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# Mitigation options to reduce phosphorus losses from the agricultural sector and improve surface water quality: A review

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## HIGHLIGHTS

- Various mitigation options to reduce phosphorus losses from agricultural land were described in terms of factsheets.
- Global budget systems and agro-environmental recommendations systems are useful tools for setting up a more sustainable agricultural management practice.
- At field scale different crop and soil management techniques are available to increase the P efficiency and reduce loss of P from the fields by erosion and runoff.
- At catchment scale the landscape and the hydrological system determines the buffer capacity, transfer and delivery of nutrients to the surface water system and several options are available to reduce P losses.
- Finally, with surface water management measures the impact of nutrient loads on surface water quality can be reduced.

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## ABSTRACT

The EU Water Framework Directive (WFD) obliges Member States to improve the quality of surface water and groundwater. The measures implemented to date have reduced the contribution of point sources of pollution, and hence diffuse pollution from agriculture has become more important. In many catchments the water quality remains poor. COST Action 869 was an EU initiative to improve surface water quality that ran from 2006 to 2011, in which 30 countries participated. Its main aim was a scientific evaluation of the suitability and cost-effectiveness of options for reducing nutrient loss from rural areas to surface waters at catchment scale, including the feasibility of the options under different climatic and geographical conditions. This paper gives an overview of various categories of mitigation options in relation to phosphorus (P). The individual measures are described in terms of their mode of action, applicability, effectiveness, time frame, environmental side-effects (N cycling) and cost. In total, 83 measures were evaluated in COST Action 869.



## 1. Introduction

The role of an excess of nutrients phosphorus (P) and nitrogen (N) in the eutrophication of surface water was recognised in the mid-20th century (Redfield, 1958; Vollenweider, 1968). Among the negative environmental effects of eutrophication are reduced functioning and biodiversity of aquatic ecosystems and decline in surface water quality (Scheffer, 1998; Smith et al., 1999). The Harmful Algal Blooms (HABs) associated with eutrophication produce toxic algal substances that kill fish (Carpenter et al., 1969; Jaworski, 1981) and cause disease in animals (Kotak et al., 1994; Main et al., 1977) and humans (Falconer, 1989; Lawrence et al., 1994). Nutrient loads to waters must be reduced to control eutrophication.

The relative concentrations of total N and P together with bioassays have been used to estimate which of these nutrients is limiting the growth of algae in aquatic systems (Atkinson and Smith, 1983; Hecky et al., 1993; Redfield, 1958; Smith, 1983). For freshwater systems the indicative N:P weight ratios are ≤4.5 for N-limitation, 4.5–6 for intermediate conditions and ≥6 for P-limitation; the equivalent

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values for marine systems are respectively, ≤5, 5–10 and ≥10 (EC, 2002). In most freshwater systems P is the limiting factor and is the nutrient present in lowest amount in relation to phytoplankton requirements (Carpenter, 2008; Herath, 1997; Lee, 1973). By contrast, in marine systems, N has often been identified as the growth-limiting nutrient, especially in summer (Anderson and Glibert, 2002; Ärtebjerg and Carstensen, 2001). These findings should be interpreted with caution, however, as the N:P ratio may not correctly indicate the limiting nutrient of the system because of the role played by the absolute concentration and other factors such as light and physical conditions. Furthermore, the bio-availability of P can change over time due to redox processes, pH changes and enzymatically-mediated hydrolysis processes (Correll, 1998; Ekholm, 2008; Reynolds and Davies, 2001).

Since the mid-20th century, nutrient losses throughout Europe have been declining, partly as an epiphenomenon of processes such as industrial decline, but also because of targeted measures. Examples of the latter are increasing the number of homes connected to sewerage, introducing denitrification of N and precipitation of P in sewage water treatment plants, purifying industrial wastewater from industries and banning phosphorus in detergents. In many countries, P losses from easily identified and targeted point sources have been reduced. Yet despite these trends and the implementation of EU directives to reduce nutrient loads (EEC, 1991a, 1991b, 1996, 2000), the water quality in many rivers, lakes and estuaries remains poor, largely due to diffuse pollution from land entering surface water systems (EEA, 2010).

Nutrient losses from agriculture mainly enter water systems as a result of nutrients being transported by surplus runoff' as it flows over or under farmland. One of the most important pathways of P loss in hilly and mountain areas occurs when erosion detaches soil particles during overland flow. In flatter or less hilly areas, the main pathways of phosphorus pollution to surface waters are leaching through the soil and artificial drainage (Chapman et al., 2005; Chardon and Schoumans, 2007; Grant et al., 1996; Heathwaite et al., 2005; Kronvang et al., 1997; Nelson et al., 2005; Ulén and Mattsson, 2003).

Organic or inorganic nutrient ions are transported by water flow, either in solution, associated with particles, or incorporated in microorganisms. In general, the amount of P bound to soil particles is much higher than the amount of bound ammonium. However, when organic particles are transported in dissolved form they carry more N than P, since the C:N:P ratio of soil organic matter is about 100:10:1 (Stevenson and Cole, 1999; Tilsdale et al., 1990). The concentration of soluble inorganic P in runoff water depends highly on the amount of adsorbed P. There is a good relationship between the amount of soluble inorganic P in surface runoff water and the soil P status of the plough layer (Allen et al., 2006), and in most cases there is a P enrichment of the fine eroded material (Schiettecatte, 2006). The magnitude of dissolved inorganic P losses by leaching seems to be strongly influenced by the phosphate saturation of the soil (Schoumans and Groenendijk, 2000; Van der Zee, 1990). The amount and composition of the soluble organic nutrients depend on factors such as the soil organic matter content, physical and chemical type of organic material, affinity of the nutrient to adsorb to the soil and soil pH (McDowell and Koopmans, 2006; Wilson and Xenopoulos, 2009). After manure or fertiliser applications, high nutrient concentrations can be found in runoff water (Allen and Mallarino, 2008; Smith et al., 2007), soil solution (Chardon et al., 2007; Van Es et al., 2004) and tile drains (Schelde et al., 2006). These losses can generate high P fluxes and concentrations (Hahn et al., 2012; Preedy et al., 2001).

As significant P losses are known to occur via particles eroded from agricultural soils, soil erosion from arable land and grassland should be reduced. Since soil and crop management in the form of tillage strategies, application of soil conditioners, crop rotation, crop management and catch or cover crops directly impact on soil erosion, it is possible to reduce P losses by adapting these management strategies for this purpose in erosion-prone areas on arable land. On grassland, avoiding poaching by intensive grazing, and soil compaction by traffic are possibilities to reduce erosion (Newell Price et al., 2011). Clayey and silty soils are frequently important sources of loss of soil particles and attached P through erosion. Such soil types furthermore often demonstrate rapid water flow (preferential flow) via preferred pathways through a fraction of the soil pores (Jarvis, 2007), which enhances leaching of P through the soil profile and transport via tile drains. Soil and crop management strategies may also be adapted with the purpose of reducing this type of leaching loss.

A large proportion of arable land in north and northwest Europe has tile drainage: 30% in UK, 40% in Denmark (Brown and Van Beinum, 2009), and over 90% in the clayey/silty soil areas of Sweden and Finland (De la Cueva, 2006). The tile drainage systems channel water directly from farmland to streams and further to the sea. Reducing P transport from such drained agricultural fields will therefore effectively reduce the P transport in streams and the load on lakes and seas. Additionally, improvement of the actual drainage system may offer a possibility for reducing P losses under certain conditions.

The Water Framework Directive (EEC, 2000) obliges catchment management authorities of the EC Member States to improve the ecological status of surface waters by 2016 or at latest by 2027. When striving to meet the Directive's targets, the focus will be on reducing nutrient losses from agriculture, because this is becoming the main source of pollution. There is thus a need to have an overview of the options for reducing nutrient losses from the agricultural sector and empirically established effectiveness under different circumstances (Cherry et al., 2008; Withers and Haygarth, 2007). Focusing on P, here we give such an overview of mitigation practices that have been tested to varying degrees. Though the overview was conducted for Europe, it has relevance to regions elsewhere in the world where loss of P from farmland is an actual or potential problem.

## 2. Methods

From 2006 to 2011 the European Commission funded COST Action 869 in which 30 countries participated (Chardon et al., 2012). One of the main objectives was to undertake a scientific evaluation of the suitability and cost-effectiveness of different options for reducing nutrient loss to surface waters at river basin scale. This included reporting their limitations in terms of applicability under different climatic, ecological and geographical conditions. Based on information from literature and an inventory amongst participating countries, 83 measures were distinguished and grouped into eight categories: (a) nutrient management; (b) crop management; (c) livestock management; (d) soil management; (e) water management within agricultural land; (f) land use change; (g) landscape management and (h) surface water management. Each measure was described in a factsheet with the following sections:

- *Description*, including whether the mitigation effect targets P (or N).
- *Rationale, mechanism of action*: describes the mechanism to retain P (and N).
- *Relevance, applicability & potential for targeting*: describes under which conditions the option can be applied.
- *Effectiveness, including uncertainty*: estimates how effective the option can be, under which conditions it will be most effective and under which conditions it is least effective.
- *Time frame*: indicates if the option is assumed to be effective in the short, medium or long term.
- *Environmental side-effects/pollution swapping*: indicates unwanted effects in other environmental compartments.
- *Administrative handling, control*: describes the ease of applying and controlling the application of an option.
- *Costs*: since actual costs can vary greatly between and even within countries, only investment and maintenance costs are defined.

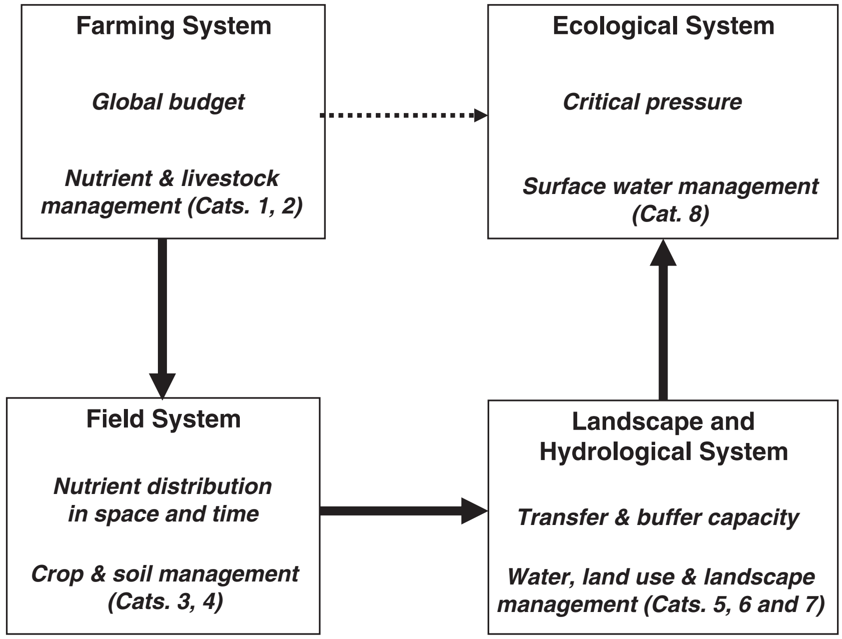


**Fig. 1.** Schematic visualization of the four systems determining the nutrient losses to surface water, with the defined categories of mitigation measures.

- *References*: the references given are preferably easily accessible, e.g. via a url.

All 83 mitigation measures are described individually on the website http://www.cost869.alterra.nl/Fs/List_of_options.htm. The complete references, links and numbering of all factsheets are given in the Supplementary Material accompanying this paper. Many factsheets include information from other evaluation studies. Important sources of information included are (Cuttle et al., 2007; Newell Price et al., 2011), from the UK, (Schou et al., 2007) from Denmark and the SERA-17 group in the USA (SERA-17; http://www.sera17.ext.vt.edu/SERA_17_Publications.htm).

In order to structure the different types of mitigation options for reducing nutrient losses, a framework was created of systems influencing the surface water quality. It comprises four systems (Fig. 1). At the farm scale, strategic socio-economic decisions are made about the production system, and often nutrient management and livestock management strategies are developed in order to comply with European and national legislation. Farm-gate nutrient budgets based on the nutrient and livestock management give valuable information about the potential pressure of nutrients in the region (I. Farm system). The distribution of the available P sources over the fields in space and time depends on the crops and the soil P status, and gives more information about the soil balances and the potential risk of losses from the fields (II. Field system). The actual P losses from the fields to surface waters strongly depend on the landscape and hydrological system in a given region, because those systems determine the transfer and buffer capacity for nutrients (III. Landscape and hydrological system). The impact of nutrient loads on the ecology of the surface waters (IV. Ecological system) depends on the actual nutrient pressure compared to the critical nutrient pressure.

Related to this framework of systems are the previously defined eight categories of measures relevant for reducing diffuse agricultural nutrient losses (see Fig. 1). They represent a complementary set of options to mitigate nutrient fluxes in space and time. Below, we discuss the most important measures for each system in more detail. The final step in COST Action 869 was to develop a web-based procedure to assist national and regional governments, water managers, intermediaries, and innovative farmers to select the best measures under specific circumstances (http://www.cost869.alterra.nl/dbase). Acknowledging

**Table 1**
Mitigation strategies for nutrient management at farm scale.

| Strategy | Aim | Measure | Factsheets |
|---|---|---|---|
| Environmentally sound fertiliser application & nutrient handling | Incorporate soil P into management strategy to achieve moderate soil P levels | Make use of available P in soils to avoid high risk hot-spots | 6, 28, 30, 36, 48, 49 |
| | Reduce P content of the soil at high risk hot spots | Don't apply manure and P fertiliser at high risk hot spots | 82, 28, 30, 34, 35, 45, 4 |
| | Increase P efficiency of crop uptake via appropriate placement and time of application | Use separated manure fractions and fertilisers with N:P-ratios in line with the N:P ratio required by crop | 59, 63 |
| | | Apply P near the roots instead of broadcast | 8 |
| | | Avoid applying manure and P fertilisers before heavy rainfall or prolonged rainfall | 81 |
| | | Phase nutrient fertilisation application over the year | 24, 25 |
| | | Create sufficient storage capacity | 83 |
| Change P input | Avoid high P content in fodder by increasing digestible P and lowering total P content in feed | Feed livestock refined fodder, taking account of their requirements given their growth phase | 1, 80 |
| | | Use feed with a lower content of phytate-P or add phytase to feed to increase digestibility of phytate-P | 1, 80 |
| Change P output | Exploit the commercial value of the manure surplus | Make products for export or for arable farms. | 46[a] |
| | | Produce secondary P resources for industries by incineration to P-ash | 46 |

[a] Schoumans et al. (2010).

the importance of an integrated approach in the design and implementation of mitigation options for reducing nutrient losses, the COST Action initially addressed both P and N. In this paper, however, we focus on P and treat N mainly as an important "side-effect".

## 3. Results

### 3.1. Farming system: nutrient and livestock management

Losses of N and P to surface waters are often a problem in regions with intensive agricultural and livestock production, due to the high availability of nutrients (Breeuwsma and Silva, 1992; Sharpley et al., 1994). Table 1 gives different mitigation strategies and associated measures related to farm management.

The EU Nitrate Directive is a crucial EU Directive: it stipulates the permitted amount of manure applications in the Nitrate Action Programmes (max. 170 kg manure N $ha^{-1}$ unless so-called derogation permits apply more manure) and the period during which manure and fertiliser can be applied. The Directive also stipulates that total N application must be based on the balance between crop requirement and manure and fertiliser N supply, taking into account N release from the soil. No application standards for P are defined in the Directive, but it makes it obligatory for Member States to prevent surface water eutrophication.

To derive agro-environmental nutrient management strategies, attention should be paid to the P balance at farm and field scales, by taking into account the amount of available soil P (FS28 Hofman, 2010). In many western European countries the amount of available soil P greatly exceeds the annual amount of P applied, because of the common agricultural practice of applying manure on the basis of crop N requirement. The average N:P ratio (w:w) of manures and composts often varies between 2 and 4, while the N:P uptake by major grain and hay crops varies between 4.5 and 9 (Eghball, 1998; Maguire et al., 2006). Thus, too much manure P is applied on agricultural land, and the P surplus accumulates in the soil. Soil samples taken in many regions with a high livestock density reveal that most of the soils sampled do not require further addition of P to support plant growth (Breeuwsma and Silva, 1992; Chardon and Schoumans, 2007; Lemercier et al., 2008; Sharpley et al., 1994).

At a high agronomic soil P status the P balance can be negative, and at low P status the P balance can be positive and this increases the soil P status to the level recommended to sustain optimal crop growth (Tunney et al., 1997). Therefore, on farms with many fields with a high agronomic soil P status, the aim should be to have a negative P balance at farm scale and to reduce the annual manure and/or fertiliser application on the fields with high soil P status, by taking account of soil nutrient status and crop response (FS06 Dana, 2010; FS30 Hofman, 2010; FS48 Newell Price, 2010; FS63 Turtola, 2010).

In so-called "critical source areas" (CSAs) (FS36 Krogstad and Bechmann, 2010; Heathwaite et al., 2000) – areas where sources and transport factors coincide with a high risk of P losses – integrated fertiliser and manure nutrient supply strategies are very important and can improve surface water quality by reducing the annual application rates of fertilisers (FS14 Garnier, 2010; FS35 Krogstad and Bechmann, 2010; FS45 Newell Price, 2010; FS49 Newell Price, 2010) and manure (FS60 Taylor, 2010; FS82 Taylor and Garnier, 2011). However, attention should also be paid to the N, P and K ratio of fertilisers in relation to the type and amount of manure that is applied to the fields and crop requirements (FS63 Turtola, 2010).

Application rates and forms of applied nutrients should fit with the crop requirement and the nutrient status of the soil; the placement and the timing of manure and fertiliser spreading are also important. Crops with large inter-row distances and/or a restricted root distribution, and crops which grow fast after fertilisation, show improved yields when fertiliser is applied in bands rather than broadcast. Depending on the crop, the recommended amount of P can be reduced appreciably (up to 75%) when P is band applied rather than broadcast (Van Dijk and Van Geel, 2012). The yield improvement is due to better nutrient availability; in addition, due to the increased uptake efficiency the nutrient losses are smaller (FS08 Delgado, 2010; Hofman et al., 1992). There are many options to deliver fertiliser nutrients in such a way that they meet the nutrient needs of different crops during the year. Strategies that are being increasingly used by farmers and should be encouraged still further are split nutrient fertiliser applications, and better timing of fertiliser applications or controlled release fertiliser, which result in more efficient nutrient uptake by plants.

Under the EU Nitrate Action Programme, EC Member States are obliged to restrict the period in which it is permitted to spread manure or fertiliser. To date, the closed period for manure and fertiliser application differs between the Member States.[1] Moreover, high nutrient losses can occur even during closed periods. A good agricultural practice guideline could be to avoid applying manure and fertiliser before predicted heavy or prolonged rainfall events (FS15 Garnier, 2010; FS16 Garnier and Harris, 2010; FS61 Taylor, 2010; FS81 Garnier et al., 2011), or to make it mandatory for the manure to be injected or ploughed in directly after application (FS25 Haygarth, 2010). This could affect the manure handling on the farm (FS24 Haygarth, 2010; FS27 Haygarth, 2010; FS44 Newell Price, 2010; FS59 Taylor, 2010) and the manure storage capacity required (FS26 Haygarth, 2010; FS50 Newell Price and Morvan, 2010; FS62 Taylor, 2010; FS83 Newell Price et al., 2011).

For farms with a large nutrient surplus, such as intensive livestock farms, additional specific measures should be taken. Total P in fodder often exceeds animal requirements and can be reduced to decrease the P content in excretions (FS80 Van Krimpen and Jongbloed, 2010). Reducing total P in animal diets will increase the manure N:P ratio. It will contribute to reducing a surplus in the P balance of a farm or region, and decrease the need for manure transport or processing. The main options for reducing P content in excretion are (Esmaeilipour et al., 2012; FS01 Bannink, 2010; FS80 Van Krimpen and Jongbloed, 2010; Maguire et al., 2005):

- feed in accordance with the growth phase of the animal;
- add phytase to feed (poultry and pig) to increase the ability to digest phytate-P;
- feed with a lower content of phytate-P;
- use or produce feed that is more digestible; and
- maximise the use of available plant phytases to predigest phytate-P before feeding feed ingredients to the animals.

As a consequence of applying the options mentioned above Jongbloed and Lenis (1998) showed that in the Netherlands total P in feed for growing–finishing pigs fell by over 2.5 g/kg (33%) from 1973 to 1996, which led to P excretion falling from 1.62 to 0.67 kg P/pig. The introduction of microbial phytase in the last decade of the 20th century stimulated the further reduction of P excretion. Furthermore, it is possible to remove phosphates from phytate-rich fodders, thereby making it possible to reduce the phosphate intake of pigs without adversely affecting their health and growth (Meesters et al., 2011).

A farm P surplus should be transported to neighbouring farms or to nearby less intensive agricultural areas (Lopez-Ridaura et al., 2009). Sometimes it is worthwhile separating a manure surplus into a liquid fraction (with high N and low P content) and a more solid fraction (high P and low N). The liquid can be applied to the land locally, and then only the more nutrient-rich solid fraction needs to be transported. Such an approach can be especially successful on a livestock farm with sufficient land of its own (Schröder et al., 2009). Instead of being transported over a long distance, the manure can be collected and processed in the region – e.g. via digestion (biogas production) – or the solid fraction can be incinerated to produce energy. Furthermore,

[1] http://ec.europa.eu/environment/water/water-nitrates/index_en.html.

**Table 2**
Mitigation strategies at field scale.

| Strategy | Aim | Measure | Factsheets |
|---|---|---|---|
| Change soil management | Avoid transport of particles or particulate P | No tillage/direct drilling: leaving more than 30% of the soil covered with plant residues or undisturbed stubble | 3, 69 |
| | | Shallow cultivation: Soil tillage to <10 cm depth. No inversion | 73, 76, 71 |
| | | Contour ploughing | 65 |
| | | Switch from autumn tillage to spring tillage | 74 |
| | | Reduce soil compaction and improve soil structure | 68, 72, 75, 66, 67 |
| | Avoid leaching of dissolved P concentrations in soils | Conventional ploughing or interspersing periods of ploughless tillage with conventional ploughing | [a] |
| | | Add chemical compounds to the soil to bind soluble P | 5, 79 |
| | Reduce nutrient budgets and increase soil storage capacity by extensification and agro-forestry | Introduce crop rotation and include more years of grass or develop mixed (perennial and annual) cropping systems | 21 |
| | | Set-aside for several years | 21 |
| | Avoid transport of particulate P in tramlines | Tillage to avoid tramlines | 78 |
| Change crop management | Avoid erosion and reduce surface runoff | Grassland instead of arable crops or grow deep-rooting crops | 52 |
| | Change cropping system | Introduce crop rotation and include more years of grass or develop mixed (perennial and annual) cropping systems | 21, 33 |
| | Avoid leaching | Apply catch crops (and harvest the products) | 51 |
| | | Crop production without fertilisation (P mining) | 4 |

[a] Ulén et al. (2012a, 2012b), Ulén et al. (2010).

the ash is rich in P that can be reused (FS46 Newell Price, 2010; Schoumans et al., 2010). Since over 80% of all P that is mined worldwide as rock is used in agriculture (in feed and fertilisers), this can contribute to sustainably closing the terrestrial P cycle. A more drastic step is to set a maximum permissible livestock density (as in Denmark) or animal production level. In general, the economic and social impacts of such severe measures make it difficult to restructure agriculture in order to create a regional/national N and P balance that reduces the risk of surface water pollution (Senthilkumar et al., 2012a, 2012b).

### 3.2. Field system: soil and crop management

Soil and crop management affect the nutrient efficiency and nutrient losses directly, and many modifications to management have been suggested as mitigation options against P losses (Table 2).

The main purpose of soil management is to improve the soil's production potential for a certain crop. A number of soil management methods are available: they can be divided into soil tillage methods and soil amendments. Examples of soil tillage methods are direct drilling (no-ploughing and only minor disturbance of the topsoil), shallow cultivation and ploughing (FS02 Bechmann and Krogstad, 2010; FS77 Ulén et al., 2010). Examples of soil amendments are organic matter (FS66 Ulén et al., 2010) and chemical additives (FS05 Chardon and Dorioz, 2010; FS79 Ulén et al., 2010).

Cropping systems without ploughing are attracting much attention in Europe, for economic reasons (such as reduction in labour and energy consumption) and because of their potential to improve soil (Holland, 2004). In cereal cropping, soil tillage contributes to increased soil erosion risk, and thereby also to the risk of particulate P (PP) losses (Lundekvam and Skøien, 1998). The risk furthermore depends on slope and soil texture, with silty and low organic content soils being more vulnerable to soil erosion than clayey and sandy soil types.

The main soil tillage mitigation options are:

1. No tillage/direct drilling: Tillage leaving >30% of the soil covered with plant residues,
2. Shallow cultivation: Soil tillage to <10 cm depth without inversion,
3. Ploughing: Soil inversion at 20–25 cm depth; and
4. Contour ploughing.

For some sites, deeper and more intense soil tillage increases the erosion risk. In areas where the soil erosion risk is highest during autumn and winter, important options to mitigate soil losses in arable cropping systems are direct drilling in autumn (FS03 Bechmann et al., 2010) or leaving stubble undisturbed instead of mouldboard ploughing in autumn. In these systems, crop residues and an intact root system may trap and retain soil particles that would be removed by erosion. Furthermore, loosening soil by tillage will increase the risk of mobilisation of soil particles. However, improving aggregate stability by adding plant material will increase biological activity, including earthworm activity, and reduce risk of erosion. The effect of soil tillage on phosphorus losses via surface runoff is related to the soil erosion processes, but the relationship between soil tillage and losses of P through leaching is much more complex. Here, in some cases ploughing may destroy macropores, thus hampering the transport of particles and dissolved substances via percolating water (Ulén et al., 2012).

In a soil tillage system with direct drilling or shallow cultivation, the topsoil is not inverted. This has great potential for reducing soil erosion and PP losses from unstable, erodible clay loams, silty and clay soils under cereals (FS73 Ulén et al., 2010; Grønsten et al., 2007). Similar results have been reported for direct drilling of other crops, e.g. sugar beet (Strauss and Smid, 2004). However, if leaving plant material on the soil surface and surface application of P fertiliser are combined with no-till for a succession of years, P accumulates in the topsoil, increasing the risk of dissolved reactive phosphorus (DRP) losses in surface runoff (FS75 Ulén et al., 2010). In a Canadian study Gaynor and Findlay (1995) reported higher losses of DRP after reduced tillage compared to conventional ploughing. They suggested that this was a result of P enrichment of the soil surface. Studies from the northern Mississippi in the US concluded that even though total P (TP) losses were considerably reduced when soil tillage was omitted, no-till DRP losses were eight times higher than with conventional ploughing (McDowell and McGregor, 1984). As DRP has a higher ecological impact than PP due to its higher bioavailability in the short term, direct drilling should preferably not be practised on areas where DRP losses from the soil are large, whether from surface water runoff or from leaching via preferential flow through the soil profile and/or tile drainage. However, since PP can at least partly be released and taken up by biota on longer term (Barko and Smart, 1980; Bole and Allan, 1978; Sharpley, 1993; Uusitalo and Yli-Halla, 1999), a balance has to be found between reducing PP loss and (the risk of) increased DRP loss.

In shallow cultivation, there is no inversion of the topsoil. The soil is usually tilled to a depth of 5–10 cm and remains covered with some crop residues. Based on field experiments reviewed in Scandinavia (Ulén et al., 2010), it can be concluded that, compared to ploughing, shallow cultivation reduces erosion and PP losses, though less so than direct drilling (FS76 Ulén et al., 2010). However, Nordic plot experiments showed large variation in the effect of shallow cultivation in autumn on soil losses between sites and years (Koskiaho et al., 2002).

Ploughing involves inversion of the topsoil, for instance by a mouldboard plough. In areas with a high runoff risk during autumn and winter, an important method for reducing erosion and PP losses is to plough in spring instead of autumn for a spring crop (FS74 Ulén et al., 2010). On erodible soils, spring ploughing rather than traditional autumn ploughing has been shown to decrease soil erosion by up to 80% (Lundekvam, 2007) and decrease TP losses by 60–80%. On less erodible soils, the effect on TP losses is smaller and may even be negative in surface runoff (Kværnø and Bechmann, 2010). Additionally, as for direct drilling, when soils are spring-ploughed, losses of DRP may be higher than during winter. The negative effect has to be weighed against the positive effect of reduced erosion and PP loss. On heavy clays, crop establishment in spring may suffer, because if these soils are not tilled in autumn their structure deteriorates.

Ploughing, cultivating and seed drilling perpendicular to slopes and along contours, is applicable for all areas at risk of surface runoff and erosion via rills (FS65 Ulén et al., 2010). In undulating fields, eroded particles in surface runoff may be trapped by grass and can settle in depressions. Therefore, grassed waterways (channels of grass within arable land, constructed in order to concentrate, control and slow down water flow) in combination with contour ploughing are desirable (FS65 Ulén et al., 2010). Another way to reduce erosion risk is reducing slope length by creating a grass buffer along the contour to break the slope (FS70 Ulén et al., 2010).

Besides alternatives for conventional tillage practices, special attention should be given to improve soil structure, especially on compacted soils (Batey, 2009; FS68 Ulén et al., 2010; FS72 Ulén et al., 2010) and in tramlines (Bailey et al., 2013; Deasy et al., 2009; FS78 Ulén et al., 2010; Silgram et al., 2007; Withers et al., 2006), because under such conditions high P losses by runoff and erosion can occur. Tramlines can be treated with a cultivator fitted with a ducksfoot tine used to disrupt the compacted surface of the tramline to circa 6 cm depth (Bailey et al., 2013).

Crop management encompasses the farmer's decisions on which crops and varieties to grow, when and how to sow or plant, and how to handle crop residues, pests and weeds. Principally, crop management options for reducing the erosion risk aim at: (1) increasing water infiltration to reduce runoff volumes and erosivity; (2) strengthening topsoil resistance against detachment of soil particles; and (3) protecting the soil surface against erosive forces with plant or residue cover (Govers et al., 2004). Cover crops can be grown in the interval between main crops to protect the soil surface against losses through erosion and surface runoff. A number of crop management options reduce the risk of nutrient mobilisation by surface runoff and soil erosion; most of these effects depend crucially on the crop type and growth stage as well as on crop rotation (Morgan, 2005).

Crop management strategies for reducing leaching losses of nutrient mainly involve associated catch crops: N is immobilised by being taken up by the catch crops in the interval between the growing seasons of two main crops (FS51 Rubæk and Jørgensen, 2010). The pronounced differences in cycling of soil P and N necessitate a different approach when using crop management to reduce P losses, because plant uptake of P reduces the concentration of dissolved P only in the immediate vicinity of the root and this reduction is counteracted by desorption of a small fraction of the P retained on the soil particle surfaces. In other words, the soil solution P concentration is buffered by the large amount of P accumulated in the solid fraction. The concentration of dissolved P in soil solution is therefore less dynamic than for N, and even a well-established crop cannot significantly lower the P concentration in the soil solution and keep the concentration low. Reducing the leaching of dissolved P via crop management is possible only through long-term P mining (Delorme et al., 2000; Eghball et al., 2003; FS04 Chardon, 2010; Koopmans et al., 2004a, 2004b).

### 3.3. Landscape and hydrological system

Despite the strong link between farm and field management and nutrient losses to surface water, losses at a catchment scale cannot be considered merely as the sum total of losses from farms or fields because of agricultural and environmental features and spatial and temporal constraints and interrelations. Diffuse agricultural pollution occurs in a complex and hierarchical matrix of fields and natural areas interacting with a hydrographical network (Forman and Godron, 1981; FS18 Gascuel-Odoux and Dorioz, 2010; Wang et al., 2004). As some areas in agricultural landscapes are nutrient sources and others are sinks, the catchment as a whole may have a net buffer capacity (Viaud et al., 2004).

A reduction in nutrient losses at catchment scale may be achieved by four means: (a) storage and trapping of water and/or sediment and nutrients within a buffer zone along watercourses; (b) water and dissolved nutrient uptake by vegetation and biota; (c) biogeochemical transformation (such as sorption, denitrification) and (d) dilution (by groundwater). The various management strategies that can be implemented to reduce nutrient losses at a catchment scale can be grouped into three main categories: field water management, land use changes and landscape management (Table 3).

Water-related measures mainly focus on changing the pathway length and the rate of the water flow from the source to receiving waters by changing the drainage conditions. The nutrient concentrations in the water flow will be reduced due to adsorption (P), denitrification ($NO_3$) and physical processes (steric effects; sedimentation) which

**Table 3**
Mitigation strategies at catchment scale.

| Strategy | Aim | Measure | Factsheets |
|---|---|---|---|
| Water management | Change runoff flow by blocking or reducing overland flow | Create ponding systems | 53 |
| | | Construct grassed waterways | 52 |
| | | Create sediment boxes | 56 |
| | | Improve surface irrigation | 9 |
| | Avoid subsurface losses through leaching | Remove trenches and ditches or allow to deteriorate | 58 |
| | | Install drains | 10, 55, 64 |
| | | Controlled drainage systems | 7, 54 |
| | | Let drainage water irrigate meadows | 57 |
| Land use management | Improve location of sinks and sources by changing agricultural use patterns | Alternate grassland and arable land Avoid certain crops in hilly areas | [a] |
| | | Locate crops with high nutrient uptake on bottom lands | 21 |
| | Protect very vulnerable areas by nature development | Afforest or set aside agricultural land | [a] |
| Landscape management | Reduce direct losses from farm yards | Minimise volume of dirty water produced and collect farm yard runoff | 47, 44, 27 |
| | Reduce direct losses from livestock | Prevent contact with surface water: fences, bridges | 11, 19 |
| | Reduce surface runoff and erosion from field to field within the catchment | Re-site gateways and paths: trails, roads, controlled access for livestock and machinery | 12, 19, 20 |
| | Intercept nutrients from runoff, erosion and subsurface losses to waters | Vegetated buffer strips | 17, 22 |

[a] Gascuel-Odoux et al. (2011).

also influence the water flow rate. However, it is important to note that changing the water flow will also change soil moisture, and can thus impact on the chemical (sorption and precipitation) and biological processes such as mineralisation or immobilisation of organic matter and nitrification/denitrification. As a result, pollution swapping can occur, e.g. if the groundwater level falls, P leaching will decrease but often the $NO_3$ concentration will increase because in dry soil less denitrification occurs. Water management measures aimed at reducing nutrient loss to surface water can be grouped into two strategies (Table 3), related to the main pathways of water flow; (a) overland and (b) subsurface.

Blocking or changing *overland water* flow can be achieved via various constructions, such as ponding systems (constructing bunds along ditches), grassed waterways and sedimentation boxes. It is important to slow down the flow rate and to increase the length of the flow path, as this will facilitate sedimentation (in lower parts of the field) and the infiltration of dissolved P. By retaining the water, sediment ponds at field edges will allow suspended material to settle and increase denitrification, resulting in reduced nutrient losses to surface water (Brown et al., 1981; FS53 Schoumans, 2010). The grass in grassed waterways acts as a filter, absorbing or taking up some of the chemicals and nutrients in runoff water (FS52 Schoumans, 2010).

During runoff events in an undulating landscape, surface runoff tends to flow preferentially towards depressions, creating rills that may result in gully erosion. To prevent this, wells for surface water can be dug in the depressions, backfilled with material that helps to collect the water efficiently. The aim is to avoid a concentrated flow of water from having so much energy that it will cause erosion in the depression (Aspmo, 1989; FS56 Schoumans, 2010).

In areas where nutrient losses are mainly caused by *subsurface drainage water* (e.g. in flat areas with shallow water tables; Chardon and Schoumans, 2007) water management measures can be taken to reduce P losses. The most important measures are: allow field drainage systems to deteriorate, remove trenches and ditches or install drains. The removal of trenches and ditches will reduce the opportunity for groundwater transporting nutrients to enter surface water: the water has to flow a longer distance underground to reach other surface water bodies (ditches, brooks, rivers) and en route the buffer capacity of the soil filters out the P. By allowing field drainage of a trench or ditch to deteriorate such an effect will be achieved at longer term. However, during the deterioration of the trench (or ditch) the water quality improves if the residence time of the water in the ditches increases (FS58 Schoumans, 2010; Olli et al., 2009; Powell et al., 2007). Since trenches and ditches are constructed in wet areas and are necessary to ensure drainage of surplus precipitation, the impact of this measure will be that high groundwater levels will occur more frequently. Therefore, such measures should be evaluated at least at sub-catchment scale. Furthermore, more frequent wet conditions can have a negative effect on the agricultural productivity of the fields.

Subsurface transport of P to trenches and ditches can also be reduced by installing artificial drains in undrained fields, because the groundwater will no longer reach the P-enriched layers and peak discharges will no longer occur from these layers (FS55 Schoumans, 2010; FS64 Turtola, 2010). Even greater reductions can be obtained via controlled drainage systems or deep controlled drainage systems (FS54 Schoumans, 2010). However, in cracking soils, artificial drainage increases nutrient losses because macropores connect the nutrient-rich topsoil with drain lines through which water and dissolved/suspended nutrients are transferred to water (FS07 Delgado, 2010). Artificial drainage is not recommended for peat areas, because when the groundwater level falls peat will oxidise and more P will become available than under anoxic conditions (FS55 Schoumans, 2010).

P losses via leaching can also be reduced by filling a mole channel, by mixing backfill, or enveloping a tile drain with a reactive material such as iron or aluminium hydroxides or oxides, or lime (CaO), depending on the soil pH (Groenenberg et al., 2013; Hanly et al., 2008; McDowell et al., 2008). A second option is to stop the water flowing through tile drains and make it flow overland, irrigating meadows or riparian areas. The general idea is that water from tile drains that is polluted with particulate and dissolved nutrients is filtered by a meadow or a riparian area (FS57 Schoumans, 2010; Stutter et al., 2012, 2009; Tanner et al., 2005).

Land use measures can be divided into three groups (Table 3). Changing the agricultural land use pattern by reallocating the land use or changing the crop can increase the buffer capacity of the catchment by rearranging the location of sinks and sources and decreasing the water flow (FS21 Gascuel-Odoux et al., 2009). Reallocating crops at farm scale, or collectively planning the land use for the entire catchment may cause changes in soil surface conditions (vegetation cover, roughness, soil structure) and therefore infiltration capacity downslope may increase, thereby reducing erosion losses (Souchere et al., 2005) and increasing nutrient uptake from the soil water. Vegetation management of buffer strips is also a critical factor in manipulating buffer conditions to remove stored P, increase buffer lifespan and prevent P being lost through leaching. Enhancing the plant uptake of the mobilised nutrients will have two benefits: removing pore water nutrients that would otherwise leach out (Lee et al., 2000) and providing a possible removal pathway via vegetation harvesting (FS22 Gascuel-Odoux et al., 2010).

Other important options available within the catchment are changing to a different agricultural land use (e.g. agro-forestry or extensification) or to a non-agricultural use. Unfortunately, catchment experiments cannot be easily performed to evaluate the effectiveness of such options, since it would take many years to test the effect of crop locations. The effectiveness of such options at catchment scale is mainly evaluated by hydrological and nutrient modelling that includes a detailed description of the agricultural landscape and its functioning. The modelling shows the effectiveness of the options on surface runoff and erosion, and thus on PP losses (Aurousseau et al., 2009; Cerdan et al., 2002; Jetten et al., 1996).

Interfaces between farm water and surface water regulate the surface flow connectivity between farm infrastructure, where wastewater is produced, and surface water. Outdoor manure heaps should not be sited over field drains or close to a watercourse because effluent from manure can contain very high concentrations of N and P (McDowell et al., 2005), and can reach surface water either directly or via the drains. Ideally, the heaps should be sited on concrete so the effluent can be collected; this is already prescribed by regulations in e.g. Switzerland (FS14 Garnier, 2010; FS26 Haygarth, 2010; FS44 Newell Price, 2010; FS60 Taylor, 2010). Although they are not considered as “agricultural”, farm septic tanks can also contribute to P losses (Withers et al., 2011) and to avoid these losses, farms should be connected to the sewer system. To attenuate the transfer of nutrients from farmyard to subsurface it is desirable to: 1) minimise the volume of wastewater produced on the farm or regulate the storage facilities; 2) separate runoff from farm roofs from runoff from farmyards and roads; 3) site waste storage facilities far from surface water drainage networks; 4) introduce buffers (such as a farm pond or filter strips) (FS47 Newell Price, 2010).

Direct access of livestock to a stream can significantly damage river banks and local aquatic ecosystems and can also lead to direct pollution with organic and inorganic nutrients and faecal contaminants. Mitigation options aim to introduce physical barriers between grazing animals and surface water (Meals, 2004), to reduce cattle grazing (FS34 Jørgensen et al., 2009), or avoid high animal density in areas very close to watercourses (FS11 Dorioz and Gascuel-Odoux, 2010; FS12 Dorioz et al., 2010). This can be achieved by: 1) fencing off rivers and streams and leaving a protective zone of a minimal width (1 to 3 m) along the stream network; 2) creating designated livestock crossings of streams and rivers by means of bridges or paths; 3) re-siting pasture gateways away from watercourses and, if possible, upslope from the watercourse. These three measures are applicable to grazing land and livestock farms. Special attention should be given to parts of fields where animals congregate, such as drinking troughs, feeding places and shelter areas near watercourses. Such places can be considered as "pollution hotspots" on the farm, as excreta will frequently be deposited there. Trampling by livestock will increase soil compaction and runoff potential, creating areas at greater risk of P loss to water (Tunney et al., 2007). It is therefore advisable to move drinking and feeding places at regular intervals, in order to reduce uneven loading and physical damage (Wilcock et al., 1999).

Interfaces located along permanent streams and ditches control direct inputs to surface water. If they have a buffer effect, they are called riparian buffers. Two types of such riparian buffers can be distinguished in terms of their hydrological conditions: 1) unsaturated: vegetated buffer strips, and 2) saturated: riparian wetlands or wet meadows. Riparian strips are generally considered to offer efficient protection against TP: much of the P transported to watercourses is bound to particles and the main physical process occurring within the buffer strips is sedimentation. Much is known about constructed vegetated buffer strips, as they have been widely studied and have been tested and calibrated with models. Hoffman et al. (2009) have reviewed the efficiencies of riparian buffers for TP retention and report that the retention efficiencies varies between 41 and 92%. The effectiveness of riparian buffers depends on many factors, like the nature of contributing sources, slope, soil type, vegetation and local flow and hydrological soil conditions. Especially, under drainage conditions with tile drains the effectiveness of riparian buffers is relatively low, since buffers mainly target the surface delivery pathway. There is uncertainty about the true effectiveness, because most of the data used in the assessment were from short-term (even single rainfall event) plot studies (Liu et al., 2008; Schmitt et al., 1999).

In wet, shallow groundwater systems, small riparian wetlands are often created in the lowlands of headwater catchments, scattered in the rural landscape and often associated with wet meadows. They are the inescapable interface between groundwater, for which they are the non-point outlet, and the water bodies, and are often the interface between intensively cultivated hill slopes and plateaus and the water bodies. Wetland efficiency at reducing nitrate pollution has been extensively studied in natural (Fisher and Acreman, 2004; Machefert and Dise, 2004) and artificial wetlands (Kadlec, 2009; Vymazal et al., 2006). Wetlands can also trap particles and thus TP. But under anaerobic conditions, reductive dissolution of ferric compounds carrying P can be an important mechanism of seasonal DRP release (Khalid and Patrick, 1974).

Finally, field boundaries can be helpful as interfaces to control the connectivity of surface runoff and subsurface flows from plot to plot (FS19 Gascuel-Odoux and Dorioz, 2010). These interfaces are very diverse: e.g. simple field boundaries or margins, vegetated to various degrees; hedges and hedgerows, all acting as a filter for surface runoff. The effect of the boundary on subsurface flow depends on the local or hill slope conditions and on the vegetation type (hedges, trees, grass) (FS20 Gascuel-Odoux and Dorioz, 2010). The root systems of plants on field boundaries can take up water and chemicals from shallow groundwater (the root systems of trees extend to a depth of several metres), and thus may affect the subsurface flow. Gateways are effectively gaps in the field boundaries and are critical areas for surface runoff in all kinds of fields, not only in pastures. At the landscape level, gateways, livestock and tractor pathways form a network of preferential flow pathways that may be hydrologically connected to the water course. Consequently, re-siting gate and livestock and tractor pathways is a simple way to decrease local and global hydrological connectivity, thus reducing pollution via these preferential flow pathways (Cuttle et al., 2007).

### 3.4. Ecological system

Surface water management is commonly deployed to retain sediment and nutrients transported to surface waters from land. The basic principles on which it is based include hydraulic retention (i.e. increasing the residence time of water in surface water systems), interaction between sediment and water, and biogeochemical processes such as denitrification and sorption. Surface water management also strives to re-establish lost biomes such as wetlands and lakes and to improve the ecological quality of streams and rivers (Kronvang et al., 2011). Three main types of surface water management can be distinguished (Table 4): (a) River maintenance and river restoration, (b) Lake re-establishment (lakes that have been drained for agricultural purposes) and (c) Wetland restoration and constructed wetlands.

In order to facilitate fast drainage of water from fields, streams have traditionally been maintained by cutting back the vegetation several times per year and by removing gravel, stones and other physical barriers that impede water flow. Streams have thus been kept in canal-like state with a low capacity of nutrient retention (FS37 Kronvang, 2010). If maintenance is limited or abandoned, the physical conditions of the streams will change (FS23 Grizzetti and Bouraoui, 2010; Sand-Jensen, 1997). Aquatic and bank-side vegetation will quickly narrow the channel profile and promote denitrification and the deposition of sediment and PP (Kronvang et al., 2005; Veraart et al., 2011). The morphological and substratum diversity of watercourses that have been straightened and channelised in order to drain agricultural land can be improved in various ways. Active restoration is a quick and direct way of achieving the required physical improvement of the channel and restoring the interaction with the adjacent riparian areas through inundation during high flow periods. A study of 13 catchments in Western Europe found that the main factor for nutrient retention in the surface water system is residence time. This can be increased by restoring floodplains and reconnecting inundation areas (FS39 Kronvang, 2010; FS41 Kronvang and Lo Porto, 2010). As an example, increasing the water travel time in a catchment by 50% may result in 15–20% more N and P retention (De Klein and Koelmans, 2011).

The P loading of lakes is in the form of dissolved or PP. PP will be deposited in lakes as a result of sedimentation, whereas part of the DRP will be taken up by the biomass of phytoplankton or macrophytes produced in the lake. The amount of P that is recirculated and immobilised depending on local conditions in the catchment and the nature of the re-established lake. The factors and processes involved are: i) periods of anaerobic conditions in lake sediments; ii) ratio of iron to P in sediments; iii) residence time of water in lakes; iv) lake depth and v) inflow of P (Hejzlar et al., 2007; Kronvang et al., 2005). The most important factor for nutrient retention in lakes is the residence time for water: the longer the residence time, the greater the

**Table 4**
Mitigation strategies in aquatic ecosystems.

| Strategy | Aim | Measure | Factsheets |
|---|---|---|---|
| River maintenance and river restoration | Increase nutrient retention capacity | Limit cutting of vegetation and reduce regular removal of gravel and impediments to flow | 37 |
| | | Re-meander, restore flood plains and reconnect inundation areas | 39, 41 |
| Lake rehabilitation and restoration | Reduce the P concentration of lake water | Control P inlet and prolong residence time of water | 38 |
| | | Apply chemicals to bind P released from sediments | 43 |
| Wetland restoration and constructed wetlands | Retain nutrient loss from upstream fields in wetlands | Create wetlands in agricultural areas with substantial P losses | 39, 40, 41, 42 |

retention of nutrients (FS38 Kronvang, 2010; Hejzlar et al., 2009, 2007; Søndergaard et al., 2003; Windolf et al., 1996).

In principle a re-established lake will start to retain nutrients from its outset. However, a period with net release of P from the rewetted soils can occur if these soils contain large amounts of "old" agricultural P and are low in iron content. Another factor that may be important is a release of P from the former terrestrial vegetation and a hydrolysis of easily decomposable organic matter in the soils submerged by the re-established lake. On a longer time scale the re-established lakes will retain nutrients like natural lakes; the retention potential for N and P can be calculated from lake nutrient models. Phosphorus-enriched sediments can release P to the water through a process known as internal loading (Søndergaard et al., 2003). As a result, the total P retention can be very variable (−265% up to +91%) as shown by Hoffman et al. (2006) for re-established Danish shallow lakes and wetlands. When sediments are contributing P to the lake, nutrient inactivation techniques called precipitation can be used to remove P from the water column and a process called inactivation can be used to retard its release from the sediments. In both cases, the aim is to prevent eutrophication or to rehabilitate water bodies that have become eutrophic due to high concentrations of soluble P, by binding the P and causing it to settle out on the lake or river bed (Svendsen et al., 1995). Aluminium, iron, or calcium salts are used to inactivate P in lake sediments. The chemical compound most commonly used to precipitate the P is aluminium sulphate (alum) $Al_2(SO_4)_3$: it is frequently used as a flocculating agent in the purification of drinking water and in wastewater treatment plants. The addition of alum helps reduce the DRP concentration in surface water (FS43 Lo Porto et al., 2010). Several studies have evaluated the effectiveness and longevity of treatments on several lakes in the USA and have concluded that in shallow lakes alum treatment effectively achieves P inactivation in most cases (Welch and Cooke, 1999). Applications in stratified lakes were highly effective and long-lasting (>80%) (FS43 Lo Porto et al., 2010).

In recent years, interest in wetland restoration programmes has been growing throughout the world (Hoffman et al., 2009; Litaor et al., 2004). Restoration of riparian wetlands is conducted on low-lying, often former organic soils that were drained at some time in recent history, usually for agriculture. They are human-made wetlands, created on areas where the physical and chemical composition of the soil had changed as a result of many years of draining and farming. In most cases, restored wetlands are established with the principal aim of retaining nutrients lost from upstream agricultural fields by denitrification, P sorption, and sedimentation (FS17 Gascuel-Odoux and Dorioz, 2010; FS40 Kronvang, 2010). The restored wetland may be fed by groundwater or surface water and be adjacent to streams and rivers, or be located in estuaries along the coast. Wetlands dominated by surface water receive both dissolved and particulate P forms that can be retained by sedimentation and sorption (Litaor et al., 2004, 2005; Sade et al., 2010) and biological uptake (Hoffman et al., 2009; Kronvang et al., 2009). Normally, restored wetlands are very effective for P sedimentation or uptake of P by vegetation. Experience with sedimentation of particulate P also shows that restored riparian wetlands can have a high retention capacity of 10–100 kg P $ha^{-1}$ inundated wetland (Kronvang et al., 2009). However, P retention is more certain for PP than for dissolved P, as some restored riparian wetlands experience a net leakage of dissolved P, due to iron-bound pools of former agricultural P in soils that is released under anaerobic conditions in amounts of up to 20 kg P $ha^{-1}$ $y^{-1}$ (Hoffman et al., 2009).

Wetlands are also constructed with the principal aim of retaining nutrients from neighbouring agricultural fields through processes such as sedimentation and sorption. Constructed wetlands are established either in small ditches and brooks or as an end-of-tile-drainpipe control. In all cases, nutrient-enriched water from fields flows through constructed wetlands for nutrient load reduction before entering surface waters downstream. The numerous configurations include: small sedimentation basins; infiltration basins with horizontal or vertical flow through the artificial substrate for sorption of P; shallow vegetative filters for storage of fine particles enriched in P and uptake of dissolved P; and small basins with material that increases the P sorption potential. The effectiveness of constructed wetlands for nutrient removal and storage is normally high, although most experience is from surface water systems in Norway, Sweden and the USA. Constructed wetlands established in brooks have been found to have an annual P retention of 1–50 g P $m^{-2}$ $y^{-1}$ of constructed wetland (Braskerud et al., 2005; FS42 Kronvang et al., 2010). Usually the absolute and relative retention performance increases concomitantly with the load. However, the P retention is more certain for PP than for dissolved P, as some constructed wetlands experience a net leakage of dissolved P (Hoffman et al., 2006). Depending on the rate of wetland terrestrialisation constructed wetlands may last for 10–50 years before they need to be dug out again. In the long run, the P-binding capacity of constructed wetlands may also become exhausted and therefore it is important to maintain them by cutting vegetation and removing sediment.

## 4. Conclusions

The list of mitigation options that COST Action 869 yielded was produced by a diverse group of scientists from 30 countries in Europe, and therefore from a range of agricultural and pedoclimatic areas. For the four systems defined in Fig. 1, the following main conclusions can be drawn:

(I) Nutrient & Livestock Management
- Nutrient management strategies, such as agro-environmental recommendations, are useful tools for setting up a more sustainable agricultural management practice.
- With respect to the selection and placement of fertilisers (NPK), the surplus of all components has to be minimised in relation to nutrient uptake of the crop and the composition of the applied manure.
- Reducing the N and/or P content of animal feed, or increasing the uptake availability, can greatly reduce a farm's nutrient surplus and the risk of losses due to over-application.
- Transport of manure from a farm with a nutrient surplus is costly; manure separation can reduce these costs if the liquid fraction with a relatively low P content can be applied locally.

(II) Soil & Crop Management
- Direct drilling and shallow cultivation reduce erosion and total phosphorus losses from high risk areas more than ploughing that results in soil inversion.
- Compared to autumn ploughing, on average a spring tillage reduces erosion risk and P losses during winter.
- Accumulated surplus P fertiliser and organic matter on or near the soil surface pose a risk of P release and therefore increased DRP concentrations in surface water.
- Cover crops can protect the soil surface against nutrient being lost through surface runoff and erosion.
- P losses related to the high P status of a soil can be reduced by using crops to mine P from the soil over several years.

(III) Water, Land Use & Landscape Management
- Controlling the water flow from fields to surface water is one of the most important options to reduce nutrient pollution of the surface water, because nutrients from agricultural land are mainly transported to surface water via flowing water.
- The P concentration of infield overland water flow can be directly changed by creating ponds or grassed waterways or by installing sediment boxes.
- Subsurface nutrient losses from agricultural land can be changed by changing the drainage system (trenches, ditches and tile drains). Controlled tile drainage systems are especially

effective at changing the depth of the water discharge to surface waters.

- Buffer areas act by storing and trapping water and/or sediment and nutrients, the uptake of dissolved nutrients by vegetation and biota, biogeochemical transformation (such as sorption and denitrification) and dilution. To be effective, they have to be properly structured and managed.

(IV) Surface Water Management

- Surface water management to increase nutrient removal and storage processes is often applied in River Basin Management Plans because it is cost-effective for both N and P.
- River maintenance and restoration can assist in increasing nutrient retention and improving stream ecology.

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