM	Generalized linear mixed model
GOPC	Grid orientated P component
ha	Hectare

HSPF	Hydrological simulation program - FORTRAN
ISIS	Irish soil information system
ISO	International organization for standarization
LMM	Linear mixed model
Mg	Magnesium
MRP	Molybdate reactive P
M3-P	Mehlich-3 phosphorus
N	Nitrogen
Na	Sodium
NH <sub>4</sub> -N	Ammonium nitrogen
NO <sub>3</sub> -N	Nitrate
NPRP	National phosphorus research project
OM	Organic matter
OSR	Oilseed rape
P	Phosphorus
Pb	Lead
Pcaci <sub>2</sub>	Calcium chloride extractable P
P <sub>m</sub>	Morgan's P
POM	Programme of measures
P <sub>ox</sub>	Ammonium oxalate-oxalic acid extractable P
PP	Particulate P
PRD	Potential rooting depth
Psat <sub>ox</sub>	Soil P saturation
PSC	Phosphorus sorption capacity
PSD	Particle size distribution
RBMP	River basin management plans
RDPRI	Runoff dissolved P risk indicator
RUSLE	Revised Universal Soil Loss Equation
SAS	Statistical analysis software
SDR	Sediment delivery ratio
SEDD	Sediment distribution delivery

SERBD	South eastern river basin district
SFD	Soil framework directive
SHETRAN	Système hydrologique Européen TRANsport
SI	Statutory instrument
SOC	Soil organic carbon
SoCo	Sustainable agriculture and soil conservation
SOM	Soil organic matter
Sq	Soil structure quality as scored by VESS
SS	Suspended sediment
STP	Soil test P
SWAT	Soil water assessment tool
TDP	Total dissolved P
TRP	Total reactive P
TP	Total P
$T_{1/2}$	Half-life
USLE	Universal soil loss equation
VESS	Visual evaluation of soil structure
VSA	Visual soil assessment
WEP	Water extractable phosphorus
WFD	Water Framework Directive
$\alpha$	Empirical parameter used to relate P sorption capacity to $(Al_{ox} + Fe_{ox})$
$\beta$	Catchment specific modeling parameter

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# Chapter 1 Introduction

## 1.1 Overview

In the Republic of Ireland, agriculture accounts for 60.8% (approx. 4.2 million ha) of the total land area (CSO, 2010) - well above the European average of approximately 40%. Of this agricultural land, approximately 80% is devoted to grass (silage, hay and pasture), 10% to rough grazing, and 10% to arable cereal and crop production, with barley as the most important cereal crop representing 4.4% (185,900 ha). Crop production is concentrated in the south of the country (Schulte et al., 2010a), where the soils are highly suitable for tillage, having a light-to-medium texture and free drainage (Gardiner and Radford, 1980). Arable farming has been identified as one of the principal factors impacting on the ecological status of streams and rivers in Ireland (Donohue et al., 2006). Arable land has higher phosphorus (P) application rates than grassland due to higher offtakes and the need for new seeding each year, and can lead to a build-up of soil test P (STP) over time. Despite having a significantly higher P export coefficient (Doody et al., 2012) and greater susceptibility to erosion (Boardman and Poesen, 2006), arable land has not received the same level of research as the more dominant grassland. Therefore, its current degradation status and contribution to surface water impairment in Ireland is as yet unknown.

A literature review highlighted knowledge gaps in relation to two key areas where tillage has the potential to impact negatively on water and soil quality: (1) Nutrient loss, in particular P, via surface runoff needs to be quantified. There is a need for risk indicators that can identify critical source areas (CSAs) of P loss, so that current agronomic guidelines on P application can be assessed with respect to their potential to impact on water quality. Remediation measures can then be focused in these areas. (2) Sediment loss from fields with reduced ground cover and impaired soil structure. There is currently no methodology available in Ireland with which to quantify the risk of erosion. There is a need for a screening method so that high risk sites can be identified and prioritised for



further investigation. While the two areas required separate examination and are reported on in separate chapters, they are intrinsically linked in their potential to impact on water and soil quality. Combining both parameters helps to build a clearer picture of the overall risk to the environment. The outcome will also be more useful to farmers and advisors, who will ultimately utilise the knowledge generated.

Nutrient enrichment causing eutrophication is the principal and most widespread pressure on the aquatic environment in Ireland. In freshwaters this enrichment is attributed primarily to excessive inputs of P (DEHLG, 2009). Overland flow events resulting from intense rainfalls could potentially transport P and sediment from vulnerable tillage soils to surface water bodies. In addition, chronic P losses from the soil as a result of a build-up of STP in excess of crop requirements can contribute to losses. As a result of the European Union (EU) Water Framework Directive (WFD) (2000/60/EC: Council of the European Union, 2000), there is increasing pressure in Europe and Ireland to develop P-based management practices that will reduce the risk of diffuse losses from agricultural land to surface waters. Furthermore, the emergence of policies such as the proposed EU Draft Soil Framework Directive (SFD) (COM (2006) 231), which deals with concern over soil degradation and anthropogenic impacts to soil is likely to increase the requirement for assessment of soil quality and identification of soils at risk from degradation (Bone et al., 2012).

The overarching objective of the present body of work was to develop two methods which can be used in parallel to screen tillage land for likelihood of erosion and P loss to surface waters. The first method uses the relationship between STP and dissolved reactive P (DRP) measured in surface runoff water to define a threshold STP level, above which DRP concentrations in runoff from tillage soils subject to agronomic guidelines have the potential to cause eutrophication. The second method uses a novel soil structure and quality assessment system to provide farmers with the necessary tools to identify tillage fields where the risk of erosion is high. This screening method will allow mitigation strategies, such as buffer zones and minimum tillage, to be targeted on a smaller number of tillage sites, leading to cost and resource efficiencies.

## 1.2 Objectives

The specific objectives of this study were:

1. To review the current state of research and the regulatory regime relating to diffuse P and suspended sediment (SS) loss for tillage soils in Ireland. Following this, the study aimed to identify the key factors contributing to surface runoff and erosion at catchment-scale.
2. To acquire experimental data on the loss of P and SS in surface runoff from a selection of tillage soils when subjected to high intensity, low energy simulated rainfall and inclined at various slopes.
3. To examine the effect of increasing overland flow rates on the mobilisation and transport of DRP, particulate P (PP) and SS by flowing water.
4. To quantify the STP threshold above which there may be a potential threat to surface water quality in Ireland.
5. To develop a novel screening toolkit for use by farmers/specialist advisors in assessment of tillage fields for likelihood of erosion and reduced soil quality.

## 1.3 Procedure

A literature review of P and SS release from Irish tillage soils and the methods used to quantify and reduce losses to surface waters was undertaken. Critical source areas of P were chosen for further study as the identification of CSAs, where specific mitigation measures can be targeted, has significant implications for the WFD river basin management plans (RBMP). The identification of an environmental soil P threshold, above which surface runoff from tillage soils will have a negative impact on water quality, will ensure that mitigation strategies employed in Ireland to meet the

requirements of the WFD are targeted in those areas where they will be most cost effective.

Following this, a runoff experiment was designed to compare the P and SS releases from 5 Irish tillage soils when subjected to high intensity ( $30 \text{ mm hr}^{-1}$ ), low energy rainfall. In this experiment, each rainfall simulation comprised 3 successive 1-hr rainfall events applied at intervals of 1 hr and 24 hr to determine the effect of storm interval on surface runoff. As the velocity of surface runoff increases with slope, each soil was examined at 2 slopes, 10 and 15 degrees, to investigate the effect of slope on the magnitude of measured losses in the runoff. Using the relationship between STP in study soils and the DRP measured in the surface runoff, a runoff dissolved P risk indicator (RDPRI) was developed to identify the STP threshold above which there is a potential threat to surface water quality. This was achieved by constructing 95% confidence limits around the DRP relationships using the upper and lower confidence bands for the linear predictor.

It is critical that the RDPRI be tested under increasing overland flow rates, because the potential for pollutant transport to surface waters increases with flow rate. To determine the effect of overland flow rate on P and SS released in surface runoff, each soil was subjected to two distinct overland flow rates of 225 and  $450 \text{ ml min}^{-1}$  (possible worst case scenarios in fields where the soil has become saturated due to high rainfall intensities), which were introduced at the top of the flume in the presence of rainfall.

The majority of P lost in runoff during the rainfall and overland flow simulation experiments was in particulate form and therefore was not accounted for in the RDPRI. To address the issue of PP loss in surface runoff from the study soils, a field-based erosion risk assessment approach was adopted with the aim of developing a screening toolkit with which, farmers/specialist advisors can assess tillage fields for likelihood of erosion and loss of soil quality. At each study site, different methods of erosion risk, and soil structure and quality assessment were conducted. Comparisons of erosion risk indicators and soil quality indicators with observed erosion levels in the study fields allowed selection of a strategic set of indicators for inclusion in a screening toolkit, which

can be used by Irish tillage farmers and specialist advisors to identify, expeditiously, fields with high erosion risk and poor soil quality.

#### **1.4 Structure of dissertation**

In Chapter 2, a review of the environmental risks posed by poorly managed cultivated soils due to weakened structure and high P status and the methods used to assess and eliminate these risks are presented. Chapter 3 details the results of a laboratory rainfall simulation study, which was used to determine the effects of land slope, overland flow rate and storm interval on P and SS losses measured in surface runoff. Chapter 4 uses the data compiled in Chapter 3 to develop a RDPRI for Irish tillage soils. Chapter 4 also ranks soil extractable P methods with respect to their potential to be used as P loss risk indicators, and identifies important parameters for which to test when attempting to predict SS, TP, and PP loss from tillage soils. Chapter 5 details the results of erosion risk, and soil structure and quality assessments carried out at each study site. Chapter 5 also details the development and testing of an erosion and soil quality screening toolkit for farmers and specialist advisors. Finally, in Chapter 6, conclusions from the study are presented and recommendations for future research work are made.

To date, two peer-reviewed papers have been published from this work:

Regan, J.T., Fenton, O. and Healy, M.G. 2012. A review of phosphorus and sediment release from Irish tillage soil, the methods used to quantify losses and the current state of mitigation practice. *Biology and Environment: Proceedings of the Royal Irish Academy* 112B: 157 - 183.

Regan, J.T., Rodgers, M., Healy, M.G., Kirwan, L. and Fenton, O. 2010. Determining phosphorus and sediment release rates from five Irish tillage soils. *Journal of Environmental Quality* 39: 185-192.

In addition, this work contributed to a review of erosion in arable soils in Atlantic Europe, contained in the following publication:

Creamer, R.E., Brennan, F., Fenton, O., Healy, M.G., Lalor, S.T.J., Lanigan, G.J., **Regan**, J.T., and Griffiths, B.S. 2010. Implications of the proposed Soil Framework Directive on agricultural systems in Atlantic Europe – a review. *Journal of Soil Use and Management* 26: 198-211.

A number of international and national conference papers have also been published describing this work. A list of outputs and manuscripts in preparation for submission to international journals can be found in Appendix A.

## Chapter 2 Literature Review

### Overview

The environmental risks posed by cultivated soils due to susceptibility to erosion and high P status, and the methods used to quantify and eliminate these risks are reviewed in this chapter. The contents of this chapter are published in *Biology and Environment: Proceedings of the Royal Irish Academy* (112B: 157 - 183, 2012).

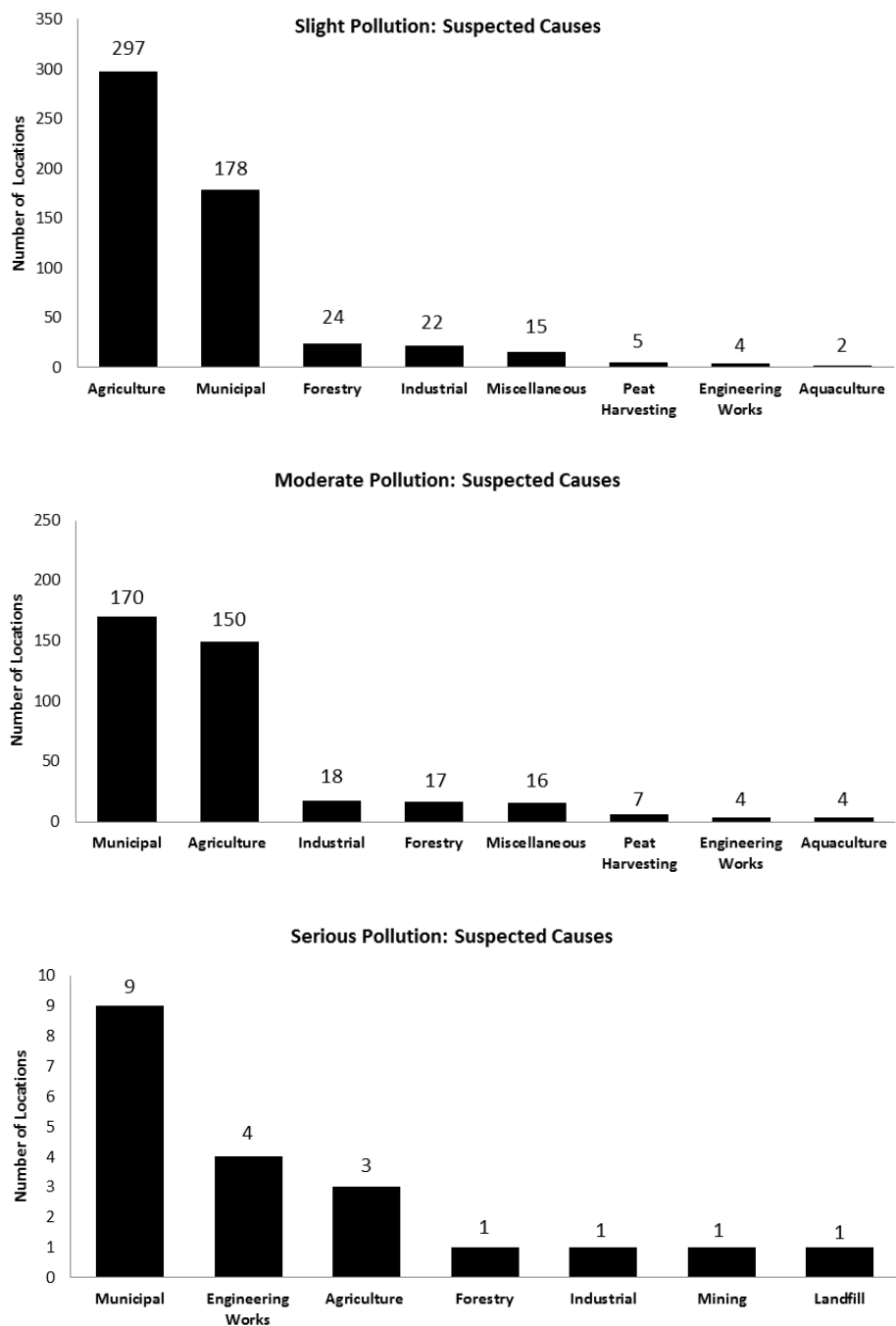
### 2.1 Introduction

In Ireland, agriculture is an important national industry accounting for 60.8% (approx 4.2 million ha) of the total land area (CSO, 2010). Of this agricultural land, approximately 0.4 million ha is devoted to arable cereal and crop production. Barley, wheat and oats are the main cereal crops representing 72% of this area. The cereal sector is small in relative terms representing < 1% of total EU production and approximately 3% of national gross agricultural output, with an annual cereal output of approximately 2 million tonnes in 2010 (Eurostat, 2012). In the southeast, cereals alone account for 17% of farmed land in County Carlow and 23% in County Wexford (Hooker et al., 2008). In the south of the country and the southeast in particular, the favourable climate provides better opportunities for seedbed preparation and harvesting. Here, there are fewer wet days, higher temperatures, less chance of frost, higher radiation receipts and more hours of bright sunshine (Collins and Cummins, 1996). Ireland's cereal yields are among the highest in the world largely due to suitable soils and climate and the technical ability of the farmers.

In Europe, de Wit et al. (2002) observed that agriculture contributed approximately one-third of all P to pollution in rivers. A biological survey of 13,177 km of Irish river and stream channels from 2007 to 2009 (McGarrigle et al., 2010) estimated that 20.7% were slightly polluted, 10% were moderately polluted, and 0.4% were seriously polluted.

However, when assessed for ecological status, according to the requirements of the EU WFD (2000/60/EC: Council of the European Union, 2000), based on the various biological and supporting physico-chemical quality elements for individual river water bodies on a one-out all-out basis, a different picture emerges, with just 52% of rivers achieving ‘good status’ (McGarrigle et al., 2010). Of the 2,515 sites surveyed in this period, the percentage of pollution attributed to agriculture was approximately 54% and 39% in rivers and streams that were slightly or moderately polluted, respectively, but only 15% in those that were seriously polluted. This data indicates that diffuse agricultural pollution causing eutrophication, accounted for 47% of the number of polluted river sites recorded over this period (Figure 2.1).

Algal growth, due to excessive P inputs, is often the primary reason for failing to achieve good ecological quality (Jeppesen et al., 2003). The WFD aims to restore polluted water bodies to ‘at least’ good ecological and chemical status ( $\leq 0.035 \text{ mg MRP L}^{-1}$  in rivers) by 2015 and prevent any further deterioration in the status of surface waters, transitional waters, groundwater and water-dependent terrestrial ecosystems and wetlands. Key to the WFD is the adoption and implementation of RBMP and Programme of Measures (POM) by 2012. These set out the actions required within each major river basin to achieve set environmental quality objectives, which will be reviewed on a six-yearly basis. These plans must include basic measures and, where necessary, supplementary measures to be implemented for a specific water body to help achieve prescribed water quality standards. The RBMP have identified agriculture as one of the main physico-chemical pressures affecting a water body. The basic regulation for agriculture in the Republic of Ireland (henceforth referred to as Ireland) is the Nitrates Directive (91/676/EEC: Council of the European Union, 1991), and is given statutory effect in the European Communities (Good Agricultural Practice for Protection of Waters) Regulations 2010 (SI 610 of 2010). The latter sets out detailed nutrient management controls for farming, including P application rates for crop production.



**Figure 2.1** Polluted river sites surveyed in 2007-2009 grouped by severity of pollution and by suspected cause (from McGarrigle et al., 2010).



Implicit in these directives and management plans is the protection and preservation of soils. The EU Draft SFD identifies the following threats to soil quality: erosion, decline of soil organic carbon (SOC), compaction, contamination, sealing, salinisation, landslides and desertification. However, to date, the directive has not been ratified. If ratified, member states will have to identify areas where soil degradation processes have occurred, or are likely to occur in the future. The identification of areas at risk of erosion will take account of the following parameters: soil type, texture and density, hydraulic properties, topography (including slope gradient and length), land cover and use, climate (including rainfall distribution and wind characteristics), hydrological conditions and agro-ecological zones. Once risk areas have been identified, member states will be required to draw up a POM, including a timetable for implementation. Ratification of the SFD will result in the unification of soil measures under one directive and provide a common approach and level playing field for member states with regard to soil protection (Creamer et al., 2010).

As part of the WFD, new environmental quality standards (EQS) for chemical compounds, and biological and hydromorphological classification systems for surface waters in Ireland have been developed by the Environmental Protection Agency (EPA) (Bowman, 2009). The proposed EQS values for chemical parameters in rivers are derived using a WFD-compliant methodology, using 3 years of biological and physicochemical data collected in Irish rivers considered to be of high and good biological status. The EQS values are necessary to define the high/good and good/moderate boundaries for the purposes of ecological classification. Rivers with molybdate reactive P (MRP) concentrations  $\leq 0.025$  and  $0.035 \text{ mg P L}^{-1}$  are classified as having high and good status, respectively (Bowman, 2009).

In agriculture, diffuse SS and P loss from soils occur primarily as a result of overland flow (Sharpley and Rekolainen, 1997; Sharpley et al., 2003a; Vadas et al., 2004) following hydrological (storm) events, and are the primary non-point-sources of pollution degrading the quality of surface waters (Daniel et al., 1998). However, P loss can also occur through drainage, with the most significant instances of downward movement of P

through the soil profile being associated with excessive application of P in manure and fertiliser (Sims et al., 1998), in particular, on soils with low P-retention properties and/or significant preferential flow pathways (e.g. cracking clay soils) (Hart et al., 2004). Recent extensive water quality surveys in Ireland revealed that diffuse P pollution originating from agricultural land, and transported by runoff and subsurface flows (e.g. leaching and artificial drainage systems) is the primary cause of the deterioration of surface water quality (Nasr et al., 2007).

## **2.2 Soil erosion**

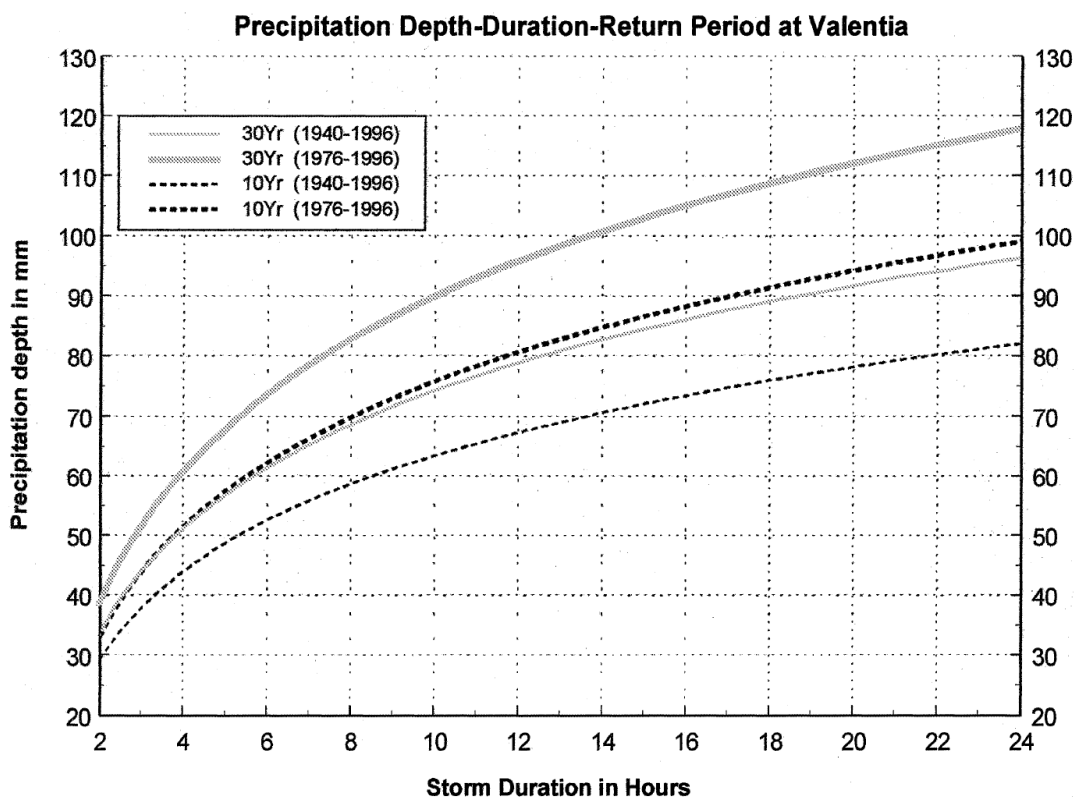
In Ireland, tillage land (cereals and root crops) only accounts for 9.6% of agricultural area utilised (CSO, 2010), but it accounts for a higher amount of fertiliser use than pastoral land (Lee, 1986). Therefore, the majority of research on the island of Ireland has focused on quantifying nutrient and, to a lesser extent, sediment losses from permanent grassland at laboratory- (Doody et al., 2006; Murphy, 2007; Murphy and Stevens, 2010), plot/field- (Tunney et al., 2007; Kurz et al., 2000, 2005 and 2006; Douglas et al., 2007; Doody et al., 2010 and 2011) and catchment-scales (Smith et al., 1995 and 2005; Scanlon et al., 2004; Jordan et al., 2005 a,b; Jordan et al., 2007). Modelling of diffuse P loss from grassland catchments has also been undertaken by Jordan et al. (2000), Daly et al. (2002), Scanlon et al. (2005), and Nasr et al. (2007) with the aim of improving management strategies to minimise P loss. Arable land is, however, more susceptible to water erosion than grassland (Van Oost et al., 2009) due to greater soil surface exposure to erosive forces during fallow and planting periods (Lal, 2001) and soil disturbance by tillage operations (Lal, 2001), which alters its structure. Furthermore, in grassland soils, higher infiltration rates can lower runoff rates (Fullen, 1991) and higher soil organic matter (SOM) levels can reduce erodibility (Fullen, 1998).

Increasing eutrophication of many surface water bodies in arable regions of the UK has been linked with increasing rates of soil erosion causing sediment and P loss from fields cropped with winter cereals and with an accumulation of soil P through continuous application of fertiliser and manures (Catt et al., 1998). Research to establish the

circumstances leading to sediment and P losses from arable land and to quantify these losses has been carried out in the UK (Speirs and Frost, 1987; Chambers et al., 1992; Catt et al., 1998; Chambers and Garwood, 2000) and throughout Europe (Kronvang et al., 1997; Verstraeten and Poesen, 2001; Miller et al., 2009) at multiple scales. Reported sediment and P losses from arable sites in these and other similar studies were significantly higher than losses from grassland, and were high enough to cause concern over eutrophication of surface water bodies in arable areas. Given the susceptibility of tillage land to erosion in general and high P applications associated with this land use in Ireland, there is a need to quantify the sediment and P losses from tillage soils to surface water bodies and monitor the effects of improving tillage practices.

Climate change will continue to make agricultural land more susceptible to P and sediment loss, with more extreme weather conditions forecast to increase the amounts of surface runoff, the detachment and transport of soil particles, and P released from fertiliser and manure amendments (Mainstone et al., 2008). Summer storm events, coupled with impervious agricultural soils, can lead to P addition to waterways during the growing season. By analysing long-term, hourly precipitation data from eight sites and daily streamflow data from four rivers in Ireland, Kiely (1999) identified a climatic and hydrologic change in 1975, after which both precipitation and streamflow increased. The study found that, since 1975, there has been an increase of about 10% in precipitation on the west-coast. Furthermore, analysis of the twenty most extreme 24-hr-duration rainfall events at the Valentia observatory for the period 1940-1993 identified 75% of them as having occurred since 1975, even though the period length since 1975 was 18 years by comparison with a length of 36 years before 1975. Kiely (1999) also noted that increases in extremes (frequency and/or magnitude) are likely to lead to increased incidence of flooding, which may also lead to more frequent incidences of deterioration in river water quality due to more frequent diffuse surface runoff. The immediate concern for river basin managers is highlighted by the precipitation depth-duration-return period curves for Valentia Observatory (Figure 2.2), which show that a storm depth that only occurred once every 30 years prior to 1975, is likely to occur approximately once every 10 years after 1975. The trend since 1975 of increasing precipitation and more extreme events is set to

continue. In the UK, in recent years, it has been observed that heavy rainfall events during the summer months are increasing (Maran et al., 2008).



**Figure 2.2** Precipitation depth-duration-frequency for Valentia for the complete (1940-1993) and partial time series (post-1975) (from Kiely, 1999).

### 2.2.1 Factors affecting soil erosion rates in Ireland

There is greater pressure on soils in Ireland due to increased erosion under modern intensive arable production systems and disposal of organic wastes to soils. Very few studies in Ireland have looked at erosion rates on tillage soils despite international research by Kronvang et al. (1997), Chambers et al. (2000), Deasy et al. (2009), Stevens et al. (2009), and Van Oost et al. (2009) concentrating specifically on such soils due to their erosion propensity. Greater demand for food and advances in farm machinery has resulted in intensified crop production, leading to higher tillage and water erosion rates in Europe. Lindstrom et al. (2001) defined tillage erosion as ‘the net movement of soil

downslope through the action of mechanical implements'. Both types of erosion can have a negative impact on productivity, with the most severe impact occurring due to a loss of topsoil depth in soils with a root restrictive layer (Lal, 2001). It is estimated that 115 million ha, or 12% of Europe's total land area, is affected by water erosion (EEA, 1995). Soil water erosion in the UK is primarily a regional phenomenon associated with sandy tillage soils in the southwest and southeast of England (Chambers et al., 2000). In Ireland, soil erosion is primarily a phenomenon associated with tillage soils and periods of intense rainfall (Fay et al., 2002). The potato growing area of Donegal is a good example. The main drivers predisposing arable soils to water erosion in Ireland and the UK are: soil type, precipitation (amount, duration and intensity), and management practice.

#### 2.2.1.1 Soil type

Soil type is important when determining the erosion risk from an arable field. The texture of a soil strongly influences SOM storage (Fullen et al., 2006). Soil organic matter breaks down faster in sandy soils than in fine-textured soils due to: (1) a lack of clay for physico-chemical binding with SOM (Fullen et al., 2006) and (2) greater oxygen availability for decomposition by microorganisms in the former. Disturbance of topsoil by tillage operations further aerates the soil which, in turn, increases soil SOM decomposition. Fullen et al. (2006) also reported that silt can play an important role in influencing organic matter (OM) storage in clay-deficient sandy soils. Sandy soils are particularly vulnerable to erosion due to low SOM content and poor structural stability, which predisposes the soil to: disaggregation under raindrop impact and a subsequent development of a surface crust, reduction of infiltration rate, and surface runoff (Quinton and Catt, 2004). Long-term arable use and modern cultivation methods can result in light textured soils capping (surface crust caused by heavy raindrops on finely cultivated soils) under rain impact (Fraser et al., 1999).

Increasing SOM content improves the cohesiveness of the soil, reduces the risk of surface crusting, lowers the risk of soil compaction, increases its water holding capacity and promotes soil aggregate formation, thereby improving structural stability and reducing

erosion. As an EU Member state, Ireland is required to monitor SOC levels in long-term (6 years or more) continuous tillage soils in order to ensure that sustainable management practices are put in place to reduce any further decline in SOC (Department of Agriculture, Fisheries and Food (DAFF) 2009a). In contrast to the UK, no work to date has been carried out in Ireland to determine the susceptibility of sandy soils to erosion under arable cropping. Findings in the UK may be indicative of potential erosion problems with sandy tillage soils in Ireland, but there is a need for Irish-specific data to establish if there is an erosion problem.

#### 2.2.1.2 Precipitation (amount, duration and intensity)

Rainfall characteristics influence processes affecting infiltration, runoff, soil detachment, and sediment and chemical transport (Truman, 2007). The risk of nutrient loss is generally greatest when soils are near field-capacity or saturation, and any further precipitation leads to water surpluses and either sub-surface drainage or overland flow (Schulte et al., 2006a). In unsaturated soils, erosion and P loss is mainly governed by the occurrence, frequency and timing of intense storm events that result in intense overland flow events. Rainfall intensity is generally considered to be one of the most important factors influencing soil erosion by overland flow and rills because it affects the detachment of soil particles by raindrop impact and enhances their transport by runoff.

A study by Chambers et al. (2000) of 13 erosion-susceptible arable catchments (containing medium silt, sand/light loam, silty clay loam, loamy sand and sandy loam soils) in England and Wales between 1989 and 1994 revealed that soil erosion can occur at any time of the year, provided the conditions suitable for erosion are present. These include: lack of ground cover vegetation ( $< 15\%$ ); loose, fluffy and very fine seed bed conditions; heavy rainfall ( $> 15 \text{ mm day}^{-1}$ ) with a high intensity ( $> 4 \text{ mm hr}^{-1}$ ) in the presence of high winds; steep slopes; presence of valley floor features that concentrate surface runoff; and compacted tramlines (unseeded wheeling areas used to facilitate spraying operations in cereal crops). The incidence of severe erosion resulting in transport of SS and P, in particular, tends to be highly dependent on hydrological storm

events (Edwards and Withers, 2008), and it has been shown that approximately 90 to 95% of soil erosion occurs during the most severe 2% of storms (Winegardner, 1996). Erosion also occurs over periods of prolonged lower-intensity rainfall (Robinson, 1999; Fraser et al., 1999).

Mean annual precipitation for Ireland ranges from 750-1000 mm on the greater part of the east coast to between 1000-1250 mm on the greater part of the west coast. The highest annual rainfall of between 1600-2800 mm occurs when Atlantic rain-bearing storms encounter landfall and mountainous terrain on the west coast. Any change in precipitation (amount, duration and intensity) over Ireland as a result of climate change, is likely to impact directly on P and sediment losses in surface runoff from agricultural soils. The 10-yr moving average for Ireland shows that rainfall amounts increased from 800 mm in the 1890s to 1100 mm in the 1990s (McElwain and Sweeney, 2006).

The general consensus from numerous climate change studies in Ireland is that winter rainfall will increase as will the frequency of intense rainfall events during summer. An EPA modelling report by Sweeney et al. (2008) on the impacts of climate change for Ireland projected an increase of 10% in winter rainfall by 2050, while reductions in summer of 12-17% are projected by the same time. Spatially, the largest percentage winter increases are forecast for the midlands, while summer reductions of 20-28% are forecast for the southern and eastern coasts. Sweeney et al. (2008) also predicted more frequent intense rainfall events during the summer. These projected changes will necessitate adaptation in water resources management in Ireland. For example, increased levels of erosion and greater suspended sediment loads will have to be managed for all Irish rivers (Sweeney et al., 2008). Increased P export in summer, resulting from high intensity rainfall events has been reported in numerous Irish grassland studies by Lennox et al. (1997), Tunney et al. (2000), Kurz (2000), Morgan et al. (2000), Kiely (2000), and Irvine et al. (2001). Overland flow events resulting from intense summer rainfalls could potentially transport P and sediment from vulnerable tillage soils to surface water bodies during the growing season.

### 2.2.1.3 Management practices

Erosion driven by management practices such as the decisions made by farmers as to what crops to grow, how to manage and prepare the land, and when to sow are also very important and easier to change in the short-term. Best management practices on steep, vulnerable slopes aims to minimise soil erosion losses, which, in turn, limit nutrient losses to a water body (Creamer et al., 2010). Research on cultivation practice in the UK by Chambers and Garwood (2000) identified valley features, lack of crop cover, wheelings (the passage over soil by wheels of a vehicle) and tramlines as the main contributors to erosion. Research by Silgram et al. (2010) on sandy loam and silty clay loam soils on 4-degree slopes in England has shown that tramlines can represent the most important pathway for P and sediment loss from moderately sloping fields. In a study of mitigation options for sediment and P loss from winter-sown arable crops on three soil types (sandy, silty and clay), Deasy et al. (2009) showed that compared to losses from cropped areas without tramlines, losses of sediment and P were between 2 and 230 and 2 and 293 times greater from tramline areas, respectively. The increase in losses due to tramlines was lower for the clay soils and greater for the silty soils, largely due to the cohesiveness of the clay soil. However, it is important to note that tramline areas normally only account for about 5% of the field area. Accelerated rates of soil erosion within agricultural landscapes are causing major modifications to terrestrial carbon (C), nitrogen (N) and P cycling (Quinton et al., 2010). Measures that can help maintain or increase SOC include: adoption of reduced tillage; straw incorporation; use of organic manures; use of cover crops; and adoption of mixed rotations (Hackett et al., 2010). Increases in SOC resulting from management changes are slow and reversible (Hackett et al., 2010).

Other contributors to erosion under modern intensive arable production systems are: ditch removal and field enlargement; use of high-powered modern traction systems, which can plough up and down slopes rather than contour ploughing (Quinton and Catt, 2004); use of heavy rollers after sowing (Boardman, 1990); and loose, fluffy and very fine seed bed conditions (Speirs and Frost, 1987; Catt et al., 1998). The removal of hedgerows, ditches



and open drains is now prohibited as part of EU Cross Compliance. Key tillage operations/practices that may impact on soil and water quality and possible mitigation options for Ireland are outlined in Section 2.2.4, Table 2.1.

### **2.2.2 Field indicators of a soil's susceptibility to runoff and erosion**

#### **2.2.2.1 Soil texture**

Soil texture and aggregate stability are the two main factors which make a soil more or less erodible (Coles and Moore, 1998). Soil texture also provides an indication of a soil's susceptibility to structural decline and compaction. While field textures, determined by hand, can approximate particle size distribution determined in the laboratory, they are influenced by cementing materials such as SOM, clay mineralogy,  $\text{CaCO}_3$  content and by iron and aluminium oxides and hydroxides, and as such must be carefully interpreted, and results treated with due caution. In general, field texture measurements are a good guide to soil behaviour (Moore, 1998) because texture controls water movement, affects chemical reactivity and nutrient availability, and is a factor in the erosion potential of a soil. As such, they can be used to infer a soil's susceptibility to erosion, nutrient leaching, waterlogging, subsurface compaction, and structural decline. While little can be done to change or improve the texture of a soil, it is an important property for guidance on how best to manage the soil.

#### **2.2.2.2 Soil colour**

Soil colour is mainly due to the presence of SOM (dark), iron oxides (yellow, brown, orange and red) and manganese oxides (black), or it may be due to the colour of the parent rock (for example, old red sandstone -red). A Munsell Soil Colour Chart (Munsell, 2009) is used in soil classification to describe the colour of each soil horizon using the notations for hue, value and chroma. Hue is the colour frequency which, for most soils, ranges from red to yellow; value refers to the lightness from white to black; and chroma defines the degree of colour saturation or intensity of hue (Fitzpatrick et al., 1999). Soil

colour is a good indicator of soil quality because it can provide an indirect measure of other more useful properties that are not so easily and accurately assessed (Shepherd, 2009). Change in soil colour can be a useful indicator of (1) changes in OM under a particular land use or management; (2) soil drainage class; (3) the amount and oxidation state of soil iron oxides; and (4) degree of soil aeration, all of which have management implications. By informing on soil drainage class, soil colour can be indicative of a particular soil's associated runoff risk.

#### 2.2.2.3 Mottling

Mottles are spots or blotches of different colours, or shades of colour interspersed with the dominant colour of the soil (FAO, 2006). Their presence in a soil indicates that it is less permeable than a whole coloured soil (Northcote, 1983), and therefore poses a greater risk of erosion arising from infiltration excess overland flow. The number and colour of soil mottles present in a soil horizon are important measurements related to soil aeration and hence waterlogging. They are also early warning signs of a decline in soil structure due to compaction under wheel traffic and over-cultivation (Shepherd, 2009). Mottling is caused by changes in the distribution, concentration, and state of oxidation of iron and manganese compounds, which are present in many minerals in the soil (Batey, 2000). The changes are caused by reducing conditions, which, in turn, result from microbial activity during anoxic conditions (Batey, 2000). Under anaerobic conditions, any iron (Fe) and manganese (Mn) present in their brown/orange oxidised ferric ( $\text{Fe}^{3+}$ ) and manganic ( $\text{Mn}^{3+}$ ) forms are reduced to grey ferrous ( $\text{Fe}^{2+}$ ) and manganous ( $\text{Mn}^{2+}$ ) oxides. If air re-enters the reduced soil, the process is reversed. As oxygen depletion increases, orange, and ultimately grey, mottles predominate. The abundance of grey mottles indicates the soil is poorly drained for a large part of the year (Shepherd, 2009), and is therefore prone to flooding during wet periods. These soils require careful management to ensure that the soil is not left vulnerable to erosion by surface runoff.

#### 2.2.2.4 Soil structure and consistence

Soil structure refers to the arrangement of soil particles into discrete soil units (aggregates or peds), which are separated from each other by pores or voids. Consistence refers to the degree of cohesion or adhesion of the soil mass. A field description of soil structure enables its classification in terms of grade (degree of distinctness and durability), class (size) and type (shape and arrangement) of aggregates. These are important soil structural characteristics that have major influence on the pathway of water movement (drainage or runoff), root penetration and development, soil trafficability, and resistance of soil to structural degradation. Soil structure is vulnerable to change by compaction and erosion, and its preservation is key to sustaining soil function (Mueller et al., 2010). Soil structure influences soil water movement and retention, erosion, crusting, nutrient recycling, root penetration and crop yield (Bronick and Lal, 2005). Soils with good structure have friable, fine, porous, sub-angular and sub-rounded (nutty) aggregates, while those with poor structure have large, dense, very firm, angular or sub-angular blocky clods that pack closely together (Shepherd, 2009). The decline in soil structure is increasingly seen as a form of soil degradation and is often related to land use and soil/crop management practices (Bronick and Lal, 2005). The influence of soil structure on soil erodibility is primarily a result of reduced infiltration of rainwater in poorly structured soils, leading to erosion caused by runoff from higher ground. Soil management practices that can negatively affect soil structure include: (1) use of power harrows and heavy machinery to produce a fine tilth in weakly structured fine sands and light silts; and (2) high axle loads and incorrect tyre pressures resulting in soil compaction (Creamer et al., 2010). Externalities such as runoff, surface- and ground-water pollution, and CO<sub>2</sub> emissions are influenced by soil structure (Bronick and Lal, 2005).

#### 2.2.2.5 Soil porosity and root development

Porosity provides an indication of the amount of soil pores, which can exist in a wide range of shapes and sizes between and within soil aggregates, and are occupied by air and water. The type, size, abundance, continuity and orientation of pores can be determined

using a hand lens as part of a detailed soil classification. Roots and macropores orientated vertically are most important for water infiltration. The permeability of soil during infiltration is mainly controlled by coarse pores, in which the water is not held under the influence of capillary forces (Beven and Germann, 1982), whereas fine pores control plant-available water and water storage capacity (Lal et al., 1999). Any reduction in the size of the coarse pores as a result of hydro-mechanical processes like slaking of aggregates and dispersion of clays or gross mechanical stresses like raindrop impact, compaction and tillage, may reduce infiltration, thereby increasing the risk of surface runoff. A reduction in voids or structural porosity will also affect root growth (Moore, 1998), and the absence of roots below the plough layer can be indicative of a plough pan. For agricultural purposes, a soil should ideally be able to withstand stress induced by vehicle traffic and to remain in a porous structural state, permitting water and air transmission for crop growth (Zhang and Hartge, 1995). Soil structure degradation following intensive agricultural activities, soil compaction, loss of structural stability and the formation of surface crusts gives rise to the loss of continuity of elongated transmission pores, thereby reducing water transport and increasing runoff and soil erosion (Pagliai et al., 2004).

Soil porosity is influenced largely by type of structure; but it is also influenced by rooting and by the activity of earthworms and other soil macro-organisms (Gardiner and Ryan, 1964). Plant root systems help to bind the soil by releasing a variety of compounds, which have a cementing effect on soil particles (Bronnick and Lal, 2005), and can play an important role in preventing soil erosion after the crop has been harvested. Soils with good structure have a high porosity between and within aggregates, but soils with poor structure may not have macropores or coarse micropores within the large clods, thus restricting their drainage and aeration (Shepherd, 2009). The act of cultivating the soil causes short-term increases in porosity but long-term decreases in aggregation (Bronnick and Lal, 2005).

### 2.2.3 Main soil properties controlling soil erosion at field-scale

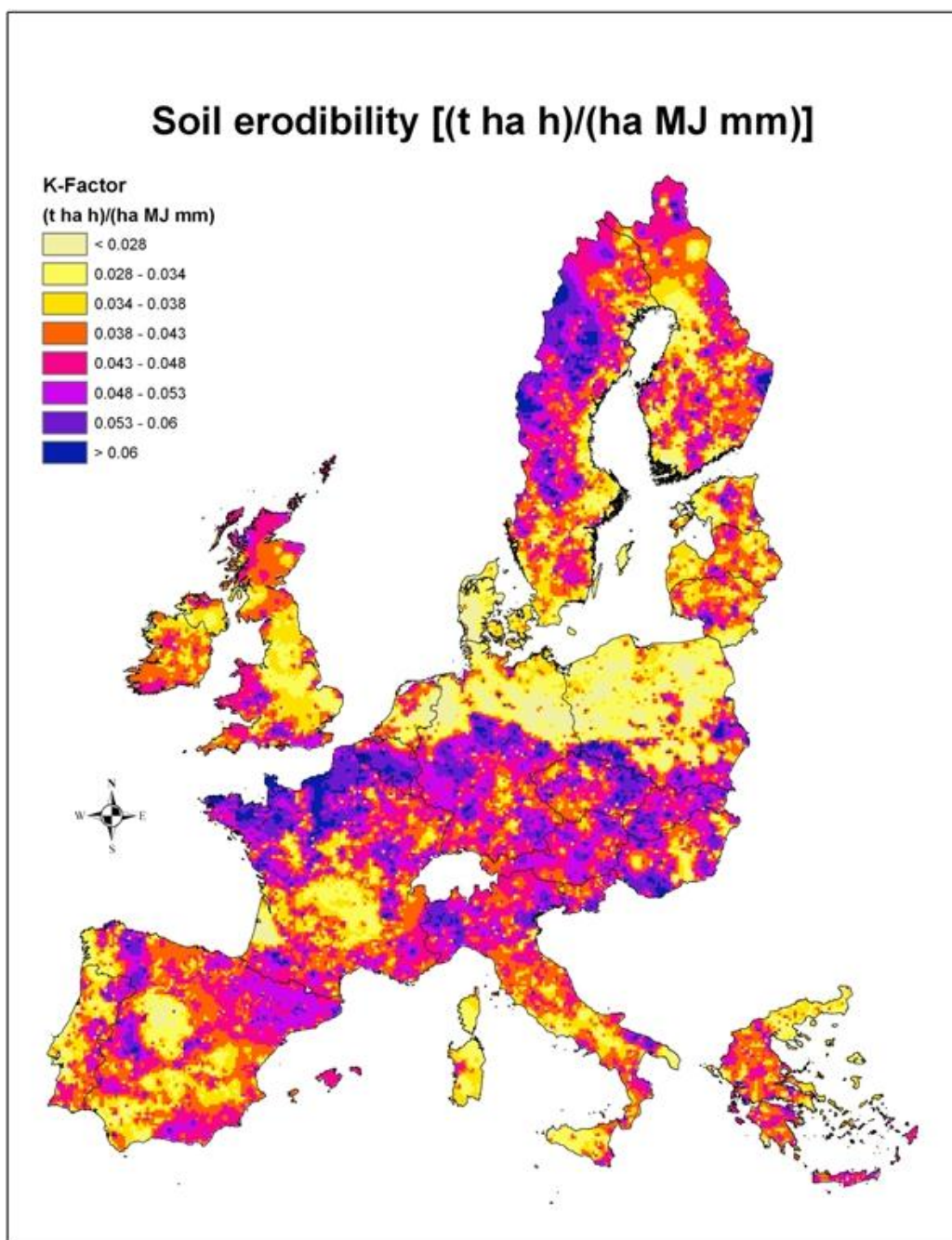
Soils identified as potentially susceptible to runoff and erosion by visual and tactile assessment of the field indicators just discussed can be further assessed by analytical methods. Results obtained by these methods must be relayed to the farmer, so that he/she is aware of the soil's vulnerability to specific threats and can then manage the soil accordingly.

#### 2.2.3.1 Particle size distribution

Particle size analysis has an advantage over texture determinations carried out by hand in the field, in that it enables the classification of sandy loams as either very fine, fine, medium, or coarse based on the exact particle size range within the sand fraction, which is difficult to determine accurately by hand estimation. The fineness of the sand particles within a sandy loam is important when assessing its susceptibility to erosion. Resistance to erosion is lowest for small non-cohesive grains, particularly silt and fine sand-sized particles with low clay content (Grimm et al., 2002). The most erodible soils identified by Romero et al. (2007) when measuring interrill and rill erodibility were those with the greatest amount of silt and very fine sand, while the most resistant to erosion were clayey soils. Fine sandy loam and silty loam textures can be highly susceptible to water erosion and may also be hardsetting (Moody and Cong, 2008).

Particle size distribution is one of the major soil properties governing soil erodibility and is one of the crucial factors required to assess soil erodibility in terms of the K-factor of Wischmeier and Smith (1978). The K-factor is a lumped parameter that represents an integrated average annual value of the soil profile reaction to the processes of soil detachment and transport by raindrop impact and surface flow (Renard et al., 1997). Most recently, Panagos et al. (2012) produced a K-factor erodibility map of Europe using data from 22,000 soil samples, collected during the first conducted LUCAS (Land Use and Cover Area Frame Survey; 2009) pan-European soil sampling campaign. Inverse distance weighted interpolation with a power parameter of 2 performed best ( $R^2$  adjusted = 0.81)

to interpolate LUCAS point data to a soil erodibility map of Europe (Figure 2.3). This represents an enormous improvement in the precise estimation of K-factor at European level compared with past methodologies, which derived this attribute based only on 5 textural classes and relatively coarser scale (Panagos et al, 2012).



**Figure 2.3 Soil erodibility (ton ha h ha<sup>-1</sup> MJ<sup>-1</sup>mm<sup>-1</sup>) across Europe based on the nomograph of Wischmeier and Smith (1978) (from Panagos et al., 2012).**

During the survey, 4,165 geo-referenced points from a standard 2 km x 2 km grid, were visited in Ireland. Soil samples (0-30 cm) were collected at 389 of these survey points for determination of sand, silt, clay and OM content, amongst other parameters. The mean K-factor determined for the land area of Ireland was 0.039, just below the European average of 0.041. Ireland had the lowest dispersion of K-factor values of all the countries surveyed, as measured by the coefficient of variation. Based on the K-factor distribution within Ireland (Figure 2.3), more than half the soils of Ireland can be considered to be at least moderately susceptible (K-factors between about 0.032 and 0.052) to detachment and transport by raindrop impact and surface flow. Soils having the highest K-factor erodibility values are located mainly in the potato growing areas of Donegal and in the tillage areas of the South of the country. These soils, if occurring on slopes  $> 3^\circ$  and left bare or degraded by compaction, may pose a significant erosion risk.

#### 2.2.3.2 Soil organic carbon/matter

Soil organic carbon also serves to bind individual soil particles into larger aggregates and is important in maintaining the aggregate stability. By keeping the aggregates in soil stable (by combining with soil minerals), it promotes infiltration, the movement and retention of water, helps develop and stabilise soil structure, cushions the impact of wheel traffic and cultivators, and reduces the potential for wind and water erosion (Shepherd, 2009). Generally, OM can hold up to 20 times its weight in water and can, therefore, directly affect soil water retention, which makes soil more resistant to drought and erosion, as well as indirectly through its positive effects on soil structure (Dick and Gregorich, 2004). The texture of a soil strongly influences SOM storage (Fullen et al., 2006), with coarse textured soils being particularly vulnerable to SOM decline. Furthermore, the decline of SOC levels has been highlighted in numerous legislative reports and scientific papers as contributing to a decline in soil quality and can result in increased soil erosion, loss of nutrients and an increased susceptibility to compaction (Van-Camp et al., 2004).

In a review of critical levels of SOC in tillage land in Ireland, Spink et al. (2010) concluded that soil function is unlikely to be adversely affected when SOC is above a threshold of 2% (equivalent to c. 3.4% SOM). Soils having less SOM than this threshold should be further assessed to see if they are in good environmental and agronomic condition. These further measures could include observation of: erosion, gullies in the field, compaction and capping (Spink et al., 2010). Whitmore et al. (2004) developed a unified framework of measurement to assess rapidly the stability of soil surface structures in England and Wales. Classification of 120 soils using this protocol, found that all aggregates classified as unstable or very unstable were from arable, light and medium soils (< 35% clay) that had low SOC (< 1.5%). In addition, practically all grassland soils, regardless of their SOC or clay content, were categorised as stable or very stable. Soil organic matter is an important component of both managed and unmanaged terrestrial ecosystems, but is especially important in influencing soil erodibility (Jankauskas et al., 2007). As an EU Member state, Ireland is required to monitor SOC levels in long-term tillage soils in order to ensure that sustainable management practices are put in place to reduce any further decline in SOC (DAFF 2009). Measurement of soil organic carbon in 1310 soil samples in Ireland showed that it varies from 1.4% to 55.8%, with a median value of 7% (Zhang et al., 2008). Most recently, Zhang et al. (2011) produced a spatial distribution map of SOC in Ireland created using geographically-weighted regression.

### 2.2.3.3 Bulk density

The natural rate of soil erosion is accelerated by increased soil bulk density (BD) resulting from vehicular traffic-induced compaction (Lal, 2001). A healthy soil bulk density is important in terms of sustainable soil productivity and environmental well-being (Merrington et al., 2006). High BD values indicate a poorer environment for root growth, reduced aeration and undesirable changes in hydrologic function, such as reduced water infiltration (FAO, 2006). Since erosion will not occur without surface runoff, soil infiltration rate is important in relation to erosion. The higher the BD in surface or sub-surface layers, the lower will be the total porosity and hence the greater the risk of surface runoff. Packing density (PD) - a derivative of BD calculated as  $PD = BD + 0.009(\% \text{ clay})$



(Hawes et al., 2010) - is a better parameter than BD for comparison of physical structure between different soils and, based on derived statistics, affords an excellent indirect estimation of soil porosity (Hall et al., 1977).

Surface compaction along tramlines and wheelings can trigger erosion problems in winter cereal crops (Deasy et al., 2009; Batey, 2009). In well-graded loam and sandy loam soils, the susceptibility to compaction can be very high because smaller particles can fit into spaces between larger particles, thus providing the ideal proportions of particles of different sizes to achieve the densest packing arrangements. In clayey soils, BD normally ranges from 1.2 to 1.5 ton m<sup>-3</sup> and in sandy soils from 1.6 to 1.9 ton m<sup>-3</sup> (Needham et al., 1998). Irrespective of soil textural group, arable soils consistently have a higher BD than other land uses. Merrington et al. (2006) proposed critical BD thresholds for broad soil and habitat groupings in England and Wales derived from a range of data sources. The proposed BD threshold for arable and horticultural function on mineral soil was 1.3 ton m<sup>-3</sup>. They also noted that for UK conditions one would expect severe hydrological degradation of a soil with bulk density > 1.5 ton m<sup>-3</sup>. Packing density classes according to Van Ranst et al. (1995) are: low < 1.40 ton m<sup>-3</sup>, medium 1.4 - 1.75 ton m<sup>-3</sup> and high > 1.75 ton m<sup>-3</sup>.

#### 2.2.3.4 Aggregate stability

Aggregate stability is one of the most complex and dynamic soil properties affecting principal physical and hydraulic soil characteristics, such as infiltration rate, hydraulic conductivity and erodibility (Dimoyiannis, 2011). Crop management practices such as application of organic fertilisers, liming, incorporation of stubble and min-till or no-till cultivation, can improve aggregate stability. Aggregation is essentially the flocculation and cementation of individual soil particles to form aggregates. The primary soil properties influencing aggregation and aggregate stability are texture, clay mineralogy, SOM, cations, sesquioxides and calcium carbonate (Le Bissonnais, 1996). Clay flocculation promotes soil aggregation and structural stability. The general consensus is that the clay fraction has a positive effect on structural stability compared to the silt and

sand fractions. Cations such as  $\text{Fe}^{3+}$ ,  $\text{Al}^{3+}$ ,  $\text{Mg}^{2+}$  and  $\text{Ca}^{2+}$  serve to: (1) stimulate the precipitation of compounds that act as bonding agents for primary soil particles and (2) form bridges between clay and SOM particles resulting in aggregation (Bronick and Lal, 2005). Dontsova et al. (2001) showed that  $\text{Ca}^{2+}$  ions were more effective than  $\text{Mg}^{2+}$  in aggregating soil clays and that if soils are prone to surface sealing, it is beneficial to manage them to high Ca:Mg ratios. High concentrations of Al and Fe oxides and hydroxides, often referred to as cementing agents, have the effect of increasing aggregate stability (Needham et al., 1998). The presence of Al and Fe oxide in soils also has a favourable effect on soil structure. Evidence of soil structural improvement is provided by increased aggregate stability, permeability, friability, porosity, and hydraulic conductivity (Goldberg, 1989). In general, Al oxides have a greater stabilising effect than spherical Fe oxides on structure because of their platy morphology (Goldberg, 1989). Dimoyiannis (2011) found that the aggregate stability of calcareous surface soils in Greece was positively affected by the presence of Al oxides, whereas Fe oxides had no significant effect. The same study found that the presence of free carbonates in soils as an excess negatively affected aggregate stability.

Water is the main cause of aggregate breakdown in most soils, either directly by rainfall or by surface runoff. Water-stable aggregation on the soil surface is therefore important when considering the inherent erodibility of a soil and its susceptibility to soil structural decline. There are four main mechanisms for aggregate breakdown: (1) slaking; (2) differential swelling; (3) mechanical; and (4) physicochemical dispersion (Le Bissonnais, 1996). Slaking is the spontaneous disintegration of a soil aggregates which have insufficient strength to withstand the stresses induced by rapid water intake. Differential swelling depends on the same properties as slaking and produces similar microaggregates, but normally occurs in soils with higher clay contents. Mechanical breakdown is a result of the kinetic energy imparted to the soil by rain drop impact. Physicochemical dispersion is the complete breakdown of aggregates into primary particles because the attractive forces holding the particles together are lessened by wetting. The Emerson aggregate test (Emerson, 1967) enables visual assessment of how aggregates breakdown in water (slaking or dispersion), and is useful for broadly defining

the stability of soils. Recommendations about how a soil should be managed are made based on the Emerson score.

#### **2.2.4 Impact of tillage farming**

Concern with the effects on water quality and river health of non-point-source delivery of nutrients and pollutants has given added impetus to research on the dynamics of erosion and erosion-driven processes, and, in particular, the size distribution of transported sediment (Hogarth et al., 2004). Soil erosion is a gravity-driven process by which soil particles are first detached from the soil mass by rainfall and runoff, and then transported by rainfall and runoff until they settle out of suspension. Concerns about present-day soil erosion are related to accelerated erosion, where the natural rate has increased as a result of human activities, which leave the land unprotected and vulnerable. These activities predominate on agricultural soils and, in particular, on tillage soils where intensification of land management, change in land cover through cultivation, and inappropriate land use (cultivation of steep slopes), coupled with naturally occurring erosive rainfalls, can lead to accelerated erosion rates. Excessive erosion can have a negative impact on-site (deterioration in soil structure, decrease in crop yield, ecosystem damage and loss of soil, OM and nutrients) and off-site (siltation and pollution of receiving waters and carbon dioxide (CO<sub>2</sub>) release from broken-down clay particles and SOM). Lal (1995) estimated that global water erosion and related processes release 1.14 Pg C/yr (1 Pg=10<sup>15</sup> g) to the atmosphere. Furthermore, extensive erosion can reduce the ability of soils to withdraw or sequester C as CO<sub>2</sub> from the earth's atmosphere. However, results from other studies indicate that erosion enhances CO<sub>2</sub> uptake (Smith et al., 2001) and that erosion and deposition reduce CO<sub>2</sub> emissions from the soil into the atmosphere by exposing low C-bearing soil at eroding sites and by burying SOC at depositional sites (Liu et al., 2003). For example, tillage erosion and deposition leads to the burial of *c.* 7 Tg C y<sup>-1</sup> in the area covered by the CO-ordination of INformation on the Environment (CORINE) (CORINE, 2000) database (assuming a 2% topsoil C content) (Van Oost et al., 2009).

Soil loss on arable agricultural land is typically an order of magnitude higher than under undisturbed native vegetation (Van Oost et al., 2009), and two orders of magnitude higher than rates of soil formation (Montgomery, 2007). There is much evidence to show that soil erosion due to rainfall and overland flow is exacerbated by tillage operations. However, of similar importance is the extent of tillage erosion resulting directly from tillage operations. This generally results in a movement of soil from convex shaped to concave shaped landscapes, and leads to a nutrient-rich soil in the latter. While water erosion is strongly controlled by soil characteristics such as soil stone level, texture and crusting potential (Van Oost et al., 2009), experimental studies have shown that tillage speed, depth, direction and implement characteristics are the primary controlling factors on tillage erosion (Van Oost et al., 2006). It is of major importance that eroded nutrients and sediment are retained in-field so as not to impact on surface water quality.

Given that rates of soil redistribution in the medium-term are influenced by tillage displacement as well as water erosion, it is necessary to separate these two components of soil redistribution in order to obtain a reliable assessment of water erosion rates (Blake et al., 1999). By using a tillage erosion diffusion-type model based on the one Lobb et al. (1999) proposed and land use databases, Van Oost et al. (2009) estimated that the mean gross tillage erosion rates for the part of Europe covered by the CORINE land use database was  $3.3 \text{ ton ha}^{-1} \text{ yr}^{-1}$ . For the same land area, they estimated the average water erosion rate was  $3.9 \text{ ton ha}^{-1} \text{ yr}^{-1}$  by using water erosion estimates for arable land, orchards and vineyards compiled in a study by Cerdan et al. (2006) of datasets from 81 experimental sites across 19 European countries. The model also used large-scale land use (CORINE), soil (Soil Geographical Database of Europe), topography (Shuttle Radar Topography Mission) (Ciat, 2004) and soil erodibility datasets for Europe. For the cropland area of Ireland, the same models estimated the average tillage and water erosion rates to be  $2.9$  and  $4.4 \text{ ton ha}^{-1} \text{ yr}^{-1}$ , respectively. These erosion rates are higher than average rates of soil formation (consisting of mineral weathering, soil biomass growth and dust deposition) which range from  $0.3\text{-}1.4 \text{ ton ha}^{-1} \text{ yr}^{-1}$ , with the lower limit being indicative of European conditions (Creamer et al., 2010). Research on tillage soils in Ireland is needed to validate erosion rates given in the model of Van Oost et al. (2009).

**Table 2.1 Key tillage operations/practices that may impact on soil and water quality and possible mitigation options**

Operation/practice	Impact on soil	Potential impact on water quality	Possible solutions
Cultivation and seeding	Machinery traffic coupled with low soil strength and high moisture content leading to soil compaction	Sediment and P loss in runoff from sloping land following heavy rainfall	Reduced axle load / larger tyres / controlled traffic on fields prone to soil erosion / avoid wet conditions / shallow cultivation to increase bearing strength
	Topsoil disturbance increases soil aeration which in turn increases SOM decomposition thus lowering soil structural stability	Soils with less than approximately 3.4% SOM can be considered erodible which can lead to sedimentation in rivers	Non-plough systems such as minimum tillage (this can give rise to problems with less reliable establishment, grass weeds and compaction) / residue and organic matter incorporation / organic manures / cover crops / mixed rotations
	Loose fluffy very fine seed bed following excessive cultivation	Rill and gully development leading to very high sediment losses in concentrated overland flow	Avoid excessive cultivation particularly on light soils
	Down-slope movement of soil by mechanical tillage on sloping land	Increased potential for P transfer to aquatic systems by water erosion	Contour tillage and cultivation / contour grass strips / proper land management reflecting site specific conditions
Application of pesticides/herbicides/fertiliser	Machinery traffic on tramlines using narrow (row crop) tyres can lead to intense compaction.	Concentrated flow path for sediment and P loss in runoff throughout the growing season.	Tramline management (disruption of the compacted tramline surface to a depth of 60 mm with a tine)
Harvesting of crop / application of slurry	Sub-soil compaction resulting from very high axle loads	May contribute to erosion and P loss particularly with the latter	Large tyres or tracks on combine harvester and large tyres on slurry spreader
Post-harvest cropping	Fallow/bare soil leading to net C loss due to an absence of C uptake	Bare soil with low SOM can have poor structure and is particularly susceptible to erosion	Reduced winter fallow by using winter and cover crops – volunteer growth also helps / crop residue incorporation
Long-term cultivation	Compaction and impaired structure	Sediment and associated P loss in runoff following heavy rainfall	Residue and organic matter incorporation / Land-use change – rotation with grass
	Reduced SOM/annual C uptake	Increased erosion with greater potential for sediment delivery to waterways	Residue and organic matter incorporation / Land-use change – rotation with grass

The potential total P (TP) losses associated with these estimates of erosion could have serious implications for water quality in Ireland if the eroded sediment reaches surface water bodies, given that the typical range for TP content of non-polluted agricultural soils in Ireland is estimated at between 0.02 to 0.2% (McGrath et al., 2001) with the median TP content of Irish soils (0-10 cm) being 0.11% (Fay et al., 2007). Applying this range of TP in Irish agricultural soils to the estimates of water erosion reported by Van Oost et al. (2009) for Ireland gives a range of 0.88 to 8.8 kg P ha<sup>-1</sup> yr<sup>-1</sup> lost from arable land. This is a conservative estimate of P loss from arable fields due to erosion, given that P has low solubility and is primarily bound to finer soil fractions like clay, which runoff preferentially transports (Quinton et al., 2001). In a study of TP export coefficients from different CORINE land cover classes in 50 experimental sub-catchments of the rivers Colebrooke and Upper Bann in Northern Ireland, Smith et al. (2005) determined the TP export coefficient from non-irrigated arable land to surface waters to be 4.88±1.12 kg P ha<sup>-1</sup> yr<sup>-1</sup> with 95% confidence limits. This export coefficient was almost twice as high as that measured during the study for any other CORINE land cover class and almost five times as high as the export coefficient for pasture.

With the exception of tillage erosion occurring adjacent to waterways, soil transported in the field by mechanical tillage operations is unlikely to reach surface waters without transportation by water erosion. Though tillage erosion does not have the same direct detrimental effect on surface water quality as water erosion, it can increase the risk of nutrient delivery to waterways by progressive accumulation of nutrient-rich sediment in low-lying areas of fields which may be exposed to concentrated overland flow and leaching, and, therefore, it must be accounted for in any assessment of soil erosion. If field- and catchment-scale research is to identify sediment sources and test mitigation options within arable areas, it must be designed to explicitly attribute losses to tillage or water erosion processes. This will require assessment of each type of erosion in isolation and while interacting with one another. This is essential if we are to understand the role played by tillage erosion in delivering sediment to surface waters.

Rainfall variability in Ireland often results in tillage field operations being carried out in less favourable conditions (soils at or near field capacity) with increased risk of soil compaction from field machinery traffic. The trend towards larger machines with increased axle loads further increase the risk of soil compaction. Compacted soils with poor structure are more prone to surface capping and poor infiltration of water due to reduced porosity and consequent reduction in hydraulic conductivity, leading to earlier saturation and thus increased surface runoff and erosion in sloping areas. Soil compaction can occur as surface compaction i.e. within the tilled layer or as subsoil compaction which occurs beneath the plough layer. Surface compaction is normally dealt with in the next tillage operation, while subsoil compaction is much more persistent and difficult to remove. While sub-soiling has been the subject of much research and can reduce bulk density and compacted layers, it is generally considered better to avoid subsoil compaction than to rely on alleviating the compacted soil layer afterwards (Alakukku et al., 2003; Spoor et al., 2003). Prevention of subsoil compaction is essential for economically and environmentally sustainable agriculture (Arvidsson et al., 2000). Compaction can be reduced by: (1) use of low ground pressure wheel equipment on machinery (Chamen et al., 2003); (2) working in good soil moisture conditions and minimising the weight of machinery (Van den Akker et al., 2003); (3) minimising the number of passes of machinery (Marsili et al., 1996) and (4) controlled traffic systems (Chamen et al., 2003).

### **2.2.5 Processes and mechanics of erosion and deposition**

The processes governing the detachment, transport and deposition of sediment are intricate and interactive in nature, and must be fully understood if rates of soil erosion are to be accurately determined.

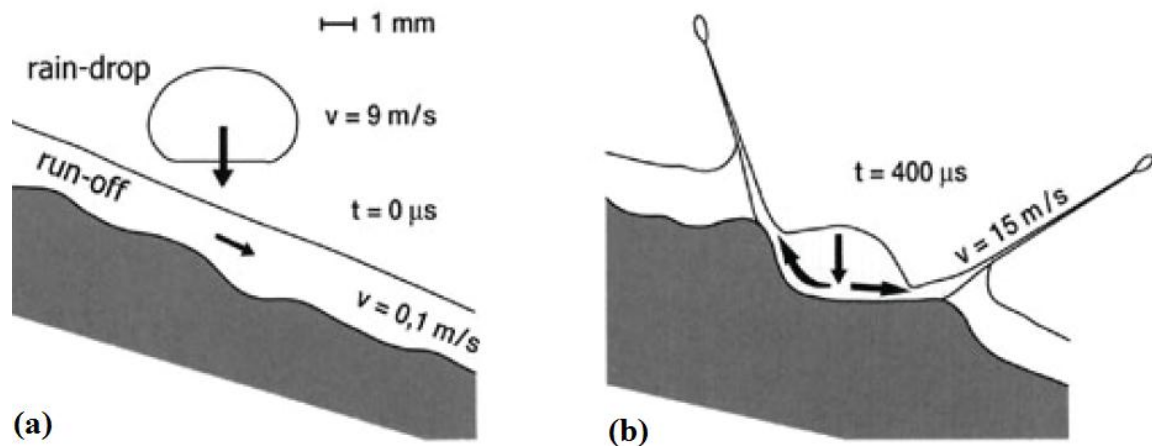
#### **2.2.5.1 Raindrop erosion**

Rainfall characteristics influence processes affecting infiltration, runoff, soil detachment, and sediment and chemical transport (Truman, 2007). Rainfall intensity (I) is generally considered to be one of the most important factors influencing soil erosion by overland flow and rills because it affects the detachment of soil particles

by raindrop impact and enhances their transport by runoff. It was reported by Guy et al. (1987) that 85% of sediment transported from inter-rill areas could be attributed to enhancement of transport capacity of runoff by raindrop impact, while only 15% was attributed to undisturbed runoff. Meyer and Wischmeier (1969) showed that in natural rainfalls, soil detachment by raindrop impact is proportional to the rainfall intensity squared ( $I^2$ ). Soil detachment resulting from raindrop impact is a function of raindrop size, mass, velocity, and kinetic energy, raindrop impact angle, soil strength, flow depth, sediment load and surface sealing (Zhang et al., 2003). Erosion due to rainfall impact can dominate erosion due to overland flow in shallow sheet flow of limited length over gentle slopes (Proffitt and Rose, 1991), or in flow between rills (Marshall et al., 1996). In inter-rill areas, particle transport by raindrop-impacted shallow flow is the dominant transport process (Zhang et al., 2003).

Ground cover, in the form of vegetation and crop residue, acts as a protective layer between the soil surface and falling raindrops by absorbing the kinetic energy that would otherwise be imparted to the topsoil. Rain intercepted by the crop canopy either evaporates or finds its way to the soil by dripping from leaves, or by running down plant stems as stemflow. The action of direct throughfall and leaf drainage on the soil produces rainsplash erosion, the most important detaching agent in the erosion process (Figure 2.4). Rainsplash erosion can also be reduced by manure application, which enhances soil aggregation. Arable land is particularly susceptible to rainsplash erosion, as it can be found bare and unprotected during stages of the arable cycle. The impact of raindrops on topsoil can contribute to erosion in two ways: first, by breakdown of the structure of the soil; and, second, by adding directly to the concentration of sediment in surface water (Rose, 2004).





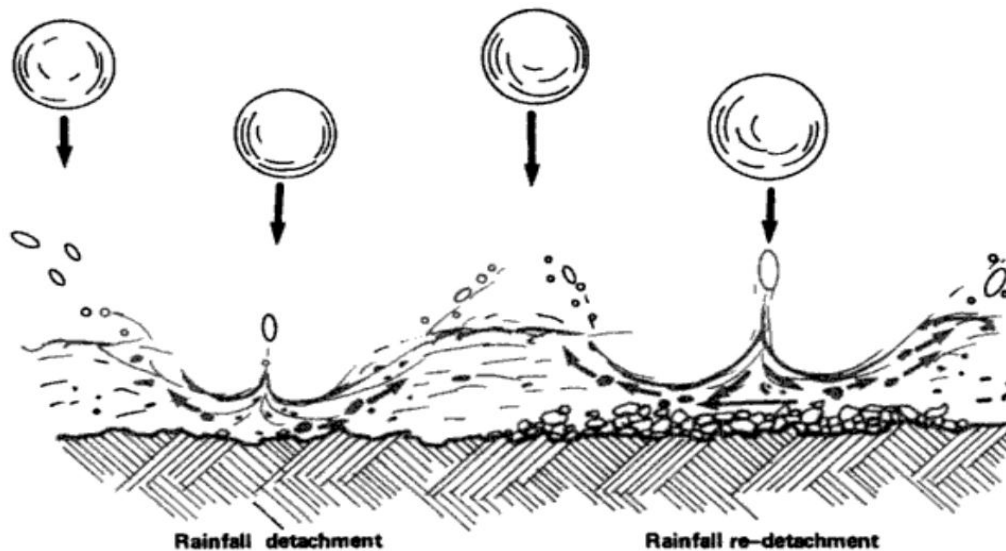
**Figure 2.4 Raindrop falling on a bare unprotected soil surface and detaching soil particles from the original soil matrix: a) raindrop velocity, b) raindrop impact producing a disrupting force in the form of laterally flowing jets (Saavedra, 2005)**

For detachment to occur, the raindrop must overcome the interstitial forces holding the primary soil particles together in bundles or aggregates. The detachability of the soil, therefore, depends not only on the soil texture, but also the shear strength of the topsoil (Cruse and Larson, 1977). Raindrops that impact on the soil bed exert shear stress which can be very high locally, compared to shear stresses commonly exerted by overland flow, given relatively shallow water depths (Rouhipour et al., 2006). A study by Poesen (1985) to determine the kinetic energy required to detach 1 kg of soil by rainfall impact showed that particles between 0.063 and 0.25 mm in size are the most detachable by raindrop impact. The coarser soils are resistant to detachment because of the weight of their larger particles, and the finer soils are resistant because the raindrop energy has to overcome the adhesive or chemical-bonding forces that link the minerals comprising the clay particles (Morgan, 2005). For this reason, silt loams, loams, fine sands and sandy loams are the most detachable soil types. The low OM content of sandy soils further disposes them to rainsplash erosion, as they have low shear strength. In some soils, the process of structural disaggregation, due to raindrop impact, can result in a surface seal or crust forming. This phenomenon decreases the infiltration rate, reduces the available water to the plant in the root zone, diminishes the natural recharge of aquifers, increases runoff and soil erosion, affects

seedlings and plant growth, and decreases crop yields (Assouline, 2004), and is particularly common in sandy soils when subjected to rainfall.

When the rainfall rate exceeds the rate of infiltration of the soil, water starts to pond in surface depressions or hollows, and runoff does not begin until the surface storage capacity is exceeded. This shallow overland flow rarely exerts enough shear force on the original soil matrix to detach particles from it, but it does carry sediment dislodged by raindrop impact (Trout, 1993). In inter-rill areas, raindrops impacting on this shallow overland flow can increase its transport capacity. Rainfall detachment decreases with increased flow depth (Moss and Green, 1983) due to cushioning of the raindrop impact by the water layer (Ferreira and Singer, 1985). It has been shown experimentally by Palmer (1965) and Ghadiri and Payne (1981) that the maximum raindrop impact occurs for water depths of less than one raindrop diameter. Furthermore, Proffitt et al. (1991) found that raindrop impact becomes negligible for flow depths greater than 3 drop diameters.

While the rainfall proceeds, runoff is continuously splashed by falling raindrops, further breaking down the soil particles it is carrying and helping to keep them in suspension. Therefore, these finer particles, which are typically more effective at absorbing the plant nutrients (P and N), will travel further (Rose and Dalal, 1988), are more likely to reach waterways, and may lead to eutrophication. Coarser particles that do settle out of solution form a deposited layer on the soil surface. The sediment in the deposited layer is much more easily eroded than the original soil matrix, as it does not have time to form cohesive bonds with neighboring sediment. The deposited layer also partially protects the original soil matrix from further raindrop impact. When sediment is removed from the deposited layer by raindrop impact, it is termed 'rainfall re-detachment' (Figure 2.5). Under the same rainfall, re-detachment of the deposited layer will occur at a much higher rate than detachment of the original soil if that soil is cohesive (Rose, 1988).



**Figure 2.5** Schematic showing the distinction between rainfall detachment of the original soil matrix, and re-detachment of previously eroded and deposited sediment (from Rose, 1993).

#### 2.2.5.2 Flow-driven erosion

For flow-driven erosion to occur, a threshold stream power value must be exceeded. The stream power is a product of the shear stress exerted by the flowing water on the soil surface and the velocity of that water, and is dependent on the nature of the soil surface. Roughness elements at the soil surface (e.g. crop residues, rock fragments, vegetation, geotextiles) strongly reduce the erosivity of overland flow (both inter-rill and concentrated overland flow) and, hence, soil detachment rates (Knappen et al., 2009). An increase in the depth or flow rate of the surface water will increase the stream power. Once the force exerted by the flow exceeds the forces keeping the soil particle or aggregate at rest, flow-driven erosion will occur.

Flow-driven erosion can be broken up into 3 processes: entrainment, deposition and re-entrainment. Entrainment is defined as the removal by flowing water of the original cohesive soil by the mutual shear stress between the soil surface and the water flowing over it. The rate of entrainment is, therefore, strongly influenced by soil physical properties, in particular, cohesion/strength, and also by hydraulic characteristics of flow. The length of time eroded soil particles remain in suspension before settling back to the soil surface in deposition is dependent on their settling

velocities, which, in turn, depends on their size. Coarser particles having higher settling velocities will settle out first while finer particles may be transported great distances before falling out of suspension. Once returned to the soil surface, these particles have insufficient time to form cohesive bonds with neighboring particles and are, therefore, much more easily removed in a process termed 're-entrainment'. It is the ease of removal of this cohesionless, deposited soil that distinguishes the re-entrainment process from that of the entrainment process (Rose, 1993). In effect, a soil particle or aggregate transported by the processes just described will make a consecutive series of hops while transitioning from a position of rest on the soil bed up into to the flowing surface water layer and back to the soil bed. The size and trajectory of these hops are dependent on the particles settling velocity in water. This hopping motion is commonly referred to as 'saltation' and is common-place in shallow overland flow.

Flow-driven erosion is commonly differentiated into sheet erosion and rill erosion (Mulqueen et al., 2006). Sheet erosion (or interrill erosion) removes a thin layer of soil, whereas rill erosion excavates a discrete centimeters-deep channel with the hydraulic power of concentrated overland flow (Whiting et al., 2001). Generally, sheet flows are a result of high-intensity, short-duration rainfall events. Sheet flow occurs as a shallow sheet of water flowing for short distances over gently sloping land. The resulting erosion is termed 'sheet erosion' and removes a uniform layer of soil from the soil surface, which is often only recognizable when this soil is deposited at the bottom of a slope or near a fenceline. Typically, sheet flow does not have sufficient depth or velocity to detach soil particles from a bare soil surface, and acts primarily as a transport mechanism for soil previously dislodged and disaggregated by raindrop impact. As such, sheet flow normally results in the loss of the finer particles, such as clay, silt and OM. Plant nutrients are generally more concentrated on these finer particles such that even modest loss of them is of considerable practical concern to both agriculture and the quality of receiving waters (Rose et al., 2006). Though rarely seen, sheet erosion accounts for large volumes of soil loss from cultivated land each year in Ireland.

Tillage disturbance, natural microtopographic variation, and drainage patterns which develop as a result of the soil erosion process itself can all produce concentrated flows (Hairsine and Rose, 1992). Therefore, sheet flow occurring over relatively rough surfaces begins to form small concentrated channels at a critical distance downslope. Once the rills become sufficiently developed, they can capture much of the rainfall-generated flow from the interrill area, and function as a sediment source and delivery system for erosion on hillslopes. In rills, the dominant processes are entrainment and re-entrainment by concentrated flow aided by mass movement of soil into the rill due to sidewall sloughing and slips, undercutting of sidewalls, and head cutting of rills (Mulqueen et al., 2006). Generally, the development of rills is accompanied by a dramatic increase in erosion. This is due to the enhanced streampower in the rill (Marshall et al., 1996). Streampower has been related to detachment rate in rills by Nearing et al. (1997) and Hairsine and Rose (1992), and was shown to be the best parameter to predict detachment capacity for rills by Elliot and Laflen (1993).

#### **2.2.6 Measuring and quantifying soil erosion on arable land**

There is a need for information on both gross and net erosion rates from agricultural land, so that the sediment delivery ratio (SDR), or proportion of the sediment mobilised by soil erosion that is transported towards local watercourses, rather than being deposited close to the original source, can be determined (Blake et al., 1999). If the level of erosion of Irish tillage soils is to be accurately determined, work must be undertaken that quantifies rates of soil movement to surface waters at the catchment scale.

Traditional monitoring techniques used to establish soil erosion rates have the inherent flaw of failing to determine the fate of eroded sediment and, therefore, give no indication of the impact of measured erosion rates on surface water quality. Blake et al. (1999) note that it is particularly difficult to assemble information on the spatial distribution of erosion and deposition rates within the landscape, and on the associated SDRs using traditional monitoring techniques. Much of the information available on erosion rates has been collected from flume and erosion plot studies; however, these only provide information on the net rate of soil loss from the bounded area, as

represented by the flux of sediment across its lower boundary. As such, plot studies typically overestimate erosion rates by failing to encompass major catchment sediment stores (Collins and Walling, 2007). These stores get larger as catchment area increases because the fraction of less steep slopes, like valley bottoms where sediment deposition occurs, also increase (Verstraeten and Poesen, 2001). It is for these reasons that the representativeness of plot results in terms of the wider landscape is often questioned. As the scale at which erosion is being studied increases from flume-to-plot and up to field- and catchment-scale, the parameters influencing this erosion change and, therefore, so must methods used to measure erosion. The use of sediment fingerprinting and composite fingerprints to determine the provenance of eroded sediment is one preferable method at larger scales which will be discussed in Section 2.6.1.

### **2.3 Phosphorus transfer from agricultural soils to surface water bodies**

The movement of P from agricultural soils to water through the action of rainfall and overland flow has serious implications for surface water quality in Ireland. Agriculturally derived P is estimated to account for 70% of P loads in rivers and estuaries in Ireland (EPA, 2004). Diffuse losses from agriculture were reported by McGarrigle and Donnelley (2003) to account for 59% of TP exported from a rural Irish catchment.

#### **2.3.1 Sensitivity of surface waters to eutrophication**

Phosphorus is a naturally occurring element in the environment essential for agricultural crop and livestock production. Concentrations of P present in streams, rivers and lakes exceeding critical values for algal growth can lead to eutrophication (Carpenter et al., 1998; Pote et al., 1999; Sharpley, 2000; Haygarth et al., 2005). As DRP is readily available for biological uptake, it poses an immediate threat for accelerated algal growth, which may negatively affect water quality in rivers and lakes (Sharpley and Smith, 1989).

In 2002, the World Health Organisation (WHO, 2002) reported that when P is the limiting factor (i.e. if N:P is greater than 7:1 in the water column), a phosphate concentration of  $0.01 \text{ mg L}^{-1}$  is enough to support plankton, and concentrations from  $0.03$  to  $0.1 \text{ mg L}^{-1}$ , or higher, will be likely to promote algal blooms. Empirical comparison of in stream phosphate levels and biological quality has demonstrated that once median phosphate concentrations exceed  $0.03 \text{ mg L}^{-1}$  P, significant deterioration is seen in Irish river ecosystems (Clabby et al., 2008). The sensitivity of surface waters to eutrophication from diffuse agricultural P losses is further highlighted by the relatively low concentrations at which eutrophication can occur:  $0.03 \text{ mg L}^{-1}$  of MRP in rivers and  $0.02 \text{ mg TP L}^{-1}$  in lakes (Lucey et al., 1999). These are an order of magnitude lower than DRP concentrations in the soil solution necessary to support plant growth ( $0.2 - 0.3 \text{ mg P L}^{-1}$ ) (Heathwaite and Dils, 2000; Aase et al., 2001; Sharpley et al., 2003a).

Prior to July 2009 water quality standards for P in Ireland were set out in the Phosphorus Regulations (SI 258 of 1998). In these regulations, rivers having annual median phosphate concentrations of  $< 0.03 \text{ mg L}^{-1}$  were classed as unpolluted. New environmental quality standards for chemical and physico-chemical elements in rivers were brought into law in the European Communities Environmental Objectives (Surface Waters) Regulations (SI 272 of 2009). These regulations address the requirements of the WFD and also repeal SI 258 of 1998. Furthermore, they define rivers with  $\text{MRP} \leq 0.035 \text{ mg L}^{-1}$  as being in good status and therefore not polluted. Therefore, for good water quality in Irish water bodies, it is considered that P additions from all sources should not give rise to a concentration in the water of greater than  $0.035 \text{ mg MRP L}^{-1}$ . A value of  $0.03 \text{ mg MRP L}^{-1}$  represents a more conservative figure as it lies midway between good ( $\leq 0.035$ ) and high ( $\leq 0.025$ ) status. The model of Donohue et al. (2006) which links catchment characteristics and water chemistry with the ecological status of Irish rivers also supports the use of  $0.03 \text{ mg MRP L}^{-1}$  as the environmental standard for river water quality.

### 2.3.2 Soil test phosphorus

It is generally accepted that there is a positive relationship between STP and P loss to water in runoff events (Tunney et al., 2000; Vadas et al., 2005). In Ireland, Morgan's extractant (Peech and English, 1944) is currently used to match P fertiliser recommendations with crop requirements. Phosphorus advice for grassland and tillage crops in Ireland is based on a four- category soil P-index system (Table 2.2). The basis of this system is a set of soil indices based on the measured Morgan's P ( $P_m$ ) in the soil and the crops response to fertiliser application as measured by field experimentation. For tillage soils at P Index 4, the addition of P is prohibited with the exception of soils planted with potatoes, beet, and turnips.

The current agronomic optimum  $P_m$  value for Irish soils is 6 mg L<sup>-1</sup> for grass production (Daly et al., 2001). In Ireland, low soil  $P_m$  concentrations of 1 mg L<sup>-1</sup> in the 1950s severely limited crop production. Since then, continual fertiliser application at high rates on agricultural land has resulted in excessive levels of plant available P in soils (Tunney, 2000). Although national sales of P fertiliser have fallen from 62,410 ton yr<sup>-1</sup> to 26,350 ton yr<sup>-1</sup> during the period 1995-2008 (DAFF, 2009b), primarily due to new farming practices, implementation of the Nitrates Directive and rising fertiliser costs, the mean  $P_m$  concentration in Irish soils is currently 8 mg L<sup>-1</sup> (Daly et al., 2001). Maintenance of the P fertility of arable soils is important as cereal crops perform better in soils of good P status (6.1-10 mg L<sup>-1</sup>  $P_m$ ) than on soils of low P status that have been supplemented with higher levels of P fertilisers (Schulte et al., 2010b). Fertiliser advice is modified for some tillage crops, according to crop yields, soil texture, or expected summer rainfall amount (Coulter and Lalor, 2008).

As soil P increases, P loss in surface runoff and subsurface flow increases (Sharpley et al., 2001b). Therefore, the higher the  $P_m$  level in fields of a catchment, the greater the risk of high concentrations of in-stream P during wet months (Lewis, 2003). Previous grassland studies at plot (Pote et al., 1999) and field (Tunney et al., 2000) scale have shown that there is a positive relationship between the  $P_m$  level in soils and DRP lost in surface runoff. Schulte et al. (2010a) developed a model of STP decline on eight principal soil series/associations representative of a range of STP concentrations for



grassland in Ireland and found that where the  $P_m$  was at  $28 \text{ mg L}^{-1}$  and with no further P inputs (estimated to be equivalent to an annual field P-balance deficit of  $30 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ), it took from 7-15 years for a soil to move from Index 4 to Index 3 (Table 2.2). Almost half the river monitoring sites sampled for phosphates in the South Eastern River Basin District (SERBD) - where tillage is common - in 2008 would not achieve good status based on this nutrient (Lucey et al., 2009). All lakes assessed from 2007 to 2009 in the SERBD were of moderate or poor ecological status largely due to TP and chlorophyll, possibly related to intensive agriculture (McGarrigle et al., 2010).

**Table 2.2 Phosphorus Index System (from SI 610 of 2010 and adapted from Schulte et al., 2010a)**

Soil P Index	Soil P ranges ( $\text{mg L}^{-1}$ )		Interpretation
	Grassland	Tillage	
1	0.0-3.0	0.0-3.0	Soil is P deficient; build-up of soil P required.
2	3.1-5.0	3.1-6.0	Low soil P status: build-up of soil P is required for productive agriculture
3	5.1-8.0	6.1-10.0	Target soil P status: only maintenance rates of P required
4	> 8.0	> 10	Excess soil P status: no agronomic response to P applications.

### 2.3.3 Phosphorus use on tillage land

While tillage land accounts for a relatively small area (9.6% of agricultural area utilised in Ireland (CSO, 2009)), it accounts for a lot of the high P status soils due to higher fertilisation rates on tillage land, and may therefore make a disproportionate contribution to the TP input to surface water systems from agricultural soils. Mean P fertiliser use in Ireland for cereals and root crops (less than 10% of tillage area) in

2008 was 20 and 46 kg ha<sup>-1</sup>, respectively, while P fertiliser use for grassland was only 5 kg ha<sup>-1</sup> (Lalor et al., 2010). These figures highlight the potential for higher P losses in surface runoff from tillage land than from grassland.

Excessive manure and fertiliser application is not only wasteful, but it can lead to a build-up of P in excess of crop requirements in the soil. The excess P may then be mobilised by surface runoff during periods of heavy rainfall. The United States Department of Agriculture estimates that about half the fertiliser used each year in the United States simply replaces soil nutrients lost by topsoil erosion (Montgomery, 2007). Soil test P, accumulated to very high concentrations, can take up to 20 years of continual crop harvesting - with no addition of P from any source - to reduce to concentrations normally recommended for agronomic production and to pose no threat to surface water quality (Sharpley and Rekolainen, 1997).

Tillage land has higher P application rates than grassland due to the higher offtakes and the need for new seeding each year. Sufficiently high available P levels are needed for satisfactory seed germination. Advice given to farmers on P application for cereal crops is based on maintaining the STP at the agronomic optimum level of Index 3 (Table 2.2). This is achieved by applying enough P to replace the anticipated crop off-take (a grain yield of 1 ton ha<sup>-1</sup> = an off-take of 3.8 kg P ha<sup>-1</sup>), based on the expected yield of the crop to be fertilised (Coulter and Lalor, 2008). Where proof of higher yield is available, an additional 3.8 kg P ha<sup>-1</sup> can be applied on soils at P Indices 1, 2 and 3 for each additional tonne above a threshold crop yield dependent on crop variety (SI 610 of 2010). Where the Soil Index is below Index 3, build-up levels are necessary in addition to anticipated crop off-take in order to raise the Soil Index to Index 3. Regular soil testing should be carried out to ensure that soils are maintained within the agronomic optimum Soil Index. Root crops, like potatoes and fodder beet, are very responsive to P and it is necessary to apply P (when sowing) even at Index 4 to achieve the agronomic optimum.

The impact of land use (agriculture) and soil characteristics (parent material and wetness) on plant available P distribution in soils is given credence by Zhang et al. (2008) in a geochemical mapping study of Ireland in which  $P_m$  was measured in 1310

surface (0-10 cm) soil samples collected from pre-determined positions - at a density of 2 samples per 100 km<sup>2</sup> - based on an unbiased grid sampling scheme. They delineated the areas having high available P using the index bands for tillage soils in the P index system (Table 2.2), which state that soils having > 10 mg L<sup>-1</sup> P<sub>m</sub> levels are in excess of crop requirement. The authors attributed high levels of available P in County Louth, east Dublin and southeast Wexford to a combination of light-textured soils, and vegetable and tillage farming in these areas. Similarly, in northwest Kerry, tillage farming on light-textured soils resulted in elevated P levels. Furthermore, they attributed high levels in east and central Cork to a combination of intensive dairying and tillage on highly fertile soils, while high levels in north Carlow and south Kildare may be due to intensive tillage on limestone-derived soils. Reducing these soil P levels may not be possible in the short term as Schulte et al. (2010a) showed that elevated soil P concentrations, resulting from agricultural land use, may take many years to be reduced to agronomically and environmentally optimum levels.

#### **2.3.4 Critical source areas of phosphorus**

The loss of P tends to be highly sporadic in nature and is often restricted to small geographic areas (Edwards and Withers, 2008). In many regions, small portions of saturated land, known as variable source areas, generate the majority of overland flow (Pionke et al., 2000), the amount of which is largely independent of rainfall intensity (Walter et al., 2000). These variable source areas commonly exist because the groundwater table is close to the land surface near the stream, causing seep zones and high soil moisture levels that limit available storage in the soil profile (Gburek and Sharpley, 1998). They can contract and expand both seasonally and during storms as a function of precipitation, topography, soil type, geology, soil moisture status, and water table level (Hart et al., 2004). Runoff generated from variable source areas is dominated by saturation excess overland flow and, to a lesser extent, rapidly responding subsurface flow (Gburek and Sharpley, 1998). Outside of these areas, infiltration and groundwater recharge are the dominant hydrological processes, and runoff generation is normally low with the exception of high intensity storm events. Hydrologically CSAs have been defined in terms of the coincidence of sources of P with variable source areas (Doody et al., 2012). However, for variable source areas

located some distance from a water course, continuous hydrological connectivity along the flowpath is necessary to link fields to water courses. Connectivity will ultimately determine whether P source areas become CSAs and create problems in receiving waters. As such, it is more appropriate to define CSAs in terms of their impact on the aquatic environment with particular emphasis on connectivity between P source and aquatic receptor. Wall et al. (2011) recently described CSAs as ‘regions where high nutrient loading or status coincides with a high propensity to be hydrologically connected to water courses’.

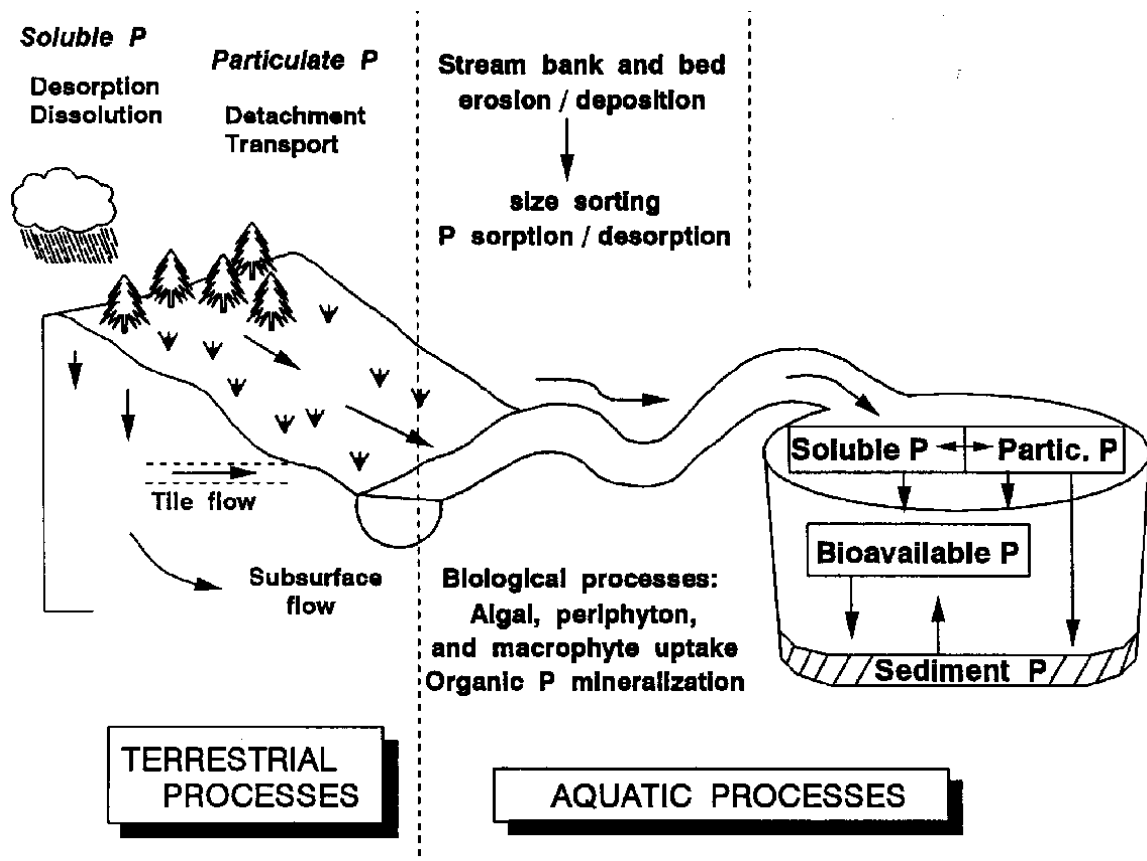
A large proportion (up to 90%) of P exported from catchments on an annual basis is generated from a relatively small portion of the catchment and during only one or two storm events (Sharpley and Rekolainen, 1997). Tunney et al. (2000) showed that 40% of the total amount of DRP lost in runoff for 1997 from four grassland fields ranging in size from 0.5 – 14.5 ha was lost when about 150 mm of rain fell in a 4-d period. In contrast, a study of nutrient and sediment loss to water from agricultural grassland catchments of the Dripsey River, Co. Cork in 2002, found that more than 80% of TP loss was for the five months of October to February, with a large proportion coming from about 10 storm events where high P concentrations occurred simultaneously with high stream flows (Lewis, 2003). This evidence suggests that, while extreme rainfall events with large return periods like that reported by Tunney et al. (2000) can be responsible for a large proportion of DRP lost over an atypical year, more normally one would expect P loss to be spread across a number of large storms throughout the year. In addition, research at plot-scale on arable land in the UK by Quinton et al. (2001) showed that more frequently occurring smaller events accounted for a greater proportion of the P lost over a 6-yr period than infrequent large events. It is important to note that losses in the study of Quinton et al. (2001) were measured at the end of an erosion plot and that even though a smaller proportion of P was lost in larger events, these events have greater transport potential and are more likely to deliver eroded sediment and P to surface waters.

The identification of CSAs, where the potential for pollution is higher, has significant implications for RBMP, because the blanket application of a specific mitigation measure across an entire catchment will not be as cost-effective as its deployment

solely in those areas where it is most appropriate. Pionke et al. (1997) suggested that effective mitigation of P losses from agriculture must focus on defining, targeting, and remediating CSAs of P loss. Focusing on CSAs of P could significantly improve the environmental efficiency and cost effectiveness of the POM adopted for the WFD (Schulte et al., 2009; White et al., 2009). Schulte et al. (2009) and Doody et al. (2009) developed a suite of cost-effective catchment specific P mitigation measures by identifying and characterising CSAs on farms in the Lough Melvin catchment using a modified P ranking scheme. Similarly, Hughes et al. (2005) used field and catchment-scale P ranking schemes to identify CSAs for P loss in Ireland. Phosphorus ranking schemes are discussed further in Section 2.4.1. Quantifying the influence of CSAs of surface and near surface runoff is a next phase in the Agricultural Catchments Programme (ACP) experimental design, targeting diffuse events of P and sediment transfer in dissimilar parts of the catchment and comparing with the continuous data at the outlets (Wall et al., 2011).

### **2.3.5 Phosphorus mobilisation**

Mobilisation is the first key step in the separation of P molecules from their source and includes chemical, biological and physical processes (Figure 2.6). These processes group into either solubilisation or detachment mechanisms, defined by the physical size of the P compounds that are mobilised (Haygarth et al., 2005). Solubilisation potential of P from soil surfaces and soil biota into soil water increases with increasing concentrations of STP, which result from long-term historical addition of fertiliser and manure in excess of crop requirements. The detachment and transfer of non-dissolved P in association with soil particles is more pronounced where farming practices generate erosion (Chambers et al., 2000), and provides a physical mechanism for mobilising P from soil into surface waters (Sharpley and Smith, 1990; Toy et al., 2002). The size threshold most commonly used to operationally define detachment is  $> 0.45 \mu\text{m}$  and has been used for the threshold between dissolved and PP (Haygarth and Jarvis, 1997).



**Figure 2.6 Processes in the transfer of P from terrestrial to aquatic ecosystems (from Sharpley et al., 1995).**

Haygarth and Jarvis (1999) have argued for the inclusion of a third mode by which P can be mobilised for transport to water - incidental transfer of dissolved P and PP occurring when fertiliser or manure applications, which are not incorporated into the soil, are coincident with onset of rainfall. They conclude that even though incidental transfer will include mobilisation and detachment, it should be kept separate from these mechanisms due to the unique circumstances leading up to its occurrence and control. The relative proportions of PP and dissolved P in surface run-off, therefore, depend on the complex interaction between climate, topography, soil type, soil P content, type of farming system, and farm management (Withers, 1999).

Particulate P encompasses all primary and secondary mineral P forms, plus organic P, P sorbed by minerals, and organic particles eroded during runoff. It constitutes the major proportion of P transported from cultivated land (75-90%) (Sharpley et al., 1995). Fang et al. (2002) reported that PP contributed from 59 to 98% of total runoff

P for unvegetated, packed runoff boxes. Unlike most DRP, which is readily available for plant uptake, PP acts as a long-term source of P for submerged aquatic vegetation and algal growth (Sharpley, 1993; Søndergaard et al., 2001), particularly in lakes where inflowing rivers deposit nutrient-enriched sediment on the lake floor. Phosphorus release at the sediment-water interface may occur in the following conditions: (1) during periods of anoxia or hypoxia (Theis and McCabe, 1978; Steinman and Ogdahl, 2008); (2) by wind-induced resuspension and bioturbation (Steinman and Ogdahl, 2008); or (3) when there is an increase in pH of the interstitial water (Sharpley and Rekolainen, 1997; Daly, 1999).

Soil cultivation is a major factor contributing to an increased risk of PP transfer to water, but when reduced cultivation such as non-plough tillage is practised to decrease losses of PP, there can be a build up of P near the soil surface, which increases the risk of DRP loss in surface runoff. Disturbance of soil structure by tillage operations also increases aggregate dispersion and the degree of interaction between soil and runoff water, thereby enabling more dissolved P to be mobilised from soils with high P (Sharpley et al., 2001a). Diffuse P loss from arable land can be as high as 1-2 kg P ha<sup>-1</sup> yr<sup>-1</sup> in the northern temperate zone, especially in areas with widespread soil erosion (Ulén et al., 1991).

### **2.3.6 Hydrological pathways of phosphorus transport**

The hydrological pathways of P movement from fields include surface runoff comprising overland flow, and subsurface flow comprising preferential flow, interflow, groundwater discharge and drainflow (Brogan et al., 2001). All of these pathways have the potential to contribute to P export depending on the connectivity with adjacent water bodies (Doody et al., 2012). It is the landscape position of a P source, both in terms of its upslope contributing area and its downslope flow path, that determine the likelihood of a connection being made to receiving waters. Surface and near surface pathways can be considered as the main link between source and delivery of P (Wall et al., 2011). Both particulate and dissolved P can be lost via these pathways. Fingerprinting studies show that the majority of SS in rivers is derived from the surface soil (Walling, 2005). In the surface pathway, overland flow is the

main pathway of diffuse P loss from agricultural soils in Ireland (Kurz et al., 2005; Tunney et al., 2007) following hydrological (storm) events. Surface runoff has a strong affinity for P transport because the surface soil has the greatest effective depth of interaction and the highest concentrations of P (Heathwaite et al., 2005). However, research has shown that P export from catchments can also occur via subsurface pathways, with the most significant instances of subsurface movement of P being associated with excessive application of P in manure and fertiliser (Sims et al., 1998). In general, the transport of PP in subsurface flow is not large. Particulate phosphorus is the dominant P fraction exported from arable land during overland flow events due to soil erosion (Doody et al., 2012).

When attempting to identify the primary flow paths of nutrient transport in fields where surface pathways dominate, attention should be paid to surface runoff, flow in tramlines and tyre tracks, and flow along roads or other impermeable features (Heathwaite et al., 2005). In cases where subsurface pathways dominate, attention should be paid to land drains, near surface interflow, deeper subsurface storm flow and groundwater flow (Heathwaite et al., 2005). These pathways may or may not be activated depending on criteria such as antecedent moisture, topography, rainfall intensity and duration (Heathwaite and Dill, 2000). The identification of primary flow paths of nutrient transport is essential for water quality protection because these routes form the critical link between P sources and P outputs measured in streamflow. Furthermore, the introduction of pipe drainage systems and tramlines, the presence of compacted headlands and gateways and/or the proliferation of ditches, tracks and roads, has greatly lengthened the distance over which sediment and P can be transported before reaching a water body (Withers et al., 2007). In situations where reducing soil P to environmentally acceptable levels will take many years of restricted fertiliser use (based on nutrient management planning and soil P testing), methods of flow path manipulation (such as buffer zones or sediment traps) that reduce connectivity between P source and receiving waters should be used, as these can immediately reduce the amount of P reaching surface waters. In a study by Schulte et al. (2009) to identify the dominant P pressure and pathway risks governing P loss in the catchment, and to evaluate and select potential mitigation measures based on an assessment of cost-effectiveness and farmer preference, installation of sediment traps



in drainage ditches was identified as the most cost-effective and popular measure aimed at reducing P transport vectors in the short term.

### **2.3.7 Rainfall simulation as phosphorus research tool**

The expensive nature of field experiments and inherent variability in natural rainfall has made rainfall simulators and laboratory microcosms a widely used tool in P transport research (Hart et al., 2004). Due to the complexity of soil erosion by water, field experimentation can be complimented by hypothesis testing in controlled reductionist laboratory experiments. Conclusions drawn from these reductionism experiments help to ‘reduce the uncertainty in explanation of complex patterns’ occurring at field- and catchment-scale (Haygarth, et al., 2005). While there are still some reservations regarding the use of simulated rainfall in place of natural rainfall (Potter et al., 2006), there is widespread support for the use of rainfall simulation experiments to obtain some estimate of the magnitude of potential losses from different land management systems, soil types, and landscapes (Pote et al., 1999; Sharpley et al., 2001b; Bundy et al., 2001; Schroeder et al., 2004; Tarkalson and Mikkelsen, 2004; Little et al., 2005). Numerous studies – outside of Ireland – have utilised rainfall simulation to evaluate nutrient losses in runoff from tillage systems (Zhao et al., 2001; Daverede et al., 2003; Franklin et al., 2007). Studies have also been conducted using laboratory rainfall simulation on runoff boxes packed with tillage soil to predict runoff of SS and PP using simple soil tests (Udeigwe and Wang, 2007), and to examine variability in mobilisation and transport of nutrients and sediment by overland flow across a range of soils (Miller et al., 2009). In addition, flume studies using concentrated overland flow as opposed to simulated rainfall have been used by Knappen et al. (2008) to show that the effect of conservation tillage on soil detachment rates is a result of soil property modifications affecting soil erodibility, rather than a result of the surface residue decreasing flow erosivity. Laboratory-scale work such as this is essential in understanding erosion processes and in selecting suitable erosion prevention measures for further testing at larger scales.

## 2.4 Phosphorus loss risk assessment tools

### 2.4.1 Phosphorus risk index

The original P risk index of Lemunyon and Gilbert (1993) was designed as a screening tool for use by field staff, catchment planners, and farmers to rank the vulnerability of sites to P loss in surface runoff. It is a simple, field-scale analysis, which integrates soil test data, soil erosion and runoff potentials; and P fertiliser or organic waste application rate, method, and timing (Sims et al., 1998). It was developed with a caveat that it would require modification to account for regional variations in agricultural management practices, climate, topography, hydrology and surface water characteristics (Hughes et al., 2005). The majority of P indices currently in use are modified versions of the Lemunyon and Gilbert (1993) index that have been made suitable for local conditions where they are being applied. The widespread adoption of the indexing concept in America (at least 49 states) shows the consensus among scientists, the fertiliser industry and policymakers with regard to the validity of the P risk index approach (Sharpley et al., 2003b). Phosphorus risk indices have also been developed in Sweden, Ireland, Norway and Denmark.

Despite its increased popularity, surprisingly few studies have been carried out to show that the ranking of a P risk index actually reflects the ranking of P transfer from fields (Bechmann et al., 2007). In Ireland, Magette (1998) developed a P ranking scheme for Irish conditions which was modified by Magette et al. (2007) and is based on the P risk index approach. Hughes et al. (2005) tested the P ranking scheme on 3 fields in Ireland and found that it correctly predicted the rank order of P losses from the fields. At the plot scale, Sharpley et al. (2001b) accurately predicted the potential for dissolved P loss ( $R^2=0.79$ ) and total P loss ( $R^2=0.83$ ) from manured plots (2 m<sup>2</sup>) using the Pennsylvania P index. As this was a plot scale study, factors associated with the off-site transport of P could not be assessed. At the subcatchment/field scale, Bechmann et al. (2007) correctly ranked catchments in terms of potential P transfers ( $R^2=0.66$ ) using the Norwegian P index which is based on the Pennsylvania P index. The Norwegian P index was also tested using monitoring data for six catchments

(ranging in size from 168 – 8700 ha) in Norway, which showed that 79% of the variation in TP losses was explained by the P index rating.

The modified P ranking scheme of Magette et al. (2007) was designed specifically for use in catchments in Ireland (the majority of which are dominated by grassland use), and, as such, the soil erosion factor used in it, only provides a coarse estimate of the risk of P loss due to erosion. The erosion risk factor is based on a field being classed as either well managed pasture (low risk), poorly managed pasture (medium risk), no-till crop systems (medium risk), or row crops under tillage (high risk). Most versions of the P risk index used in America utilise the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1994) to estimate annual soil loss in fields where a P risk index is to be determined. In presenting a ‘conceptual framework for a CSA approach to the development of supplementary measures for mitigating diffuse P export in Irish catchments under the EU WFD’, Doody et al. (2012), proposed that data collected using the modified P ranking scheme of Magette et al. (2007) be used to characterise CSAs in Irish catchments so as to identify the key factors controlling P loss to water. If the CSA approach, proposed by Doody et al. (2012), is to distinguish between key factors controlling P loss to water, from grassland and arable land, then the risk of P loss associated with eroded soil needs to be better accounted for by using RUSLE to estimate soil loss.

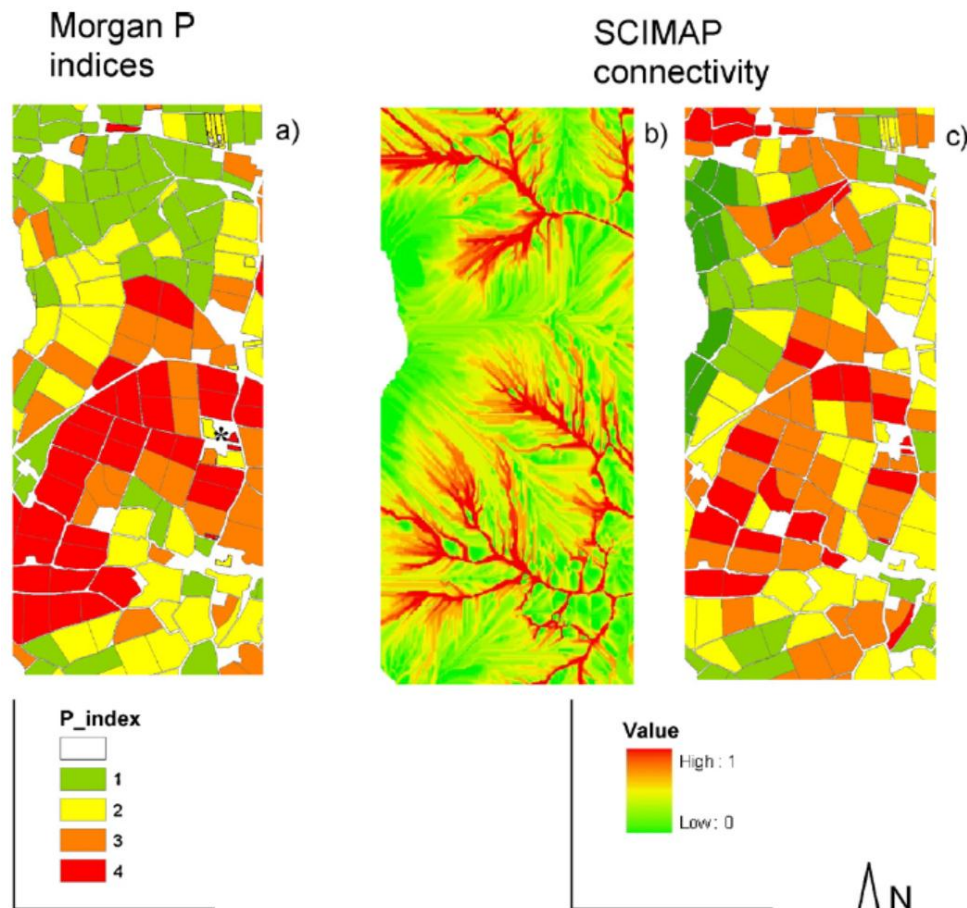
#### **2.4.2 SCIMAP (Sensitive Catchment Integrated Modelling and Analysis Platform)**

Given an observed downstream water quality degradation, and provided this can be attributed to diffuse sources, the primary challenge is to determine which parts of the landscape (subcatchments, farms, or fields) are most likely to be contributing to that degradation (Reaney et al., 2011a; Lane et al., 2006), and are therefore CSAs. The Sensitive Catchment Integrated Modelling Analysis Platform (SCIMAP) provides a framework for understanding the probable spatial origins of diffuse pollution problems within agricultural catchments. It is based upon a conception of catchments as organising entities: catchments can be conceptualised as a set of flow paths that accumulate distributed sources of possible contaminants from across the landscape

into receiving waters where, for surface waters, diffuse pollution may become ‘visible’, either through detection of temporal changes in water quality via routine monitoring (e.g. elevated nitrate concentrations) or through the more limited evidence from physical water quality deterioration (e.g. algal blooms; or long-term changes in ecological quality) (Lane et al., 2009; Reaney et al., 2011a; Milledge et al., 2012). As such, it focuses on the question of ‘where in the catchment is the pollution load coming from’ rather than aiming, as many models do, to produce estimates of the actual pollution load.

SCIMAP uses a simple approach for determining the probable relative risk of a point in the landscape producing pollution (Lane et al., 2006). The SCIMAP risk mapping framework comprises two dimensions of analysis: the delimitation of hydrologically connected source areas or CSAs; and the accumulation of these CSAs through to locations of concern (Lane et al., 2006). Surface hydrological connectivity is assessed through analysis of the potential pattern of soil moisture and saturation within the landscape. Assessment of the ability of each point in the landscape to generate saturated overland flow is done using the prediction of the spatial pattern of soil moisture and allows the probability of continuous flow to the river channel network to be assessed. The export of risk (e.g. sediment) in surface flow from a point on the landscape is dependent on each downslope point also being saturated, otherwise the risk will be captured and not reach the river channel. The total risk that a point on the landscape represents is a function of the risk of connectivity to the river channel and the point scale risk (e.g. sediment). These risks are accumulated through the catchment from field through to river so as to identify and map points on the landscape where there is a high risk of diffuse pollution impacting on the aquatic receptor. It should be noted that SCIMAP’s hydrological treatment is most suited to a surface and shallow subsurface flow regime, where residence times are short and flows are predominantly lateral rather than vertical (Milledge et al., 2012). A full description of the SCIMAP model is provided in Reaney et al. (2011a), who show how it can be used to understand the relationships between land use, hydrological connectivity and salmonid fry abundance.

A range of mitigation measures are available to reduce sediment and P loss from arable land (Kronvang et al., 2005; Deasy et al., 2009 and 2010; Stevens et al., 2009) and decision support tools are now required to help target these measure most effectively (Withers et al., 2007). The SCIMAP approach can be rapidly applied to locate mitigation measures within the landscape in a systematic, targeted and cost effective way (Reaney et al., 2011b). It is not surprising, therefore, that a high resolution digital elevation model (5 m  $\times$  5 m) and the SCIMAP approach were recently used by Wall et al. (2011) to distinguish areas within catchments of the ACP with the propensity for low to high hydrological connectivity. The authors combined soil P data with hydrological connectivity outputs from the SCIMAP assessment to get an estimate of CSAs within the catchment at a scale of less than 2 ha (Figure 2.7).



**Figure 2.7** Field-by-field soil P status (left) collated by index and showing below optimum to the north and above optimum to the south. The SCIMAP connectivity map (centre) on a 5 m pixel scale is also collated as a mean score per field (right) and indicates CSA potential in the south where the propensity for runoff (score closer to 1) is higher (from Wall et al. (2011)).

## **2.5 Mitigation measures to prevent sediment and phosphorus loss from tillage soils**

Research to evaluate the effectiveness of well-established mitigation options for prevention of soil erosion and reduction of P loss from arable land was carried out in studies by Chambers et al. (2000), Koskiaho (2002), Quinton and Catt (2004), Ulén and Jakobsson (2005), Kronvang et al. (2005), Knappen et al. (2008 and 2009), Deasy et al. (2009), Stevens et al. (2009) and Silgram et al. (2010).

### **2.5.1 Soil and land management to prevent erosion**

Various land management practices have been shown to minimise erosion risk on susceptible soils: low erosion risk crops and cover crops, tillage timing and intensity, and the use of buffer strips (Creamer et al., 2010). For example, intensively cultivated soils amended with spent mushroom compost, a bi-product of the mushroom growing industry in Ireland, exhibited improved structural stability as measured by an aggregate stability (AgSt) test (Curtin and Mullen, 2007). The UK Department for Environment, Food and Rural Affairs (Defra) highlights potatoes, winter cereals, sugar beet, maize and grazed fodder crops as having the highest erosion risk based on crop cover (Defra, 2005). To minimise erosion risk on susceptible soils, low risk crops like oilseed rape (OSR), which establish a crop cover earlier, should be sown (Chambers and Garwood, 2000). Furthermore, winter barley may be more beneficial than spring barley, as it provides winter cover. However, wet weather trafficking may offset benefits.

Minimum (or minimal) tillage, which involves shallow cultivation to a maximum depth of 10 cm using a tine cultivator, helps conserve SOM, promotes AgSt and thus reduces erosion (Quinton and Catt, 2004). In Ireland, minimum tillage normally involves: (1) shallow cultivation using a tine cultivator or disc harrow to a depth of 75-100 mm immediately followed by rolling; (2) spraying with herbicide a few days prior to sowing, following a stale seedbed period of a number of weeks (where possible) to eliminate volunteers and established weeds; and (3) sowing with a cultivator drill to a target depth of 40 mm (Forristal and Murphy, 2009). To date, the

effectiveness of minimum tillage to reduce erosion has not been investigated in Ireland. Research in the UK by Deasy et al. (2009) found that for 5 site-years, trialled losses of SS and TP decreased by an average of 151 kg SS ha<sup>-1</sup> and 0.3 kg TP ha<sup>-1</sup> under minimum tillage, compared to traditional plough cultivation. Contour grass strips have received some research attention and have been shown to reduce sediment losses (Stevens et al., 2009) by reducing slope length and by acting as a barrier to slow down overland flow. Deasy et al. (2010) found that although minimum tillage, crop residue incorporation, contour cultivation and beetle banks (raised vegetative barriers placed on the contour) all have potential to be cost effective mitigation options for SS and TP losses, tramline management (disruption of the compacted tramline surface to a depth of 60 mm with a tine) is one of the most promising treatments for mitigating diffuse pollution losses as it was able to reduce sediment and TP losses by 72-99% in four out of five site years trialled. As a management practice to reduce P loss from tillage soils in Ireland, Carton et al. (2002) advised that attention be paid to tramline compaction and that if soils become severely compacted, corrective action, such as subsoiling, should be taken where appropriate.

The Nitrates Directive (91/676/EEC), as implemented in Ireland, sets out crop cover requirements where arable land is ploughed between 1<sup>st</sup> July and 30<sup>th</sup> November. The regulations require that the owner/occupier take appropriate measures to provide for emergence of green cover from a sown crop within 6 weeks of ploughing. In the UK, as part of the cross compliance regime (Defra, 2006a), farmers are further required to carry out a field erosion risk assessment as a means of reducing risk to acceptable levels. The validity of this approach to erosion risk identification was verified by Boardman et al. (2009). Conservation tillage in autumn may reduce losses of soil and PP by improving soil structure. In Norway, ploughing and shallow cultivation of sloping fields in spring, instead of ploughing in autumn, have been shown to reduce particle transport by up to 89% on highly erodible soils (Ulén et al., 2010). Rational land use policies such as the promotion of ‘set-aside’ on erodible soils, use of grass strips on erodible arable slopes, and buffer strips in riparian zones were identified as mitigation options to reduce soil erosion by Fullen et al. (2003).

There are some preventative measures in place to prevent land degradation processes from arable agriculture (Table 2.1). In Ireland, farmers protect vulnerable tillage soils by complying with ‘good agricultural and environmental condition’ guidelines as a condition for receipt of the area-based single farm payment under the EU cross-compliance regime (DAFF, 2005). Land that’s been in continuous tillage for six years or more must be tested for OM content as a requirement for the single payment scheme. Soils having less than 3.4% SOM may require remedial action depending on soil type. As the process of building up SOM is very slow, the remedial action to be taken is set out over a 10-yr period. The remedial action will continue until such time as the OM levels are shown to have recovered to greater than 3.4%, or a level deemed acceptable for that soil type. Hackett et al. (2010) provide information on how various management practices affect SOC dynamics in arable soils. Land application of fertiliser and manures is now subject to ‘closed periods’ that coincide with the most frequent average occurrence of transport vectors. Farmers are also prohibited from applying fertilisers in close proximity to a watercourse. ‘Buffer strips’ of 1.5m and 5m for mineral fertiliser and organic fertilisers, respectively, must be observed. The effectiveness of these aspects of the regulations is currently being monitored in the ACP (Schulte et al., 2010b).

Soil data currently available in Ireland exists in variable forms and is not fully mapped at the target European scale (1:250,000). Digital soil mapping, combined with conservative ground-truthing, is currently underway in the form of the Teagasc (Irish Agriculture Food and Development Authority) and EPA funded Irish Soil Information System (ISIS). This aims to complete the soil map of Ireland at a 1:250,000 scale (Daly and Fealy, 2007), by the year 2014, by generating knowledge-based predictive soil maps using digital terrain data, subsoil maps and other geo-spatial layers in an advanced GIS technology platform. The models generated will be calibrated and verified through an intensive two-year traditional field sampling campaign which will provide hard soils data on 300 new reference profiles and over 3,000 auger points across the country. In addition to the 1:250,000 soil map of Ireland will be an associated digital soil information system which will be fully open and accessible to all. The project is ground-breaking, as no other country has adopted such a



complimentary approach of combining novel digital techniques with ground-truthing using traditional soil survey methodologies at a National scale (Creamer, 2010).

In a review of strategies to improve soil conservation in Europe, Fullen et al. (2006) identified several best management practices including: initiation of national soil conservation services; and full mapping, monitoring and costing of erosion risk by national soil survey organisations. If the SFD is eventually ratified, Ireland will be required to identify areas where erosion has occurred in the past or is likely to occur in the future. At that time, the soil information provided by the ISIS will be essential in identifying these areas.

### **2.5.2 Developing phosphorus management guidelines for water quality protection**

The relationship between STP in tillage soils and DRP concentration in runoff water needs to be adequately understood and quantified for local soils (Wright et al., 2006). To date, in Ireland, no study has investigated the link between STP and P loss to water from tillage soils. Guidelines presently used in Ireland are based on international findings and agronomic nutrient advice. Determination of upper critical limits for P in soil should consider both the STP necessary for economic crop production and the STP necessary to avoid excessive P loss due to erosion, surface runoff and leaching. This is essential for the development of P management guidelines for water quality that will satisfy the requirements of the WFD. Relationships developed between runoff P and STP have been used in Europe and the USA to establish threshold STP levels above which the potential threat of eutrophication in surface waters is unacceptable (Sibbesen and Sharpley, 1997; Sims et al., 2002).

In a study to evaluate Mehlich-3 P (M3-P) as an agri-environmental soil P test for the Mid-Atlantic USA, Sims et al. (2002) concluded that agronomic soil tests, such as M3-P, can be used to guide environmentally-based P recommendations, and that higher risks are clearly associated with M3-P values that are in excess of concentrations needed for economically optimum crop yields. As a result of the WFD, there is increasing pressure in Europe and Ireland to develop P-based management

practices that will reduce the risk of diffuse losses from agricultural land to surface waters. Modelling of P for grassland undertaken by Schulte (2006b) showed that it was possible to change the range of the target P index from 6 - 10 to 5.1 - 8 mg L<sup>-1</sup> P<sub>m</sub> (Table 2.2), while still facilitating optimum productivity and herbage quality and minimising the risk of diffuse P losses to water. Index 3 (5.1 - 8 mg L<sup>-1</sup> for grassland) in the new P-index system (Table 2.2) represents a target index that is both agronomically and environmentally sustainable for all soils (Schulte, 2006b) in Ireland. The target index for tillage crops (6 - 10 mg L<sup>-1</sup>) has not changed and it is uncertain if similar work on tillage soils is necessary, as the risk of diffuse P loss from them has not been quantified in Ireland.

The adoption of management measures in river basins requires the ability of river basin managers to quantify the importance of different P pathways, identify and map P risk areas with a certain spatial resolution, and estimate the effect of various management measures for changes in P losses (Kronvang et al., 2005). Limited resources and time will likely hinder the carrying out of a full P loss assessment (incorporating site characteristics and nutrient management practices) on all agricultural fields in a catchment. Therefore, in the interim, there is a need to identify a STP level, sometimes referred to as an environmental threshold, above which the improvement of P management practices should be a high priority.

### **2.5.3 Catchment-scale research**

Research that will quantify the P and sediment losses associated with arable land compared to agricultural grassland in Ireland is underway in the form of the ACP. This will provide a scientific evaluation of the effectiveness of the Nitrates Directive National Action Programme measures over time for the major farming and environmental stakeholders in Ireland. The programme is designed to assess effectiveness of measures well before improvements are expected to translate into improved water quality of the final aquatic receptors, which in some cases may take up to 20 yr (Schulte et al., 2010b). In the first stage, four catchments (2 arable and 2 grassland) were selected for studying from 1500 possible candidates using spatial multi-criteria decision analysis (Fealy et al., 2010). Combined, the four catchments

represent the range of intensive grassland and arable agriculture interests in Ireland across a soil and physiographic gradient that defines potential risk of P and/or N transfers (Fealy et al., 2010). A fifth catchment in a karst limestone region in the west of Ireland is also being studied. The arable catchments, having between 30 and 50% arable land use in each, are located in County Louth/Cavan on intermediately drained soils and in County Wexford on well-drained soils, enabling measurement of storm-induced diffuse transfers of P and losses of N to groundwater through leaching.

The ACP will focus on source, pathways and delivery of nutrients to waterways over time. At the outlet of each catchment, the following parameters are being monitored: TP, total dissolved P (TDP), total reactive P (TRP), DRP, total N, nitrate ( $\text{NO}_3\text{-N}$ ), turbidity, electrical conductivity, temperature, and flow rate. Particular attention is being paid to P hotspots (fields at soil P index 4) and linking these to P loads in streams. This will facilitate the identification of areas that are vulnerable to P loss and which will require measures to reduce losses. On-site bank-side nutrient analysers (Jordan et al., 2007) will enable immediate analysis of nutrients susceptible to transformation if left in sample bottles for long periods of time. Novel methodologies will be used to quantify the amount of sediment leaving a catchment and relate this to the source of the sediment and to specific areas and land uses. Measurements of turbidity and electrical conductivity are also monitored to provide ancillary information of sediment associated nutrient flux, pollution spikes and water flow pathways (surface vs. sub-surface) (Wall et al., 2011). More detailed information on the methodological design of the ACP and preliminary results can be found in Wall et al. (2011).

Information on soil erosion and P loss across different land uses (e.g. tillage and grassland) and its effect on water quality at catchment-scale will help Ireland meet the requirements of the WFD. Detailed analysis of catchment characteristics, assessment of risk to water bodies, further analysis of existing information and collection of new data are all needed to support the implementation of the WFD (Irvine et al., 2005). Given that there is still much to understand about the complex relationship between the catchment and the movement of sediment and P, and the response of the aquatic ecosystem to anthropogenic impacts, modelling that can elucidate key variables and

predict responses is a valuable tool (Irvine et al., 2005). A review of available models for modelling soil erosion and sediment and phosphorus delivery to surface waters at the catchment scale is provided in Appendix B. This includes a comparison of model predictions with measured export of P and SS from a number of Irish and international catchments.

## **2.6 Future research direction in the quantification of phosphorus and sediment loss from Irish tillage soils**

### **2.6.1 Sediment provenance**

Traditional techniques, aimed at identifying the source and the pathway of the sediment, have included methods such as risk assessments, field observation and mapping (Lao and Coote, 1993), landowner questionnaires (Krause et al., 2008), remote sensing (Vrieling, 2006), use of erosion pins (Lawler et al., 1997), and terrestrial photogrammetry (Barker et al., 1997).

Given the time and cost involved in establishing and operating plot experiments, and that data available from them is limited, attention has been directed to the use of environmental radionuclides for documenting erosion rates (Sepulveda et al., 2008). By comparing the fallout radionuclide Caesium-137 ( $^{137}\text{Cs}$ ) inventory at a particular sampling point with the reference inventory (the total  $^{137}\text{Cs}$  activity per unit surface area for a level, stable undisturbed site), the rates of soil erosion and deposition at that point can be estimated. Measurements of  $^{137}\text{Cs}$  and unsupported  $^{210}\text{Pb}$  afford a means of obtaining retrospective, medium-term (i.e. ca. 45 years for  $^{137}\text{Cs}$  and up to 100 years for unsupported  $^{210}\text{Pb}$ ) estimates of both the magnitude and spatial distribution of soil redistribution rates generated by sheet and rill erosion, by means of a single site visit (Blake et al., 1999). Due to its long retention time on soil particles once absorbed,  $^{137}\text{Cs}$  ( $t_{1/2} = 30.1$  yr) has the disadvantage of not being suitable for the investigation of erosion resulting from individual events occurring over short periods, and is unable to distinguish between tillage and water erosion. It can, however, be used to estimate changes in soil erosion rates associated with changes in soil management practices on cultivated land (Schuller et al., 2004). In contrast to  $^{137}\text{Cs}$ ,

radioactive Beryllium-7 ( $^7\text{Be}$ ) is short-lived with a half-life of only 53 days and, as such, is ideal for estimating short-term rates and patterns of soil redistribution relating to individual events (tillage or water erosion) or short periods.

Because the radionuclides  $^{137}\text{Cs}$ ,  $^7\text{Be}$ , and  $^{210}\text{Pb}$  have different distributions in the soil profile, their measurement in eroded sediment, referred to as 'sediment fingerprinting', will determine what depth in the profile the soil was eroded from and, hence, the depth and areal extent of sheet and rill erosion can be quantified as was done in a study by Whiting et al. (2001). Sediment fingerprinting is a method to allocate sediment nonpoint source pollutants in a watershed through the use of natural tracer technology with a combination of field data collection, laboratory analyses of sediments, and statistical modelling techniques (Davis and Fox, 2009). When estimating sediment erosion rates, sediment fingerprinting has the added advantage over plot studies of identifying both the source and fate of eroded sediment, which has significant implications for the development of best management practices to address soil erosion and sediment delivery to waterways.

In a study of one of Northern Ireland's prime salmon rivers (the River Bush) aimed at quantifying fine sediment loads and tracing in-stream fine sediment sources using sediment fingerprinting, Evans et al. (2006) were able to rank the four main agents generating those sources, which were (in order of importance): drainage maintenance work, bank erosion (caused by increasing flow and livestock poaching), ploughed arable land, and forestry clearfell. Ploughed arable land was found to be responsible for 36.6% of the suspended load and 7.5% of the bed load measured in the River Bush over a 1-year period. Evans et al. (2006) commented that the most likely mechanisms for transfer of topsoil to the river channel were after ploughing prior to planting and harvesting of the crop. The best management practices recommended for the Bush catchment to reduce sediment delivery from arable land by reducing bare ground were: (1) critical area planting on land prone to long-term soil erosion; (2) planting at appropriate times as assessed on the basis of storm forecasting; and (3) vehicle movement limited across fields prone to soil erosion. Unfortunately, as Evans et al. (2006) recognised, the 1-year period of monitoring in this project was too short to provide a reliable picture of sediment dynamics in the Bush catchment. An EPA

Strive funded project in conjunction with the Agri-food and Biosciences Institute Northern Ireland (AFBINI) is currently underway that will use sediment fingerprinting techniques to determine CSAs of sediment in two catchments in Co. Down and Co. Louth. A similar project is underway as part of the ACP.

## **2.7 Summary**

This chapter reviewed the current state of research and regulations on diffuse P and sediment losses from tillage soils, and provided a review of the processes controlling the mobilisation, transport and fate of P and sediment. An examination of the key threats to soil quality associated with tillage soils, and the methods used to model and quantify P loss and soil erosion was also detailed.

Modelling of water and tillage erosion rates in Ireland suggests that soil is being lost at a rate greater than it can be replenished by natural soil formation. This has significant implications for the sustainability of crop production. Furthermore, the occurrence of erosion adjacent to waterways may result in the transfer of P and sediment to them. Therefore, there is a need for laboratory- and field-scale research on tillage soils in Ireland to determine the extent of erosion and associated P loss. Given that a large proportion of P exported from agricultural catchments on an annual basis is generated from a relatively small portion of the catchment and during only one or two storm events, research to quantify P and sediment loss from Irish tillage soils should utilise high intensity rainfall typical of summer storm events.

As P is often the limiting nutrient for eutrophication in surface waters (Jarvie et al., 1998), river basin managers need to reduce P losses from agricultural land by adopting plans for mitigation strategies. The ability to identify CSAs of P loss is essential if mitigation measures are to be cost effective. The identification of an environmental soil P threshold, above which surface runoff from tillage soils may have a negative impact on water quality, will help Ireland meet the requirements of the WFD.

## **Chapter 3 Determining phosphorus and sediment release rates from five Irish tillage soils when subjected to simulated rainfall and increasing overland flow rates**

### **Overview**

A controlled laboratory flume study, used to provide experimental data on the release of P and sediment from 5 Irish tillage soils when subject to a simulated rainfall intensity of  $30 \text{ mm hr}^{-1}$  and overland flow rates of 225 and  $450 \text{ ml min}^{-1}$ , is presented in this chapter. The impact of slope, time between storm events and overland flow rate on P and sediment release from the study soils is also examined. The contents of this chapter are, in part, published in the *Journal of Environmental Quality* (39:185-192, 2010).

### **3.1 Introduction**

Phosphorus loss in surface runoff from soils is an important pathway in many agro-environments (Sims et al., 2002; Wright et al., 2006). A survey of 1151 rivers in Ireland from 2004 to 2006 (Clabby et al., 2008) estimated that the amount of pollution attributed to agriculture was approximately one-third. The loss of fertile topsoil due to soil erosion on agricultural land is a growing problem in Western Europe, and has been identified as a threat to soil quality and the ability of soils to provide environmental services (Boardman et al., 2009). Boardman and Poesen (2006) estimated that arable agriculture accounts for approximately 70% of soil erosion in Europe. It has numerous effects on soil, including thinning by removal of topsoil, textural coarsening, decline of SOM and loss of nutrients (Guerra, 1994).

Soil erosion is also associated with P transfer by overland flow, especially from arable land, where PP is the dominant P fraction exported (Doody et al., 2012). The susceptibility of arable land to P losses by erosion and overland flow is largely a result of land being left bare for periods of the year. An increase in winter cereal cropping

has exacerbated this problem, because it combines the period of maximum rainfall with long periods of bare soil (Leinweber et al., 2002). Furthermore, the disturbance of soil structure by tillage operations, increases aggregate dispersion and the degree of interaction between soil and runoff water, thereby enabling more dissolved P to be mobilised from soils with high P status (Sharpley et al., 2001a).

In Ireland, the main pathway of P loss from soils is via overland flow (Kurz et al., 2005; Tunney et al., 2007), which is greatest during storm events and is largely inactive at other times (EPA, 2008). Saturation excess overland flow (characterised by saturation of the soil over which it is moving) is the dominant type of overland flow generated under Irish conditions (Daly et al., 2000; Diamond and Sills, 2001), although research has shown that infiltration excess overland flow also occurs in Ireland (Schulte et al., 2006a; Doody et al., 2010). Infiltration excess overland flow occurs where the infiltration rate for a given soil profile is exceeded. The infiltration and saturation excess-generating mechanisms are not mutually exclusive on a watershed, nor even mutually exclusive at a point on a watershed (Smith and Goodrich, 2005). For many soil profiles, saturation excess overland flow is a special case of infiltration excess overland flow whereby infiltration is occurring, albeit at a negligible rate, because of the low hydraulic conductivity of the underlying strata (Nash et al., 2002). Where saturation excess and infiltration excess conditions combine, the result is a complex pattern of P loss (McDowell et al., 2012).

Both saturation and infiltration excess overland flow can occur on tillage soils provided the conditions necessary for it to occur are present. Tillage increases the initial infiltration rate, loosens the topsoil, disrupts soil aggregates and compacts the subsurface soil (Coles and Moore, 1998). This can result in a subsurface soil with much lower hydraulic conductivity than the surface soil, and may lead to saturation of the topsoil. The loose topsoil is then susceptible to erosion by saturation excess overland flow. Infiltration excess overland flow is common with cultivated soils, or where surface soil structure has degraded or consolidated to form a 'seal', but can also occur on unsealed soil surfaces, especially with high rates or amounts of rainfall (Rose, 2004). Factors that increase the volume, velocity and turbulence of overland flow, such as impaired infiltration, high intensity storms, run-on, reduced soil cover,



cultivation and high slopes, increase detachment compared with dissolution (Nash et al., 2002). Furthermore, steeper slopes increase the potential for runoff-dominated erosion due to faster flow threads and lower surface area connectivity (Armstrong et al., 2011).

The objective of this chapter was: (1) to quantify the amount of DRP, PP, TP and SS released into overland flow from 5 tillage soils of varying  $P_m$  when subject to a rainfall intensity of  $30 \text{ mm hr}^{-1}$  and overland flow rates of 225 and  $450 \text{ ml min}^{-1}$  applied in 3 successive events; and (2) to determine the impact, if any, of slope, time between storm events and overland flow rate on P and sediment release from the study soils.

### **3.2 The tillage soils, laboratory flume set-up, and analysis methods used in this study**

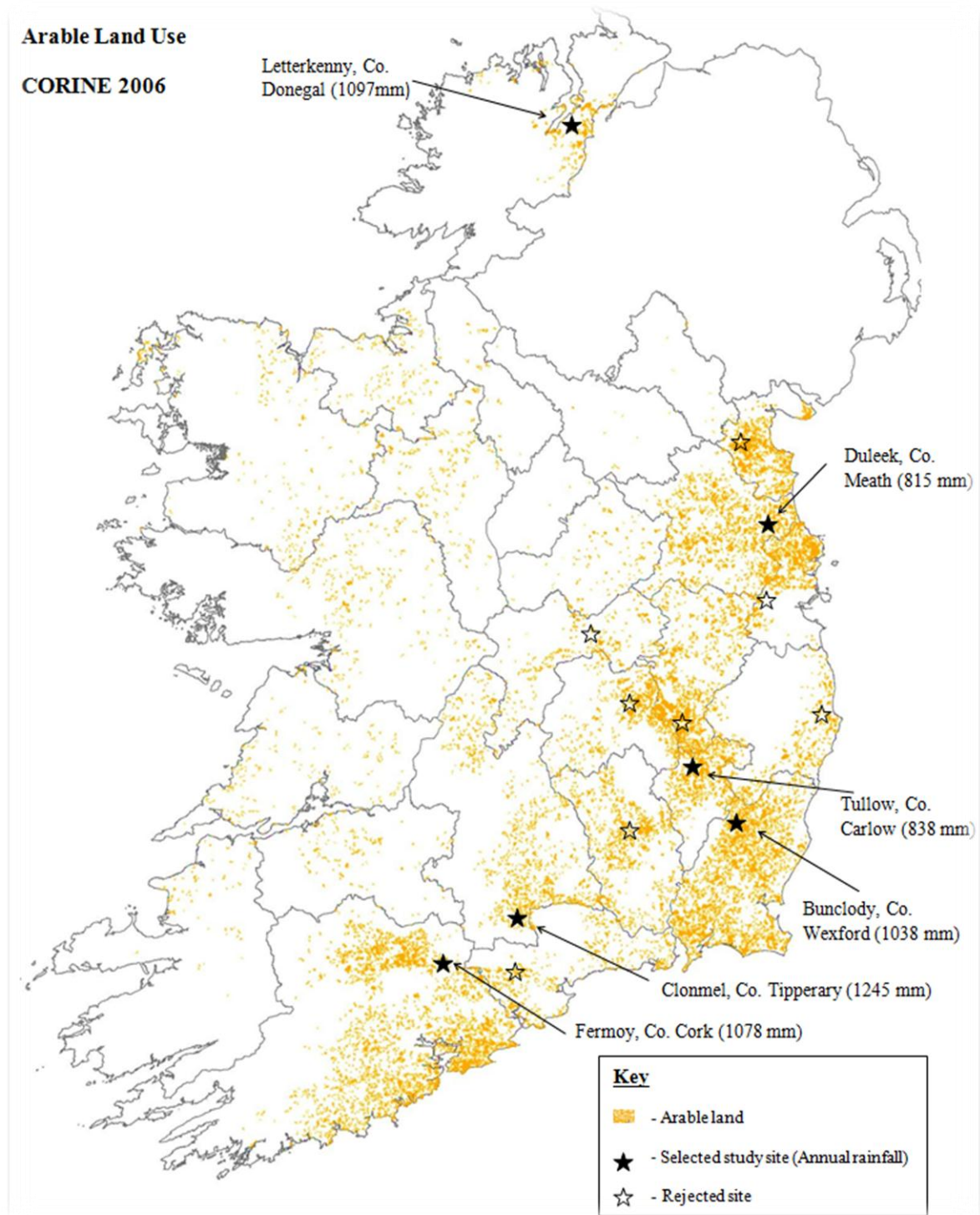
The USEPA National Phosphorus Research Project (NPRP, 2001) protocol uses soil-packed runoff boxes subjected to simulated rainfall to investigate the relationship between STP and DRP in surface runoff. Runoff boxes containing homogenised soil minimise significant variability in physical and chemical characteristics that may occur in field plots. They also facilitate large numbers of replications not possible at field-scale. Kleinman et al. (2004) found that regression coefficients of DRP in runoff and M3-P were consistent between grassed field plots and soil-packed boxes, but noted that runoff boxes may not be fully representative of field conditions. However, they concluded that despite large differences in rainfall, hydrology, and erosion between field plots and packed boxes, both can be used to produce comparable P extraction coefficients for process-based models and P site assessment indices.

#### **3.2.1 Soil collection and preparation**

Fourteen tillage field sites, spread across Ireland, were investigated to find suitable soils with wide ranging physical and chemical properties. After preliminary characterisation, 6 soils were then selected based on soil type, STP, particle size distribution (PSD), tillage history, and evidence of prior erosion problems. The soils

selected were from: (1) Tullow, Co. Carlow; (2) Clonmel, Co. Tipperary; (3) Letterkenny, Co. Donegal; (4) Bunclody, Co. Wexford; and (5) Fermoy Co. Cork; (6) Duleek, Co. Meath (Figure 3.1). The Duleek soil was not used in the simulated rainfall/overland flow study and is only considered in Chapter 5. The sites selected were in tillage for a minimum of 15 years. The soils had a  $P_m$  index of 1 to 4 and ranged from 2.8 to 17.5 mg L<sup>-1</sup> (Table 3.1). The  $P_m$  values broadly reflected the P fertiliser history of the 5 sites. The soils (Fermoy and Letterkenny) with low  $P_m$  (2.8 and 4.8 mg L<sup>-1</sup>, respectively) received P fertiliser below the recommended agronomic levels over their rotation history, while those (Bunclody, Clonmel and Tullow) with medium and very high  $P_m$  (7.1, 15.8 and 17.5 mg L<sup>-1</sup>, respectively) received P fertiliser above agronomic levels. The Clonmel and Tullow soils had a history of receiving above 40 kg P ha<sup>-1</sup> yr<sup>-1</sup> in excess of crop requirement.

A sample of the plough layer of each of the 5 tilled soils was collected, air-dried, sieved (< 5 mm), and thoroughly mixed for use in a rainfall simulation/overland flow study. This strategy was similar to that adopted by Miller et al. (2009). Other studies by Sharpley (1980) and Fang et al. (2002) used soils sieved to less than 4 mm in flume studies to determine the effect of storm interval on DRP in runoff and to estimate runoff P losses, respectively. Subsamples of each soil were further sieved (< 2 mm) for physical and chemical characterisation (Section 3.2.3).



**Figure 3.1** Arable land use in Ireland (CORINE, 2006) and selected/rejected soil sampling sites.

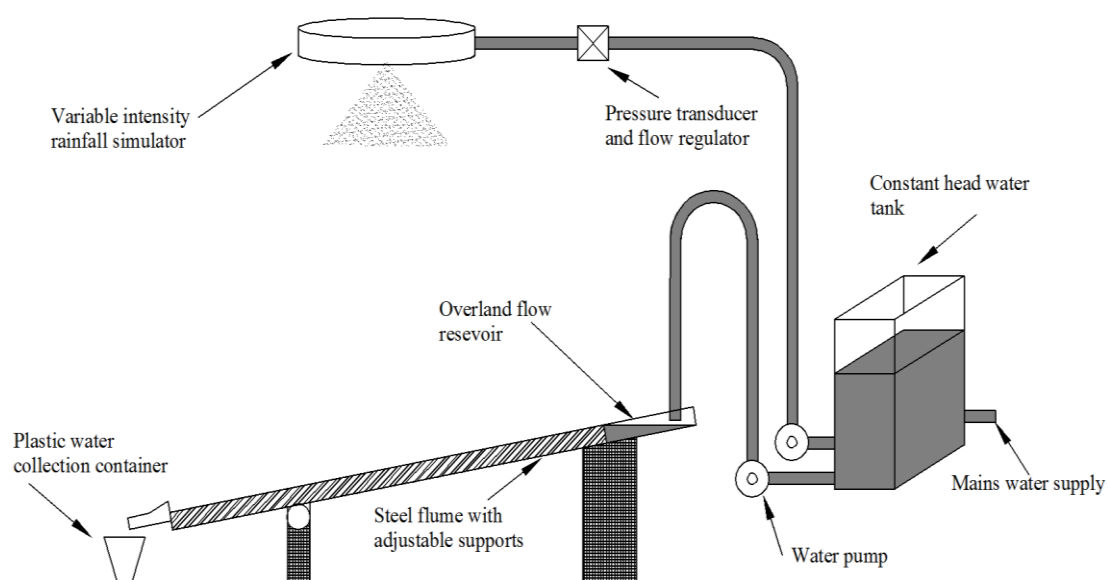
**Table 3.1 Chemical and physical properties of selected Irish tillage soils**

Location	Soil Type	pH	P <sub>m</sub> <sup>1</sup>	CEC <sup>2</sup>	AgSt <sup>3</sup>	CaCO <sub>3</sub>	OM <sup>4</sup>	Sand	Silt	Clay	M3-P <sup>5</sup>	Pcacl <sub>2</sub> <sup>6</sup>	WEP <sup>7</sup>	P <sub>ox</sub> <sup>8</sup>	Al <sub>ox</sub> <sup>9</sup>	Fe <sub>ox</sub> <sup>10</sup>	Psat <sub>ox</sub> <sup>11</sup>
			mg L <sup>-1</sup>	cmol kg <sup>-1</sup>	%			g kg <sup>-1</sup>						mg kg <sup>-1</sup>			%
Tullow, Co. Carlow	GBP <sup>12</sup>	6.9	17.5	13.4	96.4	5	49	579	267	154	96.3	3.0	11.5	566	1033	3482	36.2
Clonmel, Co. Tipperary	GBP	6.7	15.8	11.2	90.0	5	42	528	306	167	89.4	2.1	6.6	457	903	3867	28.8
Buncloidy, Co. Wexford	BP <sup>13</sup>	7.7	7.1	13.9	92.9	26	71	410	387	203	58.7	1.0	3.5	414	2560	4755	14.8
Letterkenny, Co. Donegal	BP	6.5	4.8	13.7	98.5	17	55	491	395	114	52.1	1.3	2.8	592	1700	5468	23.7
Fermoy, Co. Cork	ABE <sup>14</sup>	6.4	2.8	13.1	97.8	4	51	569	286	145	29.1	1.4	2.3	273	1227	3886	15.4

<sup>1</sup>P<sub>m</sub>, P determined by Morgan's extraction; <sup>2</sup>CEC, cation exchange capacity; <sup>3</sup>AgSt, aggregate stability; <sup>4</sup>OM, organic matter by loss on ignition; <sup>5</sup>M3-P, Mehlich-3 extractable P; <sup>6</sup>Pcacl<sub>2</sub>, calcium chloride extractable P; <sup>7</sup>WEP, water extractable P; <sup>8</sup>P<sub>ox</sub>, acid ammonium oxalate extractable P; <sup>9</sup>Al<sub>ox</sub>, acid ammonium oxalate extractable Al; <sup>10</sup>Fe<sub>ox</sub>, acid ammonium oxalate extractable Fe; <sup>11</sup>Psat<sub>ox</sub>, soil P saturation as determined by acid ammonium oxalate extraction; <sup>12</sup>GBP, grey brown podzolic; <sup>13</sup>BP, brown podzolic; <sup>14</sup>ABE, acid brown earth.

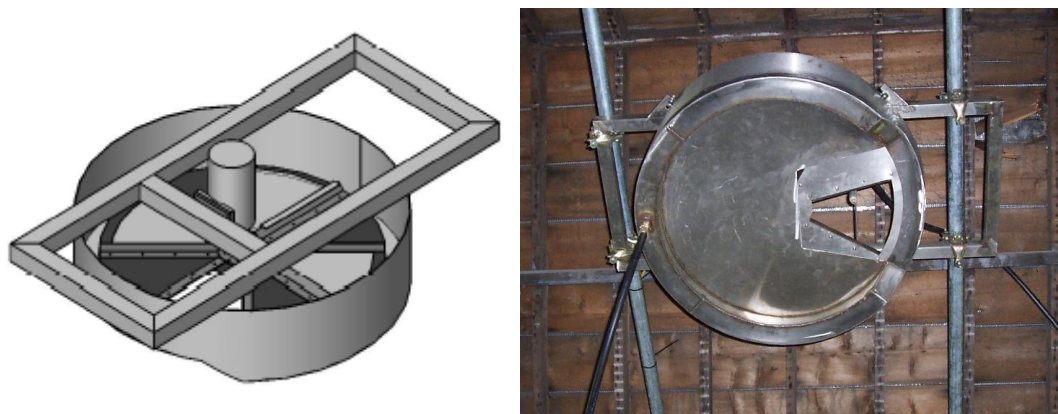
### 3.2.2 Simulated rainfall experiment

A laboratory scale runoff experiment was designed to compare the nutrient and sediment releases from the 5 study soils inclined at slopes of 10 and 15 degrees, when subjected to high intensity ( $30 \text{ mm hr}^{-1}$ ) simulated rainfall. In order to minimise the effects of soil variables on P release to runoff and give better control over hydrological and soil surface conditions, laboratory rainfall simulations were chosen to compare the nutrient and sediment releases from the 5 soils. As the study was focused on understanding process rather than soil management, this was considered to be a reasonable approach. The layout of the flume set-up is illustrated in Figure 3.2 (the overland flow reservoir shown is only used in the experiments described in Section 3.2.3). This experiment used two laboratory runoff boxes ('flumes'), 200-cm-long by 22.5-cm-wide by 5-cm-deep with side walls 2.5 cm higher than the soil surface, and 5-mm diameter drainage holes, drilled in triplicate and located at 300-mm-centres, in the base. Cheese cloth was placed at the base of each flume before packing to prevent soil loss through drainage holes. Runoff water was collected at the lower end of the sloped flume by a U-shaped aluminium trough equipped with a canopy to prevent rainfall water entering the runoff collection containers.



**Figure 3.2 Laboratory flume set-up for rainfall simulator/overland flow experiment.**

A rotating disc, variable-intensity rainfall simulator (after Williams et al., 1997), was constructed and calibrated for use in the rainfall simulation study. The rainfall simulator (Figure 3.3) consisted of a motorised rotating disk module for regulating the rainfall intensity and a single 1/4HH-SS14SQW nozzle (Spraying Systems Co., Wheaton, IL), which, at a pressure of 100 kPa, creates a distribution of drop sizes approximating natural rainfall (Bubenzer et al., 1985), with similar raindrop impact energy; i.e.  $260 \text{ kJ mm}^{-1} \text{ ha}^{-1}$  (nozzle) and  $240 \text{ kJ mm}^{-1} \text{ ha}^{-1}$  (natural). The rainfall simulator was attached to a 2.5 m by 2.5 m by 4.5-m-high metal frame, and calibrated prior to each experiment, to ensure that the rainfall intensity had not changed since the last experimental run. During calibration, the two flumes were placed under the rainfall simulator equidistant on either side of the nozzle so that each received approximately  $30 \text{ mm hr}^{-1}$  of rainfall. The total mass of water in each flume was used to determine how much rain had fallen on each flume (area receiving rainfall =  $0.45 \text{ m}^2$ ). The rainfall distribution in each flume was calculated by collecting rainfall for three 30-min periods in 18 identical cylindrical containers spread across the area of the flume. The mass of water in each container was determined and converted to a depth value ( $\text{mm h}^{-1}$ ) to obtain the intensity and distribution of the rainfall. The Christianson coefficient ( $C_u$ ) of application uniformity (Christianson, 1942):  $C_u = (1 - \text{average deviation from mean} / \text{mean depth of applied water}) \times 100$ , was used to evaluate depth distribution. A uniform depth distribution generates a  $C_u = 100$ . A  $C_u > 85\%$  was achieved in all calibrations.



**Figure 3.3 Rainfall Simulator (isometric drawing and photo of underside)**

Soils were packed to achieve an approximate bulk density of  $1.3 - 1.5 \text{ g cm}^{-3}$  (Figure 3.4). The packed soil was then saturated using the simulator, and left to drain for 24 hr



before the experiment commenced. A furnace filter was placed on the soil surface to protect the soil from raindrop impact during saturation. The furnace filter was removed before the start of the first rainfall event. All soils were approximately at field capacity before the first rainfall event (field capacity was determined to have been achieved once drainage from the base of the runoff box had ceased).



**Figure 3.4 Soil in laboratory flume before and after rainfall simulation.**

The return period for a  $30 \text{ mm hr}^{-1}$  rainfall event ranged from 30-100 years across the selected soil locations. This was based on the depth-duration-frequency model of Fitzgerald (2007). However, these return periods may be overestimated, given the ongoing changes in precipitation and storm frequency due to climate change. The 10-yr moving average for Ireland shows that rainfall amounts increased from 800 mm in the 1890s to 1100 mm in the 1990s (McElwain and Sweeney, 2006). Furthermore, Sweeney et al. (2008) modelled the effect of increased global emissions on the hydrology of nine Irish river catchments and concluded that the magnitude and frequency of flood events will increase due to climate change, with the greatest increases associated with floods of a higher return period. By the 2020s, three of the catchments modelled, in which there is a significant area under tillage (Blackwater, Suir and Barrow), showed an increase in the frequency of the 50-yr flood, with the same flood expected every 8.4 - 12.6 yr under a medium-to-low emission scenario and every 3.8 - 7.4 yr under a high-to-medium emission scenario. In effect, this means that high intensity rainfalls, such as the  $30 \text{ mm hr}^{-1}$  intensity investigated in this study, could potentially occur every 5 - 10 yr in the period 2020 - 2030. These projected decreases in the time period between extreme floods are likely to result in greater levels of erosion in tillage areas in Ireland. Large rainfall return periods are not

uncommon when investigating the effect of high-intensity storm events. A study by Vadas et al. (2005) used simulated rainfall intensities that represented storm return periods ranging from 5 - 50 yr.

The source water for the rainfall simulations was potable tap water with DRP, NO<sub>3</sub>-N, and ammonium-N (NH<sub>4</sub>-N) concentrations of < 0.005, 0.036, and 0.038 mg L<sup>-1</sup>, respectively. The tap water had an electrical conductivity = 0.421 dS m<sup>-1</sup>, measured using a conductivity meter, and a calcium cation (Ca<sup>2+</sup>), magnesium cation (Mg<sup>2+</sup>), and sodium cation (Na<sup>2+</sup>) concentration of 3.11, 2.24, and 22.55 mg L<sup>-1</sup>, respectively, measured by atomic absorption spectrophotometry. The Sodium Adsorption Ratio (SAR = Na/[(Ca + Mg)/2]<sup>1/2</sup>, where all concentrations are expressed in meq/liter) of the tap water was 2.38. Annual mean concentrations (in mg L<sup>-1</sup>) of Ca<sup>2+</sup>, Mg<sup>2+</sup>, and Na<sup>2+</sup> in rainwater were 0.85, 0.93, and 6.78, respectively, between 1992 and 1994 for the island of Ireland (Jordan, 1997). Over the same period, the SAR was 1.21 and the electrical conductivity ranged from 0.029 - 0.176 dS m<sup>-1</sup>. More divalent cations present in tap water than in natural rainfall may encourage larger aggregates in surface runoff (Aase et al, 2001). If this occurs, P losses in runoff using tap water could potentially be lower than for natural rainfall, since finer soil particles and aggregates have higher P concentrations than larger soil particles. The higher electrical conductivity of the tap water used here, compared to natural rainfall in Ireland, may result in lower P desorption from the study soils. However, Aase et al. (2001) found that average DRP concentrations in runoff from a calcareous soil using two different water sources, with electrical conductivities of 0.02 and 0.4 ds m<sup>-1</sup>, were equivalent to each other. Tap water has been used previously by Penn et al. (2006) when estimating dissolved P concentrations in runoff from three physiographic regions of Virginia and by McDowell and Sharpley (2002) when investigating P transport in overland flow. Most recently, tap water (0.005 mg P L<sup>-1</sup>) was used by Wang et al. (2010) when estimating DRP concentration in surface runoff from major Ontario soils.

Each rainfall simulation comprised 3 successive 1-hr rainfall events at time zero (Rainfall 1), 1 hr (Rainfall 2) and 24 hr (Rainfall 3) to determine the effect of storm interval on surface runoff. Previously, Sharpley (1980) used storm intervals of 5 and 30 min and 1-day when comparing the effects of short intervals and 1-day intervals on



the concentration of soluble P in runoff. During the rainfall simulation, 6 drainage holes remained open to better replicate field conditions. This limited drainage scenario is designed to replicate a tillage field where subsurface compaction has impeded drainage and resulted in the topsoil becoming saturated. It is also representative of areas at the base of slopes or along rivers where the water table is near the surface. As the risk of surface runoff increases with slope, each soil was examined at 2 slopes, 10 and 15 degrees, to investigate the effect of slope on nutrient and sediment losses in the runoff. Surface runoff samples were collected when runoff began: once every 2.5 min for the first 20 min and in each subsequent 5-min interval to evaluate changes in runoff volume, and nutrient and sediment concentration over time.

### **3.2.3 Overland flow experiment**

In this laboratory-scale overland flow experiment, soils were prepared and tested in exactly the same way as in the simulated rainfall experiment described in Section 3.2.2, with the exception of the introduction of two distinct overland flow rates via an overflow reservoir (Figure 3.2) located at the top of the runoff box. Two separate experiments were conducted, in which an overland flow of either 225 or 450 ml min<sup>-1</sup> was added at the top of the runoff box (when inclined at a 10 degree slope) in the presence of rainfall (30 mm hr<sup>-1</sup>), in order to investigate the effect of increasing overland flow rates on nutrient and sediment release from the study soils. The overland flow was generated by pumping water into a reservoir at the top of each flume using a Cole-Parmer Masterflex<sup>®</sup> L/S<sup>™</sup> peristaltic pump, which was calibrated prior to each experimental run. The water was allowed to flow over a metal plate that was level with the soil surface. It was envisaged that this approach would best replicate sheet flow arriving at the top of the flume. The two overland flow rates applied represent possible worst case scenarios in fields, where the soil has become saturated due to high intensity rainfall. Increases in high intensity storm events due to climate change are likely to result in tillage soils being subject to higher volumes of overland flow. The approximate surface runoff rates at the end of the runoff boxes for the 3 conditions of rainfall only (simulated rainfall experiment), rainfall and overland

flow at  $225 \text{ ml min}^{-1}$ , and rainfall and overland flow at  $450 \text{ ml min}^{-1}$ , were 200, 425 and  $650 \text{ ml min}^{-1}$ , respectively.

Each simulation comprised 3 successive 1-hr rainfall/overland flow events at time zero, 1 hr (this event began 1 hr after the first event finished) and 24 hr (this event began 24 hr after the 2<sup>nd</sup> event finished) to determine the effect of storm interval on surface runoff. At each time interval, each soil was subjected to either rainfall and overland flow at  $225 \text{ ml min}^{-1}$ , or rainfall and overland flow at  $450 \text{ ml min}^{-1}$ , to investigate the effect of increasing overland flow rate on nutrient and sediment losses in the runoff. Similarly, Hairsine (1988) introduced clear water at the top of the flume in the presence of rainfall when investigating erosion of a cohesive soil in a flume testing facility. Shallow depths of overland flow permit raindrop impact to have a significant influence on the removal of sediment from a soil bed and its subsequent displacement downslope (Hairsine, 1988). Following the commencement of surface runoff, water samples were collected as described in Section 3.2.2.

### 3.2.4 Soil analysis

The soil characteristics measured in each of the 5 test soils were: (1) pH (1:1 soil/solution ratio); (2) PSD by sieve and pipette analysis; (3)  $P_m$  was determined by adding 8 ml of dried and sieved ( $< 2\text{mm}$ ) soil to 40 ml of Morgan's Reagent (Morgan, 1941) (1480 ml of 40% sodium hydroxide and 1444 ml of glacial acetic acid to 20 L distilled water at pH 4.8) and shaking for 30 min on a Brunswick Gyratory shaker. The filtered extracts were analysed colorimetrically for P; (4) SOM by loss on ignition at  $550^\circ\text{C}$  (Byrne, 1979); (5) ammonium oxalate-oxalic acid extractable P ( $P_{ox}$ ), aluminium ( $Al_{ox}$ ), iron ( $Fe_{ox}$ ) measured by inductively coupled plasma-atomic emission spectroscopy. Soil P saturation ( $Psat_{ox}$ ) was calculated as  $P_{ox} \text{ (mmol kg}^{-1}\text{)}$ , divided by  $\alpha[Al_{ox} + Fe_{ox}]$  ( $\alpha = 0.5$  for non calcareous sandy soils), and multiplied by 100 (Schoumans, 2009); (6) WEP was measured by shaking 0.5 g of soil in 40 ml of distilled water for 1 hr, filtering ( $0.45 \mu\text{m}$ ) the supernatant water and determining P colorimetrically; (7) M3-P (Mehlich, 1984) (8) cation exchange capacity (CEC; Bascomb (1964)); (9) calcium carbonate ( $CaCO_3$ ) was determined by the volumetric method (ISO, 1995; ISO 10693) using a Scheibler apparatus; (10) calcium chloride

extractable P (  $P_{CaCl_2}$  ) by extraction with 0.01 M  $CaCl_2$ ; (11) AgSt was determined using the wet sieving apparatus (Eijkelkamp Agrisearch Equipment, The Netherlands). Selected soil chemical and physical properties are shown in Table 3.1. To ensure homogeneity of the individual soils, 3 subsamples of each soil were tested for  $P_m$  to determine the coefficient of variation (standard deviation divided by mean  $P_m$  concentration) for each soil. The coefficient of variation was  $< 0.05$  for the 5 soils.

### 3.2.5 Runoff analysis

Immediately after collection, runoff water samples were filtered (0.45  $\mu m$ ) and analysed colorimetrically for DRP using a nutrient analyser (Konelab 20, Thermo Clinical Labsystems, Finland). Total reactive phosphorus was determined on unfiltered samples, which were immediately frozen after collection. During defrosting, SS settled out facilitating the extraction of 2 ml of clear water by syringe from just below the surface of the water sample which was then analysed colorimetrically for TRP using the nutrient analyser (data not shown). Runoff water samples were frozen at  $-20\text{ }^{\circ}C$  until TP was conducted. Total phosphorus was determined for every second runoff sample after acid persulphate digestion. Total phosphorus comprises both PP and TDP. As in other studies by Randall et al. (2005) and Udeigwe and Wang (2007), PP was calculated by subtracting DRP from TP. As tests indicated that TDP was similar to DRP in the runoff water, and two orders of magnitude smaller than TP, this simplification was deemed appropriate. All digested samples were analysed colorimetrically for P using the nutrient analyser. Suspended sediment concentrations were determined for all samples by vacuum filtration of 50 ml of well-mixed runoff water through Whatman GF/C (pore size 1.2  $\mu m$ ) filter paper. All samples were tested in accordance with the Standard Methods (APHA, 2005) by the candidate at the Department of Civil Engineering, NUI, Galway.

### 3.2.6 Data analysis

Flow-weighted mean concentration (FWMC) values for nutrients and sediment in runoff from the flumes were determined by dividing the total mass load for the 1-hr runoff event by the total flow volume for the same period.

### 3.2.7 Statistical methods

In the case of the rainfall only experiment, a generalized linear mixed model (GLMM) was fitted to each surface runoff response to test whether the effect of soil type depends on slope and rainfall event. A random effect with a first-order autoregressive variance-covariance structure was fitted to account for non-independence of successive rainfall events (The GLIMMIX Procedure, SAS V9.1). A log link function was required for all surface runoff responses to satisfy the assumption of normality of residuals. When investigating P transport in surface runoff from packed soil boxes, Kleinman et al. (2004) also logarithmically transformed P concentrations as did Little et al. (2005), when investigating P losses from a plot experiment.

The inclusion of soil type as a factor in the GLMM allows quantification of the variation due to soil, so that we can then determine how much of this variation is accounted for by the different soil characteristic parameters. For DRP and TRP, the effect of soil type differed depending on slope and rainfall event. A stepwise regression selection procedure was subsequently conducted for each slope-rainfall combination to determine which characteristics were important in explaining variation in DRP and TRP. For SS, TP and PP, the stepwise procedure was performed separately for each slope, as the effects of soil type differed depending only on slope.

In the case of the simulated rainfall/overland flow experiments, a linear mixed model (LMM) was fitted to each surface runoff response to test whether the impact of soil type on DRP, TP, PP and SS concentrations in surface runoff was affected by flow rate and rainfall event. A random effect with either a compound symmetry or a first-order autoregressive variance-covariance structure was fitted to account for non-independence of successive rainfall events (The GLIMMIX and MIXED Procedures, SAS, 2004). A log transformation was required for all surface runoff responses to satisfy the assumption of normality of residuals. The analysis was conducted as a factorial combination of overland flow rate and rainfall event, with soil type as a blocking factor. The general classification by soil type allowed testing of the effects of the overland flow rate and event, but a number of covariates were recorded as a characterisation of the soil type. A series of models were fitted by removing the soil

type category and substituting mostly continuous variables in an attempt to improve understanding of the processes involved. Covariates were fitted initially in a hypothesis-based set of tests and subsequently best-fit models were obtained using a combination of hypotheses and stepwise selection of regressor variables. All covariate testing was carried out on an analysis model incorporating the experimental factors so that the full data set could be used without any bias due to the structure of the treatments.

### **3.3 Experimental results from simulated rainfall/overland flow experiments**

The results from the simulated rainfall/overland flow events on flumes packed with Irish tillage soils are presented here. These data provided information on amounts of P and sediment lost in surface runoff from the study soils.

#### **3.3.1 Characteristic properties of the study soils**

The selected study soils covered a  $P_m$  index of 1 - 4 and ranged from 2.8 to 17.5 mg  $P_m L^{-1}$ . Mehlich-3 P ranged from 29.1 to 96.28 mg  $kg^{-1}$  and was well correlated with  $P_m$  ( $r = 0.98$ ), WEP ( $r = 0.89$ ),  $Psat_{ox}$  ( $r = 0.86$ ), and  $P_{cacl_2}$  ( $r = 0.80$ ). Water extractable P ranged from 2.3 to 11.5 mg  $kg^{-1}$  and was well correlated with  $P_m$  ( $r = 0.92$ ),  $Psat_{ox}$  ( $r = 0.9$ ) and  $P_{cacl_2}$  ( $r = 0.95$ ). Calcium chloride extractable P ranged from 1.0 to 3.0 mg  $kg^{-1}$  and was well correlated with  $P_m$  ( $r = 0.87$ ) and  $Psat_{ox}$  ( $r = 0.93$ ). Soil P saturation ranged from 14.8 to 36.2% and was well correlated with  $P_m$  ( $r = 0.87$ ). If an agronomic soil P test is to be used for environmental purposes, it is important that it be well correlated with the forms of soil P most susceptible to losses in surface runoff and with  $Psat_{ox}$  (Sims et al., 2002). Soil pH, OM, and AgSt ranged from 6.4 to 7.7, 41.7 to 70.5 g  $kg^{-1}$ , and 90 to 98.5%, respectively.

Particle size analysis showed that sand was the dominant size fraction across the 5 soils, and ranged from 410 to 579 g  $kg^{-1}$ . The silt fraction ranged from 267 to 395 g  $kg^{-1}$ , and the clay fraction ranged from 114 to 203 g  $kg^{-1}$ . The basic soil textural class ranged from loam to sandy loam, with sandy loam dominating. This was representative of tillage which predominates in the east and south of Ireland, where

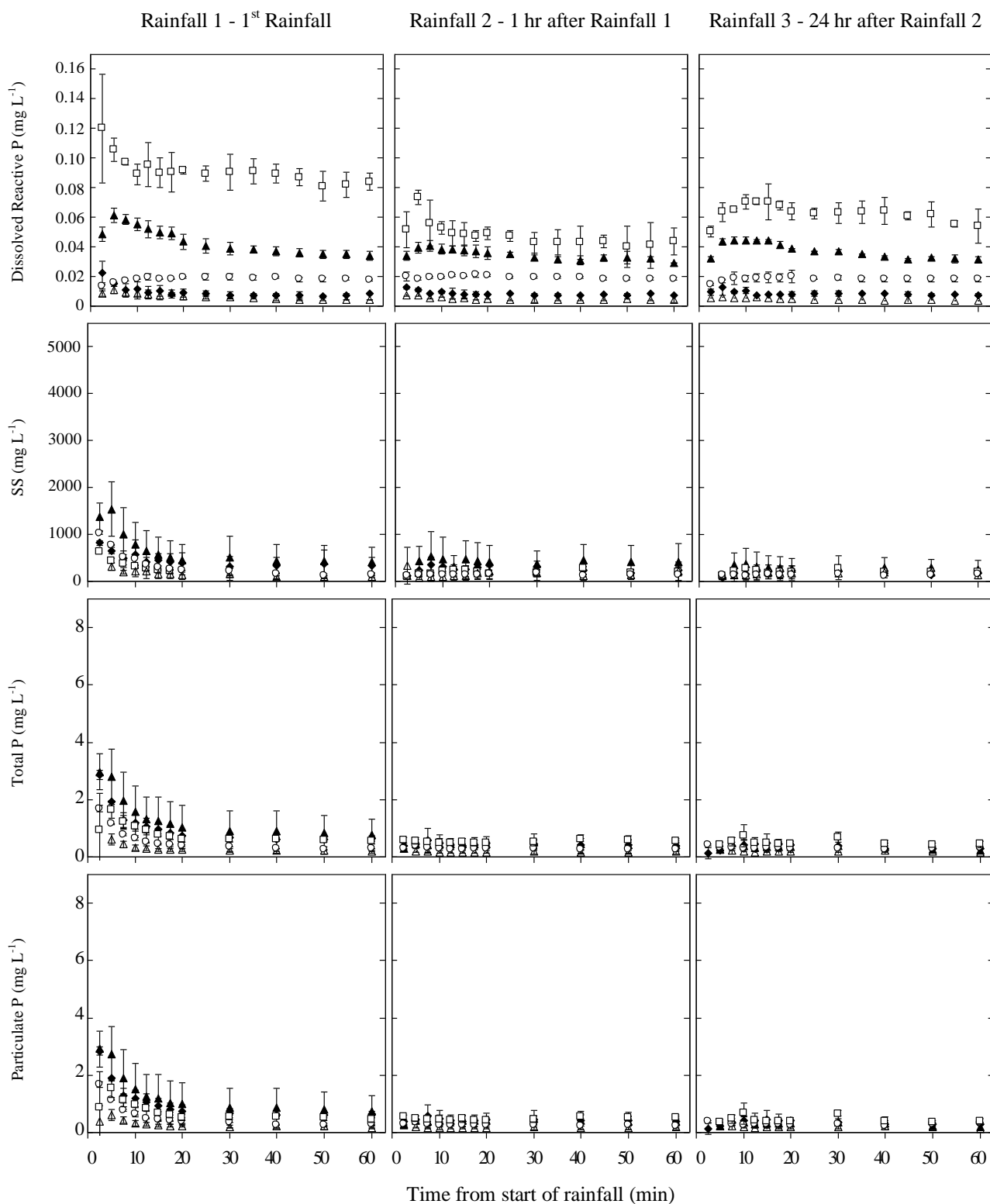
soils are highly suited to tillage, and generally have a light-to-medium texture, friable consistence and free drainage (Gardiner and Radford, 1980). Heavier textured soils are less suitable for tillage in Ireland as they generally occur at higher elevations leading to slope problems, have a weak structure, and are imperfectly drained (Gardiner and Radford, 1980). Soil CEC ranged from 11.2 to 13.9 cmol kg<sup>-1</sup>.

### **3.3.2 Suspended sediment and phosphorus concentrations in runoff from simulated rainfall events**

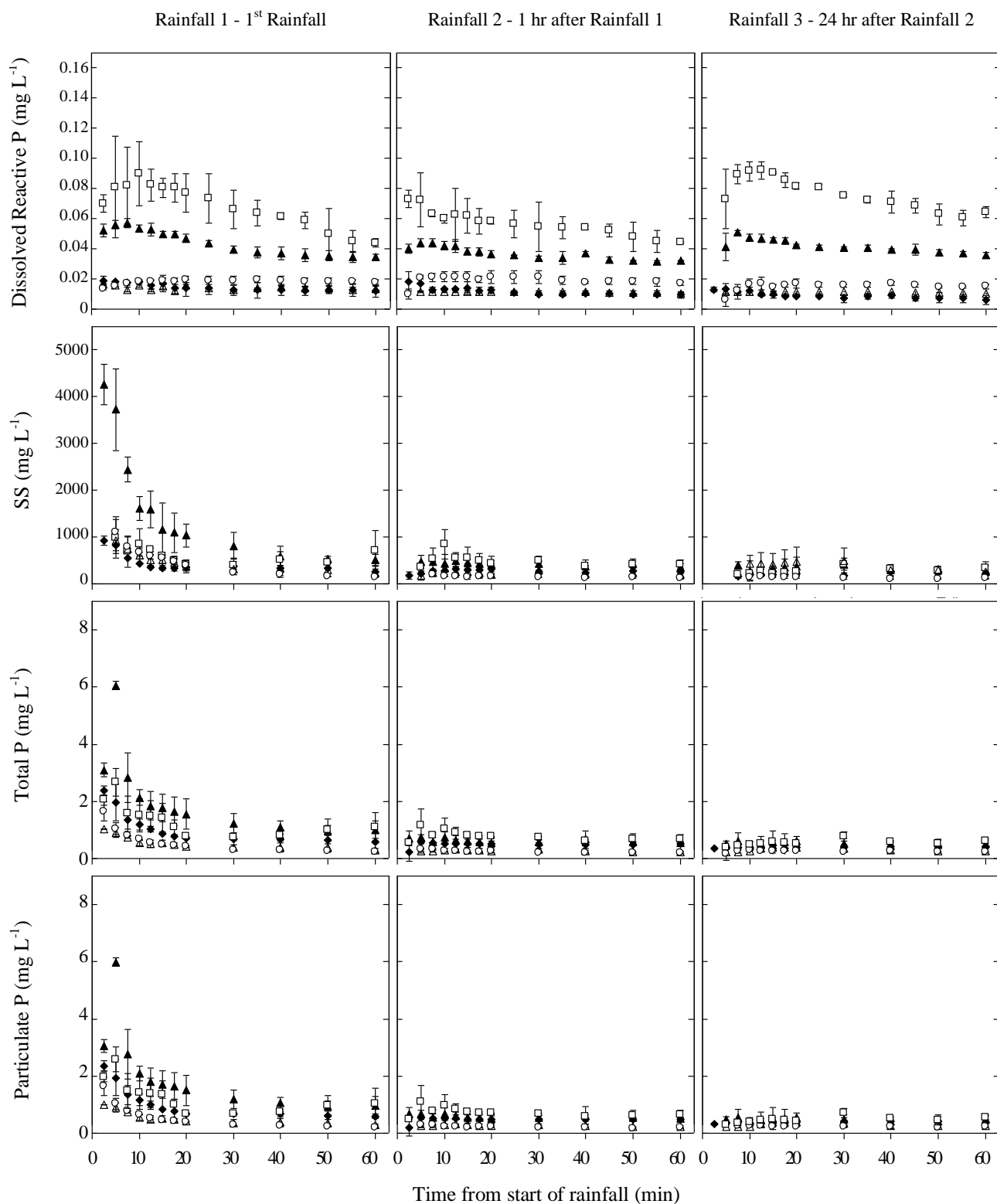
Generally, the highest SS and P concentrations occurred within 15 min of the commencement of surface runoff from the flumes and had reached steady-state 30 min after the commencement of the first rainfall event (Figure 3.5 and Figure 3.6).

The high DRP concentration in runoff at the start of a storm event may partly be explained by dilution as a function of runoff rates, which did not reach equilibrium until 5 min into a rainfall event. In addition, given that soils were pre-wet for up to 24 hr before rainfall commenced, a larger portion of the readily soluble and more slowly soluble P forms may have reached solution, thus elevating P levels in the soil water. As the rainfall event took place, the older pre-wet water became diluted by clean simulation water, resulting in a reduction in runoff DRP concentration. During the remainder of the rainfall event, the DRP measured in runoff was controlled by the pool of P that was freely available for desorption, and was rapidly desorbed and transferred into overland flow. Soils with high concentrations of extractable soil P had the highest concentration of DRP in surface runoff (Figure 3.5). This is a result of there being more freely available P in the soil solution at higher extractable soil P levels.

The FWMC of DRP in surface runoff was highest from the Tullow and Clonmel soils, which peaked at 0.09 and 0.042 mg L<sup>-1</sup>, respectively, during the first rainfall event and may have negatively affected water quality. These FWMC equated to DRP loads in surface runoff from the Tullow and Clonmel soils of 0.893 mg (or 19.85 g ha<sup>-1</sup>) and 0.338 mg (or 7.52 g ha<sup>-1</sup>), respectively, during the 1 hr rainfall events. In contrast, the



**Figure 3.5 Phosphorus and sediment losses over time from tillage soils inclined at a 10 degree slope. Clonmel (▲), Tullow (□), Letterkenny (◆), Bunclody (○), Fermoy (△).**



**Figure 3.6 Phosphorus and sediment losses from selected tillage soils inclined at a 15 degree slope. Clonmel (▲), Tullow (□), Letterkenny (◆), Bunclody (○), Fermoy (△).**



threat to surface water quality posed by the Fermoy soil, as a result of its potential to release DRP, was much lower, as evidenced by a peak FWMC of DRP of only  $0.014 \text{ mg L}^{-1}$ , which occurred at a 15 degree slope, during the 1<sup>st</sup> rainfall event. This equates to a DRP load of  $0.127 \text{ mg}$  (or  $2.82 \text{ g ha}^{-1}$ ). As such, the Tullow soil (Soil P Index 4), when subjected to simulated rainfall, has the potential to transfer 7 times as much DRP into surface runoff as the Fermoy soil (Soil P Index 1) and therefore poses a much greater risk to surface water quality. Whether this risk ultimately translates into impairment of a surface water body depends primarily on connectivity between the location where surface runoff occurred and a water body that is sensitive to pollution from P inputs. If connectivity between the P source and the pollution sensitive water body can be shown, then the area is termed a CSA. As DRP is readily available for uptake by aquatic plants, the likelihood of it resulting in eutrophication upon reaching a water body is far greater than that of an equivalent mass of PP, the availability of which can be as low as 10%. In general, the peak value of DRP observed during the first rainfall event on each soil reduced in subsequent rainfall events. Similar trends were noticed in the SS, TP and PP concentrations of runoff from the flumes.

The potential for particulate losses in surface runoff was found to be high for the 5 soils, with the highest coming from the Clonmel soil when inclined at a slope of 15 degrees, where peak SS and PP concentrations of  $4263 \text{ mg L}^{-1}$  and  $5.99 \text{ mg L}^{-1}$ , respectively, were measured during the first rainfall event (Figure 3.6). The SS and PP loads measured in surface runoff from the same 1 hr event were  $8.67 \text{ g}$  (or  $192.6 \text{ kg ha}^{-1}$ ) and  $12.8 \text{ mg}$  (or  $284 \text{ g ha}^{-1}$ ), respectively. The environmental risk posed by the Fermoy soil as a result of its low susceptibility to particulate losses, was lower than that of the other soils as evidenced by PP loads measured in surface runoff at 10 and 15 degree slopes of  $1.58 \text{ g}$  (or  $35.1 \text{ kg ha}^{-1}$ ) and  $3.92 \text{ g}$  (or  $87.1 \text{ kg ha}^{-1}$ ), respectively, during the first rainfall event. Particulate P contributed 84 to 99% of total runoff P. This is in close agreement with Fang et al. (2002), who reported that PP contributed from 59 to 98% of total runoff P for unvegetated packed boxes. The greater contribution of PP to total runoff P in this study is probably a result of the steeper slopes investigated. Similarly, Sharpley et al. (1994) reported that PP contributed 75 to 95% of total runoff P from conventionally tilled land. As PP is generally bound to the minerals (particularly Fe, Al, and Ca) and organic compounds contained in soil, it

constitutes a long-term P reserve of low bioavailability. The availability of PP to plants and algae is variable, ranging from 10 to 90% of the TP, but it can represent a long-term source of P for algae and plant uptake from surface water bodies, in particular lakes. Reducing dissolved P loss is far more difficult than reducing P loss associated with erosion (McDowell and Sharpley, 2001) and control measures are mainly limited to preventing soil P accumulation to environmentally sensitive levels (Sibbesen and Sharpley, 1997). Albeit nutrients lost in surface runoff from packed boxes are broadly consistent with those lost from field plots, the exposed bare soils of packed boxes are vulnerable to erosion, resulting in greater PP concentrations in runoff (Kleinman et al., 2004). The PP and SS losses measured in surface runoff during this study may represent a worst case scenario because of the steep slope (typical of sites where a P loss risk assessment is necessary), high rainfall intensity (typical of storm events), and bare soil with reduced AgSt compared to *in situ* soil. It is unlikely that SS and PP edge of field losses from these soils *in situ* would be so high given the variable surface slope and proximity to surface waters. Recognised management practices, namely, riparian buffer strips, conservation tillage, and contour ploughing, where used, are also effective in controlling PP loss from agricultural fields.

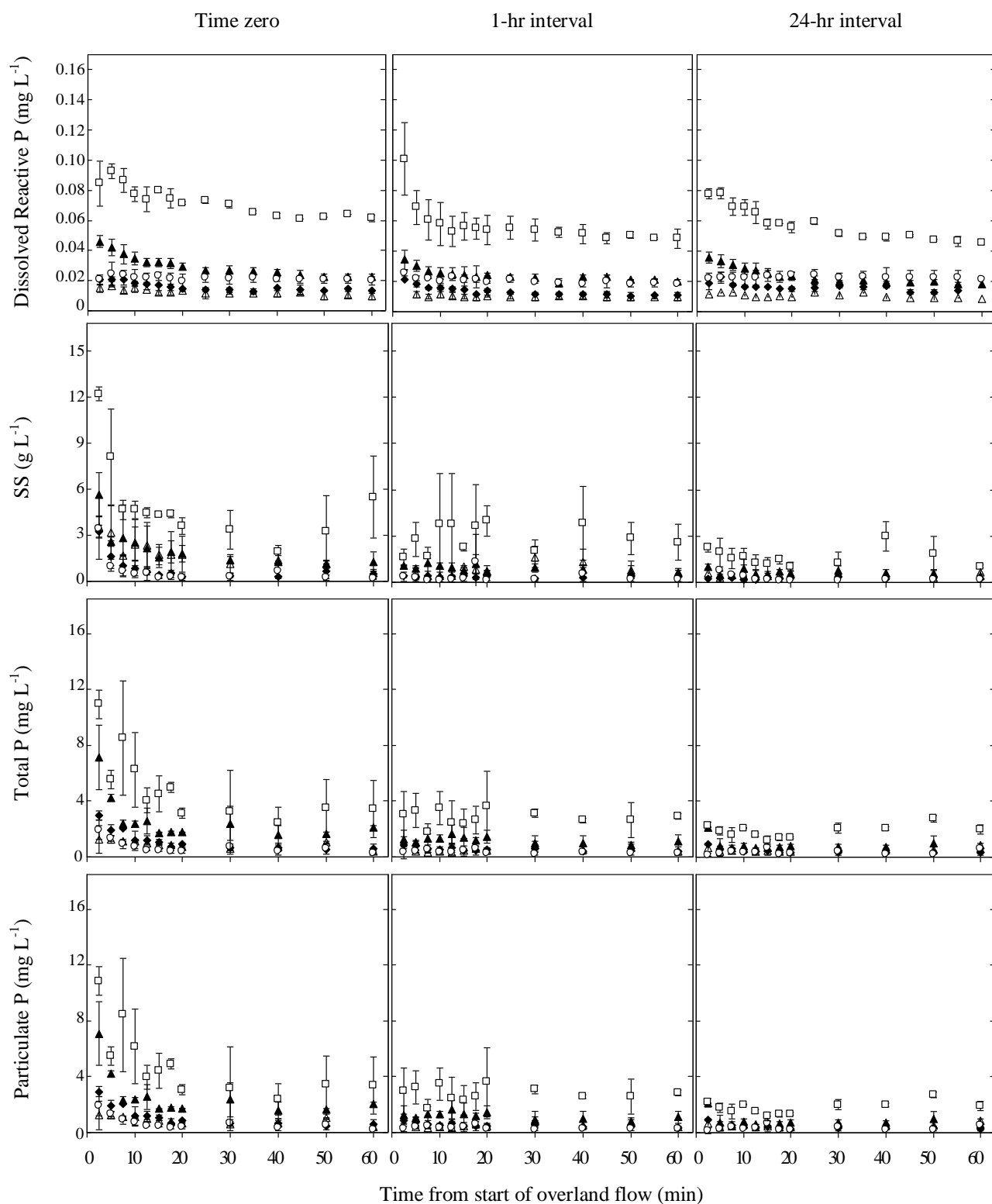
### **3.3.3 Suspended sediment and phosphorus concentrations in runoff from simulated rainfall/overland flow events**

Generally, the highest SS and P concentrations across the 5 soils, for an overland flow rate of  $225 \text{ ml min}^{-1}$ , occurred within 15 min of the commencement of the zero hr, 1 hr and 24 hr events, and reached steady-state no later than 30 min after commencement of runoff (Figure 3.7) with the exception of the Tullow soil, which did not achieve steady-state for particulate losses. Introducing overland flow rates of 225 and  $450 \text{ ml min}^{-1}$  at the top of the flume had the effect of increasing the runoff rate at the end of the flume from approximately  $200 \text{ ml min}^{-1}$  (for rainfall only) to 425 and  $650 \text{ ml min}^{-1}$ , respectively.

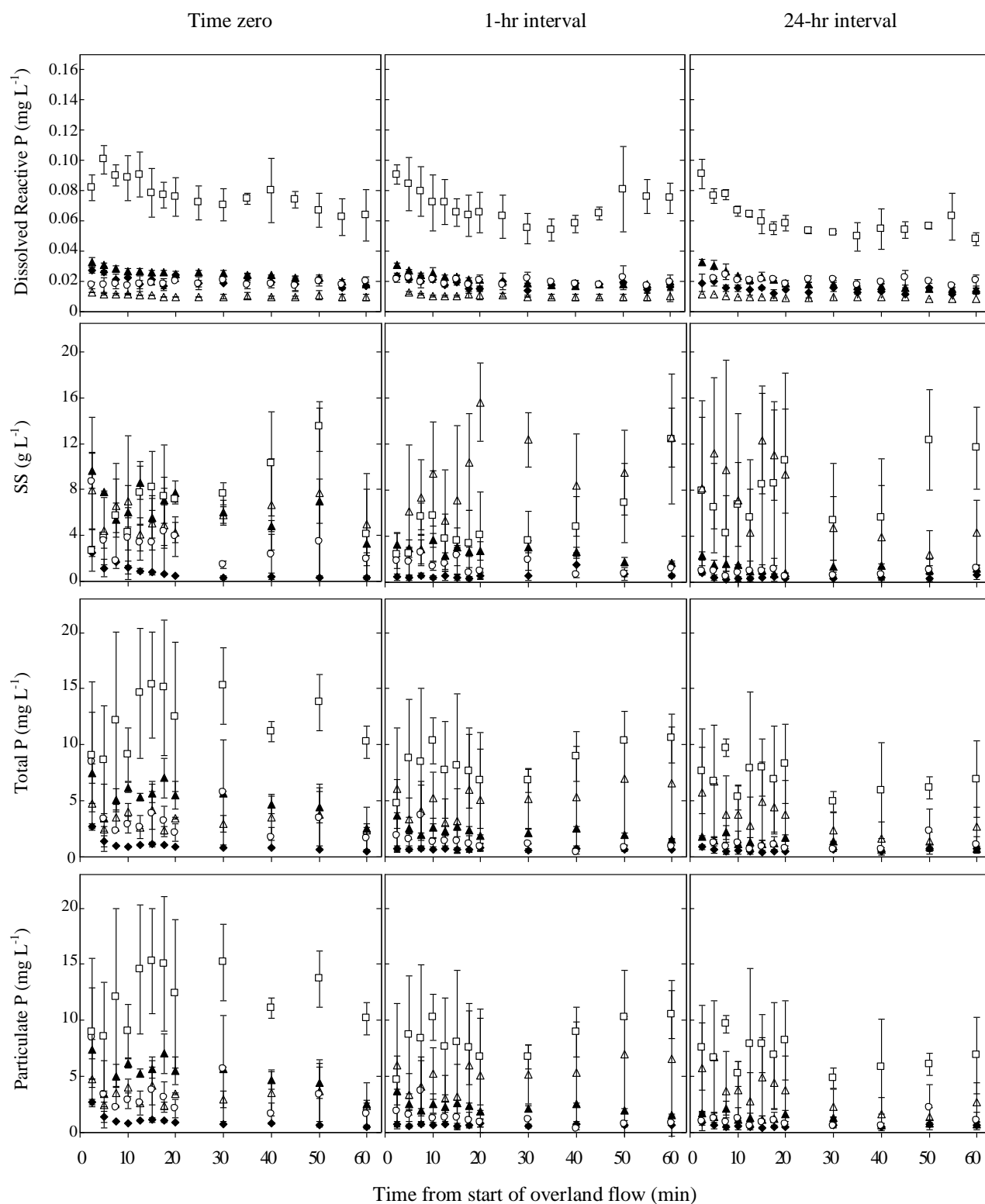
For the higher flow rate of  $450 \text{ ml min}^{-1}$ , nutrient and sediment concentrations only achieved steady-state for some soils (Figure 3.8). An increase in overland flow rate (0

up to  $225 \text{ ml min}^{-1}$  and up to  $450 \text{ ml min}^{-1}$ ) resulted in an increase in concentrations of SS, PP and TP in surface runoff across all soils ( $p < 0.05$ ). These increases in concentrations were generally not proportional to the increase in runoff rate measured at the end of the flume. This is to be expected given the vulnerable nature of the soils, after being sieved and then packed into flumes. As might be expected, there was a strong relationship ( $r = 0.92$ ,  $p = 0.0001$ ) between SS and PP concentrations measured in surface runoff across the 5 soils. In general, soils that experienced higher SS losses at  $450 \text{ ml min}^{-1}$  had higher levels of variability in nutrient and SS concentrations between replicate samples. An increase in extractable soil P resulted in an increase in concentrations of DRP in surface runoff ( $p < 0.05$ ) across all soils. The FWMCs of DRP in surface runoff (overland flow rate =  $225 \text{ ml min}^{-1}$ ) from the Tullow and Clonmel soils peaked at  $0.07$  and  $0.028 \text{ mg L}^{-1}$ , respectively, during the time zero events. Therefore, runoff from these soils has the potential to cause eutrophication on reaching a sensitive water body. These FWMCs equated to DRP loads in surface runoff from the Tullow and Clonmel soils of  $1.79 \text{ mg}$  (or  $39.8 \text{ g ha}^{-1}$ ) and  $0.732 \text{ mg}$  (or  $16.3 \text{ g ha}^{-1}$ ), respectively, during the 1-hr rainfall events.

The potential for particulate losses in surface runoff was very high for the 5 soils, with the highest losses coming from the Tullow soil (overland flow rate =  $450 \text{ ml min}^{-1}$ ), where FWMCs for SS and PP of  $3.86 \text{ g L}^{-1}$  and  $4.25 \text{ mg L}^{-1}$ , respectively, were measured (Figure 4.3) for the time zero event. These high concentrations were to be expected given the worst case scenario being investigated. The impeded drainage of the flume, high rainfall intensity, overland flow run-on, soil similar to that of a finely harrowed field, and 10 degree slope investigated are all conducive to increased rates of soil detachment. While FWMCs of PP measured in surface runoff across the study soils were far greater than FWMCs of DRP, it must be borne in mind that PP is attached to sediment which may settle out of suspension if the runoff transporting it, encounters less steep slopes than those in which it initially entrained the sediment. Similarly, if runoff encounters vegetation capable of reducing its velocity, the larger sediment will settle out, thereby reducing the concentration of PP being transported towards the aquatic receptor. There is greater potential for PP concentrations measured in surface runoff from the study soils to be reduced, as runoff makes its way toward the aquatic receptor; this must be considered when comparing the risk posed



**Figure 3.7** Phosphorus and sediment concentrations in runoff water from tillage soils subjected to rainfall and overland flow ( $225 \text{ ml min}^{-1}$ ) when inclined at a 10 degree slope. Clonmel (▲), Tullow (□), Letterkenny (◆), Bunclody (○), Fermoy (△).

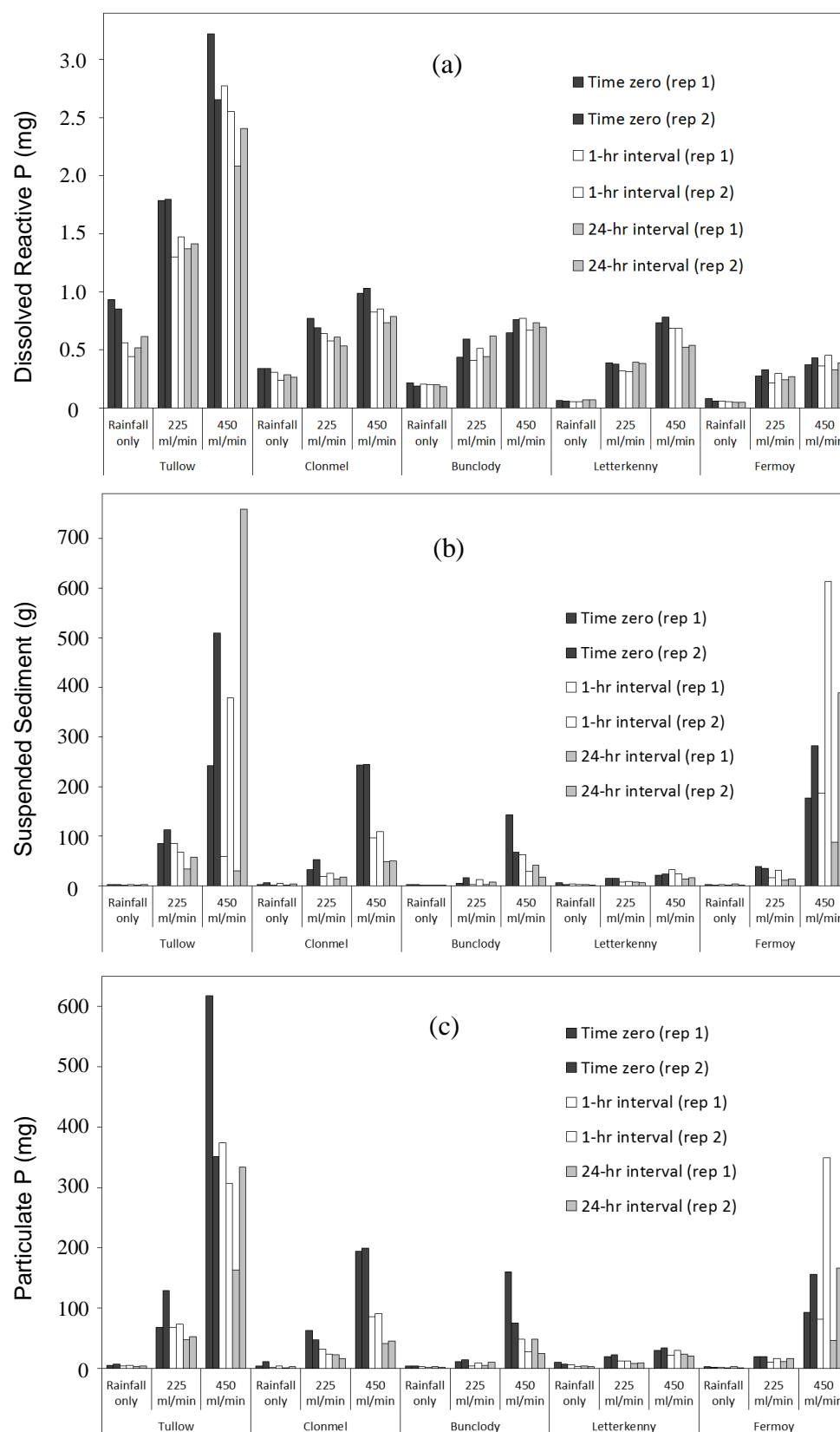


**Figure 3.8** Phosphorus and sediment concentrations in runoff water from tillage soils subjected to rainfall and overland flow ( $450 \text{ ml min}^{-1}$ ) when inclined at a 10 degree slope. Clonmel (▲), Tullow (□), Letterkenny (◆), Bunclody (○), Fermoy (△).

by PP and DRP. Furthermore, PP is of limited availability for uptake by aquatic plants, whereas DRP is generally considered to be 100% available.

While the effect of overland flow rate on DRP concentrations measured in overland flow across the 5 soils tested was variable, its effect on mass losses was quite clear (Figure 3.9a). As the overland flow rate increased, there was an almost proportional increase in DRP lost in surface runoff, indicating that any dilution effect of higher flow rates was minimal. This is discussed further in the Section 4.3.3. Mass losses of DRP were highest for the Tullow soil, where 2.9 mg (or 64.4 g ha<sup>-1</sup>) was released for time zero at an overland flow rate of 450 ml min<sup>-1</sup> compared to 0.893 mg (or 19.85 g ha<sup>-1</sup>) when subjected to 30 mm hr<sup>-1</sup> rainfall only.

Increases in mass losses of SS, PP and TP (data not shown because it is indistinguishable from PP) due to increased overland flow rates were more pronounced (Figure 3.9b-c) than those for DRP, with the exception of the Letterkenny soil, which was more resistant to degradation. For example, mass losses of SS from the Tullow soil at time zero were 2.67 (59.33 kg ha<sup>-1</sup>), 99.3 (2206 kg ha<sup>-1</sup>) and 375.8 (8352 kg ha<sup>-1</sup>) g when subjected to rainfall only, 225 ml min<sup>-1</sup>, and 450 ml min<sup>-1</sup>, respectively. While FWMCs of SS from the Tullow soil at time zero were 0.269, 3.86 and 9.63 g L<sup>-1</sup> when subjected to rainfall only, 225 ml min<sup>-1</sup>, and 450 ml min<sup>-1</sup>, respectively. In contrast, mass losses of SS from the Letterkenny soil at time zero were significantly lower (Figure 3.9).



**Figure 3.9 Mass loss of phosphorus and sediment in runoff from selected tillage soils subjected to rainfall and two overland flow rates (225 and 450 ml min<sup>-1</sup>) while inclined at a 10 degree slope.**

### **3.3.4 Effect of slope, time between events and overland flow rate on concentrations in surface runoff**

The GLMM (Table 3.2) used for the rainfall only experiment, indicated that the effect of soil type on the FWMC of DRP ( $p = 0.013$ ) and TRP ( $p = 0.007$ ) depended on both slope and time between rainfall events. The effect of soil type depended only on surface slope for the FWMCs of SS ( $p = 0.044$ ), TP ( $p = 0.014$ ) and PP ( $p = 0.022$ ) in surface runoff.

The LMM analyses (Table 3.3) used for simulated rainfall/overland flow experiments, indicated that the effect of soil type on the FWMC of DRP interacted with/depended on both overland flow rate and time between overland flow events ( $p = 0.0351$ ). The effect of soil type depended only on overland flow rate for the FWMCs of SS ( $p < 0.0001$ ), TP ( $p = 0.0015$ ) and PP ( $p = 0.0013$ ) in surface runoff. These results were as expected given the small storm intervals being investigated. Larger storm intervals that allow the soil time to dry might be expected to impact on the levels of SS, TP and PP lost in runoff. There was significant interaction of soil type with at least one of the experimental factors for each of the variables examined, indicating the importance of soil type in assessing the potential to release DRP, SS, TP and PP into overland flow (Table 3.3).



**Table 3.2 Overall Anova for responses from GLMM analyses (rainfall only)**

<b>Dissolved Reactive Phosphorus</b>			
<b>Source</b>	<b>DF</b>	<b>F Value</b>	<b>Pr &gt;F</b>
Soil type	4	768.58	<.0001
Slope	1	67.38	<.0001
Rainfall interval	2	12.94	<.0001
Soil*Slope	4	240.97	<.0001
Soil*Rainfall interval	8	3.54	0.005
Slope*Rainfall interval	2	1.68	0.203
Soil*Slope*Rainfall interval	8	3.01	0.013
<b>Total Reactive Phosphorus</b>			
Soil type	4	904.11	<.0001
Slope	1	86.33	<.0001
Rainfall interval	2	15.19	<.0001
Soil*Slope	4	43.99	<.0001
Soil*Rainfall interval	8	5.14	0.001
Slope*Rainfall interval	2	2.53	0.097
Soil*Slope*Rainfall interval	8	3.4	0.007
<b>Total Phosphorus</b>			
Soil type	4	33.82	<.0001
Slope	1	10.62	0.002
Rainfall interval	2	32.7	<.0001
Soil*Slope	4	3.49	0.014
<b>Particulate Phosphorus</b>			
Soil type	4	27.51	<.0001
Slope	1	9.62	0.003
Rainfall interval	2	30.33	<.0001
Soil*Slope	4	3.13	0.022
<b>Suspended Sediment</b>			
Soil type	4	9.56	<.0001
Slope	1	14.87	0.001
Rainfall interval	2	9.65	0.001
Soil*Slope	4	2.66	0.044

**Table 3.3 Overall ANOVA for responses from LMM analyses (rainfall and overland flow)**

<b>Dissolved Reactive Phosphorus</b>			
<b>Source</b>	<b>DF</b>	<b>F value</b>	<b>Pr &gt;F</b>
Soil type	4	639.33	<.0001
Overland flow	2	17.71	0.0001
Overland flow*soil type	8	29.95	<.0001
Event	2	32.69	<.0001
Event*soil type	8	5.24	0.004
Overland flow*event	4	3.89	0.0116
Soil type*overland flow*event	16	2.14	0.0351
<b>Total Phosphorus</b>			
Soil type	4	23.17	<.0001
Overland flow	2	83.87	<.0001
Overland flow*soil type	8	6.00	0.0015
Event	2	65.05	<.0001
Event*soil type	8	4.04	0.0011
Overland flow*event	4	1.24	0.3053
<b>Particulate Phosphorus</b>			
Soil type	4	21.27	<.0001
Overland flow	2	83.3	<.0001
Overland flow*soil type	8	6.16	0.0013
Event	2	80.73	<.0001
Event*soil type	8	5.23	0.0004
Overland flow*event	4	1.49	0.2310
Soil type*overland flow*event	16	1.75	0.0898
<b>Suspended Sediment</b>			
Soil type	4	5.29	0.0073
Overland flow	2	35.13	<.0001
Overland flow*soil type	8	9.89	<.0001
Event	2	36.46	<.0001
Event*soil type	8	1.66	0.1348
Overland flow*event	4	1.91	0.1240

### 3.4 Summary

This study quantified the amount of DRP, PP, TP and SS released into overland flow for 5 tillage soils when subject to a rainfall intensity of  $30 \text{ mm hr}^{-1}$  and overland flow rates of 225 and  $450 \text{ ml min}^{-1}$  applied in 3 successive events. The main conclusions from this study were:

1. Increasing the overland flow rate over the soil surface in the presence of rainfall had the effect of increasing the concentrations of SS, PP and TP (but not DRP) in surface runoff across all soils. This increase in concentration varied in magnitude across soils and was highest for the Tullow soil.
2. Overall, there was no evidence of a relationship between overland flow rate and DRP concentration measured in surface runoff. This implies that rainfall can be used in isolation when developing relationships between soil P level and potential DRP lost in runoff.

Chapter 4 uses the data compiled in this chapter to determine threshold STP values above which surface runoff may cause eutrophication and to identify potential risk indicators for estimating P and sediment release from tillage soils.

## **Chapter 4 Threshold values and potential risk indicators for estimating phosphorus and sediment release from Irish tillage soils**

### **Overview**

Using the relationship between STP level in the 5 soils and the DRP measured in surface runoff, a runoff dissolved phosphorus risk indicator (RDPRI) was developed to quantify the Morgan's P ( $P_m$ ), Mehlich 3-P (M3-P), water extractable P (WEP), calcium chloride extractable P ( $P_{CaCl_2}$ ) and soil P saturation ( $P_{sat_{ox}}$ ) levels for 5 Irish tillage soils, above which there may be a potential threat to surface water quality. A statistical analysis of the experimental data collected in Chapter 3 is used to rank soil extractable P methods with respect to their potential to be used as P loss risk indicators. Finally, this chapter identifies important parameters for which to test when attempting to predict SS, TP and PP loss from tillage soils. The contents of this chapter are, in part, published in the Journal of Environmental Quality (39:185-192, 2010).

### **4.1 Introduction**

In tillage soils excessive organic and inorganic fertiliser application can lead to a build up of P in excess of crop requirements. This may result in DRP loss in runoff, which is readily available for biological uptake, and poses an immediate threat for accelerated algal growth, which may negatively affect water quality in rivers and lakes. Summer storm events, coupled with impervious agricultural soils, can lead to P addition to waterways during the aquatic growing season. While PP loss can be minimised through the use of buffer zones and minimum tillage, reducing dissolved P loss is far more difficult (McDowell and Sharpley, 2001) and control measures are mainly limited to preventing soil P accumulation to environmentally sensitive levels (Sibbesen and Sharpley, 1997). The relationship between STP and DRP loss to water in runoff events needs to be adequately understood and quantified for local soils

(Wright et al., 2006) in order to determine upper critical limits for P in soil that will reduce the risk of diffuse losses from tillage land to surface waters. A laboratory flume study was chosen over a field study, as soils in flume studies can be homogenised minimising variability in soil physical and chemical characteristics. It is also less expensive and facilitates testing under standardised conditions including surface slope, soil conditions, rainfall intensity and overland flow rate.

Soil P saturation has been suggested as a method to characterise the potential for P loss from agricultural soils (Maguire et al., 2001). It is thought to be a more meaningful indicator of potential losses to water than STP, since it describes soil in terms of P sorption sites already saturated and is therefore independent of soil type (Daly et al., 2001). Furthermore, its use in soil studies and for environmental purposes is becoming more frequent (Beauchemin and Simard, 1999; Kleinman et al., 2000; Schroeder et al., 2004; Randall et al., 2005; Penn et al., 2006; Little et al., 2007; Wang et al., 2010). It has been shown to be well correlated with runoff DRP (Pote et al., 1996; Pote et al., 1999). Soils with higher  $Psat_{ox}$  pose a greater risk of P loss because eroded soil particles will be enriched in potentially desorbable P (Pautler and Sims, 2000). Dilute calcium chloride has also been proposed for use as an environmental P test because it represents more readily desorbable forms of P in soil (Daly and Casey, 2005) and has been shown to relate well to the concentration of DRP in surface runoff from soil using rainfall simulators (McDowell and Sharpley, 2001). These proposed environmental P loss risk indicators need to be assessed with respect to their potential to predict DRP loss in surface runoff from Irish tillage soils.

The aims of this chapter were: (1) to investigate the relationships between soil extractable P measured in the study soils and DRP measured in surface runoff; (2) to rank a number of soil extractable P methods with respect to their potential to be used as P loss risk indicators for a selection of Irish tillage soils; (3) to rank the importance of soil physical and chemical parameters, in the prediction of P and sediment release from soil to overland flow; and (4) to develop a RDPRI for tilled soils in Ireland.

## 4.2 Data analysis methods

A RDPRI was first developed using the relationship between the FWMC of DRP lost in surface runoff (generated by using simulated rainfall events on soils inclined at 10 and 15 degree slopes – Section 3.2.2) and each of  $P_m$ , M3-P and WEP measured in the study soils. This was achieved by constructing 95% confidence limits around the DRP relationships using the upper and lower confidence bands for the linear predictor. The resulting confidence limits were then back-transformed from the log-linear model to the original scale to identify the level of each P measure above which there might be a potential threat to surface water quality. An improved RDPRI was developed using the relationship between FWMC of DRP lost in surface runoff (generated by using simulated rainfall/overland flow events on soils inclined at 10 degree slopes – Section 3.2.3) and each of  $P_m$ , M3-P, WEP,  $P_{CaCl_2}$ , and  $P_{sat_{ox}}$ .

In order to rank the relative importance of the soil parameters, each parameter was assessed individually in its effect on the surface runoff response, averaged across the overland flow rates and flow events. The Akaike Information Criterion (AIC) is a statistic that gives a measure of the goodness of fit of a model. As the models were not nested, the extractable soil tests were ranked according to the AICs of their individual models. Each P indicator was added, in turn, to a model with the experimental factors and then assessed to determine which of them produced the best increase in the goodness-of-fit of the model used for determining the critical value of DRP.

## 4.3 Analysis of experimental results from simulated rainfall/overland flow experiments

### 4.3.1 Soil properties affecting phosphorus release from soil to water

Selected soil chemical and physical properties are presented in Chapter 3, Table 3.1 and briefly discussed in Section 3.1.1. A more detailed discussion of study soil properties affecting the release of P from soil to runoff is provided here. In particular, attention is paid to Al-oxides, Fe-oxides and  $CaCO_3$ , as these properties play an

important role in the relationship between each of  $P_{\text{cacl}_2}$  and  $P_{\text{sat}_{\text{ox}}}$  and DRP in surface runoff.

Soil pH ranged from 6.4 - 7.7 with most of the soils in the neutral or slightly acid range, as is common for tillage soils in Ireland where the median pH value is 6.4 (Fay et al., 2007). Only the Bunclody soil had a pH > 7. The  $\text{CaCO}_3$  content of the soils ranged from 0.4 - 2.6%, with the Letterkenny and Bunclody soils being classed as slightly calcareous ( $\text{CaCO}_3 > 1\%$ ) in nature according to Defra (2006b). While  $\text{CaCO}_3$  is the soil parameter most notably associated with P sorption in calcareous soils (Lindsay, 1979), some studies have found that P sorption in these soils is more closely related to Fe-oxide, Al-oxide and clay contents (Leytem and Westermann, 2003). Generally, soils with higher pH would normally contain greater amounts of extractable Ca, as well as lower extractable Al and Fe (Brady and Weil, 2007). The Bunclody soil was an exception in the studied soils, for while it had high  $\text{CaCO}_3$ , it contained relatively large amounts of  $\text{Fe}_{\text{ox}}$  and  $\text{Al}_{\text{ox}}$ , even though its pH was 7.7. This high pH may be a result of the use of beet factory sludge lime as a fertiliser in the past. Reactions that reduce P availability occur in all ranges of soil pH, but can be very pronounced in alkaline soils (pH > 7.3) and in acidic soils (pH < 5.5) (Busman et al., 1998). In acidic soils, P is largely sorbed to Al-oxides and Fe-oxides, whereas in neutral to alkaline soils, P occurs primarily as Ca-phosphates and Mg-phosphates often precipitated, or sorbed, onto Ca and Mg carbonates (McDowell et al., 2012). The presence of free  $\text{CaCO}_3$  in calcareous soils reduces the amount of soluble P present in soil and prevents it from being released into runoff (Torbert et al., 2002). In this study, the  $\text{CaCO}_3$  levels present in the soils were sufficiently low that the more likely controllers of P retention were Fe and Al.

Distilled water and dilute calcium chloride solutions are not considered to be soil test indices like  $P_m$  and M3-P, but are thought to be good predictors of dissolved P in runoff and subsurface drainage, respectively (Pote et al., 1996, 1999; McDowell and Sharpley, 2001; Maguire and Sims, 2002) and their use as environmental indicators of P loss has been proposed by Irish researchers (Daly and Casey, 2005). Calcium chloride extractable P, WEP and  $P_m$  extracted the least amount of P from the soil, while M3-P extracted larger amounts because it is more acidic than the soil solution

and able to dissolve calcium phosphates, a fraction of P which is normally of low availability. Both distilled water and 0.01 M  $\text{CaCl}_2$  possess extraction matrices that closely mimic soil solution and their limited propensity to extract P, measuring primarily the soluble and easily desorbable P forms results in lower P extraction (Schindler et al., 2009). Distilled water extracts more P than 0.01 M  $\text{CaCl}_2$  because  $\text{Ca}^{2+}$  enhances P sorption in soils. Acid ammonium oxalate extracted much larger amounts of soil P than other extractants, suggesting that most of the P in the study soils was sorbed or precipitated on amorphous oxides of Fe and Al (Pote et al., 1996). There is general agreement in the literature that these forms of Fe and Al are the most important in terms of P retention. Levels of  $\text{Fe}_{\text{ox}}$  measured in all the soils of this study were higher than  $\text{Al}_{\text{ox}}$ , as was the case for Irish soils studied by Doody et al. (2006), Maguire et al. (2001) and Evans and Smiley (1976). While reports differ on their relative importance in terms of P retention, Evans and Smiley (1976) showed that  $\text{Al}_{\text{ox}}$  was two and a half times as effective as  $\text{Fe}_{\text{ox}}$  in retaining P in Irish soils. Schroeder et al. (2004) demonstrated that areas with larger  $\text{Fe}_{\text{ox}}$  to  $\text{Al}_{\text{ox}}$  ratio may produce proportionally greater P loss as the STP increases than areas with lower  $\text{Fe}_{\text{ox}}$  to  $\text{Al}_{\text{ox}}$  ratio.

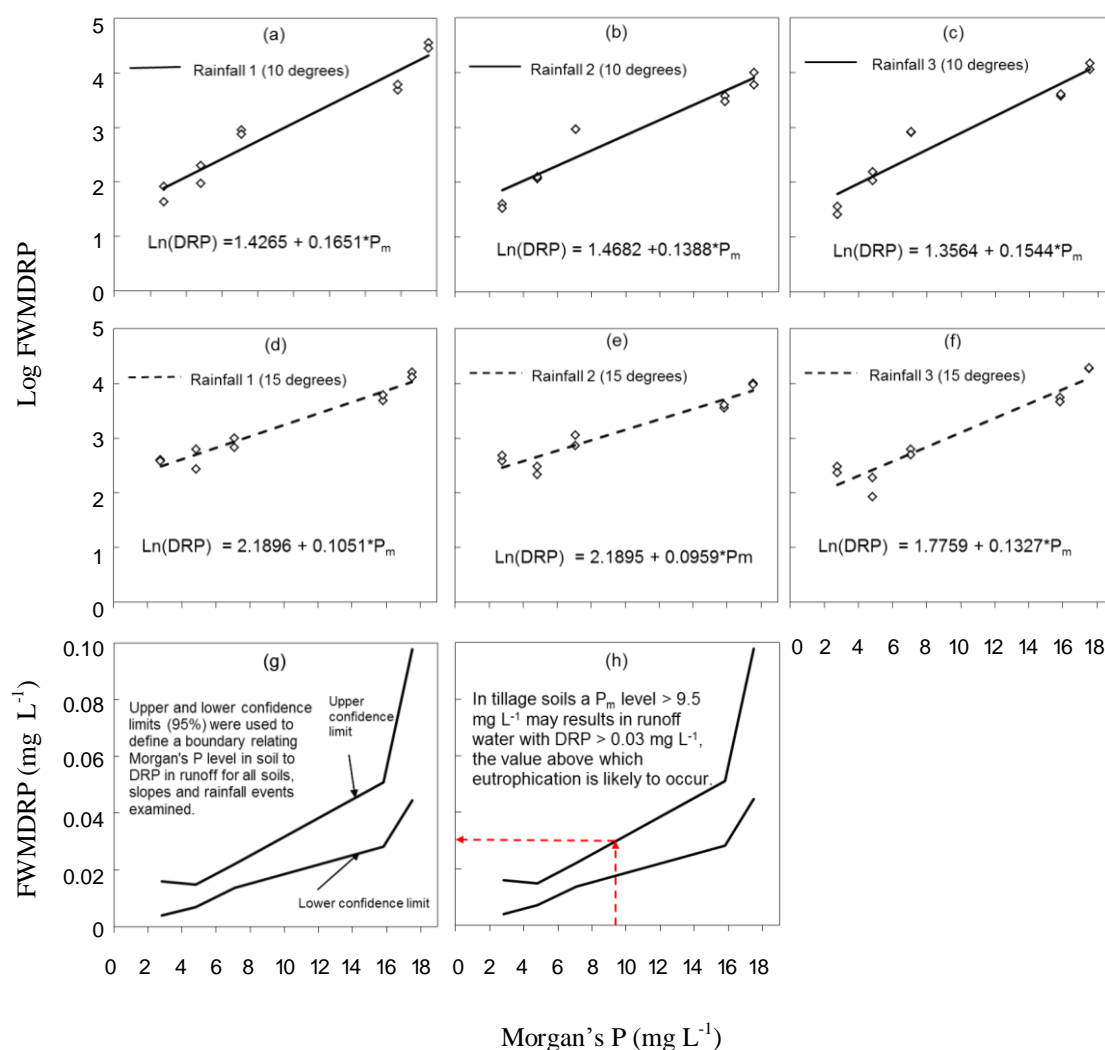
Soil P saturation has been identified as a potential P loss risk indicator for agricultural soils because it has a strong relationship with runoff P concentrations (Maguire et al., 2001; Sims et al., 2002). It is different from other soil P tests because it not only considers the quantity of P present in a soil, but also includes the capacity of the soil to retain additional P. Soils with higher  $\text{Psat}_{\text{ox}}$  maintain higher solution P concentrations, and any eroded soil particles will be enriched in potentially desorbable P (Pautler and Sims, 2000). Based on  $\text{Psat}_{\text{ox}}$  levels measured across the study soils, the Tullow and Clonmel soils had the highest risk of P desorption, with  $\text{Psat}_{\text{ox}}$  levels of 36.2 and 28.8%, respectively. Maguire et al. (2001) found that for a selection of Irish and American soils with high P levels, a single oxalate extraction for Al, Fe, and P proved to be most useful for predicting long-term P desorption, through calculation of the  $\text{Psat}_{\text{ox}}$  and for predicting the ability of the soils to sorb more P by calculating free  $[\text{Fe}_{\text{ox}}$  and  $\text{Al}_{\text{ox}}]$ . In Section 4.3.5, the effect of  $\text{Fe}_{\text{ox}}$  and  $\text{Al}_{\text{ox}}$  on DRP in runoff is investigated.



### **4.3.2 A method for identifying the critical soil test phosphorus threshold above which simulated rainfall-induced surface runoff may pose a threat to surface water quality**

Researchers have shown that there is a relationship between STP and runoff DRP concentrations from both grassland (Pote et al., 1996; Torbert et al., 2002) and cultivated soils (Cox and Hendricks, 2000; Vadas et al., 2005). The retention of P in rivers is dominated by physical processes such as flow velocity, discharge, and water depth. The long-term storage of P in the water column through chemical processes, such as assimilation by river sediments, is inhibited by the rapid mobilisation and transport that occurs during hydrological storm events. Furthermore, biological uptake can account for the majority of dissolved P transformations in streams (Reddy et al., 1999), thereby increasing the population of algae and aquatic plants, affecting the quality of the water and disturbing the balance of organisms present within it. Eutrophic symptoms in rivers are commonly linked to transient increases in DRP concentrations during times of ecological sensitivity (Jarvie et al., 2006). A critical STP threshold must exist above which runoff water will negatively affect surface water quality. In order to estimate defensible, upper critical soil P limits that are environmentally sensitive, it is necessary to first develop analytical methods that measure soil P availability relevant to the release of soil P to runoff; quantify the relationship between soil and runoff P; and identify transport potential for a site (Sibbesen and Sharpley, 1997). Given that eutrophication has been shown to occur in rivers with  $> 0.03 \text{ mg MRP L}^{-1}$  and lakes with  $> 0.02 \text{ mg TP L}^{-1}$ , runoff water with  $> 0.03 \text{ mg P L}^{-1}$  entering rivers and lakes is likely to contribute to further deterioration in surface water quality. A RDPRI was developed to identify a STP threshold (measured in terms of  $P_m$ , M3-P and WEP) above which runoff water across the 5 soils tested will have a DRP concentration greater than  $0.03 \text{ mg P L}^{-1}$ . Present nutrient advice for tilled crops in Ireland prohibits the application of fertiliser or manure at  $P_m$  concentrations above  $10 \text{ mg L}^{-1}$  (SI 610 of 2010) with the exception of potatoes, beet, and turnips. In Delaware, a M3-P concentration  $> 100 \text{ mg kg}^{-1}$  is considered above optimum and therefore no further P addition is recommended. This limit is based on crop yield response to fertiliser P, but needs to be assessed from an environmental standpoint. The RDPRI enables this assessment to be performed.

The logarithm of the FWMC of DRP in surface runoff from each soil was linearly related to the  $P_m$  of each soil for all rainfall events and slopes examined (Figure 4.1a - f). A RDPRI was developed by constructing 95% confidence limits around the  $P_m$  - DRP relationships using the upper and lower confidence bands for the linear predictor. The resulting confidence limits were then back-transformed from the log-linear model to the original scale to identify the  $P_m$  level above which there might be a potential threat to surface water quality (Figure 4.1g - h).



**Figure 4.1** Runoff Dissolved Phosphorus Risk Indicator using Morgan's P for selected tillage soils at 10 and 15 degree slopes under a 30 mm hr<sup>-1</sup> rainfall. Rainfall 1 - 1st rainfall event; Rainfall 2 - 1hr after Rainfall 1; Rainfall 3 - 24 hr after Rainfall 2.

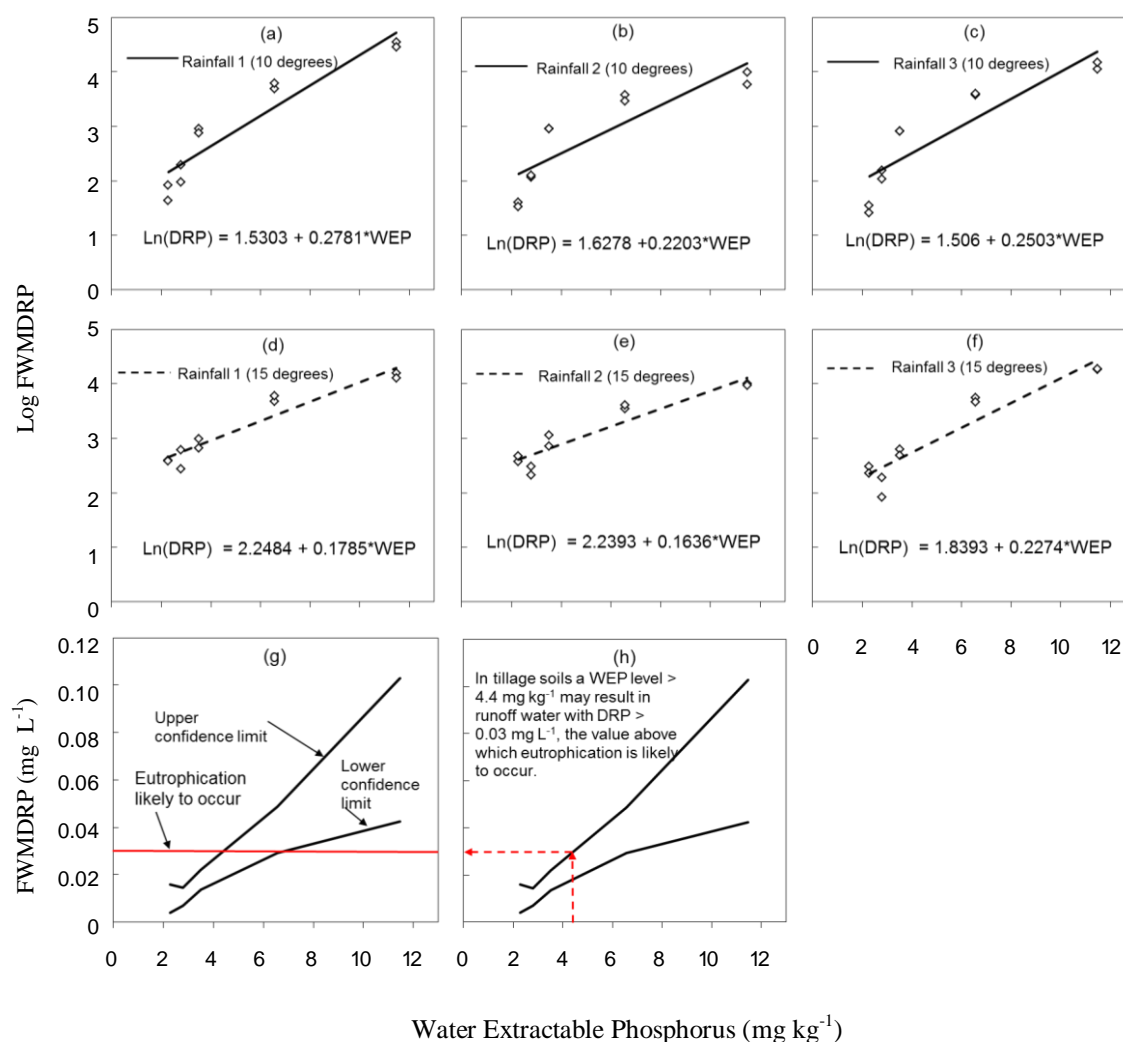
The logarithm of the FWMC of DRP in surface runoff from each soil was also linearly related to WEP and M3-P levels in each soil for all rainfall events (Figure 4.2 and Figure 4.3).

The DRP lost to runoff water increases log-linearly as  $P_m$ , WEP and M3-P increase. For both 10 and 15-degree slopes, the 1<sup>st</sup> rainfall event had the highest FWMC of DRP in runoff from the 5 soils. The FWMC of DRP in surface runoff was lower during the 2<sup>nd</sup> rainfall event, but was higher in the 3<sup>rd</sup>. This increase may be attributed to more soluble P becoming available to runoff during a 24-hr rainfall cessation than during a 1-hr cessation. Sharpley (1980) found that when the time interval between rainfall events exceeds 1 d, the initial soluble P concentration in a runoff event increases.

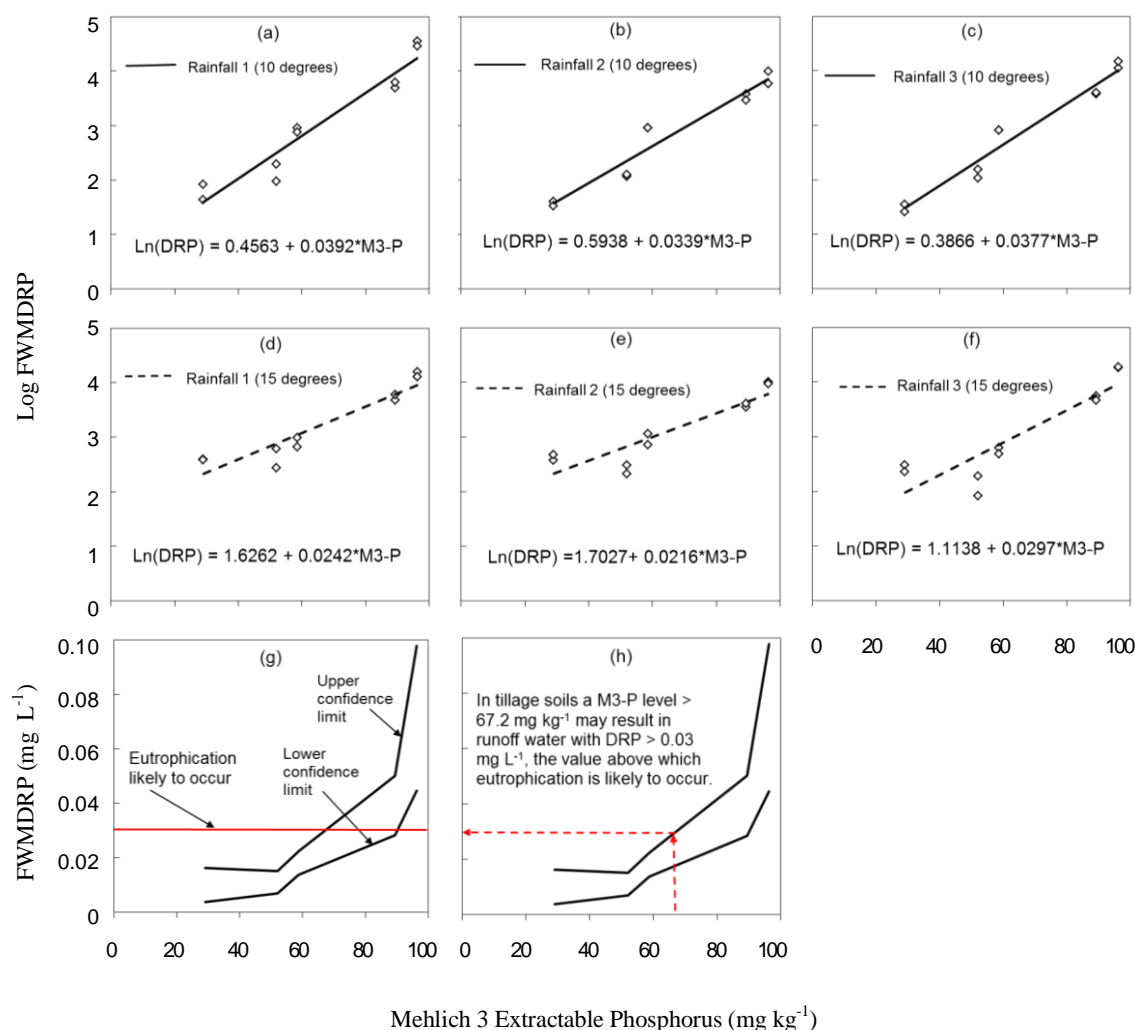
Research by Foy et al. (2002) and Pote et al. (1999) found a positive relationship between  $P_m$  and DRP lost in surface runoff. A positive relationship between surface runoff DRP and soil WEP was also reported by Penn et al. (2006), McDowell and Sharpley (2001), Wang et al. (2010) and Pote et al. (1996) using simulated rainfall on packed soil boxes. The WEP method closely mimics the interaction between rainwater and soil particles, and provides a good indication of DRP in runoff.

Under the test conditions, the 95% confidence bands indicated that provided the  $P_m$  does not exceed  $9.5 \text{ mg L}^{-1}$ , WEP does not exceed  $4.4 \text{ mg kg}^{-1}$  and M3-P does not exceed  $67.2 \text{ mg kg}^{-1}$ , for tillage soils, the concentration of DRP in runoff will be within the currently acceptable P range for surface water quality of  $< 0.03 \text{ mg L}^{-1}$ . The finding for  $P_m$  in this study reinforces the statutory requirements of SI 610 of 2010, which prohibit fertiliser application to tillage soils with a  $P_m > 10 \text{ mg L}^{-1}$  (Figure 4.1 h). In contrast, agri-environmental interpretation of M3-P in Delaware indicated that improved P management was necessary to reduce potential for nonpoint P pollution when  $M3-P > 150 \text{ mg kg}^{-1}$  (Sims et al., 2002). However, some states are adopting a critical M3-P of  $65 \text{ mg kg}^{-1}$  (Sharpley, 1995) as a cutoff point in their P indexing systems for rating the potential for P loss in runoff. A M3-P of  $65 \text{ mg kg}^{-1}$  represents the level at which no yield response to fertiliser P addition is expected (Sharpley, 1995) and is therefore comparable to the  $P_m$  of  $10 \text{ mg L}^{-1}$  used in Ireland which is also

based on there being no response to fertiliser P addition above this value. Fang et al. (2002) deemed a M3-P of  $65 \text{ mg kg}^{-1}$  to be more consistent with environmental levels of P that produce eutrophication than are higher values used by many other states such as Delaware. The threshold M3-P of  $67.2 \text{ mg kg}^{-1}$  determined in this study is in close agreement with the critical M3-P of  $65 \text{ mg kg}^{-1}$  now being adopted in some states.



**Figure 4.2 Runoff Dissolved Phosphorus Risk Indicator using water extractable P for selected tillage soils at 10 and 15 degree slopes under a  $30 \text{ mm hr}^{-1}$  rainfall. Rainfall 1 - 1st rainfall event; Rainfall 2 - 1 hr after Rainfall 1; Rainfall 3 - 24 hr after Rainfall 2.**



**Figure 4.3 Runoff Dissolved Phosphorus Risk Indicator using Mehlich-3 extractable P for selected tillage soils at 10 and 15 degree slopes under a 30 mm hr<sup>-1</sup> rainfall. Rainfall 1 - 1st rainfall event; Rainfall 2 - 1hr after Rainfall 1; Rainfall 3 - 24 hr after Rainfall 2.**

Under the test conditions, the 95% confidence bands indicated that provided the  $P_m$  does not exceed 9.5 mg L<sup>-1</sup>, WEP does not exceed 4.4 mg kg<sup>-1</sup> and M3-P does not exceed 67.2 mg kg<sup>-1</sup>, for tillage soils, the concentration of DRP in runoff will be within the currently acceptable P range for surface water quality of < 0.03 mg L<sup>-1</sup>. The finding for  $P_m$  in this study reinforces the statutory requirements of SI 610 of 2010, which prohibit fertiliser application to tillage soils with a  $P_m > 10$  mg L<sup>-1</sup> (Figure 4.1 h). In contrast, agri-environmental interpretation of M3-P in Delaware indicated that improved P management was necessary to reduce potential for nonpoint P pollution when M3-P > 150 mg kg<sup>-1</sup> (Sims et al., 2002). However, some states are adopting a

critical M3-P of  $65 \text{ mg kg}^{-1}$  (Sharpley, 1995) as a cutoff point in their P indexing systems for rating the potential for P loss in runoff. A M3-P of  $65 \text{ mg kg}^{-1}$  represents the level at which no yield response to fertiliser P addition is expected (Sharpley, 1995) and is therefore comparable to the  $P_m$  of  $10 \text{ mg L}^{-1}$  used in Ireland which is also based on there being no response to fertiliser P addition above this value. Fang et al. (2002) deemed a M3-P of  $65 \text{ mg kg}^{-1}$  to be more consistent with environmental levels of P that produce eutrophication than are higher values used by many other states such as Delaware. The threshold M3-P of  $67.2 \text{ mg kg}^{-1}$  determined in this study is in close agreement with the critical M3-P of  $65 \text{ mg kg}^{-1}$  now being adopted in some states.

The predictions of the RDPRI may be affected by soil management decisions. Changes in soil management resulting in the deterioration or improvement of soil drainage may have the effect of reducing the predictive power of the RDPRI by increasing or decreasing, respectively, the overland flow volume generated in upslope soil areas.

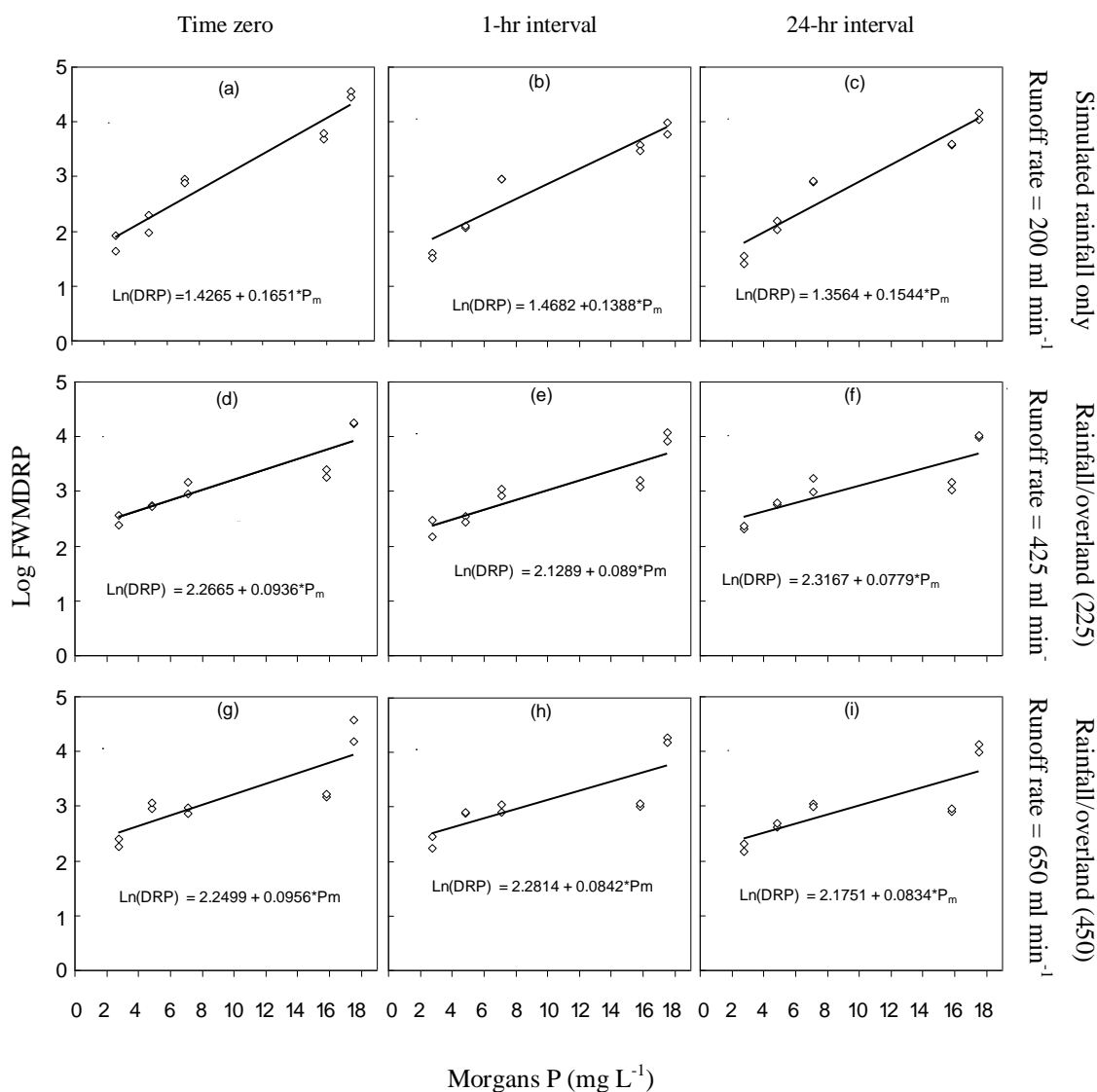
The next step is to examine the effect of increasing overland flow rates, which may result from intense summer rainfalls (more frequent intense rainfall events during the summer have been predicted for Ireland by Sweeney et al. (2008)), on the critical soil test P thresholds developed using the RDPRI.

#### **4.3.3 Investigating the effect of increasing overland flow rates on the critical soil test phosphorus threshold above which runoff may pose a threat to surface water quality**

The logarithm of the FWMC of DRP in surface runoff from each soil was linearly related to the  $P_m$  of each soil for all overland flow rates (Figure 4.4). The logarithm of the FWMC of DRP in surface runoff from each soil was also linearly related to WEP, M3-P,  $\text{Pcacl}_2$  and  $\text{P}_{\text{sat}_{\text{ox}}}$  concentrations in each soil for all overland flow/simulated rainfall events (Appendix C).

As was the case for simulated rainfall only experiments, the DRP lost in surface runoff when both simulated rainfall and overland flows were applied to the soil,

increased log-linearly as  $P_m$ , WEP, M3-P,  $P_{\text{cacl}_2}$  and  $P_{\text{sat}_{\text{ox}}}$  increased. For both overland flow rates on all 5 soils, the time zero event had the highest FWMC of DRP in runoff, while the FWMCs of DRP in surface runoff for the 1-hr and 24-hr events were lower than the time zero event, and were generally not significantly different in magnitude from each other.



**Figure 4.4 Log FWMDRP against Morgan's P for selected tillage soils at a 10 degree slope under a 30  $\text{mm hr}^{-1}$  rainfall and subjected to two distinct overland flow rates (225 and 450  $\text{ml min}^{-1}$ ).**

A critical value was derived for each of the soil extractable P methods above which surface water quality may exceed 0.03  $\text{mg L}^{-1}$  DRP, the concentration above which water quality may deteriorate (Figure 4.4). The 95% confidence limits/intervals

(Figure 4.5 and Figure 4.6) indicate that provided the  $P_m$  does not exceed  $7.83 \text{ mg L}^{-1}$ , WEP does not exceed  $4.15 \text{ mg kg}^{-1}$ , M3-P does not exceed  $61 \text{ mg kg}^{-1}$ ,  $\text{Pcacl}_2$  does not exceed  $1.2 \text{ mg kg}^{-1}$  and  $\text{Psat}_{\text{ox}}$  does not exceed 17.1% for tillage soils, the concentration of DRP in surface runoff will be below  $0.03 \text{ mg L}^{-1}$ . While these new values for  $P_m$ , WEP, and M3-P are lower than those determined using the RDPRI in the previous section, the confidence intervals of the two sets of values overlap by more than 25% (Table 4.1), indicating no significant difference. Furthermore, in an attempt to improve on the model used in the previous section, an all-in approach was adopted which allowed testing of any change from one combination of experimental factors to the next. The estimates of noise/error, on which confidence intervals are based, are better when all available data are used (as in the case of an all-in approach) and this may, in part account for the new lower values and narrower confidence intervals (Table 4.1) of the new model. The larger data set used in this model also improved its precision. Overall, there was no evidence of a linear relationship between overland flow rate and DRP concentration measured in surface runoff ( $p = 0.125$ ). As such, the new lower values for  $P_m$ , WEP, and M3-P are primarily a result of improvements in the model used to predict DRP in runoff. Table 4.1 shows the 95% confidence limits for each P extraction method at water quality limits of 0.03 and  $0.035 \text{ mg L}^{-1}$  (the new limit as set by SI 272 of 2009).

The finding for  $P_m$  in this study is in close agreement with the agronomic optimum ( $P_m = 6.1\text{-}10 \text{ mg L}^{-1}$ ) for plant growth and crop yields. It is also in close agreement with the statutory requirements of SI 610 of 2010, which prohibits fertiliser application to tillage soils with a  $P_m > 10 \text{ mg L}^{-1}$  (Figure 4.5a) and suggests that given the worst case storm scenarios tested, a change in the statutory  $P_m$  limit is unwarranted at this time. However, the results also suggest that the limit may have to be reviewed in future should the predicted climate change storms materialise. The threshold M3-P of  $61.2 \text{ mg kg}^{-1}$  determined in this study is in close agreement with the critical M3-P of  $65 \text{ mg kg}^{-1}$  now being adopted in some states in America (Figure 4.5c). A M3-P value of  $45\text{-}50 \text{ mg kg}^{-1}$  in soil is generally considered to be optimum for plant growth and crop yields (Sims, 2000). The findings of this study suggest that keeping the M3-P level in the study soils close to this agronomic optimum will ensure that the concentration of DRP in overland flow will be below  $0.03 \text{ mg L}^{-1}$ . Caution



must be exercised when interpreting STP results in an environmental context as they comprise only a small percentage of the total soil P reservoir and do not account for potential detachment (Haygarth and Condron, 2004) or dissolution from eroded sediments.

**Table 4.1 Comparing RDPRI outputs (Section 4.3.4) with new model outputs (this section) to identify thresholds for each of Morgan's P,  $P_m$ ; water extractable phosphorus, WEP; Mehlich-3, M3-P; calcium chloride extractable phosphorus,  $P_{cacl_2}$ ; and Soil P saturation,  $Psat_{ox}$ , above which DRP in surface runoff may exceed  $0.03 \text{ mg L}^{-1}$  (old limit - SI 258 of 1998) and  $0.035 \text{ mg L}^{-1}$  (new limit - SI 272 of 2009)**

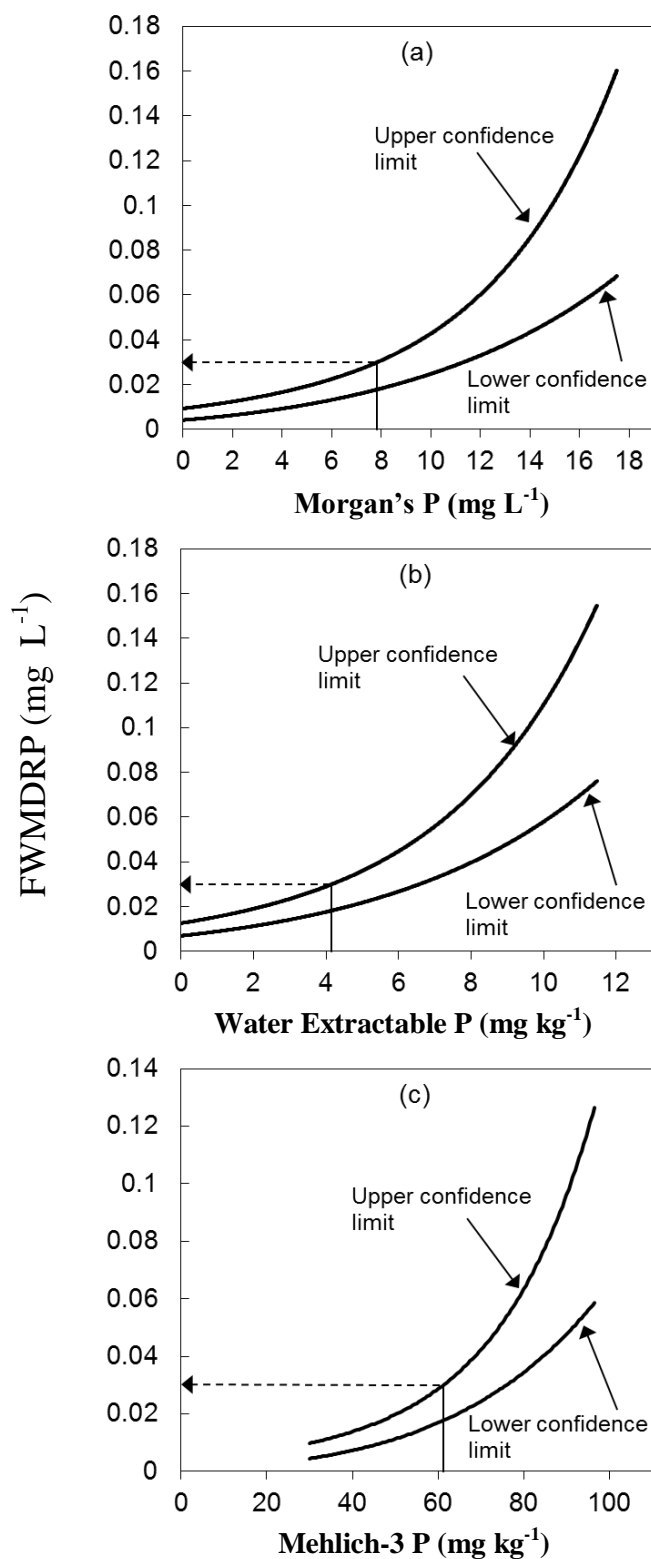
	Upper 95% CL	Lower 95% CL	% Overlap of intervals
Phosphorus $< 0.03 \text{ mg L}^{-1}$ (Old limit - SI 258 of 1998)			
RDPRI $P_m$ ( $\text{mg L}^{-1}$ )	9.5	16	52
New $P_m$ ( $\text{mg L}^{-1}$ )	7.83	11.31	
RDPRI WEP ( $\text{mg kg}^{-1}$ )	4.4	6.83	89
New WEP ( $\text{mg kg}^{-1}$ )	4.15	6.57	
RDPRI M3-P ( $\text{mg kg}^{-1}$ )	67.2	89.3	59
New M3-P ( $\text{mg kg}^{-1}$ )	61.2	76	
$P_{cacl_2}$ ( $\text{mg kg}^{-1}$ )	1.2	1.79	-
$Psat_{ox}$ (%)	17.1	24.1	-
Phosphorus $< 0.035 \text{ mg L}^{-1}$ (New limit - SI 272 of 2009)			
$P_m$ ( $\text{mg L}^{-1}$ )	8.78	12.42	-
WEP ( $\text{mg kg}^{-1}$ )	4.87	7.35	-
M3-P ( $\text{mg kg}^{-1}$ )	65.2	80.5	-
$P_{cacl_2}$ ( $\text{mg kg}^{-1}$ )	1.36	2	-
$Psat_{ox}$ (%)	18.9	26.44	-

CL, Confidence limit

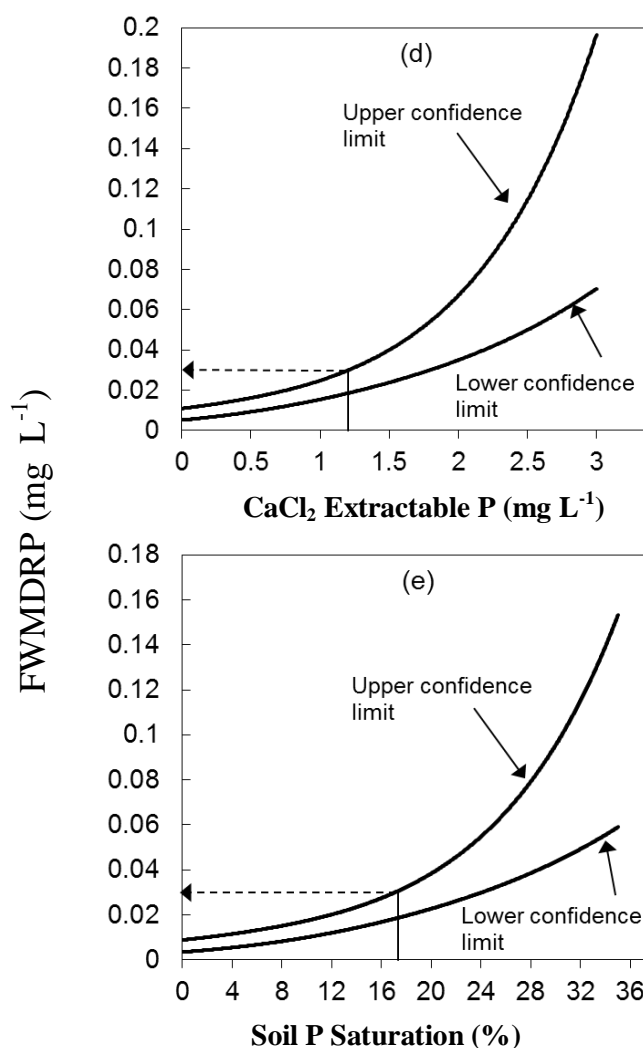
Soil P tests developed for environmental purposes such as WEP and  $P_{cacl_2}$  are less affected by soil type than agronomic soil tests like M3-P and  $P_m$  (Self-Davis et al., 2000), and can be valuable for estimating labile forms of P (Simard et al., 1995).

Their ability to either better mimic the interaction between soil and runoff or better represent the likelihood of P release from soil to runoff (Vadas et al., 2005), makes them ideal P loss risk indicators. The relationships developed in this study, between runoff DRP and each of WEP and  $P_{\text{CaCl}_2}$  are shown in Figure 4.5b and Figure 4.6a, respectively. In Section 4.3.5, soil P tests are compared to determine the best environmental P loss indicator.

A  $\text{Psat}_{\text{ox}}$  value of 25%, or more, has been established on the basis of laboratory data for non-calcareous, sandy soils from the Netherlands, as a critical value above which the potential for P losses through runoff and leaching become unacceptable. The water quality standard used in the Netherlands in determining this critical value is 0.1 mg ortho-P  $\text{L}^{-1}$  (Breeuwsma et al., 1995). The findings of the present study suggest that a lower limit of 17.1%  $\text{Psat}_{\text{ox}}$  would better protect water quality in Ireland by ensuring that DRP losses from Irish tillage soils remain below 0.03 mg P  $\text{L}^{-1}$ . Soils with  $\text{Psat}_{\text{ox}}$  values that exceed this threshold, such as the Tullow and Clonmel soils, are more heavily saturated with P and are vulnerable to losses to overland flow by P desorption. This lower  $\text{Psat}_{\text{ox}}$  threshold, as determined for Irish tillage soils subjected to overland flow and rainfall in this study, may be a result of the P sorption capacity (PSC) used in the determination of  $\text{Psat}_{\text{ox}}$  for the study soils. The PSC of soils varies widely depending on clay content, clay mineralogy, OM content, exchangeable Al, Fe, and Ca concentrations, and soil pH (Tisdale et al., 1993). Furthermore, results from Beauchemin and Simard (1999) indicate that the relationship between PSC and  $[\text{Fe}_{\text{ox}} + \text{Al}_{\text{ox}}]$  contents may vary among soil groups. Therefore, the extension of the 25%  $\text{Psat}_{\text{ox}}$  threshold to other soil types and other water quality standards may not be appropriate (Beauchemin and Simard, 1999).



**Figure 4.5** Runoff Dissolved Phosphorus Risk Indicator using Morgan's P, water extractable P, and Mehlich-3 P for selected tillage soils at a 10 degree slope, under a 30 mm hr<sup>-1</sup> rainfall and subjected to two distinct overland flow rates (225 and 450 ml min<sup>-1</sup>).



**Figure 4.6 Runoff Dissolved Phosphorus Risk Indicator using calcium chloride extractable P and soil P saturation for selected tillage soils at a 10 degree slope, under a 30 mm hr<sup>-1</sup> rainfall and subjected to two distinct overland flow rates (225 and 450 ml min<sup>-1</sup>).**

#### **4.3.4 Phosphorus and sediment loss risk indicators for Irish tillage soils subjected to simulated rainfall**

Soil extractable P can be measured in numerous ways in an attempt to predict DRP available to runoff. A stepwise regression selection procedure was used to identify the soil extractable P method best suited to predicting DRP loss from Irish tillage soils when subjected to simulated rainfall. Measurement of soil WEP was selected as important when predicting DRP in runoff across all soils, slopes and rainfall events.

This is in agreement with other studies where WEP provided the strongest correlation with DRP concentrations in runoff when compared to other STP methods such as M3-P and  $P_m$  (Pote et al., 1996), and was able to simulate actual runoff DRP concentrations (Yli-Halla et al., 1995). Soil parameters also selected as important by stepwise regression to test for when predicting DRP in runoff were: STP, AgSt, pH,  $P_{\text{cacl}_2}$ , and clay. However, these were only selected for certain combinations of slope and rainfall event, and are less reliable indicators.

The effect of soil type on SS, PP and TP loss to water differs depending on slope and, consequently, the stepwise procedure was performed separately for each slope. The soil parameters selected as important (by stepwise regression) in predicting SS loss to water were: OM, AgSt and clay. Soil P saturation was selected as important when predicting PP and TP lost in runoff.

#### **4.3.5 Phosphorus loss risk indicators for Irish tillage soils subjected to simulated rainfall/overland flow**

This study's findings show that, for soils subjected to both simulated rainfall and overland flow, WEP (AIC = -12.3) and  $P_{\text{cacl}_2}$  (AIC = -6.9) performed better than the agronomic soil P tests,  $P_m$  (AIC = 6.4) and M3-P (AIC = 7.1) in predicting DRP in overland flow. Only large differences (here, at least 4) between AIC scores for soil extractable P tests are taken as indicating a difference in the goodness of fit of the model used for determining the critical value of DRP. The addition of WEP to the model produced the best increase in goodness of fit (as is evidenced by WEP receiving the lowest AIC score of -12.3 and the difference between it and the next lowest AIC being > 4) and therefore performed better than the other measures of soil P when predicting DRP in runoff. These results are in agreement with Pote et al. (1999), Penn et al. (2006) and Wang et al. (2010), who reported that WEP had consistently stronger relationships with DRP concentrations in surface runoff than other measures of soil P. The authors attributed this to the fact that the extracting solution for WEP (distilled water) is more similar to the simulated rainfall water (tap water) than other extracting solutions.

Soil P saturation ( $AIC = 20.6$ ) ranked lowest, which is probably due to some of the study soils being slightly calcareous in nature and the fact that soil texture ranged from sandy loam to loam. In other studies, the calculation of  $Psat_{ox}$  using  $P_{ox}/0.5(Al_{ox} + Fe_{ox})$  has produced strong correlations with soluble P when the range of soils used was homogeneous, but the relationship weakens if a wider range of soil types is considered (Beauchemin and Simard, 1999). In a study of published data from 17 studies, Vadas et al. (2005) concluded that for noncalcareous soils, a test for soil P saturation (determined by acid ammonium oxalate extraction) may provide a more universal prediction of dissolved P in runoff than Mehlich-3, Bray-1, or water extractions. Soil P saturation measures the degree to which soil P sorption sites have been filled and has been found to be a good indicator of P availability to runoff (Kleinman and Sharpley, 2002).

Due to the limited range in some of the soil parameters measured across the 5 soils (this was largely due to tillage in Ireland being conducted primarily on sandy loam or loam soils), it was difficult to ascertain which parameters had a significant effect on DRP in overland flow. Although higher levels of  $Fe_{ox}$  and  $[Fe_{ox} + Al_{ox}]$  measured in the study soils were found to have a significant lowering effect ( $p < 0.05$ ) on the concentration DRP measured in overland flow. This further emphasises the important role these parameters play in determining the PSC of a soil.

#### 4.4 Summary

The RDPRI, developed in this chapter, identified the levels of extractable soil P above which surface runoff, resulting from both high intensity simulated rainfall and overland flow on tillage soils, may, under the conditions tested, pose a threat to surface water quality. The main conclusions from this study were:

1. Tilled soils, subjected to simulated rainfall only, may produce surface runoff P concentrations in excess of  $0.03 \text{ mg L}^{-1}$  (the value above which eutrophication of rivers is likely to occur) if their  $P_m$ , WEP, and M3-P concentrations exceed  $9.5 \text{ mg L}^{-1}$ ,  $4.4 \text{ mg kg}^{-1}$ , and  $67.2 \text{ mg kg}^{-1}$ , respectively.

2. The RDPRI developed here, using both simulated rainfall and overland flow, showed that provided the  $P_m$  does not exceed  $7.83 \text{ mg L}^{-1}$ , WEP does not exceed  $4.15 \text{ mg kg}^{-1}$ , M3-P does not exceed  $61 \text{ mg kg}^{-1}$ ,  $P_{\text{cacl}_2}$  does not exceed  $1.2 \text{ mg kg}^{-1}$  and  $P_{\text{sat}_{\text{ox}}}$  does not exceed 17.1% for tillage soils, the concentration of DRP in surface runoff will be below  $0.03 \text{ mg P L}^{-1}$ .
3. The finding for  $P_m$  in this study is in close agreement with the agronomic optimum ( $P_m = 6.1\text{-}10 \text{ mg L}^{-1}$ ) used in Ireland for plant growth and crop yields. It is also in close agreement with the statutory requirements of SI 610 of 2010, which prohibits fertiliser application to tillage soils with a  $P_m > 10 \text{ mg L}^{-1}$ .
4. Of the five soil extractable P methods investigated, WEP was identified, using stepwise linear regression, as having the greatest potential to be used as an indicator of the risk of P movement from soil into runoff water via dissolution. Despite its apparent advantage over  $P_m$  in determining environmental risk, it would appear to be impractical and costly to run two soil P tests side by side given that  $P_m$  gives a good approximation for both agronomic and environmental purposes.

Chapters 3 and 4 have shown that if current guidelines for P application to tillage soils are adhered to, the risk of P loss via dissolution should be minimal. They have also demonstrated that P loss from tillage soils, when subjected to high intensity rainfall and overland flow under controlled conditions, is primarily associated with eroded sediment and is therefore a potential long-term source of P for algae and plant uptake from surface water bodies, in particular lakes. The next step, therefore, is to determine the erosion risk posed by the study soils in their natural field conditions. The identification of tillage fields posing a high erosion risk will facilitate the deployment of mitigation measures solely in those fields, thereby greatly reducing the number of sites susceptible to PP loss in Ireland. Chapter 5 details the development and preliminary testing of a novel screening toolkit with which, the farmer/specialist advisor can identify fields where the erosion risk is high and soil quality is an issue.

## **Chapter 5 Physical, chemical and visual evaluation of six Irish tillage soils to assess soil quality and susceptibility to erosion**

### **Overview**

In this chapter, a novel screening toolkit is developed with which farmers/specialist advisors can screen tillage fields for likelihood of erosion and reduced soil quality. At each study site, detailed soil classification results and simple soil quality assessments are used in conjunction with observed erosion levels to select the most appropriate indicators for assessing erosion risk and soil quality status. A preliminary validation of the screening toolkit showed it to be effective at identifying fields where the erosion risk is high.

### **5.1 Introduction**

Soil is a vital, non-renewable resource which requires sustainable management to ensure the viability of food and fibre production, nutrient retention and cycling, and filtration of water (Creamer et al., 2010). Emergence of policies, such as the proposed SFD which deal with concern over soil degradation and anthropogenic impacts to soil is likely to increase the requirement for assessment of soil quality and identification of soils at risk from degradation (Bone et al., 2012). The SFD aims to establish a common framework to protect, preserve and prevent further degradation to soil and its associated functions. Under the SFD, seven main threats to soil quality are recognised: erosion (water, wind and tillage), decline of SOC, compaction, contamination, salinisation, landslides and desertification (Soil Strategy in 2006 (COM (2006) 231). The European project ‘Sustainable Agriculture and Soil Conservation (SoCo) - Case Studies’ identified the main concerns for sustainable agriculture on soils in Northern and Western Europe to be: soil erosion by water, decline in SOM; diffuse soil contamination, in particular contamination associated with nitrates and agrochemicals; and compaction. The case studies, which comprised



the Northern and Western European area of the SoCo project, were concerned primarily with arable farming systems under intensive management conditions. It is in these systems primarily, that efforts must be focused in order to reduce soil erosion losses closer to tolerable levels.

Despite having a significantly higher P export coefficient (Doody et al., 2012) and greater susceptibility to erosion (Boardman and Poesen, 2006), arable land, which comprises 10% of agricultural area utilised in Ireland, has not received the same level of attention as the more dominant grassland and therefore its current degradation status and contribution to surface water impairment in Ireland is as yet unknown. Research in England and Wales by Chambers et al. (1992 and 2000) and Chambers and Garwood (2000) has shown that soil erosion by water is most prevalent in the southwest and southeast of the country where tillage cultivation on light sandy soils predominates. Tillage land is similarly distributed in Ireland, with 48% of crop production concentrated in the south of the country (Schulte et al., 2010a), where the soils are highly suitable for tillage, having a light-to-medium texture. Unlike in England and Wales, where extensive research of soil erosion in tillage areas has been conducted at multiple scales over the past 40 years, research incorporating tillage in Ireland is only recently underway in the form of the ACP. The ACP includes two catchments (9.4 and 11.2 km<sup>2</sup>) with high proportions of winter wheat or spring barley cropping (Wall et al., 2011), and will serve as the first assessment of nutrient and sediment loss from agricultural catchments with high proportions of tillage land in Ireland.

Ireland has a valuable resource in terms of its land and soil quality, and promoting sustainable soil management is one of the areas of action included in Food Harvest 2020 (DAFF, 2010), the national strategy for the development of the agri-food sector. Agricultural activities that can negatively impact on soil quality must be tackled if Ireland is to meet the ambitious growth targets set out in this vision. However, the main focus of current agricultural and environmental policy, such as the cross-compliance regulations that accompany the Common Agricultural Policy and agri-environment schemes, tends to be on control of diffuse water pollution rather than protecting or conserving soil *in situ* (Posthumus et al., 2011). As such, focus in

Ireland has been on maintaining good water quality with limited attention being given to soil quality. Just as soils are regularly tested for nutrients, their physical condition must receive similar and appropriate attention (Batey, 2009) to ensure that management practices are sustainable. Soil structure is a crucial soil property that affects several processes important to soils productive capacity, environmental quality and agricultural sustainability (Lal, 1991). Visual and tactile assessments of soil and its structure carried out in the field can identify areas where soil threats like compaction (vehicle or cultivation induced), decline of SOC (due to intensive cultivation and removal of crop residues particularly on coarse textured soils), and increased risk of surface runoff (due to reduced soil porosity) leading to erosion, may be occurring. These threats to soil quality - if not prevented or their effects at least mitigated - can put the soil's productivity and surface water quality at risk, thereby hindering the achievement of the growth targets set out in Food Harvest 2020 and delaying Ireland's progress towards achievement of at least "good status" for all waters by December 2015 as required by the WFD (2000/60/EC: Council of the European Union, 2000).

Given time and capital constraints, it would be prohibitive for countries to use detailed quantitative approaches to assess all soils, because there is a possibility that the highest priority soils would not be reached if such methods are employed (Bone et al., 2012). Moreover, as government alone cannot take all the steps necessary to safeguard our soil resource for future generations, farmers and other land managers have an essential role to play in managing agricultural soils sustainably (Defra, 2009a). Methods of erosion risk assessment and soil structure and quality assessments that are reliable, quick, inexpensive and conductible by specialist advisors and farmers have been developed in other countries in recent years. In England, the system of risk-assessment that farmers must follow as part of the 'cross-compliance' regime is set out in "Controlling Soil Erosion: a Manual for the Assessment and Management of Agricultural Land at Risk of Water Erosion in Lowland England" Defra (2005). The efficacy of the Defra (2005) risk assessment scheme was tested by Boardman et al. (2009) and shown to be 90% successful (18/20 cases) at identifying high risk sites given the land-use at the time. Visual Soil Assessment (VSA; Shepherd, 2009) and Visual Evaluation of Soil Structure (VESS; Guimarães et al., 2011; Ball et al., 2011)

based on the Peerlkamp test (Peerlkamp, 1959)) were developed in New Zealand and the UK, respectively, to enable farmers to assess soil structure and quality. Mueller et al. (2009) showed that soil structure scored by VSA and VESS was significantly correlated with certain soil physical parameters (dry bulk density, soil strength, and infiltration rate) affecting runoff, erosion and crop yields. Using these methods, soil structure can be assessed quickly on site and the land user may take a positive part in the evaluation (Batey and McKenzie, 2006), and by doing so, gain a better understanding of the cultivation practices or management decisions that led to the degradation. This is in keeping with the Department of Agriculture, Fisheries and Food's Harvest 2020 vision which states that "primary producers have a valuable role to play as guardians of the rural environment".

While detailed quantitative assessment methods such as those employed to classify soils in the Soil Survey of Ireland (Gardiner and Radford, 1980) are inappropriate for assessing erosion risk and soil quality on a field by field basis, they do provide valuable information on the erodibility (as determined using the K-factor (Wischmeier and Smith, 1978)) of a mapped soil series. The K-factor is related to four crucial soil properties triggering erosion: SOM, soil texture (particle size analysis), soil structure, and permeability (Panagos et al., 2012). These properties were identified through nationwide studies performed by the United States Department of Agriculture - Natural Resources Conservation Service using rainfall simulation tests (USDA-NRCS, 2005). Data collected during the Soil Survey of Ireland and data currently being collected as part of the new ISIS (Creamer et al., 2010) represents the only soil physical data available in Ireland, and can be used to predict the soils potential for erosion by water. If the SFD is eventually ratified, Ireland will be required to identify areas where erosion has occurred in the past or is likely to occur in the future. At that time, the soil information provided by ISIS will be essential in identifying these areas.

There is currently no standard for the assessment of erosion risk or soil quality in Ireland. The aim of this study was to develop and conduct a preliminary validation of a novel, easy-to-use screening toolkit for use by farmers/specialist advisors in assessment of tillage fields for likelihood of erosion and reduced soil quality. Using different methods of erosion risk, and soil structure and quality assessment applied to

a representative selection of Irish tillage soils in which different levels of erosion were observed, I investigated the erodibility of these soils and developed and conducted a preliminary validation of a screening toolkit containing a set of erosion and soil quality indicators, which can be used by Irish tillage farmers to identify, expeditiously, fields with high erosion risk and poor soil quality. The next step would be to make the screening toolkit part of a larger-scale erosion study in which measured edge-of-field sediment losses can be used to validate the erosion risk rankings assigned using the screening toolkit.

## **5.2 Site information and soil assessment methods**

### **5.2.1 Site selection**

As previously mentioned in Section 3.2, the soils used in this study were broadly representative of the major tillage areas in Ireland (Figure 3.1). Initially, fourteen tillage field sites, spread across the major tillage areas of Ireland, were visited and the most suitable sites (6 based on soil type, tillage history, slope and evidence of prior erosion problems), which exhibited a wide range of physical and chemical properties, were chosen. Tillage areas are predominantly in the midlands, south and east of Ireland (Figure 3.1), where there is a drier climate (average annual rainfall (AAR) of between ca. 657 and ca. 1400 mm) than other areas of the country and soil types are free draining, making them more suitable for spring cultivations and less likely to be damaged by harvesting machinery. This leads to better opportunities for seedbed preparation and harvesting, as well as potentially higher grain yields (Collins and Cummins, 1996). The physical characterisation of the soils, presented in Section 5.4, was carried out between May and November 2007. The topography of the sites ranged from undulating lowland at the Bunclody and Tullow sites to hilly at the Fermoy site. The remaining sites had rolling lowland topography.

Exact location, tillage history, and observed erosion features are given in Table 5.1. There was no identifiable reduction in grain yield over time at any of the sites according to the land users, while yields were deemed to be improving at the Clonmel site as a result of better husbandry. The Fermoy and Duleek sites relied solely on

chemical fertiliser to supply crop nutrients because of a lack of easily accessible organic fertilisers. On sites where organic fertilisers such as turkey litter, cheese sludge and beet factory sludge lime were applied, the  $P_m$  levels in the soil were higher than those sites that relied solely on chemical fertiliser. At some sites (Table 5.1), subsoiling to a depth of 36 cm was carried out along headlands of fields where tillage compaction had occurred. On sites where OSR was planted, subsoiling was carried out over the whole field area to shatter soil below plough depth, thereby eliminating compact layers which inhibit the deep rooting crop.

### **5.2.2 Management history of the sites**

Tramlines at the study sites were normally used 4-7 times per year for crops like OSR and winter oats and up to 10 times per year for winter wheat. The greater number of spraying operations associated with winter wheat and winter oat crops increases the risk of compaction of tramlines and headlands. The planting of OSR facilitated the use of minimum tillage in the following year. Minimum tillage was practiced in some years on some sites (Table 5.1). In Ireland, minimum tillage normally involves: (1) shallow cultivation using a tine cultivator or disc harrow to a depth of 75-100 mm immediately followed by rolling; (2) spraying with herbicide a few days prior to sowing, following a stale seedbed period of a number of weeks (where possible) to eliminate volunteers and established weeds; and (3) sowing with a cultivator drill to a target depth of 40 mm (Forristal and Murphy, 2009). The only dramatic erosion event reported across all sites occurred at the Clonmel site, where there was a large wash out of soil in November 2010, leaving a gully behind; gully dimensions were approximately 35 m in length, 0.3 m in breath and 0.35 m in depth. The farmer used earth moving equipment to return the soil after the crop was harvested.

**Table 5.1 Information on selected tillage sites.**

Study site	Lat/long <sup>1</sup>	Tillage history to present day	Cultivation Method	Erosion Features	Subsoiling	Organic Fertiliser	Farmers Comments	Typical yields (at 20% MC)
Buncloody, Co. Wexford	52°36'59" N, 6°33'46" W	1980- : rotation of beet (replaced by beans/OSR after 2005) every 3 yr and cereals in between	Ploughed to 18 cm; some min till	Moderate erosion on longer slopes during prolonged heavy rainfall	2010 (36 cm)	BFSL (2001 and 2003)	Moderate compaction on headlands/tramlines prior to subsoiling	t/ha SW = 9.9; WB = 8.9; SB = 7.4
Tullow, Co. Carlow	52°48'00" N, 6°50'07" W	1981- : rotation of beet (replaced by WO/OSR after 2005) every 3 yr and cereals in between	Ploughed to 23 cm; some min till	Moderate erosion on longer slopes during prolonged heavy rainfall	No	Turkey litter, FYM, slurry, and BFSL	Moderate compaction on headlands/tramlines	WO = 8.6; WW = 11.1-12.4
Fermoy, Co. Cork	52°08'00" N, 8°12'08" W	1985- : rotation of beet (beans/OSR after 2005) every 4 yr and oats, barley and wheat in between	Ploughed to 20 - 25 cm; one-pass system; some min till	Accumulation of soil at bottom of slope over time	No	Some straw incorporation	P deficiency; cultivation and rolling carried out across the contour	WW = 9.9
Clonmel, Co. Tipperary	52°16'51" N, 7°47'07" W	1979- : rotation of beet (replaced by WO/OSR after 2005) every 3 yr and cereals in between	Ploughed to 20 cm; one-pass system	Erosion on slopes > 4°; gully eroded in 2011 due to mismanagement	2010 (36 cm)	Cheese sludge	Ploughed at 35° angle to slope direction; rolled across slope; compaction on headlands/tramlines	WW = 11.1-12.4; WO = 8.0; OSR = 5.2
Letterkenny, Co. Donegal	54°59'55" N, 7°32'57" W	1995- : rotation of potatoes every 4 yr and cereals in other years	Ploughed to 18 cm; one-pass system	Erosion during heavy rainfall in particular on tramlines; flooding	2007 (36 cm)	FYM (2007 and 2008)	Prone to compaction and cracking; poor growth at bottom of slope due to deposition of sand/clay	WB = 8.9
Duleek, Co. Meath	53°38'57" N, 6°23'14" W	1990- : planted with beans or left fallow every 4 yr with SB in between	Ploughed to 18 cm; seed spreader	No	No	Difficult to access	Prone to clodding (irregular blocks created by artificial disturbance)	SB = 4.9 – 6.2

<sup>1</sup> Lat/long, latitude and longitude; MC, moisture content; OSR, oilseed rape; BFSL, beet factory sludge lime; SW, spring wheat; WB, winter barley; SB, spring barley; WO, winter oats; FYM, farmyard manure; WW, winter wheat;

### 5.2.3 Methods used to assess erosion risk and soil quality at the study sites

No erosion risk or soil quality assessment standard currently exists in Ireland. Therefore, the methods of erosion risk (Defra, 2005), soil structure (VESS) and soil quality (VSA) assessment developed in the UK and New Zealand in recent years were applied to the study soils in order to develop a risk assessment for erosion in Ireland by improving on the Defra (2005) erosion risk assessment. While the primary objective was to develop an improved erosion risk assessment by incorporating the elements of VESS and VSA that affect surface runoff generation and soil erosion into Defra (2005), it was also envisaged that these newly added elements would give an indication of soil structural quality, as that was their original function in the VESS and VSA methods. The methods of soil structure and soil quality assessment are compared with the Soil survey of Ireland soil classification method in Table 5.2 and advantages and limitations of each method are outlined in Table 5.3.

**Table 5.2 Methods used in visual and tactile assessment**

<b>Technique</b>	<b>Soil Survey of Ireland profile description (depth = up to 155cm)</b>	<b>Visual soil assessment (depth = top 20 cm)</b>	<b>Visual evaluation of soil structure (depth = top 25 cm)</b>
<b>Soil textural class</b>	Hand assessment	Hand assessment	-
<b>Size distribution (fragmentation)</b>	-	Drop shatter test (photo comparison)	-
<b>Soil unit morphology</b>	Aggregates are assessed for size, grade, shape, consistence, porosity and roots	Aggregates are assessed for size, shape, porosity and strength	Aggregates are assessed for size, shape, porosity, roots and strength (photos)
<b>Biological activity</b>	Record size, quantity, and orientation of roots	Record earthworm number and species type, soil smell, and roots	Check for anaerobic zones and presence of large worm holes
<b>External porosity</b>	Record quantity, size, shape, orientation and continuity of pores using a hand lens	Examine an exposed soil face for spaces, holes, cracks and fissures between aggregates and clods (photos)	Check for abundance and clustering of roots, macropores, and cracks between aggregates
<b>Internal porosity</b>	Check quantity, size shape, orientation and continuity of pores within aggregates	Check for pores within clods and aggregates	Check for pores and roots within aggregates
<b>Colour</b>	Assess soil colour and mottling according to Munsell soil colour charts	Comparison with soil colour under fenceline and quantification of mottling (photos)	Grey-blue colour recorded if present
<b>Water content</b>	Assess drainage condition	Assessment of degree of surface ponding (photos)	-
<b>Rooting</b>	Potential rooting depth – record quantity, size and location of roots in each horizon and look for clustering, thickening and deflection of roots	Potential rooting depth – record overthickening or forced horizontal growth of roots, firmness and tightness of soil and hard pans (in top 80cm)	Look for clustering, thickening and deflection of roots

**Table 5.3 Advantages and limitations of respective methods**

Method	Advantages	Limitations
<b>Soil Survey of Ireland profile description</b>	<ul style="list-style-type: none"> <li>• Soil assessed to full depth of profile</li> <li>• Parent material and profile drainage are determined</li> <li>• Sensitive enough to detect slight changes in soil structure</li> </ul>	<ul style="list-style-type: none"> <li>• Time consuming and labour intensive</li> <li>• Replication difficult for reasons above</li> <li>• Agronomists and farmers must be trained in its use</li> <li>• Not suitable for identifying effects of cropping systems</li> </ul>
<b>Visual soil assessment</b>	<ul style="list-style-type: none"> <li>• Special training or technical skills are not required</li> <li>• Objective method of assessment</li> <li>• Can identify effects of cropping systems on soil structure and rooting</li> <li>• Links soil condition to plant performance</li> <li>• Relatively quick</li> <li>• Takes into account surface cover, erosion and nutrient loss</li> </ul>	<ul style="list-style-type: none"> <li>• Only deals with top 20cm of profile</li> <li>• No assessment of profile drainage or parent material</li> </ul>
<b>Visual evaluation of soil structure</b>	<ul style="list-style-type: none"> <li>• Special training or technical skills are not required</li> <li>• Objective method of assessment</li> <li>• Can identify effects of cropping systems on soil structure and rooting</li> <li>• Low cost, rapid and flexible test</li> <li>• Large numbers of replicates are possible for statistical analysis</li> </ul>	<ul style="list-style-type: none"> <li>• Only deals with top 25cm of profile (can be adapted to assess deeper layers)</li> <li>• Potential rooting depth and profile drainage not determined</li> <li>• No assessment of erosion features</li> <li>• Limited measurement of biological activity</li> <li>• No assessment of soil texture which is important for erosion risk</li> <li>• No size distribution of aggregates</li> </ul>

#### 5.2.3.1 Field-Scale Defra (2005) Assessment

The Defra (2005) method is divided into two stages: first, each field is assessed by the farmer on the basis of soil texture and slope (Table 5.4) and a map of erosion risk is produced for the farm; secondly, the cropping regime or land use is classified by degree of erosion susceptibility. The method followed in determining the soil texture in each of the study fields is outlined in Appendix D. The slope of each of the study fields in the present study was determined using an inclinometer. The observation of rills, gullies, or sediment transport at the study sites during assessment overrode the classifications given in Table 5.4.

Additional factors used in the assessment to upgrade or downgrade a study site included: soil structure; SOM; valley features which tend to concentrate runoff water; long, unbroken slopes and very steep slopes (i.e. greater than 11 degrees). If a study site was reported to flood, on average, once in every 3 years or more frequently, it was deemed to be highly vulnerable. The land uses deemed by Defra (2005) to leave the soil in the most erosion susceptible condition include: late sown winter cereals;



potatoes; sugar beet; field vegetables; and grazed fodder crops. These land uses should be avoided on very high or high erosion risk fields unless erosion control measures are in place. Spring cereals are generally at lower risk from erosion than winter cereals, as seedbeds are not exposed to winter rainfall.

**Table 5.4 Water erosion risk-assessment (from Defra, 2005)**

Soils	Steep slopes > 7°	Moderate slopes 3° - 7°	Gentle slopes 2° - 3°	Level ground < 2°
Sandy and light silty	Very high	High	Moderate	Lower
Medium and calcareous	High	Moderate	Lower	Lower
Heavy	Lower	Lower	Lower	Lower

Very high = rills are likely to form in most years and gullies may develop in very wet periods; High = rills are likely to develop in most seasons during wet periods; Moderate = sediment may be seen running to roads, ditches or watercourses and rills may develop in some seasons during very wet periods; Lower = sediment rarely seen to move but polluting runoff may enter ditches or watercourses.

In 2009, Defra published a new soil protection review (Defra, 2009b), the purpose of which is to tackle degradation threats to soil. It explains about risks, and guides farmers through the process of identifying the risks on their soils and how to address them. It requires the farmer to produce a map of his land identifying the fields at risk of degradation (erosion, compaction and SOM decline) based on measured soil texture, observation of runoff, erosion, compaction, waterlogging and SOM decline, and knowledge of the site conditions and locations of fields. Measurement of slope is not required in the new soil protection review approach to soil risk mapping, which attempts to classify the field in terms of its susceptibility to three degradation threats as opposed to focusing on erosion as the Defra (2005) approach does. While there are certainly merits to this new approach, it was deemed more appropriate to use the Defra (2005) approach in the present study, as the focus was on identifying sites with a high risk of erosion which is difficult if slope is omitted. Furthermore, Boardman et al. (2009) showed that the Defra (2005) approach incorporating slope was 90% successful at identifying high risk fields, whereas the effectiveness of the new approach in which slope is omitted, is unknown. The suitability of the Defra (2005) erosion risk assessment for use in Irish conditions is discussed below.

Any proposed approach to erosion risk assessment on tillage land in Ireland should be one that requires limited investment of resources and, if possible, results in higher productivity through improved soil structure and quality. Ireland is similar to England in that erosion is not widespread but occurs in many localised areas and therefore requires a field-scale reconnaissance approach to identify risk areas where mitigation measures are warranted. The application of the Defra (2005) scheme to tillage land in Ireland would be a positive first step towards quantifying the actual erosion risk associated with tillage soils here. Given that (1) more frequent, intense rainfall events in summer are projected and (2) winter rainfalls are expected to increase by approximately 10% in Ireland by 2050 as a result of climate change (Sweeney et al., 2008), there is an increasing need for erosion risk assessments that can identify areas where erosion is occurring or is likely to occur under future climatic conditions, with a view to ultimately developing measures to mitigate the effects of erosion. Boardman et al. (2009) note that in the long-term, risk assessment procedures will have to take into account these predicted climate changes. The application of the Defra (2005) scheme to Irish conditions should not pose great difficulties because of its simple design - requiring only minimal input by farmers. Furthermore, the soil types, topography, rainfall and land uses do not differ greatly between the two countries so adjustments to suit Irish conditions should also be minimal. Its adoption in Ireland would present an opportunity to improve the scheme by including a simple assessment of soil structure and quality, conductible by the farmer. The need to assess soil structure when determining erosion risk was highlighted in the Defra (2005) scheme; however, no procedure for assessing it was provided. If soil structure and quality assessments are included in the scheme, the farmers will be made more aware of the links between poor soil structure and erosion, and good soil quality and productivity. This could increase uptake by farmers by creating a 'win-win' situation in which the farmer is rewarded for better soil management through improved crop yields while the local environment benefits from lowered erosion levels. This may also reduce the reliance on monetary incentives to ensure high uptake of agri-environmental schemes by farmers.

### 5.2.3.2 Visual Soil Assessment

All VSAs were carried out in the present study when the soil was moist or slightly moist. Consultation with the farmer and a visual inspection at each site identified specific locations where a VSA should be conducted. At each location, the VSA was carried out in triplicate using the equipment in Figure 5.1. Areas of the field thought to be susceptible to erosion, such as tramlines and areas assumed to be less susceptible because they were under crop and less compacted, were studied. The scorecard, tables and visual aids presented in the VSA field guide of Shepherd (2009) and used during this study to score indicators of soil quality, are summarised in Appendix E. The VSA indicators of soil quality are texture, structure, porosity, number and colour of soil mottles, colour, earthworms, smell, potential rooting depth (PRD), surface ponding, surface cover and surface crusting and erosion.



**Figure 5.1** Equipment used to carry out visual soil assessments.

Soil fragmentation and friability (the ability of a solid substance to be reduced to smaller pieces with little effort) were examined following a drop shatter test (Shepherd, 2009). In this test, a 200 mm cube of topsoil was dropped a maximum of 3

times from a height of 1m (for sandy loam soils the sample was dropped once from a height of 0.5 m) onto a firm board. The aggregates were then ordered from largest to smallest on a wooden board for comparison with sample photographs before scoring the soil structure in the following manner: 2 = good condition, 1 = moderate condition and 0 = poor condition (Appendix E - soil structure). For the porosity score, a spade slice of soil was removed from the hole and broken in half so that the fresh horizontal face could be compared with sample photographs (Appendix E - soil porosity). The other VSA indicators of soil quality were scored according to the tables and visual aids provided in Appendix E. On completion, each visual indicator of soil quality was multiplied by its designated weighting and the products were added to give the soil quality index (Appendix E - scorecard).

### 5.2.3.3 Visual Evaluation of Soil Structure

The locations identified for VSA were also assessed using VESS. A block of soil with a depth of 25 cm, a length of 20 cm and a breadth of 20 cm, was first extracted using a spade and placed on a tray for examination. The block was examined for horizontal layering (layers of differing structure) and, if present, each layer was scored separately. The block was then broken up by hand to reveal any cohesive layers or clumps of aggregates, and to separate the soil into natural aggregates and man-made clods. The major aggregates were then broken apart until a piece of aggregate, 1.5-2.0 cm in size, remained. These aggregates were then assessed with regard to their structural quality as outlined in Appendix F. Soil structure quality (Sq) was scored between Sq 1 (friable) and Sq 5 (very compact). As is recommended in the VESS method, scores were cross-checked regularly between two assessors. The scores were confirmed by checking for factors that increase the score, such as larger, more angular, less porous aggregates; and clustering, thickening and deflection in roots.

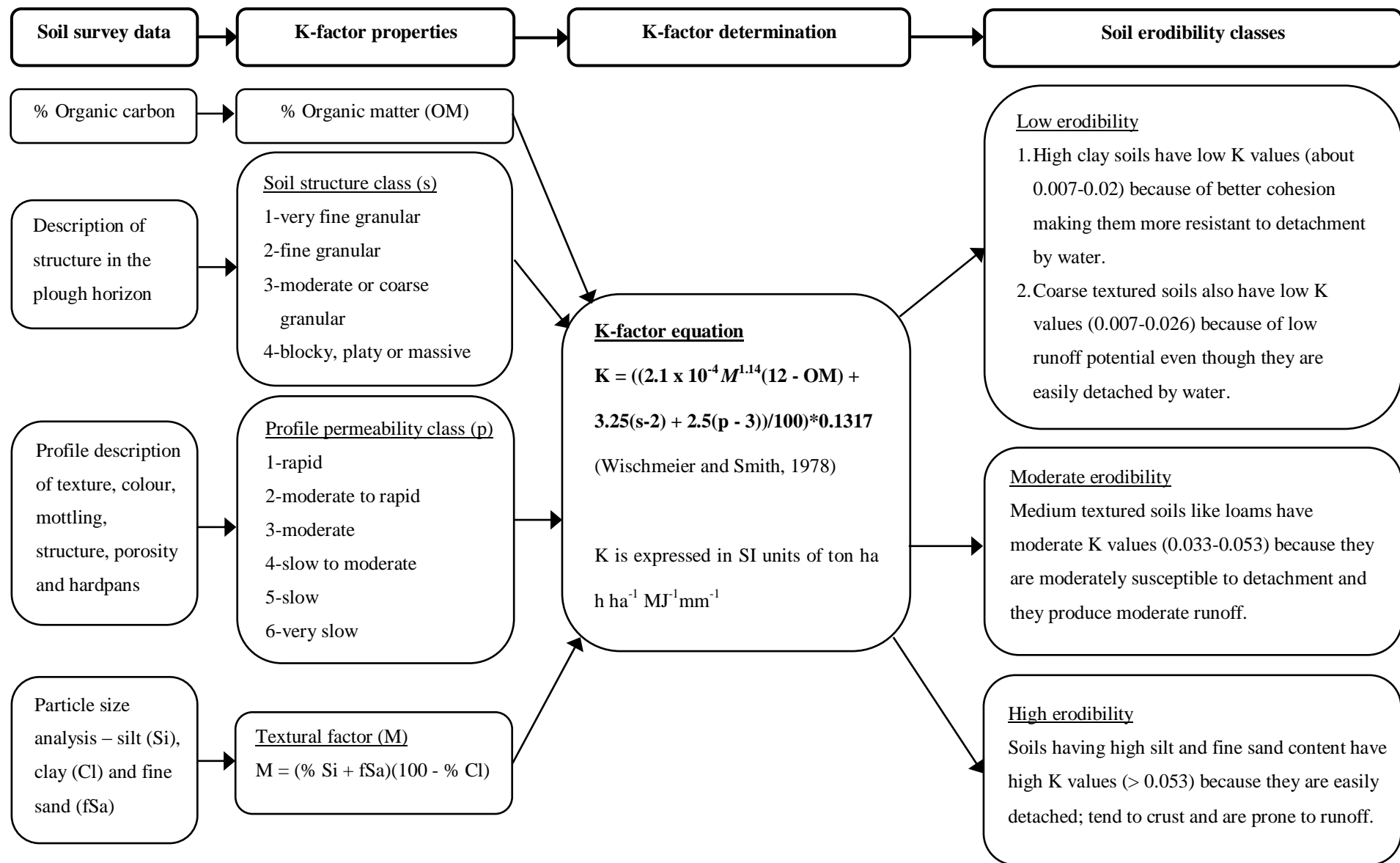
### 5.2.3.4 Soil Survey of Ireland assessment

The methods used in the Soil Survey of Ireland to formally classify soils according to the United States Soil Taxonomy (Soil Survey Staff, 1975 and 1993) classification systems and to evaluate the suitability of land for specific agricultural enterprises and

crops were applied to each of the 6 study sites. Soil augering was first carried out to ensure that the selected pit location was as representative as possible of the field as a whole. A pit was then excavated at each of the 6 field sites so that a fresh vertical soil face (no smearing) was achieved to the required depth (0.7 - 1.55 m). A detailed description of the general characteristics (drainage condition and pattern of horizon development) of the profile was carried out. Properties of individual soil horizons such as texture, structure, porosity, consistence, colour (assessed using a Munsell colour chart (Munsell, 2009)), mottling, amount of SOM, stoniness, presence of hardpans and root development, were described. Other less important criteria assessed were the formation of saturated zones, changes in soil moisture with depth, anaerobic zones and the pattern of roots. A bulk sample was taken from each soil horizon for physical and chemical analysis.

The soil properties measured in each soil horizon as part of the morphological description were: (1) pH (1:1 soil/solution ratio); (2) PSD by sieve and pipette analysis; (3) CEC; Bascomb, 1964; (4) SOC determined by sulfochromic oxidation (ISO, 1998; ISO 14235); and (5) Free iron (Fe) (hydro)oxides using the dithionite-citrate-bicarbonate method. Measurement of SOM was by loss on ignition at 550°C after Byrne (1979). Only after the analytical data became available were the soils formally classified by a soil scientist. The full classification for each site included determination of soil type, soil series (where available) and parent material.

The soil properties described and measured using the Soil Survey of Ireland method can also be used to estimate the erodibility of the study soils. Some of these properties influence the soils capacity to infiltrate rain, and therefore help determine amount and rate of runoff; some influence its capacity to resist detachment and transport by rainfall and overland flow, and thereby determine the soil content of the runoff (Wischmeier and Mannering, 1969). The methodology used to determine the study soils erodibility using the soil-erodibility nomograph of Wischmeier and Smith (1978):  $K (\text{ton ha h ha}^{-1} \text{ MJ}^{-1} \text{ mm}^{-1}) = ((2.1 \times 10^{-4} M^{1.14}(12 - \text{OM}) + 3.25(s-2) + 2.5(p - 3))/100) \times 0.1317$ , which ranks the soils erodibility based on the % OM, soil structure class (s), permeability class (p) and textural factor (M: percentage silt (Si) + fine sand fraction (fSa) content multiplied by 100 - clay fraction (Cl)) is outlined in Figure 5.2.



**Figure 5.2 Methodology for determining soil erodibility using Soil Survey of Ireland data.**

### 5.3 Experimental results from erosion risk and soil quality assessments

The results from the erosion risk and soil quality assessments conducted at the study sites are presented here. These data provided information on the likelihood of erosion and the loss of soil quality across the study sites.

#### 5.3.1 Field-Scale Defra Assessment

The application of the Defra (2005) erosion risk assessment to the 6 study sites (Table 5.5) identified three of the sites as having high or very high erosion risk based on soil texture (determined by hand as outlined in Appendix D) and slope (determined using an inclinometer). According to the assessment, rills are likely to form in most years and gullies may develop in very wet periods in fields classed as very high risk. In high risk areas, rills are likely to develop in most seasons during wet periods. The Bunclody and Duleek fields were less erodible due to the clay loam texture of their soils which makes them more resistant to detachment by rainfall and runoff than the sandy loam textured soils of the Tullow, Fermoy, Clonmel, and Letterkenny fields. The erosion risk of the Tullow field was classed as moderate due to a low slope of 2 degrees. The Duleek field was also classed as a moderate risk due to a moderate slope of 4 degrees and a medium soil texture of clay loam. According to Defra (2005), sediment may be seen running to roads, ditches or watercourses, and rills may develop in some seasons during very wet periods in fields classed as moderate risk. The Fermoy site was classed as having very high risk of soil erosion due to a combination of erosion-susceptible soil and steep slope. Only the Letterkenny site had a high land use risk (Table 5.5) because of the planting of potatoes in the rotation, which is highlighted in the Defra (2005) assessment as a highly erosion-susceptible land use. The planting of potatoes at the very high risk Letterkenny site should be avoided in future unless precautions are taken to control erosion.

Based on the erosion features observed in the fields during the study period (Table 5.5) and the anecdotal evidence provided by the farmers (Table 5.5), the Defra (2005) assessment was effective in identifying fields where more severe erosion is likely to occur, with the exception of the Fermoy field in which the assigned risk class was too

**Table 5.5 Results of the Defra (2005) erosion risk assessment**

Study site	Annual rainfall mm	Soil texture	Slope °	Slope Length m	Erosion history of sites	Observed erosion features	Land use rotation	Defra erosion risk	Defra land use risk
Buncloody	1038	Clay loam	2	120	Sheet erosion on longer slopes during prolonged heavy rainfall	None	OSR, WB, SB, SW	Lower	Moderate
Tullow	838	Sandy loam	2	200	Sheet erosion on longer slopes during prolonged heavy rainfall	None	OSR, WO, WW	Moderate	Moderate
Fermoy	1078	Sandy loam	9	100	Greater depth of soil in footslope areas	Greater depth of soil in footslope areas	WB, WW, SB	Very high	Moderate
Clonmel	1245	Sandy loam	3	200	Erosion on slopes > 4°; gully eroded in 2011	Rilling; gully; runoff in tramlines	WW, WO, OSR	High	Moderate
Letterkenny	1097	Sandy loam	5	75	Erosion during heavy rainfall in particular on tramlines; flooding	Rilling; runoff and erosion in tramlines; sand and silt deposits at field ending	WB, potatoes	High	High
Duleek	815	Clay loam	4	120	No past erosion	None	SB, beans, fallow	Moderate	Moderate

SB, Spring barley; SW, Spring wheat; WO, Winter oats; Winter barley; WW, Winter wheat; OSR, Oilseed rape



high when compared with the observed erosion levels. The Fermoy site was classed as very high risk and therefore rilling and even gullying should have been observed; however, no erosion features were observed during the study or reported by the farmer with the exception of a greater depth of soil measured in footslope areas compared to the crest of the hill. In contrast, the Letterkenny and Clonmel fields were classed as high risk and were the only fields in which rilling and sediment deposits were observed. Further assessment of the Fermoy field is required to determine what factors (that can be easily assessed by the farmer), outside of slope and texture can be used to help explain the observed low erosion levels. Once identified, these factors should be included in the assessment so that more accurate determination of erosion risk on fields such the Fermoy field is possible. It is recommended in the Defra (2005) assessment to upgrade or downgrade a field's erosion risk based on the following factors: soil structure; SOM; valley features which tend to concentrate runoff water; long, unbroken slopes and very steep slopes (i.e. greater than 11 degrees). However, no clear method is outlined for the farmer with which he/she can assess soil structure and upgrade or downgrade the risk class accordingly. These and other factors influencing soil erosion are investigated in this section using simple farmer friendly methods that are suitable for inclusion in an improved erosion risk assessment method for tillage soils in Ireland.

### **5.3.2 Visual Soil Assessment**

Results of VSAs conducted at the 6 study sites are presented in Table 5.6. The cropped areas of the Bunclody, Fermoy and Tullow sites were shown to be in good condition (denoted by VSA structure scores of 2; Table 5.6), dominated by friable, fine aggregates with no significant clodding (irregular blocks created by artificial disturbance; e.g., tillage or compaction - Figure 5.3). Tramline areas of these sites were shown to be in moderate-to-moderately good condition (denoted by VSA structure scores of between 1 and 1.75) and contained a mix of coarse clods and friable fine aggregates (Figure 5.3). Soil structure is vulnerable to change by compaction and erosion, and its preservation is key to sustaining soil function (Mueller et al., 2010). Tramline areas of the Clonmel and Letterkenny soils were in poor condition (denoted by VSA structure scores of 0), and were dominated by coarse

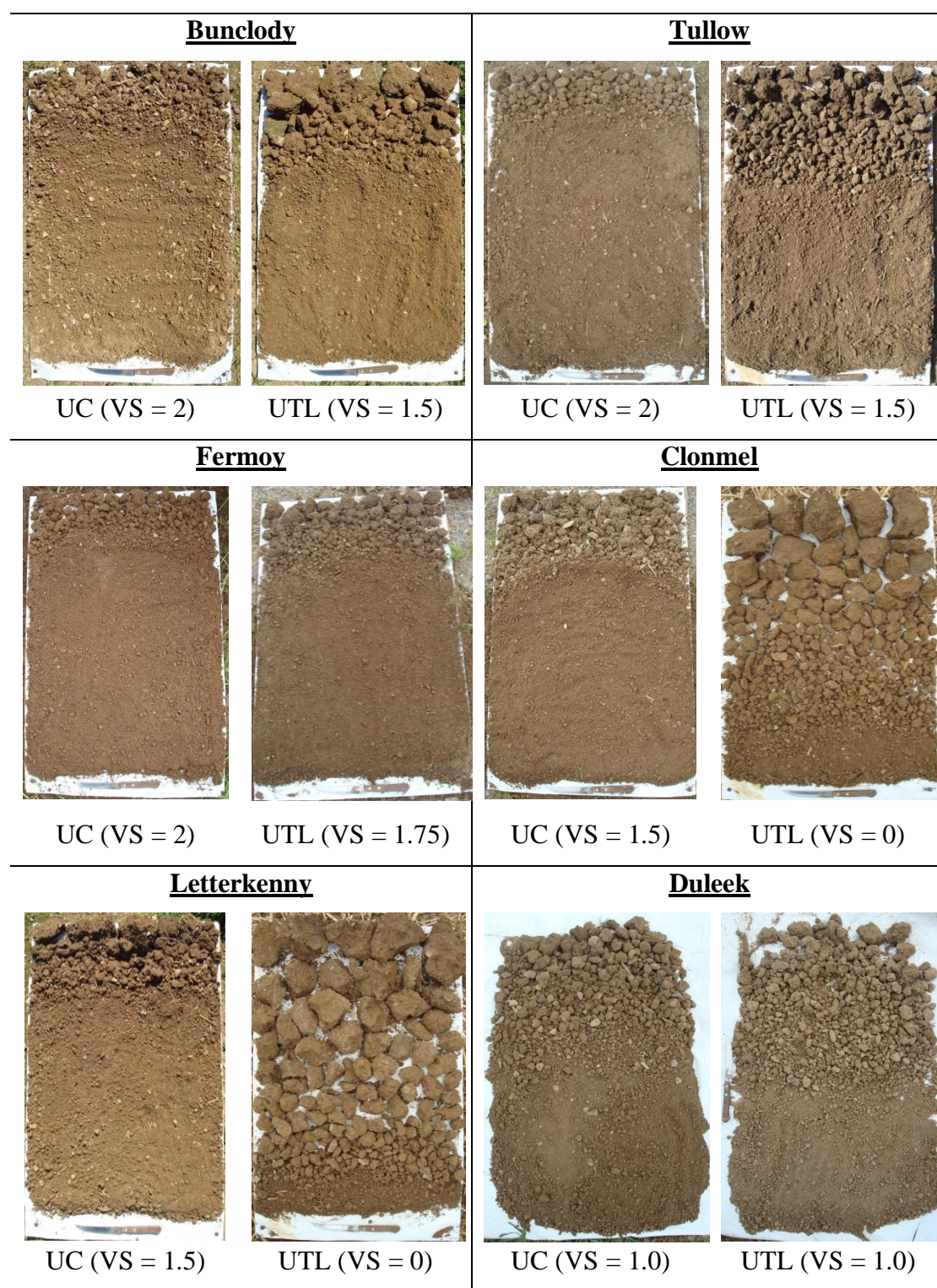
**Table 5.6 Results of Visual Soil Assessments**

Visual indicators of soil quality and respective weightings in brackets														
Study site	Land use	Texture (3)	Structure (3)	Porosity (3)	Mottling (2)	Colour (2)	Earthworms (3)	Smell (2)	Potential rooting depth (3)	Ponding (3)	Cover and crusting (2)	Erosion (1)	Visual score ranking <sup>1</sup>	Soil Quality Index <sup>2</sup>
Buncloody	OSR	1.5	2	1.5	2	2	0	1	2	2	2	2	43	Good
	Tramline	1.5	1.5	0.5	2	1.5	0	1	0	1	0	1	23.5	Moderate
Tullow	OSR	1	2	1.5	2	2	0	1	2	2	2	2	41.5	Good
	Tramline	1	1.5	0.5	2	2	0	1	0	1	0	1	23	Moderate
Fermoy	WB	1	2	2	2	1.5	0.5	1	2	2	2	2	43.5	Good
	Tramline	1	1.75	1	2	1.5	0	1	0	1	0	1	24	Moderate
Clonmel	WW	1	1.5	0.78	2	1	0.5	1	0.5	1	1	1	28	Moderate
	Tramline	1	0	0	1.5	1	0	1	0	0	0	0	10	Poor
Letterkenny	WB	1	1.5	1.5	1.5	1	0	1	0.5	0.5	1	1	25	Moderate
	Tramline	1	0	0	1.5	1	0	1	0	0	0	0	10	Poor
Duleek	SB	1.5	1	1.5	1.5	1	0.5	1	2	2	0.5	2	35.5	Moderate
	Tramline	1.5	1	1	1.5	1	0	1	0	0.5	0	1	20	Moderate

SB, Spring barley; WB, Winter barley; WW, Winter wheat; OSR, Oilseed rape

<sup>1</sup> The score for each visual indicator of soil quality was multiplied by its designated weighting and the products were added to give the visual score ranking, after Shepherd (2009).

<sup>2</sup> Good = > 37; Moderate = 20-37; Poor = < 20. See Appendix E for more details.



UC, Undercrop; UTL, Under tramline; VS, Visual scoring

**Figure 5.3 Visual scoring of soil structure during visual soil assessments at the 6 study sites.**

clods with very few finer aggregates. The coarse clods were very firm, angular or sub-angular in shape, and had very few or no pores (Figure 5.3). The presence of these larger aggregates reduces intra-aggregate porosity (Shepherd, 2009), thereby lowering soil permeability. Visual soil assessment structure scores identified poorer soil structure in cropped areas at the Letterkenny, Clonmel and Duleek sites when compared to the other sites.

The poorer structure at the Duleek site was most likely due to tillage operations being conducted when the soil was drier than the optimum water content. Tillage conducted when the soil is wetter or drier than this optimum, results in the production of a greater number of large clods that must be broken down in one or more subsequent tillage operations (Dexter and Birkas, 2004). The Duleek site had the lowest annual rainfall (Table 5.5) of all the sites and was subject to drought in the past. As a result, tillage operations may have to be conducted when the soil is drier than the optimum. In contrast, the Clonmel and Letterkenny sites received high annual rainfall and were subject to periodic flooding, which may have resulted in tillage operations being conducted when the soil was wetter than the optimum. The lower degree of compaction in the tramlines of the Duleek site, when compared with the cropped area of the Duleek site and tramline areas in other sites, was likely due to the drier condition of the soil during the course of the year. Vehicle traffic operations conducted when soil moisture is high result in greater soil compaction. Ideally, soils should only be trafficked upon when soil moisture conditions are < 60% of field capacity (Raper, 2005). However, this is generally not possible under Irish weather conditions and tramline compaction on sandy soils such as Clonmel and Letterkenny often results. The susceptibility of these soils to compaction is further highlighted by the moderately poor PRD scores obtained in the cropped area of each soil (Table 5.6). The apparent resistance of the Bunclody and Tullow soils to compaction in tramline areas could be due to their better soil structure in cropped areas. Good soil structure reduces the susceptibility to compaction under wheel traffic (Shepherd, 2009).

The soil quality index determined using visual scores of indicators of soil quality is in close agreement with VSA structure scores (Table 5.6). This is to be expected given the strong influence soil structure has on soil quality indicators like porosity, mottling,

ponding, PRD and erosion. Soil structure is known to be a crucial criterion of agricultural soil quality. The soil quality at the Bunclody, Tullow, and Fermoy sites was good in cropped areas and moderate in tramline areas where some compaction had occurred. Soil quality was only moderate in the cropped areas and poor in the tramline areas of the Clonmel and Letterkenny sites, which may partly explain the observed higher erosion levels at these sites.

### **5.3.3 Visual Evaluation of Soil Structure**

Visual evaluation of soil structure scores for the study soils ranged from 1 - 2 for the respective land use being practiced on each soil and from 3 - 5 in tramline areas and are presented in Table 5.7. These scores reflect the surface compaction that has taken place in tramline areas as a result of vehicular traffic during spraying and fertilising operations. This affects the structural quality of the soil by changing it from friable or intact under crop to firm, compact or very compact in tramlines areas. The reduction in soil porosity associated with this compaction of the surface soil can significantly reduce its permeability in these areas, thereby increasing the risk of surface runoff being generated during wet periods. This risk was particularly high in tramline areas of the Letterkenny and Clonmel fields as these soils had the poorest structural quality, with both being classed as very compact. Observed surface runoff from tramline areas during wet weather at the Clonmel and Letterkenny sites allied with the absence of runoff from cropped areas shows that soil structure quality, as measured by VESS, can be indicative of areas where runoff risk is high. Unless traffic is eliminated, good timing of operations is the most effective way to preserve soil structural quality (Ball et al., 1997). Surface compaction can be alleviated in the next cultivation cycle.

**Table 5.7 Results of Visual Evaluation of Soil Structure**

Study site	Land use	Structural Quality Score	Structural Quality
Buncloody	OSR	1	Friable
	Tramline	3	Firm
Tullow	OSR	1	Friable
	Tramline	3	Firm
Fermoy	WB	1	Friable
	Tramline	3	Firm
Clonmel	WW	2	Intact
	Tramline	5	Very Compact
Letterkenny	WB	2	Intact
	Tramline	5	Very Compact
Duleek	SB	2	Intact
	Tramline	3	Firm

SB, Spring barley; WB, Winter barley; WW, Winter wheat; OSR, Oilseed rape; Structural quality as determined by VESS (see Appendix F); Scores of 1 - 3 are usually acceptable, whereas scores of 4 or 5 require a change of management.

### 5.3.4 Soil Survey of Ireland assessment

Detailed results of the soil profile examinations carried out to classify the study soils according to the Soil Survey of Ireland method can be found in Appendix G. Profile descriptions carried out to the full depth of each study soil are provided in Appendix G (Table G.1 and Figure G.1). Soils containing high proportions of silt and very fine sand, identified by the presence of small pockets or thin layers of pale sand grains within the cultivated layer or concentrated at its base (Batey, 2000), are most erodible (Smolen et al., 1988). Thin layers of pale sand grains were identified in the cultivated layer of the Letterkenny soil (Appendix G, Figure G.2). These are the result of the disintegration of aggregates into their component particles (Batey, 2000) which, due to their lack of cohesion and low settling velocity, are easily eroded. The observation in the Letterkenny field of deposits of fine sand particles at the bottom of sloping tramlines (Appendix G, Figure G.3) was further evidence of the disintegration of aggregates and the transport of their component particles by concentrated water flow in compacted tramlines.

The PSD results for the Letterkenny soil (Appendix G - Table G.2) confirmed the findings of the field analysis. Having 60% fine sand within its sand fraction, the Letterkenny topsoil was the only topsoil to be classed as a fine sandy loam, with the other study soils falling in the medium and coarse sandy loam and loam classes. Medium and coarse sandy loams are normally less susceptible to erosion than fine sandy loams, as they are very fast draining and contain larger particles with higher settling velocities. The soils tested in this study, which had topsoil textures ranging from loam to sandy loam and clay contents between 10 and 21% (Table 5.8), fell within the range of soils susceptible to structural decline and erosion.

In general, permeability decreases as subsoil colour changes from red to brown to yellow to dark (black to very dark brown) to grey (light grey to bluish and greenish greys) (Moore, 1998). Based on the Munsell colour description of each horizon for the soils tested, the Letterkenny soil had by far the worst drainage class of all the soils, going from moderately well-drained (yellow-brown) in the topsoil to near permanent waterlogging (grey/olive) in the lower horizons (Appendix G, Table G.1). This was indicative of a substantial risk of overland flow due to saturation excess.

The lack of mottles in the horizons of the Bunclody, Tullow, Fermoy and Duleek soils indicated that they had good drainage and were unlikely to generate overland flow when managed properly. The presence of few-to-many mottles in the lower horizons of the Clonmel soil indicated that the soil was moderately well-drained and may be susceptible to infiltration excess overland flow when subjected to high intensity rainfall. The presence of many medium, distinct olive-to-dark olive 5Y 5/8 mottles in the B21g horizon of the Letterkenny soil (Appendix G, Figure G.4) may be indicative of prolonged waterlogging. Care should be taken when using mottle patterns to assess the wetness class of a soil profile, as they can persist long after the condition responsible for their formation is gone (Batey, 2000).

Stable surface soil structure is important for facilitating rapid water infiltration, controlling soil erosion, and reducing water runoff of soil contaminants to nearby surface waters (Franzluebbers, 2002). The surface horizon of each soil was friable (except the Clonmel soil, which was firm) in consistence and had a structural mix of

crumb in the upper few cm grading to fine and medium sub-angular blocky. The plough horizon of the Clonmel soil had the poorest structure of the 6 plough horizons. The lower horizons of the Clonmel and Letterkenny soils had structures normally associated with poor drainage and waterlogging. Hard pans such as those identified in the Clonmel and Letterkenny soils impede the movement of water through the soil profile, increasing the susceptibility to waterlogging and erosion by rilling and sheet wash (Shepherd, 2009). Based solely on soil structure, the Letterkenny and Clonmel soils had a higher associated erosion risk due the greater risk of runoff being generated on these soils.

The SOM level in the plough horizon of the study soils was above 3.4% (soils with SOM levels above this threshold are considered not to be vulnerable or depleted (DAFF, 2009a)) with the exception of the Letterkenny soil, where it was only 3.23% (ca. 1.9% SOC) (Appendix G, Table G.2). These low values of SOC are not unusual for sandy tillage soils. Based solely on the SOC levels measured in the plough horizon of the study soils, the Letterkenny, Tullow and Clonmel soils posed the greatest erosion risk.

Both the Clonmel and Letterkenny soils were classified as Gleys by a soil scientist (Appendix G - Table G.3) based on the results of the profile examinations. Gleys are soils in which the effects of drainage impedance dominate and which have developed under conditions of permanent or intermittent water-logging (Finch et al., 1983). In general, Gley soils have weak structure and poor drainage which makes them difficult to cultivate. The potential for surface runoff due to saturation excess is higher in these soils than in the better drained Brown Earth (Bunclody, Fermoy and Duleek) and Grey Brown Podzolic (Fermoy) soil types.

In Table 5.8, the soil properties determined analytically or estimated by visual and tactile assessment as part of the soil classification procedure are used to produce K-factor values for the cropped and tramline areas of the study sites. The K-factor values for the Letterkenny soil identified it as having the highest erodibility of all the soils, which was in agreement with the observed erosion across the study sites. The Fermoy soil, when cropped, had the lowest erodibility as estimated by the K-factor. The low



**Table 5.8 K-factor erodibility values and associated risk.**

Study site	FSa + silt %	Clay %	SOM %	Land use	Permeability	Structure	K-factor ton ha h ha <sup>-1</sup> MJ <sup>-1</sup> mm <sup>-1</sup>	Erodibility Risk
Buncloody	57	21	4.81	OSR	Mod. to rapid (2)	Fine crumb (2)	0.0258	Low to moderate
				Tramline	Slow (5)	Med/coarse crumb (3)	0.0399	Moderate
Tullow	53	14	3.6	OSR	Mod. to rapid (2)	Fine crumb (2)	0.0311	Low to moderate
				Tramline	Slow to mod. (4)	Med/coarse crumb (3)	0.042	Moderate
Fermoy	56	14	4.08	WB	Rapid (1)	Very fine crumb (1)	0.0237	Low
				Tramline	Slow to mod. (4)	Med/coarse crumb (3)	0.0422	Moderate
Clonmel	54	16	3.57	WW	Slow (5)	Med/coarse crumb (3)	0.0452	Moderate
				Tramline	Very slow (6)	Massive (4)	0.0528	Moderate to high
Letterkenny	69	10	3.23	WB	Slow (5)	Med/coarse crumb (3)	0.0620	High
				Tramline	Very slow (6)	Massive (4)	0.0696	High
Duleek	59	20	4.08	SB	Mod. to rapid (2)	Med/coarse crumb (3)	0.0348	Moderate
				Tramline	Slow to mod. (4)	Med/coarse crumb (3)	0.0414	Moderate

FSa, fine sand; SB, Spring barley; WB, Winter barley; WW, Winter wheat; OSR, Oilseed rape; SOM, Soil organic matter;

Low risk = low runoff potential (K-factor is 0.007-0.02 (high clay), 0.007-0.026 (coarse texture)); Moderate risk = moderate potential to produce runoff and moderately susceptible to detachment (K-factor is 0.033-0.053 (medium texture)); High risk = easily detached, tend to crust and prone to runoff (K-factor is >0.053 (silt/fine sand)).

K-factor in this case was largely due to the rapid permeability of the Fermoy soil profile and the fine crumb structure of its plough layer, both of which lower its potential to produce runoff.

#### **5.4 An erosion and soil quality screening toolkit for Irish tillage soils**

Integration of erosion risk indicators (Defra, 2005) and soil quality indicators with observed erosion levels in the study soils (Table 5.9) allowed the selection of a strategic set of indicators for inclusion in a screening toolkit designed to assess fields for likelihood of erosion and loss of soil quality (Table 5.10). It is proposed that the specific data required to support indicators deemed appropriate for inclusion in the screening toolkit be collected by the farmer and an erosion risk class assigned (Figure 5.4 and Figure 5.5). Sites identified as high and very high risk can then be further assessed by specialist advisors/consultants. The justification for inclusion of each indicator in the screening toolkit is now provided.

##### **5.4.1 Development of the screening toolkit**

The Defra (2005) erosion risk assessment utilises field slope and soil texture primarily to determine a field's erosion risk class. Certain visual indicators of soil quality such as structure, ponding, and porosity, are also good indicators of erosion risk and were therefore used in conjunction with SOM and AAR values to upgrade or downgrade the erosion risk class generated using the Defra assessment (Figure 5.4 and Figure 5.5). Visual indicators receiving scores of 2 were in good condition and indicated that the risk of soil erosion occurring was lower and hence the erosion risk class was downgraded. In contrast, visual indicators receiving scores of less than 1 were in poor condition and therefore required upgrading to a higher risk class. Soil structure is the critical parameter here and its appraisal was supported by assessment of porosity, ponding, SOM and AAR. Soil structure was weighted twice as high as the other parameters used in the methodology for upgrading/downgrading the Defra risk class (Figure 5.5).

**Table 5.9** Integration of Defra (2005) erosion risk rankings with selected soil quality indicators

Study site	Defra (2005) erosion risk	<sup>1</sup> AAR (mm)	<sup>2</sup> SOM (%)	Land use Structure	VSA Porosity	VESS	VSA Ponding	VSA Erosion	K-factor Erodibility Risk	Observed erosion features	
Buncloody	Lower	1038	4.81	OSR	2	1	1.5	2	2	Low to moderate	None
				Tramline	1.5	3	0.5	1	1	Moderate	None
Tullow	Moderate	838	3.6	OSR	2	1	1.5	2	2	Low to moderate	None
				Tramline	1.5	3	0.5	1	1	Moderate	None
Fermoy	Very high	1078	4.08	WB	2	1	2	2	2	Low	Greater depth of soil in footslope areas
				Tramline	1.75	3	1	1	1	Moderate	Greater depth of soil in footslope areas
Clonmel	High	1245	3.57	WW	1.5	2	0.78	1	1	Moderate	Rills (shallow)
				Tramline	0	5	0	0	0	Moderate to high	Rills (moderate), small gully, runoff
Letterkenny	High	1097	3.23	WB	1.5	2	1.5	0.5	1	High	Rills (deep), fine sand and silt deposits
				Tramline	0	5	0	0	0	High	Rills (deep), runoff carrying sediment,
Duleek	Moderate	815	4.08	SB	1	2	1.5	2	2	Moderate	None
				Tramline	1	3	1	0.5	1	Moderate	None

SB, Spring barley; WB, Winter barley; WW, Winter wheat; OSR, Oilseed rape

<sup>1</sup>AAR, average annual rainfall; <sup>2</sup>SOM, soil organic matter

Soil structure determines the porosity, strength and stability of a soil. Improved soil structure enhances nutrient recycling, water availability and biodiversity while reducing water and wind erosion, and improving surface and ground water quality (Bronick and Lal, 2005). Effective erosion control therefore requires the maintenance of good soil structure. The Defra (2005) erosion risk assessment recommends that farmers assess their soil's structure when assigning an erosion risk class but does not provide a method with which to conduct the assessment. The methods of soil structure assessment used in VESS and VSA are proven methods with which the farmer can score a soil's structure. In the present study, poor soil structure scores of 0 (VSA structure score) and 5 (VESS score = very compact) in the tramlines of the Letterkenny and Clonmel fields coincided with the observation of erosion features such as rills and surface runoff carrying eroded soil (Table 5.9). Deposits of fine sand and silt observed at tramline endings in the Letterkenny field are further evidence of the effect of poor soil structure on erosion levels. Study field areas receiving good soil structure scores of 2 (VSA structure score) and 1 (VESS score = friable) were free of erosion features (Table 5.9) and were not seen to generate surface runoff at times when surface runoff was recorded in adjacent areas, which had received worse structure scores. These findings support the inclusion of an assessment of soil structure, either by VESS or by VSA, in a screening toolkit (an improved version of the Defra (2005) erosion risk assessment).

In general, VESS scores showed good agreement with VSA structure scores. However, it is recommended that the farmer assess soil structure using the drop shatter test from the VSA method rather than breaking the soil by hand as is done in the VESS method, because the former is less subjective and more effective at demonstrating to the farmer the effect management decisions have on soil structure. Guimarães et al. (2011) compared the two methods and concluded that breaking the soil by hand, if done as recommended, gives a result comparable to the method of break-up by dropping. However the authors also noted that the user might need some training in breaking up the aggregates along their natural failure planes (boundaries). They found that this was generally not an issue after dropping the soil (as is required in the VSA method) because of well-defined crack lines formed on impact.

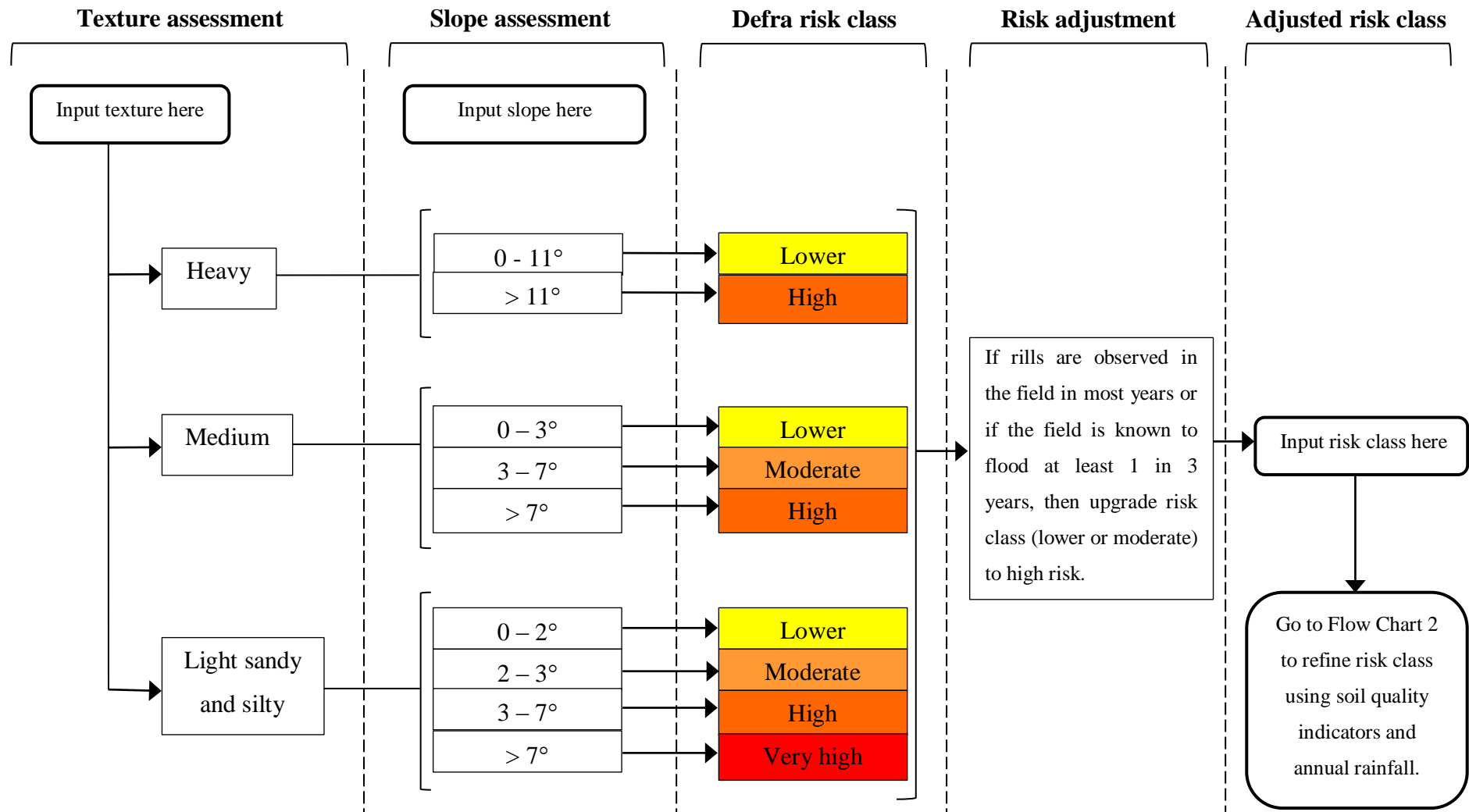
**Table 5.10 Proposed toolkit indicators for use in screening sites for erosion risk and reduced soil quality.**

Screening indicator	Proposed assessment method	Interpretation of results
Soil texture	Defra (2005) hand texturing method	Heavy soils = lower erodibility; medium soils = moderately erodible; sandy and light silty soils = highly erodible.
Slope angle	Measure slope as accurately as possible using a clinometer; record position in landscape	< 2° = level ground; 2 – 3° = gentle slope; 3 – 7° = moderate slope; > 7° = steep slope; > 11° = very steep slope.
Erosion features	VSA soil erosion (Shepherd, 2009)	Observation of rilling or gullying is used to upgrade sites classed as lower and moderate risk to high risk,
Soil structure	VSA drop shatter test (Shepherd, 2009)	2 (soil dominated by friable, fine aggregates with no significant clodding) = good; 1 (soil contains significant proportion (50%) of both coarse clods and friable fine aggregates) = moderate; 0 (soil dominated by coarse clods with very few finer aggregate) = poor.
Ponding	VSA ponding (Shepherd, 2009)	2 (no surface ponding of water evident after 1 day following heavy rainfall on soils that were at, or near, saturation) = good; 1 (moderate surface ponding occurs for 2 days after heavy rainfall on soils that were at, or near, saturation) = moderate; 0 (significant surface ponding occurs for 4 days or more after heavy rainfall on soils that were at, or near, saturation) = poor.
Porosity	VSA porosity (Shepherd, 2009)	2 (many macropores and coarse micropores between and within aggregates associated with good soil structure) = good; 1 (fewer macropores and coarse micropores between and within aggregates and moderate consolidation evident) = moderate; 0 (no soil macropores and coarse micropores are visually apparent within compact massive structureless clods) = poor.
% SOM <sup>1</sup>	% loss on ignition	< 3.4% (ca. 2% SOC) = soil may be vulnerable or depleted; > 3.4% soil considered not to be vulnerable or depleted.
AAR <sup>2</sup>	AAR amount (low, moderate, high)	< 850 mm = low rainfall risk ; 850 – 1200 mm = moderate rainfall risk; > 1200 mm = high rainfall risk
Land use	Defra (2005) land use risk categories	Avoid erosion susceptible land uses such as late sown winter cereals, potatoes, and field vegetables, on very high or high risk sites unless precautions are taken to limit erosion. Land uses such as early sown winter cereals, OSR <sup>3</sup> and spring sown cereals, can be carried out with care on these sites.

<sup>1</sup>SOM, soil organic matter; <sup>2</sup>AAR, average annual rainfall; <sup>3</sup>OSR, oilseed rape

Given that the permeability of soil during infiltration is mainly controlled by coarse pores (Beven and Germann, 1982), which can be seen by the naked eye, the assessment of porosity using the VSA approach can give an indication of the study soil's ability to transmit water away from the soil surface. Soils with high porosity are therefore less likely to generate overland flow and associated erosion than are soils with low-to-moderate porosity. Furthermore, subsoils with low-to-moderate porosity are likely to have problems with waterlogging (Moore, 1998). Study soils receiving VSA porosity scores of  $< 1$  (indicative of low porosity) were also the soils in which overland flow and rilling were observed (Table 5.9). For this reason, porosity was deemed suitable for inclusion in the screening toolkit.

Soil organic matter is an important indicator of soil quality and productivity and is important in influencing soil erodibility (Jankauskas et al., 2007). Furthermore, a reduction in SOM levels has been highlighted in numerous legislative reports and scientific papers as contributing to a decline in soil quality, and can result in increased soil erosion. Increasing SOM content improves the cohesiveness of the soil, reduces the risk of surface crusting, lowers the risk of soil compaction, increases its water holding capacity and promotes soil aggregate formation, thereby improving structural stability and reducing erosion. The primary role of SOM in reducing soil erodibility is by stabilising the surface aggregates (Stott et al., 1999). Poor stability in surface aggregates can predispose the soil to: disaggregation under raindrop impact and a subsequent development of a surface crust, reduction of infiltration rate, and surface runoff (Quinton and Catt, 2004). Guerra (1994) found that soils on eroded sites in England had OM contents of 2.1 - 3.8%. In the present study, OM contents ranged from 3.23 to 4.81% and as such were mostly higher than the range of OM contents measured at eroded sites by Guerra (1994). The lowest levels of SOM recorded across the study sites were measured at the Letterkenny (3.23%) and Clonmel (3.57%) sites, and may in part account for the higher levels of erosion observed at these sites (Table 5.9) when compared to the other study sites. Soil organic matter was deemed to be an important parameter when screening fields for likelihood of erosion because: (1) previous studies by Kay and Angers (1999) and Whitmore et al. (2004) have demonstrated its importance in maintaining soil structural stability and (2) it has been shown to be one of the crucial factors in determining a soil's inherent erodibility. The



**Figure 5.4 Methodology for determining erosion risk (adapted from Defra (2005)).**

Flow Chart 2

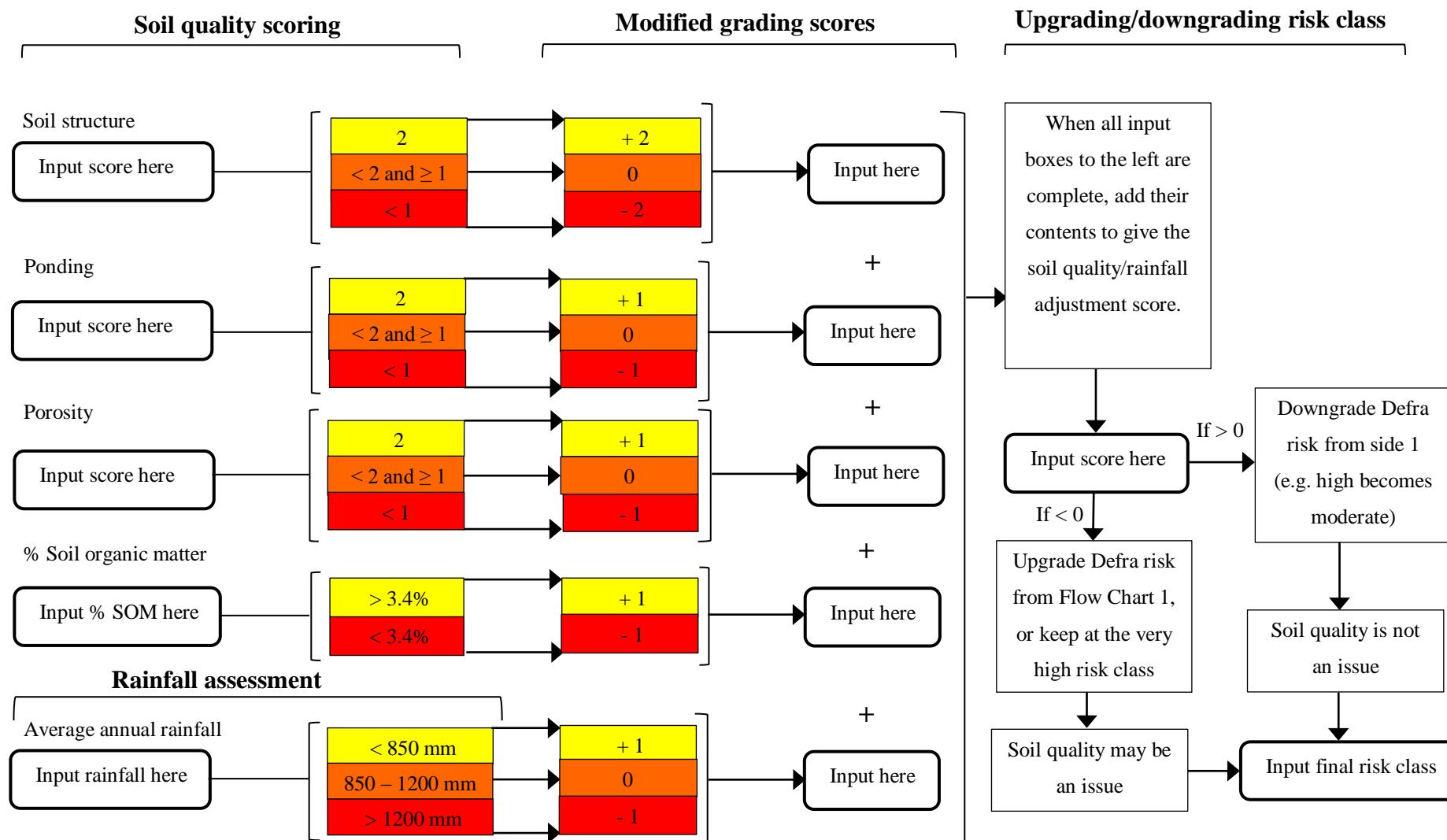


Figure 5.5 Methodology for upgrading/downgrading Defra risk class using soil quality indicators and average annual rainfall.



inclusion of SOM in the screening toolkit will not increase the burden on Irish tillage farmers as they are already required to monitor SOM levels on long-term (6 years or more) continuous tillage land to ensure that levels stay above a threshold of 3.4%. Soils with SOM levels above 3.4% (ca. 2% SOC) are considered not to be vulnerable or depleted (DAFF, 2009a). In Ireland, where OM levels are found to be below the threshold value of 3.4%, the farmer is obliged to seek advice from a specialist advisor and, where appropriate, follow the programme of remedial actions. The programme of action required depends on soil type and on-going practices.

Land use is included in the screening toolkit so that high and very high erosion risk crops in each field's cropping regime can be identified by the farmer and the necessary precautions taken to ensure that soil erosion is kept to a minimum. The precautions taken in each case will depend on the land use risk class of the planted crop and the erosion risk class of the field under assessment. The farmer should consult with his/her specialist advisor/consultant to determine the best course of action. If the precautions taken are ineffective and erosion persists, then the land use should be ceased. The AAR in areas where tillage land is mainly concentrated (i.e., the midlands, south and east of Ireland) varies from ca. 657 mm to ca. 1400 mm. As such, the risk of erosion occurring in a farmer's field is strongly influenced by its location. Previously, Unwin (2001) observed that erosion problems in England were worse in areas where the AAR exceeds a threshold of 800 mm. This resulted in the country being divided into areas above and below a mean annual rainfall threshold of 800 mm when assessing the risk of erosion occurring. Boardman et al. (2009) pointed out that such an approach ignored the problem of rainfall variability (both spatially and temporally) and failed to consider 'at-risk' periods where high daily/monthly totals, for example, may have coincided with the most vulnerable periods for erosion (sowing and post-harvesting); areas that normally fall below 800 mm could be at serious risk in wetter years. While the problems identified by Boardman et al. (2009) are valid, the use of an AAR threshold in erosion risk assessment is still appealing, as it accounts for the role of rainfall amount (if not intensity and duration) in soil erosion. As such, a similar AAR threshold as that used by Unwin (2001) is proposed for use in Ireland to upgrade or downgrade a field's erosion risk class. Average annual

rainfall thresholds are included among the proposed toolkit indicators (Table 5.10) and can be obtained by the farmer for his location, from his/her specialist advisor.

#### 5.4.2 Toolkit testing

In Table 5.11, the methodology developed in Figure 5.4 and Figure 5.5 is applied to the study soils examined in the present study. There was better agreement between erosion risk classes assigned using the screening toolkit developed in this study and the observed erosion in study fields than there was between Defra (2005) erosion risk classes and the observed erosion in study fields. The Fermoy soil of the present study received scores of 2 for each of structure, porosity and ponding (Table 5.6), which are assigned modified positive gradings of 2, 1, and 1, respectively, in Table 5.11. These gradings are indicative of the good drainage class and lower runoff potential of the Fermoy soil. Similarly, SOM received a positive grading of 1 because it is above the threshold of 3.4%, while AAR received a neutral grading of 0. The total grading score for the Fermoy soil of + 5 is evidence of good soil quality and can be used to downgrade the Defra erosion risk class (determined by the farmer using Figure 5.4) of the Fermoy field from very high to high, which is more in line with observed erosion in the field. In the case of fields still flagged as high risk after downgrading, the farmer should seek advice from his/her specialist advisor/consultant. Detailed quantitative assessment of the Fermoy soil using the Soil Survey of Ireland method and the K-factor equation (Table 5.8) confirmed that its low erodibility was largely due to the rapid permeability of the soil profile and the fine crumb structure of its plough layer, both of which lower its potential to produce runoff.

The same grading system was used to downgrade the risk class of the Tullow and Duleek fields from moderate to lower risk, and to upgrade the Letterkenny and Clonmel fields from high risk to very high risk, thereby ensuring the prioritisation of areas where detailed investigation is required. According to the Defra (2005) assessment, sediment is rarely seen to move in fields classed as lower risk, but polluting runoff may enter ditches or watercourses. Good soil structure (+2), an absence of surface water ponding (+1), moderate porosity (0) and healthy SOM (+1) levels in both the Tullow and Duleek fields (Table 5.11) and a low AAR (+1) in the

former, provides the evidence to support the downgrading of the Defra (2005) risk class from moderate to lower in both fields. No erosion was observed in the Tullow or Duleek fields; therefore, their downgrading from a Defra (2005) erosion risk class of moderate to one of lower, using Chart 2 of the screening toolkit (Figure 5.5), is justified. In the case of the Letterkenny field, moderate soil structure (0), the presence of surface water ponding (-1), moderate porosity (0), and lower than desirable SOM levels (-1) (Table 5.11) provides the evidence required to support the upgrading of the Defra (2005) risk class from high to very high. Similarly, the moderate soil structure (0), low porosity (-1), and high AAR (-1) recorded at the Clonmel field is evidence enough to support the upgrading of the Defra (2005) risk class from high to very high. The upgrading of the Letterkenny and Clonmel fields to very high risk is justified by the observation of rills in most years in both. The K-factor erodibility values (Table 5.8) for the cropped areas of the Clonmel and Letterkenny fields were higher (indicating greater risk) than in other study fields due to the poorer structure of their plough horizons and slow permeability of their soil profiles. The newly developed screening toolkit for refining Defra erosion risk classes proved effective (after a preliminary validation) and has the added advantage of bringing to the farmer's attention the soil quality status. The next step would be to make the screening toolkit part of a larger scale erosion study in which measured edge-of-field sediment losses can be used to further validate the erosion risk rankings assigned by farmers using the screening toolkit. Comparisons of the amounts of SS initially mobilised by rainfall and overland flow with the amounts measured in flow at appropriate monitoring points (for example in drain flow, at the edge of fields or at catchment outlets) provides a means of assessing the delivery factor in sediment transport (Beven et al., 2005). The larger-scale erosion study should include a workshop where farmers and specialist advisors can test the toolkit and make suggestions on how it can be further refined based on their experience with soil erosion and soil quality.

**Table 5.11 Application of the methodology for upgrading/downgrading Defra risk classes to the study soils.**

Risk indicators	Modified gradings					
	Fermoy	Tullow	Buncloody	Duleek	Clonmel	Letterkenny
Structure (-2,0,+2)	+ 2	+ 2	+ 2	0	0	0
Ponding (-1,0,+1)	+ 1	+ 1	+ 1	+ 1	0	- 1
Porosity (-1,0,+1)	+ 1	0	0	0	- 1	0
% SOM (-1,0,+1)	+ 1	+ 1	+ 1	+ 1	+ 1	- 1
AAR (-1,0,+1)	0	+ 1	0	+ 1	- 1	0
<b>Total grading score</b>	<b>+ 5 (downgrade)</b>	<b>+ 5 (downgrade)</b>	<b>+ 4 (downgrade)</b>	<b>+ 4 (downgrade)</b>	<b>- 1 (upgrade)</b>	<b>- 2 (upgrade)</b>
Defra (2005) risk class (from Table 5.5)	Very high	Moderate	Lower	Moderate	High	High
	↓	↓	↓	↓	↓	↓
<b>New refined erosion risk class</b>	<b>High</b>	<b>Lower</b>	<b>Lower</b>	<b>Lower</b>	<b>Very high</b>	<b>Very high</b>

SOM, soil organic matter; AAR, average annual rainfall

## 5.5 Summary

This study developed a screening toolkit of erosion risk and soil quality indicators for use by farmers and specialist advisors in assessing sites for likelihood of erosion and reduced soil quality. The toolkit was developed by comparing results from quick and easily conductible onsite assessments and more detailed quantitative assessments with observed erosion features. Indicators proposed for inclusion in the toolkit included: texture, slope, erosion features, structure, ponding, porosity, % SOM, AAR and current land use. Assessment of these indicators at each study site using the grading system developed in this chapter made it possible to correctly identify the sites where the erosion risk was high enough to justify further investigation by a specialist advisor/consultant. The grading system also informs the farmer of the impact of soil quality on the assigned erosion risk class.

The screening method developed in this study can, if adopted by Irish tillage farmers, ensure that the highest priority sites are identified expeditiously and with little cost incurred by the farmer. Resources can then be focused on these sites ensuring the cost effectiveness of mitigation measures employed to reduce SS and associated P loss, such as buffer zones and minimum tillage. The screening method will have the added benefit of bringing to the farmer's attention any issues with soil quality on his holding.

## **Chapter 6 Conclusions and recommendations**

### **6.1 Overview**

The aims of this study were to develop methods capable of identifying tillage fields where STP and erosion risk levels are sufficiently high to cause concern over potential losses of DRP, PP and SS in surface runoff following heavy rainfall, and to develop a methodology for farmers/specialist advisors to identify tillage fields which may be susceptible to erosion.

### **6.2 Integration of RDPRI findings with results from the screening toolkit**

Integration of the results from the RDPRI and the screening toolkit, developed in this study, is necessary in order to estimate the risk of sediment and phosphorus mobilisation from Irish tillage soils, and subsequent delivery to surface water bodies. The RDPRI and screening toolkit developed in this study are concerned primarily with estimation of the risk of phosphorus and sediment being mobilised from Irish tillage soils by rainfall and overland flow. The RDPRI goes somewhat further by determining the soil test phosphorus threshold above which dissolved reactive phosphorus mobilised from the soils, by simulated rainfall and overland flow, has the potential to cause eutrophication of a surface water body. Therefore, tillage fields identified by routine soil testing as having soil test phosphorus levels in excess of this threshold can, based on the findings of the RDPRI, be termed critical source areas of dissolved reactive phosphorus if the fields in question are hydrologically connected to a surface water body. Sharpley and Tunney (2000) observed that threshold soil P values have little meaning unless they are used in conjunction with an estimate of a site's potential for surface runoff and erosion. The screening toolkit developed in this study provides such an estimate by combining site factors (slope angle, land use and annual average rainfall) and soil properties (texture, porosity, ponding, % soil organic matter, and structure) that are known to affect runoff and erosion rates. As such, integration of the RDPRI and the screening toolkit provides a more complete picture of the risk of phosphorus and sediment being transported from tillage fields.

The screening toolkit informs the farmer of the risk of erosion occurring on a particular field. It does not inform him/her about the risk of an eroding field impacting on a surface water body. The same is true of the Defra (2005) erosion risk assessment. To address this, Boardman et al. (2009) proposed a simple extension of the Defra scheme to account for the issue of connectivity between an eroding field and a river. They proposed that the Defra scheme be adapted by taking account of high or very high risk fields with a river within 200 m downslope. This resulted in certain fields no longer being classed as at risk and others being confirmed as at risk of delivering sediment to the river (i.e. high connectivity). Applying the same connectivity extension to the study fields that were identified as high and very high risk using the screening toolkit, results in the Fermoy and Letterkenny fields no longer being classed as at risk (no river 200 m downslope) and highlights the Clonmel field as being a critical source area of sediment and phosphorus (river within 200 m downslope). The Morgan's P, water extractable phosphorus and Mehlich-3 phosphorus measured in the Clonmel field greatly exceeded the respective thresholds determined for each by the RDPRI and therefore the Clonmel field is also a critical source area of dissolved reactive phosphorus. Given that reducing soil phosphorus to environmentally acceptable levels in the Clonmel field will take many years of restricted fertiliser use, methods of flow path manipulation (such as buffer zones or sediment traps) that reduce connectivity between phosphorus source and receiving waters should be used, as these can immediately reduce the amount of phosphorus reaching surface waters.

### 6.3 Conclusions

The main conclusions of the study are as follows:

1. Tilled soils, when subjected to high intensity, simulated rainfall and overland flow, may produce DRP concentrations in excess of  $0.03 \text{ mg L}^{-1}$  (the value above which eutrophication of rivers is likely to occur) if their  $P_m$ , WEP, and M3-P concentrations exceed  $7.83 \text{ mg L}^{-1}$ ,  $4.15 \text{ mg kg}^{-1}$ , and  $61.2 \text{ mg kg}^{-1}$ , respectively.

2. Water extractable P in both rainfall only and rainfall and overland flow simulations was identified by statistical methods as being a better indicator of the study soils potential to release DRP in surface runoff than  $P_m$ , the national soil P test.
3. This study showed that, while tillage soils with higher levels of soil P generate higher losses of DRP in surface runoff, an important mechanism of P loss is by detachment and transport of eroded soil particles. Therefore, the identification and remediation of sites susceptible to erosion and associated PP loss should be a high priority.
4. This study developed a methodology that can be used by Irish tillage farmers with minimal training to identify fields at possible risk of soil erosion and reduced soil quality. These sites can then be assessed in more detail by specialist advisors. The method, which uses a combination of erosion risk and soil quality indicators, correctly identified the study fields in which erosion was observed as high risk.
5. The adoption of the screening method, developed in this study, by tillage farmers in Ireland will enable the quantification of the extent of erosion risk and soil quality degradation associated with tillage soils. This will allow remediation measures to be prioritised for the most vulnerable sites, which is likely to result in cost and resource savings for farmers and advisory services.

#### **6.4 Recommendations for future work**

1. Cereal growers in Ireland will face pressure to increase yields substantially into the future to maintain margins and satisfy the demand for home grown concentrate feed due to the projected expansion in the dairy, drystock and pig sectors by 2020 (DAFF, 2010). The intensification required to meet these yield demands may impact negatively on soil quality and increase erosion risk, particularly where the soil type is prone to damage and/or the site is in long-term tillage. These problems can lead to a reduction in productivity unless the



specific sites are identified and remedial action is taken. It is proposed the screening toolkit developed in this study can be used by farmers to identify sites with soil quality/erosion issues. This should be done systematically in conjunction with regular soil testing so that any soil quality problems identified by farmers can subsequently be discussed with their advisors when soil tests are being interpreted. The required course of action to reverse any degradative trend including any identified in the soil test results such as excessive soil P status ( $> 10 \text{ mg L}^{-1} \text{ P}_m$ ) can then be set out in a management plan for the farm.

2. If predicted increases in the magnitude and frequency of storm events in Ireland due to climate change are accurate, then soil erosion may become a significant threat to soil and water quality in tillage areas by the 2020s, through removal of topsoil, decline of SOM and loss of nutrients. The proposed SFD has recognised erosion as a threat to soil quality and, if ratified, will require member states to identify areas prone to soil degradation processes like erosion. The screening toolkit may be used to assist with this work and also help identify the causes of the erosion.
  
3. Soils in long-term tillage and/or under intensive management conditions are prone to problems relating to soil quality. Further research is needed to determine 'best management practice' for inclusion in remedial plans that will improve soil quality and productivity, and reduce erosion levels and associated P loss on fields identified as high risk by the farmer using the screening toolkit. In order to develop comprehensive best management strategies that can control P loss from fields and/or catchments, all hydrologic implications, particularly variable source area concepts of runoff generation, must be incorporated. These best practice measures will also be required for inclusion in the POM, which Ireland will be required to implement in identified risk areas should the SFD be ratified.

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## **APPENDICES**

## **Appendix A List of publications**

### **JOURNAL PAPERS (Accepted)**

Regan, J.T., Rodgers, M., Healy, M.G., Kirwan, L. and Fenton, O. 2010. Determining phosphorus and sediment release rates from five Irish tillage soils. *Journal of Environmental Quality* 39: 185-192.

Regan, J.T., Fenton, O. and Healy, M.G. 2012. A review of phosphorus and sediment release from Irish tillage soil, the methods used to quantify losses and the current state of mitigation practice. *Biology and Environment: Proceedings of the Royal Irish Academy*, 112B: 157 - 183. DOI: 10.3318/BIOE.2012.05.

Creamer, R.E., F. Brennan, O. Fenton, M.G. Healy, S.T.J. Lalor, G.J. Lanigan, **J.T. Regan**, and B.S. Griffiths. 2010. Implications of the proposed Soil Framework Directive on agricultural systems in Atlantic Europe – a review. *Journal of Soil Use and Management* 26: 198-211.

### **MANUSCRIPTS IN PREPARATION**

Regan, J.T., Fenton, O., Walsh, M. and Healy, M.G. Physical, chemical and visual evaluation of 6 Irish tillage soils to assess soil quality and susceptibility to erosion (Target journal: *Soil and Tillage Research*).

Regan, J.T., Fenton, O., Grant, J. and Healy, M.G. Estimating phosphorus and sediment release rates from five Irish tillage soils when subjected to increasing overland flow rates (Target journal: *Science of the Total Environment*).

### **INTERNATIONAL CONFERENCE PAPERS**

Regan, J., Rodgers, M., Healy, M.G., Fenton, O. 2008. Sediment and nutrient loss from five Irish tillage soils at a 30 mm hr<sup>-1</sup> rainfall intensity. EUROSOIL Congress, University of Technology Vienna, Austria, 25 -29 August, 2008 (Oral presentation).

## NATIONAL CONFERENCE PAPERS

Regan, J., Rodgers, M., Healy, M.G., Fenton, O., Walsh, M. 2008. Soil erosion and nutrient loss from Irish tillage soils. Agricultural Research Forum, Tullamore, Co. Offaly, 10-13 March, 2008 (Oral presentation).

Regan, J., Rodgers, M., Healy, M.G., Fenton, O. 2008. Sediment loss and surface runoff from Irish tillage soils. ESAI Colloquium, Dundalk IT, February, 2008 (Oral presentation).

Regan, J., Rodgers, M., Healy, M.G., Fenton, O. 2007. Soil erosion and surface runoff from Irish tillage soils. Teagasc Walsh Fellowship Seminar, RDS, Dublin, November 14<sup>th</sup>, 2007 (Poster presentation).

Regan, J., Rodgers, M., Healy, M.G., Walsh, M., Fenton, O. 2007. Enrichment ratios for a range of Irish soils. ESAI colloquium, IT Carlow, January, 2007 (Oral presentation).

## OTHER PUBLICATIONS

Regan, J.T., Rodgers, M., Healy, M.G., Kirwan, L. Fenton, O.. 2008. Sediment and nutrient loss from five Irish tillage soils at a 30 mm hr<sup>-1</sup> rainfall intensity. Teagasc grassland and EU Water Framework Directive conference, Johnstown Castle, Wexford (Poster presentation).

Regan, J., Healy, M.G., Rodgers, M. and Fenton, O. 2008. Soil erosion and surface runoff from Irish tillage soils. Farmfest, Teagasc, Athenry, Galway (Poster presentation).

Fenton, O., Hyde, B., Ó hUallacháin, D., Healy, M.G., **Regan, J.**, and Rodgers, M. 2007. Tackling nutrient loss head on: catching the nutrients that got away. TRResearch 2(2): 32-34. ISBN: 1649-8917.

Regan, J., Rodgers, M., Healy, M.G., Fenton, O., Walsh, M. 2007. Soil erosion and surface runoff from Irish tillage soils. Pp. 25-26. In: J.A. Finn, K. Richards, G. Shortle (eds). Ireland's rural environment. Johnstown Castle Environment Research Centre. ISBN: 18 4170 477 6.