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# A REVIEW OF PHOSPHORUS AND SEDIMENT RELEASE FROM IRISH TILLAGE SOILS, THE METHODS USED TO QUANTIFY LOSSES AND THE CURRENT STATE OF MITIGATION PRACTICE

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## ABSTRACT

Throughout the European Union (EU), agricultural soils with high phosphorus (P) status due to a surplus input of fertiliser have been identified as a landscape pressure impacting on water quality. In the Republic of Ireland, approximately 80% of agricultural land is devoted to grass, 11% to rough grazing, and 9% to arable cereal and crop production. Consequently, the majority of erosion research has focused on quantifying nutrient and sediment losses from grassland. Tillage soils are, however, more susceptible to erosion than grassland soils and, in general, have higher levels of soil P. This paper reviews the current state of research, and the regulatory regime relating to diffuse P and sediment loss for tillage soils. It identifies the key threats to soil quality associated with cultivated soils, and proposes the targeting and remediation of critical source areas for effective mitigation of P losses from tillage soils. A multi-scaled approach is recommended, in which catchment and field-scale monitoring is complemented with controlled laboratory and small plot-scale rainfall simulation experiments to identify areas where P loss and soil erosion are at critical

levels and may pose a threat to water quality. Catchment scale research will help to link critical source areas of sediment and P loss with hydrological pathways to surface waters in the catchment. These areas can then be targeted for remediation in the River Basin Management Plans.

*Keywords:* tillage soil; phosphorus; suspended sediment; cultivation practices; water Framework Directive.

**INTRODUCTION**

To date in Ireland, the extent of erosion and phosphorus (P) loss from tillage land is unknown, as research has concentrated on quantifying losses from the more dominant grassland. Increasing eutrophication of many surface waterbodies 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 and Quinton 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.  Further 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), Deacy *et al.* (2009), Stevens *et al.* (2009) and Silgram *et al.* (2010).

Diffuse P loss from arable land can be as high as 1-2 kg P $ha^{-1}$ $yr^{-1}$ in the northern temperate zone, especially in areas with widespread soil erosion (Ulén *et al.* 1991). This review determines the current state of research on diffuse P and sediment losses relating to tillage soils in Ireland. It examines the associated key threats to soil and water quality and identifies knowledge gaps and future needs in erosion research on Irish tillage soils.

## AGRICULTURE AND WATER QUALITY IN IRELAND

In the Republic of Ireland, agriculture accounts for 60.8% (approx 4.2 million ha) of the total land area (CSO 2009) - well above the European average of approximately 40%. Of this agricultural land, approximately 80% is devoted to grass (silage, hay and pasture), 11% to rough grazing, and 9% to arable cereal and crop production with barley as the most important cereal crop representing 4.4% (185,900 ha). Most of the grain is used for the production of animal concentrate feedstuffs. 48% of 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). 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. There are fewer wet days, higher temperatures, less chance of frost, higher radiation receipts and more hours of bright sunshine (Collins and Cummins, 1996).

While tillage land (cereals and root crops) accounts for a relatively small area (9.6% of agricultural area utilised in the Republic of Ireland (CSO 2010)), 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 total P input to surface water systems from agricultural soils. Mean P fertiliser use in the Republic of Ireland for cereals and root crops (less than 10% of tillage area) in 2008 was 20 and 46 kg ha$^{-1}$, respectively, while P fertiliser use for grassland was only 5 kg ha$^{-1}$ (Lalor *et al*. 2010). These figures highlight the potential for higher P losses in surface runoff from tillage land than from grassland. 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 mg P L$^{-1}$ (Bowman 2009). To date, arable land in Ireland has received limited attention for its potential to impact on water quality (Doody et al. 2012).

Continual fertiliser application at high rates on agricultural land in the past resulted in excessive levels of plant available P in soils (Tunney 2000). Excess P may then be available to surface runoff following heavy rainfall events (Culleton *et al*. 2002). As soil P increases, P loss in surface runoff and subsurface flow increases (Sharpley *et al*. 2001b). Therefore, the higher the Morgan's P ($P_m$) (the national soil P test in the Republic of Ireland for determining plant available P) 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 dissolved reactive phosphorus (DRP) lost in surface runoff.

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 phosphorus, 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).

In the Republic of Ireland, Morgan's extractant (Peech and English 1944) is currently used to match P fertiliser recommendations with crop requirements. Fertiliser advice is modified for some tillage crops, according to crop yields, soil texture or expected summer rainfall amount (Coulter and Lalor 2008). Phosphorus advice for grassland and tillage crops in the Republic of Ireland is based on a four- category soil P-index system (Table 1). The basis of this system is a set of soil indices based on the measured $P_m$ in the soil and the crops response to fertiliser application as measured by field experimentation. For tillage soils in P Index 4, the addition of P is prohibited with the exception of soils planted with potatoes, beet, and turnips. 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 mg $L^{-1}$ and with no further P inputs (estimated to be equivalent to an annual field P-balance deficit of 30 kg $ha^{-1}$ $yr^{-1}$), it took from 7-15 years for a soil to move from Index 4 to Index 3 (Table 1).

The current agronomic optimum $P_m$ value for Irish soils is 6 mg $L^{-1}$ for grass production (Daly *et al.* 2001). In Ireland, low soil $P_m$ concentrations of 1 mg $L^{-1}$ in the 1950s severely limited crop production. Although national sales of P fertiliser have fallen from 62,410 t $yr^{-1}$ to 26,350 t $yr^{-1}$ during the period 1995-2008 (DAFF 2009b), primarily due to new farming practices, implementation of the Nitrates Directive (91/676/EEC: Council of the European Union 1991) and rising fertiliser costs, the mean $P_m$ concentration in Irish soils is currently 8 mg $L^{-1}$ (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 $L^{-1}$ Morgan's P) than on soils of low P status that have been supplemented with higher levels of P fertilisers (Schulte *et al.* 2010b).

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 1). This is achieved by applying enough P to replace the anticipated crop off-take (a grain yield of 1 tonne $ha^{-1}$ = an offtake of 3.8 kg P $ha^{-1}$), based on the expected yield of the crop to be fertilized (Coulter and Lalor 2008). Where proof of higher yield is available, an additional 3.8 kg P $ha^{-1}$ can be applied on soils at P Indices 1, 2 and 3 for each additional tonne above a threshold crop yield dependant on crop variety (S.I. No. 610 of 2010). Where the Soil Index is below Index 3, build-up levels are necessary in addition to anticipated crop offtake 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 $km^2$ - 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 1), which state that soils having > 10 mg $L^{-1}$ $P_m$ 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 results 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.

A biological survey of 13,188 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 Water Framework Directive (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 water bodies achieving 'good status' (< 0.035 mg P $L^{-1}$ in rivers) (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. Diffuse losses from agriculture were reported by McGarrigle and Donnelley (2003) to account for 59% of total phosphorus (TP) exported from a rural Irish catchment. Almost half the rivers 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).

LEGISLATIVE BACKGROUND

The WFD aims to restore polluted water bodies to 'at least' good ecological and chemical status (< 0.035 mg P $L^{-1}$ 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 Programmes of Measures (POM) by the end of 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 waterbody to help achieve prescribed water quality standards. The RBMP have identified agriculture as one of the main physico-chemical pressures affecting a waterbody. The basic regulation for agriculture in the Republic of Ireland is the Nitrates Directive and is given statutory effect in the European Communities (Good Agricultural Practice for Protection of Waters) Regulations 2010 (S.I. No. 610 of 2010). The latter sets out detailed nutrient management controls for farming, including P application rates for crop production.

Implicit in these directives and management plans is the protection and preservation of soils. An EU Draft Soil Framework Directive (SFD; COM (2006) 231: Commission of the European Communities 2006) 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 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).

RESEARCH TO DATE ON SEDIMENT AND P LOSS

To date, the majority of research in 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) 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. Tillage soils are, however, more susceptible to water erosion than grassland soils (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 organic matter levels can reduce erodibility (Fullen 1998).

In an analysis of studies investigating the relationship between dissolved P (DP) concentration in runoff and soil test P (STP), Sibbesen and Sharpley (1997) noted that for the same level of STP, generally less P was lost from grassland than from cultivated land. This may result from less interaction of surface runoff with surface soil for grass than for tillage crops, due to a better vegetative cover and surface-soil protection by grass (Sharpley 1995). Disturbance of soil structure by tillage operations also increases aggregate dispersion and the degree of interaction between soil and runoff water, thereby enabling more DP to be mobilised from soils with high P (Sharpley *et al.* 2001a). 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 P and sediment losses from tillage soils to surface waterbodies and monitor the effects of improving tillage practices.

**FACTORS AFFECTING SOIL EROSION RATES IN IRELAND**

Very few studies in Ireland have looked at soil 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. 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). Greater demand for food and advances in farm machinery has resulted in intensified crop production, leading to higher tillage and water erosion rates. 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.

## SOIL TYPE

Soil type is important when determining the erosion risk from an arable field. The texture of a soil strongly influences soil organic matter (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 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). 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 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).

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) tillage soils in order to ensure that sustainable management practices are put in place to reduce any further decline in SOC (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.

## 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 in 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 mm $day^{-1}$) with a high intensity (> 4 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 east coast to between 1000-1250 mm on the west coast. The highest annual rainfall of between 1600 and 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-year 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 Environmental Protection Agency (EPA) 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. 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.

## 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. 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° 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, nitrogen and phosphorus 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 2010). Increases in SOC resulting from management changes are slow and reversible (Hackett 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 Table 2.

## IMPACT OF TILLAGE FARMING

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 controls 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 Mg $ha^{-1}$ $yr^{-1}$. For the same land area, they estimated the average water erosion rate was 3.9 Mg $ha^{-1}$ $yr^{-1}$ 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 Mg $ha^{-1}$ $yr^{-1}$, 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 - 1.4 Mg $ha^{-1}$ $yr^{-1}$, with the lower limit being indicative of European conditions (Creamer *et al*. 2010). Research on tillage soils in Ireland is needed to validate the estimates of erosion rates given by the model of Van Oost *et al.* (2009).

The potential TP losses associated with these estimates of erosion could have serious implications for water quality in Ireland if the eroded sediment reaches surface waterbodies, 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 (0-10cm) 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 $ha^{-1}$ $yr^{-1}$ 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 $ha^{-1}$ $yr^{-1}$ 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 favorable 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 (Allakukku *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).

## 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, or proportion of the sediment mobilized 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 water 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 sediment delivery ratios 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 later.

## MEASURING AND QUANTIFYING PHOSPHORUS TRANSFER FROM ARABLE SOIL TO WATER

Soil cultivation is a major factor contributing to an increased risk of particulate phosphorus (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. 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; Murphy 2007), in particular, on soils with low P-retention properties and/or significant preferential flow pathways (e.g., cracking clay soils) (Hart *et al*. 2004).

## PHOSPHORUS MOBILIZATION

Mobilization is the first key step in the separation of P molecules from their source and includes chemical, biological and physical processes. 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). The detachment and transfer of non-dissolved P in association with soil particles is more pronounced where farming practices generate erosion (Chamber *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 µm and has been used for the threshold between dissolved DP and PP (Haygarth and Jarvis 1997). 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 DP 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 mobilization 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 DP in surface runoff is dependent on the complex interaction between climate, topography, soil type, soil P content, type of farming system and farm management (Withers 1999).

Particulate phosphorus includes 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 - 98% of total runoff P for un-vegetated packed flumes. 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 Oghahl 2008); or (3) an increase in pH of the interstitial water (Sharpley and Rekolainen 1997; Daly 1999).

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 work in controlled laboratory experiments. 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 (Andraski *et al.* 1985; Zhao *et al.* 2001; Daverede *et al*. 2003; Franklin *et al*. 2007). Studies have also been conducted using laboratory rainfall simulation on flumes packed with tillage soil to predict runoff of SS and PP using simple soil tests (Udeigwe and Wang 2007), and examine variability in mobilization and transport of nutrients and sediment by overland flow across a range of soils (Miller and Quinton 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.

## PHOSPHORUS SOURCE AREAS

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 (VSAs), generate the majority of overland flow (Doody *et al*. 2006), the occurrence of which is largely independent of rainfall intensity (Walter *et al*. 2000). This type of runoff is known as saturation excess runoff. A VSA can contract and expand both seasonally and during storms as a function of precipitation, topography, soil type, soil moisture status, and water table level (Hart *et al.* 2004). The occurrence of high STP within a VSA results in a critical source area (CSA) of P (Gburek and Sharpley 1998).

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 four-day 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. Hughes *et al.* (2005) used field and catchment-scale ranking schemes to identify CSAs for P loss in Ireland. Outside of the VSAs, runoff may be triggered when the infiltration capacity of the surface soil is exceeded, usually following high-intensity storm events. Both saturation and infiltration-excess runoff occur in Ireland and, though the latter is less common, research has shown that it does occur (Schulte *et al*. 2006a). At field-scale, runoff collection is complicated in areas with a perched watertable. The divisions between saturated and infiltration runoff become difficult as a VSA is generated.

## MODELLING SOIL EROSION AND SEDIMENT AND P DELIVERY TO SURFACE WATERS AT THE CATCHMENT SCALE

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 surface and groundwaters, 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).

Many different kinds of models are available for use to simulate soil erosion and sediment and P delivery to waterways at the catchment-scale. In general, these models fall into three main categories: (1) empirical (2) conceptual and (3) physical or process based. However, the difference between the model categories is not always clear, and making the distinction can be somewhat subjective (Merritt *et al*. 2003). For example, it has been argued by Lowe (2006) that the Hydrological Simulation Program – Fortran (HSPF) (Bicknell *et al.* 1997), which has been classed as a conceptual model by many studies is, in fact, a physically-based model. Previous work by Merritt *et al*. (2003) provides a comprehensive review of erosion and sediment transport models. For the purposes of this review, the focus will be on catchment-scale models that have been used in Ireland to estimate soil erosion, and P and sediment delivery to waterways. These are empirical models (Revised Universal Soil Loss Equation (RUSLE) and Sediment Distribution Delivery (SEDD)) and physically-based models (Hydrological Simulation Program – Fortran (HSPF) (Bicknell *et al*. 1997), Soil Water Assessment Tool (SWAT) (Arnold *et al*. 1998), Système Hydrologique Européen TRANsport (SHETRAN) (Ewen *et al*. 2000)) and a modified version of TOPMODEL (Scanlon *et al*. 2005). Where possible, the losses estimated using these models are compared with losses from the same models applied in other countries and with measured losses from Irish and international catchments (Tables 3 and 4). Caution is required when comparing results from these tables given the effect of catchment size on suspended sediment yield.

## EMPIRICAL MODELS

These models are generally considered to be the simplest of the three model types and are frequently used in preference to more complex models as they can be implemented in situations with limited data and parameter inputs, and are particularly useful as a first step in identifying sources of sediment and nutrient generation (Merritt *et al*. 2003). They are derived from the analysis of field observations and endeavour to characterise response from these data.

The Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978) is a program used widely in America and worldwide that estimates the long-term water erosion from interill and rill areas. It is represented by the equation: $A = RKLCSP$, where $A$ is the estimated soil loss per unit area, $R$ is the rainfall erosivity factor, $K$ is the soil erodibility factor, $L$ is the slope-length factor, $S$ is the slope-steepness factor, $C$ is the cover and management factor and $P$ is the support practices factor. The USLE was revised (RUSLE) (Renard *et al*. 1991) and revisited (Renard *et al.* 1994) to take into account additional information that had become available since its development. Although developed for application to small hillslopes, the USLE and its derivatives have been incorporated into many catchment-scale erosion and sediment transport modelling applications (Merritt *et al*. 2003). The SEDD model is based on the USLE model. It discretizes a catchment into morphological units (areas of defined aspect, length and steepness) and determines a sediment delivery ratio (SDR) for each unit (Fernandez *et al*. 2003). The SDR is the ratio of sediment reaching a continuous stream system to the total amount of sediment eroded by sheet and channel erosion. The magnitude of the SDR for a particular catchment will be influenced by a wide range of geomorphological, hydrological, environmental and catchment factors (Fu *et al*. 2006).

The combined use of Geographical Information Systems (GIS), RUSLE and SEDD has been shown to be an effective method for estimating water erosion and sediment yield by Fernandez *et al*. (2003) and Fu *et al*. (2006), and for estimating the impacts of no-till practice on soil erosion and sediment yield by Fu *et al.* (2006). A case study in Ireland by He (2010) to estimate soil erosion and sediment yield using GIS, RUSLE and SEDD, predicted that the average SS delivered from arable land to waterways was 0.22 Mg $ha^{-1}$ $yr^{-1}$. However, this finding should be treated with caution because the catchment-specific parameter β was only estimated for the Dripsey catchment (using an inverse modelling approach employing observed SDR values from fields) and sensitivity analysis of β by increments of 1.0 to a maximum of 20.0 was carried out to infer possible values of β for the Bandon, Dromcummer, Duarrigle and Mallow catchments. Fu *et al*. (2006) tested β between 0.5 and 2.0 with an increment of 0.1 and found that the sediment delivery ratio was very sensitive to the values of β, varying from 0.6 ($\beta = 0.5$) to 0.27 ($\beta = 2.0$).

## PHYSICALLY BASED MODELS

Physically based models are those in which the model equations are based on physical laws and relationships. They are more complex than empirical models and require more measurement and calibration of model parameters. Complex physical models applied with the necessary expertise or user support can be far superior where there is a need to address spatial and temporal complexities (Irvine *et al.* 2005). According to the DPSIR conceptual framework (Drivers, Pressures, State, Impact and Response) (Irvine *et al.* 2005) that guides the selection of modelling techniques in Ireland, it is likely that the most useful models will be of the physically based or mechanistic type (Nasr *et al*. 2007).

Nasr *et al*. (2007) tested three widely used physically based models (SWAT, HSPF and SHETRAN coupled with the grid orientated P component (GOPC) (Nasr *et al.* 2005) of diffuse P pollution, in three Irish grassland catchments, to explore their suitability in Irish conditions for future use in implementing the WFD. These models range from semi-empirical (SWAT) to fully physically based (SHETRAN/GOPC) in how they represent the relevant hydrological, chemical and bio-chemical processes transforming the P compounds both in the soil and during its transport by water (Nasr *et al*. 2007). The catchments were selected based on data availability and different climate, land use and soil type. SWAT is a continuous model working at the basin-scale to look at the long-term impacts of management and also timing of agricultural practices within a year on water, sediment, and agriculture chemical yields in large un-gauged basins (Arnold *et al*. 1998). HSPF is a lumped-parameter model that simulates hydrology and water quality processes on a continuous basis in natural and man-made water systems (Im *et al*. 2003). SHETRAN/GOPC is a fully physically-based model which relies on relationships derived from physical and chemical laws. The three models also differ in their representation of the spatial variation within the catchment and the time steps at which they can simulate. In order of ability to match the measured discharge hydrographs from each catchment in this study, the models performed (from best to worst) as follows: HSPF, SWAT, SHETRAN. The best simulation for daily TP loads in the study catchments was by SWAT. In the short term, Nasr *et a*l. (2005) recommended using SWAT for TP load estimation. The SWAT model recently showed good potential for predicting TP losses from arable land in a Swedish study by Ekstrand *et al*. (2010) (Table 4).

TOPMODEL is a process based semi-distributed catchment model (Irvine *et al*. 2005) in which the major factors affecting runoff generation are the catchment topography and the soil transmissivity, which diminishes with depth (Parsons *et al.* 2001). In this model, overland flow generation follows the VSA concept while groundwater discharge is from a permanent water table. TOPMODEL is not intended to be a traditional modelling package, but a collection of concepts to help in understanding and predicting the hydrological behaviour of basins (Parsons *et al.* 2001). It was used for this purpose by Scanlon *et al*. (2005) when a modified version of TOPMODEL was developed and applied to a 14.26 ha grassland catchment in Ireland in order to infer the significant pathways of soil-to-stream P transport. In this study, a physically-based hydrological model generated pathway-specific information for three components of discharge: overland flow, shallow subsurface flow and groundwater discharge. An independent comparison of the hydrological model output and stream water P measurements allowed the authors to infer the relative contributions from individual pathways to the overall P transport. They found that the fraction of modelled stream discharge deriving from overland flow and shallow subsurface flow was a reliable descriptor of the observed TP concentrations. Shallow subsurface flow was inferred to be the dominant P transport mechanism, primarily due to much greater volumetric contributions to stream discharge deriving from it than from overland flow. These model results challenge the commonly held assumption that the majority of P transport occurs via surface runoff and could have important implications for the design and implementation of remedial measures (Scanlon *et al*. 2005).

## MITIGATION MEASURES TO PREVENT EROSION ON TILLAGE SOILS

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 test (Curtin *et al*. 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, 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 aggregate stability 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 *et al.* 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 $ha^{-1}$ and 0.3 kg TP $ha^{-1}$ 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 1st July and 30th 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 2006), 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). In the Republic of Ireland, farmers protect vulnerable tillage soils by complying with ‘good agricultural and environmental conditions’ (GAEC) guidelines as a condition for receipt of the area-based single farm payment under the EU cross-compliance regime (Department of Agriculture and Food 2005). Land that has been in continuous tillage for six years or more must be tested for organic matter 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 organic matter 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 2m 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 Agricultural Catchments Program (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). Spatial soil mapping, combined with conservative ground-truthing, is currently underway in the form of the Irish Soil Information System (ISIS). This aims to complete the soil map of Ireland at a 1:250,000 scale (EPA 2009). 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.

## FUTURE RESEARCH DIRECTION IN THE QUANTIFICATION OF NUTRIENT AND SEDIMENT LOSS FROM IRISH TILLAGE SOILS

### 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 ($^{137}Cs$) inventory at a particular sampling point with the reference inventory (the total $^{137}Cs$ 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 $^{137}Cs$ and the fallout radionuclide unsupported $^{210}Pb$ afford a means of obtaining retrospective, medium-term (i.e. ca. 45 years for $^{137}Cs$ and up to 100 years for unsupported $^{210}Pb$) 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, $^{137}Cs$ ($t_{1/2}$ = 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 $^{137}Cs$, $^{7}Be$ 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 $^{137}Cs$, $^{7}Be$, and $^{210}Pb$ 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). 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. Sediment fingerprinting correlating landuse with river sediment appears to offer a valuable alternative to direct monitoring for elucidating the provenance of SS and the relative importance of spatial zones or sub-catchments comprising larger (>500 km$^2$) drainage basins (Collins *et al.* 1998).

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 most 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) recognized, the 1-year period of monitoring in this project was too short to provide a reliable picture of sediment dynamics in the Bush catchment.

## P LOSS AND STP IN TILLAGE SOILS

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 M3-P as an agri-environmental soil P test for the Mid-Atlantic United States of America, 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 $L^{-1}$ $P_m$ (Table 1) while still facilitating optimum productivity and herbage quality and minimising the risk of diffuse P losses to water. Index 3 (5.1 - 8 mg $L^{-1}$ for grassland) in the new P-index system (Table 1) 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 $L^{-1}$) 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.

Using the relationship between STP in five Irish tillage soils and the DRP released in the surface runoff, Regan *et al.* (2010) developed a runoff dissolved phosphorus risk indicator (RDPRI) to identify the STP level above which there may be a potential threat to surface water quality. The results of this study complemented the agronomic guidelines of the Nitrates Directive, as they indicate that tillage soils may produce surface runoff P concentrations in excess of 30 µg $L^{-1}$ (the median phosphate concentration above which significant deterioration is seen in river ecosystems) if their $P_m$ and M3-P concentrations exceed 9.5 mg $L^{-1}$ and 67.2 mg $kg^{-1}$, respectively.

## 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 (NAP) measures over time for the major farming and environmental stakeholders in Ireland. The program 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 years (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 will be monitored: TP, total dissolved P (TDP), total reactive P (TRP), DRP, total N, $NO_3$-N, turbidity, electrical conductivity, temperature, and flow rate. Particular attention will be 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 bankside 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. The use of turbidity as a surrogate technique for the measurement of SS is also being investigated in the catchments.

## CONCLUSIONS

The vast majority of research on soil erosion and nutrient loss in Ireland has concentrated on grassland due to its predominance. There is little research dedicated to tillage areas, although internationally, tillage areas have been identified as risk areas. Estimating the environmental risk associated with Irish tillage areas based on international findings is difficult because rainfall, soil type and cultivation practices differ and therefore not all international research findings are relevant within an Irish context. Modelling of water and tillage erosion rates across Europe 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. The main conclusions from this review are:

1. As P transfer to surface water may occur in dissolved form, the relationship between STP in tillage soils and DRP concentration in runoff water needs to be adequately understood and quantified for Irish tillage soils.
2. The ability to identify CSAs of P and the hydrological pathways connecting these areas to surface waterbodies is essential if mitigation measures are to be cost effective.
3. 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 a number of large storm events, research to quantify P and sediment loss from Irish tillage soils should utilise a laboratory-, field- and catchment-scale approach that can identify contributing portions of land within a catchment posing a risk, and also identify and quantify actual releases from the catchment as a whole.
4. Catchment scale research will help to link critical source areas of sediment and P loss with hydrological pathways to surface waters in the catchment. Remedial initiatives set out in the RBMP can then be targeted in these areas and their effectiveness evaluated.
5. Research conducted at laboratory-scale can contribute valuable information towards understanding the mechanisms controlling sediment and P loss, and provides an estimate of future losses due to climate change and potential losses at larger scales. However, field-scale research offers a real life situation where ground-truthing of laboratory findings can take place. At the catchment-scale, diffuse sediment losses can be traced from source to receptor using constantly improving techniques such as sediment fingerprinting.

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Table 1. Phosphorus Index System (from S.I. No. 610 of 2010 and adapted from Schulte et al., 2010a)

| **Soil P Index** | **Soil P ranges (mg/l)** | | **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. |

Table 2. 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 |

Table 3. Erosion and sediment yield in a selection of Irish and international studies

| Catchment | Area (Km$^2$) | Land use | Assessment method | Study Period (months) | Precipitation (mm) | Runoff (mm) | Erosion (ton km$^{-2}$ yr$^{-1}$) | Sediment yield (ton km$^{-2}$ yr$^{-1}$) | Reference |
|---|---|---|---|---|---|---|---|---|---|
| Bush, Co. Antrim, Ireland | < 340 | Arable (36.6%) | In stream sampling and sediment fingerprinting | 12 | - | - | - | 51.4 – 107 [a] | [1] |
| Dripsey Co. Cork, Ireland | 14 | Grassland dominated | In stream sampling | 12 | 1833 | 1037 | - | 13.60 | [2] |
| Clarianna, Co. Tipperary, Ireland | 29.8 | | | 12 | 1091 | 434 | - | 8.40 | [2] |
| Oona Water, Co. Tyrone, Ireland | 88.5 | | | 12 | 1366 | 817 | - | 41.0 | [2] |
| Gelbaek, Central Jutland, Denmark | 11.6 | Arable | In stream sampling | 12 | 932 | 369 | - | 7.1 | [3] |
| Belmont, Herefordshire, England | 1.5 | Arable (61%) | In stream sampling, turbidity sensors, sediment fingerprinting, $^{137}$Cs measurements | 12 | 660 | 268 | 466.6 | 81.9 | [4] |
| Lower Smisbey, Leicestershire | 2.6 | Grassland/arable | | 12 | 660 | - | 400.5 | 80.3 | [4] |
| Pang, Berkshire, England | 166 | Arable dominated | | 24 | 647 - 706 | - | 507 (ploughed)<br>140 (grass) | 2.4 | [5] |
| Lambourn, Berkshire, England | 234 | Grassland/arable | | 24 | 698 - 793 | - | 437 (ploughed)<br>95 (grassland) | 3.7 | [5] |
| Ireland | - | Arable | RUSLE and SEDD models | - | - | - | 1978 | 22.0 [b] | [6] |
| Pataha Creek Watershed, Washington, USA | 327 | Arable | RUSLE and SEDD models | - | 250 - 1000 | - | 1766 | 711 | [7] |
| Lawyers Creek, Watershed, Idaho, USA | 308 | Arable | RUSLE and SEDD models | - | 533 - 737 | - | 2150 | 660 | [8] |
| Dripsey Co. Cork, Ireland | 14 | Grassland | RUSLE and SEDD models | - | - | - | - | 909.0 | [6] |

[1] Evans et al. 2006; [2] Jordan et al. 2005a; [3] Kronvang et al. 1997; [4] Walling et al. 2002; [5] Walling et al. 2006 ; [6] He et al. 2010; [7] Fu et al. 2006; [8] Fernandez et al. 2003

[a] This value is given in ton yr$^{-1}$ because the exact catchment area was not given in this paper

[b] Model prediction for the total tillage area of Ireland

Table 4. P delivery to streams in a selection of Irish and international catchments

| **Catchment** | **Area ($Km^2$)** | **Land use** | **Study period (months)** | **Annual Precipitation (mm)** | **SWAT (kg P $yr^{-1}$)** | **HSPF (kg P $yr^{-1}$)** | **GOPC (kg P $yr^{-1}$)** | **Measured (kg P $yr^{-1}$)** | **Reference** |
|---|---|---|---|---|---|---|---|---|---|
| Dripsey, Co. Cork | 15 | Grassland | 12 | 1833 | 1371 | 1530 | 1389 | 1719 | [1] |
| Clarianna, Co. Tipperary | 23 | Grassland | 8 | 1091 | 231 | 136 | 243 | 289 | [1] |
| Oona water, Co. Tyrone | 96 | Grassland | 12 | 1366 | 33285 | 25717 | 12519 | 27496 | [1] |
| Gelbaek, Central Jutland, Denmark | 11.6 | Arable | 12 | 932 | - | - | - | 371.2 | [2] |
| Sagån, Sweden | 864 | Arable | 36 | 618 | 31104 | - | - | 36288 | [3] |
| Lake Fork, East Central Illinois, USA | 365 | Arable | 72 | 960 | - | - | - | 17650 | [4] |
| Belmont, Herefordshire, England | 1.5 | Arable (61%) | 48 | 660 | - | - | - | 405 | [5] |

[1] Jordan et al. 2005a; [2] Kronvang et al. 1997; [3] Ekstrand et al. 2010; [4] Gentry et al. 2007; [5] Withers and Hodgkinson 2009