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The effect of buffer strip width on cost efficiency: a Swedish case study

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ABSTRACT

The objective of this study is to analyze the cost-efficiency of buffer zone schemes in Sweden in the context of European Directives, national agri-environmental support programmes and environmental targets to be achieved. The study includes a comprehensive overview of the scientific literature on the effectiveness of buffer zones in retaining pollutants and a modelling exercise. The exercise evaluates the effect of buffer zone width on reducing phosphorus losses using the model ICECREAM DB and Swedish data. The study then analyzes how the parameters that influence the effectiveness of different widths may impact the cost-effectiveness of policy alternatives. What emerges is that the width of a buffer might not be that influential as other site conditions when looking at the cost-efficiency in reducing the load of phosphorous from agricultural fields. Eventually, a more down-scaled and differentiated payment scheme for buffer zones (of different designs) based on more localized and easy-to-establish parameters (such as climate and soil type, but also the load of P and reduction targets) would produce a more cost-effective approach.

Keywords: buffer zones, width, cost-efficiency, Sweden

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1. Introduction

"A buffer zone is a piece of land where there are farming restrictions with a view to protecting water. Such restrictions can include: no fertiliser applied, no plant protection products applied, no cultivation, no livestock grazing allowed, no farming at all is allowed, particular plants or types of plant must be grown or allowed to grow" (Rouyer, 2010).

A buffer zone is generally seen as a multiple-objective structure identifiable in riparian areas whose main purpose is to address problems that mainly arise from diffuse source pollution and adversely affect the quality of water bodies and jeopardize the overall riparian ecosystem functioning. Other side benefits are identifiable in controlling environmental pressures such as lack of water retention capacity, bank erosion, loss of in-stream and terrestrial biodiversity, release of pathogenic micro-organisms excreted by livestock and organic pollutants from manure.

The awareness of buffer zones importance finds acknowledgment in the increasing interest by stakeholders and traditional user rights, which eventually gives the buffer zone thinking a more socio-economic perception and value (Ebregt and Greve, 2000). The rationale behind such belief is that the social marginal benefits provided by installing a buffer zone (prevention of water quality degradation and increased biodiversity) weigh more than its social marginal costs (loss of revenue due to the conversion of productive land into a buffer zone). When a buffer is implemented, a scheme of subsidies is generally endorsed in order to compensate farmers. Due to the inner difficulties in determining values of non-market goods (such as the environmental services provided), the amount of the compensation is usually quantified upon the income foregone.

In order to achieve the best compromise among such variables, in terms of having the most cost-efficient buffer zone design, a cost-efficiency analysis (CEA) can prove a viable economical tool. A CEA is defined as an applied appraisal technique that assesses and provides a ranking of alternative measures on the basis of their costs and effectiveness for achieving environmental objective (Balana et al., 2011). Such ranking of measures is often based upon several parameters, but often the trade-offs between the many indicators are not very clear. Its importance has been also suggested in the Water Framework Directive (Article 11[1] and Annex III) for the developing of the river basin management plans[2] (RBMPs). Referring to a study carried out by Balana (2011), the methodology for performing a CEA consists of 4 major elements: 1) review of potential environmental pressure and impacts; 2) estimation of the potential effectiveness of measures; 3) prediction of the costs of the measures and 4) assessment of the overall cost-effectiveness.

An important parameter in assessing costs and benefits is the size of the buffer zone. The larger the area of the buffer zone, the higher the potential loss of income as well as the higher potential benefits due to the greater effectiveness of the buffer zone in reducing the loss of harmful substances to the receiving water body.

The ultimate objective of this study is thence to analyze the cost-efficiency of different buffer zone designs as well as evaluate the importance of the variable "width" within different environmental

[1] *"Identification of cost-effective programme of measures"*

[2] Member States are clearly required to set water quality standards and identify the most cost-effective set of mitigation measures in order to achieve them

Swedish contexts. The first section below is an overview of the scientific literature on the effectiveness of buffer zones. This is followed by a summary of environmental policy at the European level, schemes in Sweden and neighboring countries. The following section presents a field scale model which is used to evaluate the effect of buffer zones in the reduction of P under Swedish environmental conditions. This section begins with a description of the model and followed by a study of the effect of individual modeling parameters on their P reduction effect on buffer zones. This section concludes with an evaluation of the impact on cost effectiveness of one of these parameters, the width of the buffer zone. The thesis ends with conclusions based on the results from the modelling exercise.

2. Background

The concept of installing buffer zones is not something totally new. The idea evolved already in the early 1970s from the intention of better protecting core areas within conservation areas or conservation area as a whole by minimising the negative impacts of human activities on nature (Ebregt and Greve, 2000). Subsequently, in the 1980s, scientists began to realise that receiving water quality was not often closely related to the quality of surface and groundwater as it left the "*edge of the fields*" (Correll, 2005). Nowadays one of the most impacting environmental pressures on water resources comes from agriculture[3]. Enhanced levels of nutrients in the water compartment usually contribute to increased primary production and induction of phenomena such as algal blooms, increased water turbidity, oxygen depletion and fish kills (Hoffmann et al., 2009).
Buffer zones are generally most effective in intercepting surface runoff from fields other than sub-superficial flows. Through processes of deposition, absorption, plant uptake and denitrification, riparian buffer zones interact and reduce the load of sediments, organic matter and nutrients (e.g. nitrates and phosphate) which might affect the water quality and the overall ecological value of receiving compartments (Ducros and Joyce, 2003). They can also serve as spray drift and runoff control when it comes to pesticides.

The efficiency of a buffer very much depends on a combination of factors, such as runoff volume, topographic features of the area originating the runoff (e.g. slope, size, land use) as well as the type of buffer, in terms of its components, age and width (Borin et al., 2010). The presence of vegetation also contributes to increase the filtering function of the buffer itself by raising surface roughness and thus improving infiltration by decreasing flow volumes and speed. This reduces the transport capacity of runoff and encourages sediment deposition in the buffer strip (Rose et al., 2003). An other possible way to maximise the functions of a riparian buffer is to improve its design establishing a combination of different vegetation types such as trees, shrubs and grass, subsequently planted in zones parallel to the water body. Trees (closest to the stream) grow deep roots and increase bank stability, shrubs also offer a recurring root system and a long-term nutrient tank close to the stream as well as adding biodiversity and wildlife habitat and eventually grass, which slows surface runoff allowing infiltration and sediment deposition in addition to increase the organic matter content of the soil. A study dated back to 1995 by Iowa State University already provided directives on how to properly set such a system, whose functionality and potential has been again arisen in a recent workshop on the multi-benefits aspect of buffer zones (Stutter, COST

[3] EEA, 2011 (http://www.eea.europa.eu/themes/agriculture/about-agriculture)

Action 869, Scotland, April 2010). Lee et al. (2003) evaluated the efficiency in removing sediment and nutrient of a switchgrass/woody buffer compared to a simple switchgrass buffer and found out that the removal rates were significantly higher (over 20%) in all the parameters considered (total suspended solids, total nitrogen, nitrate, total phosphorous and phosphate) though a larger width was required (16.3 meters compared to a 7.1 meters). Another contribution in favour of such combined approach comes from Mankin et al. (2007): the grass-shrubs buffers studied proved significant mass reductions of sediment (99%), total phosphorous (>85%) and nitrogen (>85%) for widths varying between 8.3 and 16.1 meters. Efficiency is also hampered over time due to changes of initial conditions, thus requiring constant performance checking and a maintenance scheme over both short time scale (i.e. after individual storms and snowmelt events) and long time scale (seasonal and year-to-year monitoring).

The information available in the scientific literature concerning buffer zones is considerably large. Nevertheless, which is the optimal mixture of vegetation, which the right size and the design of a buffer, which the environmental pressure to be targeted primarily (single or multiple objectives), are still questions difficult to be answered. Besides the knowledge of cost-effectiveness of different buffer widths – a consideration which is seldom included in the literature reviews – remains vague, though much is being implemented in the national and european legislations about the concept (of buffer zones). In order to avoid possible misuses and misunderstandings of the term "buffer zones"[4], for the purposes of the present study the term is preferentially used for indicating riparian[5] vegetated buffer strips between agricultural fields and aquatic ecosystems (e.g. streams, rivers, lakes).

2.1 Literature review

Despite buffer zones flexibility in targeting simultaneously multiple environmental pressures, few studies performed a comprehensive analysis on their combined potential, whereas the majority of them focused on single functions at time (reduction of nitrogen, phosphorous or addressing biodiversity). In the following sections, the most updated information on the effectiveness of buffer zones with regard to reduction of nutrients, sediments, pesticides control and biodiversity has been collected and reviewed. Considerations of different environmental conditions and site characteristics like type of vegetation present, climate, topography, hydrology, soils and width have been laid out.

2.1.1 Nutrients

High nutrient load - a problem significanly affecting the Baltic Sea - is appointed as a main cause for eutrophication, yet some sort of disagreement remains among scientists whether either phosphorous or nitrogen contribute the most (Bergström et al., 2007). Buffer zones can target both type of nutrients, however, an optimum situation for P is not necessarily the same as for N. Such

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[4] The terms buffer zones, riparian buffer, riparian zone, buffer strip, filter strip, and vegetated filter strip have been sometimes considered as synonyms in some scientific literature. Wetlands (usually referred to as wet buffers) and buffer strips in forested areas are not contemplated in the study.

[5] Riparian zones are usually defined as natural or semi-natural ecosystems at the interface of terrestrial and aquatic ecosystems.

difference mainly depends on the physical status in which the nutrient is transported. Phosphorous tends to be present more in particulate form, thence the effectiveness in retention being influenced more by features of the buffer that enhance deposition, infiltration and decreasing of flow velocity (width in the forefront). Conversely, the retention of nitrogen, which is mostly in solute form, is more effectively achieved thanks interception by vegetation (and consequent plant uptake, microbial immobilization and denitrification). In the following sections, the processes affecting both nutrients are presented and discussed.

Phosphorous

The dynamics of phosphorus in a grass buffer strip are somehow different from nitrogen and most pesticides, because no biogeochemical transformation is able to reduce significantly the quantity of P stored within the buffer (Dorioz et al., 2006). Thus, phosphorus can accumulate in the buffer strip until its concentration is so high that the soil and vegetation can no longer contain additional P. Despite the potential unfeasibility of grass buffer strips to function as a component of a long-term strategy for phosphorus reduction, recommendations to utilise grass buffer strips to control diffuse P transfer are becoming a common practice (Dorioz et al., 2006). The capacity of storing P and keeping such function over long time is dependent on pH, organic matter content, and seasonal soil conditions, including moisture content, redox potential, and temperature dynamics.
Most of the total phosphorus (total-P) stored within ecosystems is found associated with particles (particulate-P) because of its strong affinity for the solid phase. Sorption and precipitation are considered the most important long-term P sequestration mechanisms. On the other hand, microbial and plant uptake are considered temporary P pools (Hoffmann et al., 2009).
The capacity of retention and abatement of phosphorous has been found to be pretty significant (within a range of effectiveness between 60 and 98%) according to a set of studies (Duchemin and Madjoub, 2004; Borin et al., 2005). Borin (2005) inferred that total P, which is mainly sediment-bound, is reduced while passing through the buffer due to the effect of sediment trapping. According to Lee et al. (2000), the reduction is less than proportional to sediment reduction. Generally, the effectiveness of grass buffer strips with regard to particulate-P and sediments is very similar given their close functional relationship. The situation is very different for the dissolved forms of P: although dissolved-P is not the dominant form of P in agricultural runoff (experience from a finnish case study exhibited that the proportion of dissolved P in surface runoff ranged between 17% and 45% - Uusi-Kämppä, 2005), these contrasting retention values result in large differences in the retention of total-P (between 8 and 97% according to Dorioz et al.). This large range results from the very different dynamics associated with the physical and biological processes governing soluble and particulate species moving in subsurface and surface flows. Variation in hydrologic conditions in different buffer zones might also result in a greater contrast in soluble transfers of P, including remobilization. According to the afore-mentioned study of Syversen et al. (2005), phosphorous reduction efficiency in experiments both simulated and natural runoff, ranged between 60 and 89%.
Buffer width is often given as the paramount intrinsic factor controlling the efficiency of P and suspended solids retention (Hoffmann, 2009). However, some experiments have showed that the effectiveness of grass buffer zones in reducing sediment losses from fields does not increase linearly with width. This suggests that there is an optimum width, beyond which there is little further increase in effect (Castelle et al., 1994; Gilliam et al., 1996; Abu-Zreig et al., 2003).

Kronvang et al. (2000) found that no sediment and particulate-P leached across a 29 m wide grass buffer zone[6]. Narrow zones can thence be also quite effective. Abu-Zreig et al. (2003) for instance obtained a 31% P retention with a grass buffer strip only 2 m wide. Vallières (2005) tested a 1 m grass buffer strip and found a good retention (60–80%) of both total-P and bio-P during runoff events of medium intensity.

The effectiveness in retaining phosphorous also changes in relation to the type of vegetation in place. An increase in the ratio of plant cover reduces the speed of runoff and thus the energy available for the transport of particles. This results in increased retention of the particles and their associated P load. Mankin et al. (2007) for instance, found phosphorus levels in their plot study decreased 91.8% by mass when using a seven-year-old grass-shrub buffer. Mowing grass and harvesting biomass are procedures that reduce P release from leaching of litter in autumn. Maintaining a good vegetation cover contribute to decrease the total P accumulation in the soil of the buffer (this is specifically important if the filter has been implemented on a formerly cultivated area, rich in P) (C. Gascuel et al., 2010). Hoffmann et al. (2009) reported about a study where P uptake by woody vegetation was higher than uptake in herbaceous vegetation. A similar conclusion was stated in a recent Danish study on the effectiveness of buffer zones : converting 10% of the buffer zones acreage from grass to trees, would boost the reduction of P per year from 6-38 tonnes to 17-120 tonnes. However, for buffer having same width, Dorioz et al. (2006) demonstrated that vegetation might not be a key factor, yielding gains in retention of P of at most 20%. (Uusi-Kämppä, (2005) also inferred that dormant vegetation in Nordic countries is not that effective in retaining nutrients during snow melting.

Soil texture is also an important factor to be considered. Schwer and Clausen, (1989) found a large difference in retention of total-P and dissolved-P between two grass buffer strips, one established on a sandy soil (retention 92 and 98%, respectively) and the other on a silty clay (33 and 12%). Soil characteristics do also affect phosphorous behaviour: during anoxic conditions and in presence of ferrous (III) oxides, release of adsorbate phosphate have been demonstrated (Hoffmann et al., 2009).

The temporal and spatial dynamics of P in the runoff moving across the grass buffer strip and the effectiveness of the retention of this P, are the result of a chain of physic-chemical and biological processes and which are triggered by the local changes in flow conditions due to the hydrological properties of the buffer. For instance, when the input of runoff exceeds the capacity of the buffer, the retention effectiveness for phosphorus and sediments declines because of saturation of the soil within the buffer and shorter residence times during the transport process (Dorioz et al., 2006). Over a period of many years, grass buffer zones could become a source of P to adjacent surface waters. A problem which might be exacerbated due to climate change, as a consequence of more rainfall events and frequent cycles of freezing/thawing especially in the Nordic countries. Dorioz et al. (2006) eventually agree asserting that grass buffer zones have a role in controlling diffuse phosphorus pollution, but that this role is both specific to place and limited in duration. Other studies also agree that grass buffer strips are a practical way of managing agricultural fields. By time, they can significantly reduce (by at least half) the flows of sediment and particulate-P transferred by diffuse runoff, without requiring special practices, large amounts of space, or specialized maintenance.

[6] However, 45% of P was retained in the first 4 meters and all of it within the first 12 meters

Nitrogen

Nitrogen fertilizers, manure application, nitrogen fixation by legumes and mineralization of soil nitrogen are the primary sources of NO3-N in agricultural watersheds. Part of the NO3-N are utilized by crops and other plants and excess of it become available to be carried by the surface and groundwater flow into the river and other water bodies as pollutants (Sahu e Gu, 2009). Nitrogen speciation occur through various processes such as plant uptake, microbial immobilization, soil storage, groundwater mixing and denitrification (Lowrance et al., 1997). Nitrogen removal effectiveness varies widely. A recent investigation on buffer zones effectiveness literature performed by Mayer et al. (2007), over 45 published studies and data from 89 individual riparian buffers, seemed to confirm such a trend. They found out that the mean nitrogen removal effectiveness (from surface runoff) in buffers larger than 50 m was significantly higher than in narrower buffers (0–25 m), suggesting that buffer width is an important consideration for nitrogen management in watersheds (Table 2). Riparian zones in any watershed can be either considered as a net source or a net sink of nitrate whenever it is released or retained. This is somehow confirmed by analysis of stream chemistry over both short and longer time scales which have shown that temporal and spatial variations in nitrate concentration result mainly from temporal changes in flow paths (Ranalli and Macalady, 2010). When in presence of water table rise, nutrients stored near or at the soil surface tend to be mobilized and leached out to the receiving water body (Creed et al., 1996). On the other hand, when the main flow pattern is a deep sub-superficial flow, nitrogen seems to accumulate in the soil. Dillaha et al. (1988) were among the first ones to conduct detailed experiments. Their study showed that retention of nitrogen in the buffer zones varied as a function of the width of the buffer strip: the larger the width of the buffer, the higher the efficiency in the removal of nitrogen, with rate of 73% for a strip width of 9,1 meters. The study confirmed also a decrease in the rate of efficiency due to nutrient saturation in the filter strip.

Nitrogen removal effectiveness also differed as the flow patterns change (Mayer et al., 2007). In their meta-analysis on nitrogen removal, they found out that subsurface removal of nitrogen was much more efficient than surface removal and that the former did not appear to be related to buffer width. Nevertheless, a small but important proportion of the variance in surface removal of nitrogen was explained by buffer width. Hence, the wider the buffer is, the higher the removal of nitrogen in the surface runoff. Based on their model, they estimated 50, 75, and 90% nitrogen removal efficiencies in surface flow in buffers of approximately 27, 81, and 131 meters of width respectively.

The buffer effectiveness in reducing nitrate in the groundwater compartment traversing the riparian zone, also varies in relation to the riparian-zone width. This variability was found to depend upon two factors such as the position of the riparian zone with regard to local, intermediate, and regional groundwater flow systems, and variation of the hydrogeologic properties of the riparian zone itself (Ranalli, 2010). In a previous study, Hill (1996) observed that riparian zones that are effective in removing nitrate also share similar hydrogeologic features (when in presence of shallow subsurface flow caused by permeable surface soils and sediments that are underlain at a depth of 1–4 m by an impermeable layer). Under these conditions, small amounts of groundwater follow not too deep, horizontal flow paths that increase the water residence time and contact with vegetation roots and organic-rich riparian soils, thus giving opportunity for both denitrification and plant-uptake (Burt et al., 1999). If shallow aquiclude are not present in riparian zones, the thicker surficial aquifer allows groundwater to follow deeper, longer flow paths bypassing riparian vegetation and soils (Vought et al., 1994; Mayer et al., 2005) and hence reducing the buffering capacity. A more recent study performed by Borin et al. (2010) came to the conclusion that a buffer strip of 6 meters (provided with perennial plants such as trees) could achieve rates of abatements of nutrients (in the chemical form of NO3-N) close to 100%. The width of the buffer, however, has been a reason of controversy throughout the years, with authors suggesting several widths as optimal size in their studies. Syversen, (2005) demonstrated, after her study with both simulated and natural runoff experiments, an average removal efficiency for total nitrogen which varied between 37% and 81%. Such variability would have to be correlated to variations in parameters such as local conditions such as climate, soil type and topography.

Studies on the effects of soil type on nitrogen removal have been performed by Gold et al. (2001). They investigated hydric soils in glaciated watershed and found out that the rates of groundwater nitrate removal were higher than 80%, whereas areas with nonhydric soils, which had steeper slopes and a greater depth of the water table had nitrate removal rates lower than 30%. Riparian zones with high groundwater nitrate-removal capacity presented a width larger than 10m (of hydric soil) and an absence of groundwater seeps. This produced a slow rate of groundwater flow and led to longer residence times in biologically active soils with significant nitrate-transformation rates. A fairly recent study by Vidon and Hill (2004) of stream riparian sites on glacial till showed also a high mean nitrate-removal efficiency (larger than 90%) for seven of eight sites investigated. The removal efficiency varied accordingly to width and soil type with rates of more than 90% in presence of loamy sand soils and 15 meters of width. In highly permeable sediments, shorter residence times of groundwater in contact with aquifer sediments restricted the development of anaerobic conditions and decreased the amount of nitrate removed. Also, coarse and grained sediments usually contain only small amounts of organic matter, which reduces denitrification processes. Although nitrate attenuation in groundwater occurs in riparian zones, the degree to which this attenuation affects stream concentrations and loads has still to be assessed on a watershed scale (Vidon and Hill, 2004). A riparian zone can have a high nitrate-removal efficiency measured in relation to the decline in concentration of nitrate; however, if the amount of nitrate that flows to the water body is small, the concentrations in the stream and relative loads will be scarcely affected. On the other hand, some buffer zones may have a lower capacity to reduce nitrate concentrations but receive high loads of nitrate over the year. This being the reason why, when considered at the catchment scale, these buffer zones with lower nitrate-removal capacity can exert a greater influence on stream conditions (Ranalli, 2010).

Syversen (2005) also argues that it is important to evaluate the most essential design criteria for having an efficient buffer strip. Uptake of nutrients by vegetation, for instance, will depend on type of vegetation. Trees with higher total biomass than grass, for example, will normally have higher nutrient uptake. Some of the nutrients incorporated in the leaves through the growing season will, however, recycle back to the soil during fall. The rate of plant growth through the growing season may enhance uptake of nutrients compared to withered vegetation. Nevertheless Syversen, as well as Vought et al. (1994) found no significant difference in retention efficiency for nitrogen between grass, brush/grass and beech forest buffer zones. Similar conclusion was achieved by Mayer et al. (2007) who agreed on the fact that there is no significant difference between woody and herbaceous vegetation as with regard to controlling nutrient movements in the riparian zone. Clément et al. (2002) found also no difference in the rates of denitrification among three vegetation types (forest, understorey vegetation, and grass), due to the fact that each vegetation type contains enough organic carbon for supporting denitrifying bacteria. Slightly aside from this flow of belief stands

Ranalli (2010) whose conclusions indicate a significantly lower nitrogen-removal efficiency in grass riparian zones relatively to forest, forested wetland, and wetland and that forests were slightly, but significantly, more effective than all the other vegetation types.

2.1.2 Sediment

Sediment is also considered a major agricultural pollutant. It is a nonpoint source of pollution for surface water worldwide (Liu et al., 2008). It mainly affects water quality by increasing turbidity and thus hindering light penetration and the related primary production processes. Chemicals have also been found to be linked to sediment transport (Munoz-Carpena and Parsons, 2004). The main buffer strip features believed to have the strongest influence on sediment deposition and retention are the width of the buffer (Liu et al., 2008; Abu-Zreig et al., 2004), slope (Liu et al., 2008) and vegetation type (density, stiffness and hight) (Syversen, 2005). The sediment trapping efficacy is mainly a function of size and mass. Liu et al. (2008) carried out a review study over 80 scientific articles whose efficiency rates of trapping related ranged from 45 to 100%. As for P, since buffers will also act as sediment sink (Hickey and Doran, 2004) long term issues will arise and maintenance practices need to be considered in order to keep the buffer effective.

2.1.3 Pesticides

Several experimental studies have demonstrated that grass buffer zones are also effective in reducing pesticide transfer from agricultural field to water bodies though with a certain variability (Lacas et al., 2005). As for nutrients evaluation, the effectiveness variability suggests that a wide range of physical and biochemical processes are involved in the functioning of grassed strips and that their relative importance can vary from one situation to another as a function of numerous parameters, leading to a final consideration that the prediction of the interception effectiveness of a given strip still seems unattainable with the present state of knowledge (Lacas et al., 2005). What is also argued by this author is that little work has been so far performed on the fate of the products intercepted by the buffer system and that attention shall be focused on by-products from degradation (other than only parent compounds) as well as on a better understanding of sub-surface flow processes. A comprehensive study carried out by Borin et al. (2004) tried to address the performance of a narrow buffer strip in abating herbicides such as terbuthylazine, alachlor, nicosulfuron, pendimethalin, linuron. The buffer system evaluated showed a good herbicide degradation potential, even if the abatement was not sufficient to satisfy the EU limit for environmental and drinking water. The reason of such pattern was related to insufficient buffer width: while a 6 m wide buffer is sufficient for controlling nutrients, a wider buffer would be recommended to increase herbicide abatement. The effectiveness in reduction varied from 60% (alachlor) to 90% (nicosulfuron). Arora et al. (2010) reviewed and performed an estimation on the average pesticides mass retention of both carrier phases such as runoff volume and sediment mass. Runoff volume retention averaged (with ranges) 45 (0-100) % across the different studies under both natural and simulated experimental conditions, whereas the sediment mass retention averaged 76 (2-100) %. The overall pesticide retention by buffer strips from natural and simulated studies for weakly (Koc[7] < 100 l ⁄ kg), moderately (100 < Koc < 1,000 l ⁄ kg), and strongly sorbed pesticides (Koc > 1,000 l ⁄ kg) averaged (with ranges) 61 (0-100), 63 (0-100), and 76 (53- 100) %,

[7] Sorption coefficient

respectively. Nevertheless, the same authors suggest the limitation of such estimations due to the different conditions the studies reviewed were performed upon. Syversen e Bechmann (2004) performed instead an analysis of particle-bound pesticides in situations of simulated surface runoff finding average removal efficiency rates in the amount of 39, 71 and 63% for glyphosate, fenpropimorph and propiconazole.

According to Vianello et al. (2005) infiltration is thought to be among the most important herbicide removal mechanism associated with vegetated buffer zones, especially for soluble or weakly adsorbed pesticides. Besides, plants in the vegetated strip confer a higher organic matter content to the filter zone than in the adjacent cultivated field. This organic matter accumulation should increase adsorption capacity and microbial activity for herbicide degradation, so reducing the amount of herbicide in surface runoff and leaching (Staddon et al., 2001). Higher herbicide removal would be thence due both to enhanced degradation and the formation of non-extractable (bound) residues, which can become a long-term store inside the filter (Benoit et al., 2000). In the study performed by Vianello et al. dealing with herbicides such as metolachlor, terbuthylazine and isoproturon, efficiency reduction rates were found out to range from 85.7 to 97.9% in the monitored events, those percentages being similar to those observed in other studies conducted under different climatic, pedological and agricultural conditions. Additionally total herbicide losses by runoff were acknowledged as low.

2.1.4 Biodiversity

Though to present date no biodiversity obligations for buffers are included in the legislative apparatus, some studies have evaluated its implications within such a context. Lankoski and Ollikainen (2003) offer an important overview of the status-of-art of effects of agricultural practices on species diversity. Referring to Bäckman et al. (1999) and Wossink et al. (1999), he infers that the largest number of species of both flora and fauna are found at the field boundary, which can be easily included among the most important semi-natural habitats sustained by agriculture. Species-richness can be described in terms of species-area relationship[8]. To this end, Ma et al. (2002) demonstrated that by widening other than lengthening the buffer strip, more floral species per unit area could be detected. However, biodiversity may be negatively affected by an increase in the nutrient content due to increase competition between species. Nevertheless, establishment of buffer zones is also expected to maintain biodiversity by increasing the amount of seminatural open areas in agricultural landscapes and by providing refuge, breeding and foraging habitat for riparian-obligate taxa (Hansen et al., 2010). Experience from the Finnish context (Tattari et al., 2003) shows that biodiversity tends to increase when former crop fields are left uncultivated though the role of buffer zones in terms of biodiversity has not eventually been assessed. Buffer zones with a width of more than 15 meters represent large uncultivated areas with potentially high significance for farmland biodiversity. Their value in maintaining biodiversity is likely to depend on the way they are managed (Meek et al., 2002). Buffers containing woody vegetation are also thought of exhibiting greater species richness than buffer covered merely in grass (Freemark et al., 2002). Connectivity is also another important aspect to consider: longitudinally intact riparian habitat provide continuous pathways for movement of wildlife, habitat corridors and vegetation continuity, though the degree to which riparian zones facilitate movement in un-modified landscapes is still

[8] Tendency of species richness to increase in function of the area

poorly quantified (Hansen et al., 2010). Plant species diversity strongly favours insect and bird species diversity though the number of variables affecting bird diversity is smaller than for plant diversity (higher insect diversity is also believed to increase bird diversity). Uncertainty still lies regarding reduced openness as a factor contributing or reducing bird diversity. The availability of water is also critical to riparian plant diversity, and variability in precipitation and flow, especially during dry spells and droughts, determines the broad-scale patterns of floodplain forest development (Ward et al., 2002). The relatively constant supply of water in riparian areas supports the proliferation of a greater assortment of plant types (Arthington et al., 2006). When considering the aquatic compartment, vegetation in riparian areas can help in regulating light and temperatures (through shading effect and increased surface and soil moisture). Such conditions can heavily affect aquatic organisms lives and dynamics. High in-stream temperatures directly affect aquatic biota through ecosystem respiration, which reduces dissolved oxygen availability and pH, (Davies *et al.*, 2004). Reductions in dissolved oxygen thus affect basal metabolic rate, and consequently, fitness parameters such as growth and reproduction. Changes in thermal regime can impact fish reproduction and increased temperatures can also reduce their tolerance to other toxicants, e.g. ammonia (Bradley James Pusey, 2003). Inputs of wood debris to the streams are also important in mediating the channel flow and the retention of sediments and nutrients, thus increasing the variability in the current and the complexity of microhabitats, which in turn sustain macroinvertebrate communities and the overall food chain (Hansen et al., 2010).

Table 2 Buffer zones efficiency summarized from literature with regard to retention of nitrogen, phosphorous, pesticides, sediments and enhancement of biodiversity (*) (N=nitrogen, P=total phosphorous/particulate-P, Pe=Pesticides, S=sediment, B=biodiversity)

Study	Environmental target	Efficiency reduction rates (%)	Features (**)
Borin et al. (2010)	N	~100	6 m buffer with perennial trees (N in form of nitrate)
Dillaha et al. (1989)	N	73	Buffer width of 9,1m
Mayer et al. (2007)	N	~58 ~71 ~85	0-25m 25-50m >50m
Rosenblatt et al. (2001); Gold et al. (2001)	N	>80	~10m width, hydric soils
Syversen (2005)	N	37-81	5 and 10m buffers with simulated and natural runoff
Vidon and Hill (2004)	N	>90	Within the first 15m on loamy sand soils
Abu-Zreig et al. (2003)	P	31 79	2m buffer 15m buffer
Borin et al. (2005)	P	80	6 m buffer
Kronvang et al.	P	~100	29 m width

(2000)			
Mankin et al.	P	92	7-year old grass-shrub buffer
Schwer and Clausen (1989)	P	92 33	Sandy and silty clay soil respectively
Syversen et al. (2005)	P	60-89	5 and 10m buffers with simulated and natural runoff
Uusi-Kämpä (2005)	P	41	10 m buffers (mowed grass and shrubs/gruss)
Vallieres (2005)	P	60-80	1m buffer
Arora et al. (2010)	Pe	61 63 76	Respectively for weakly, moderately and strongly sorbed pesticides
Borin (2004)	Pe	60-90	6m buffer
Syversen and Bochmann (2004)	Pe	39 71 63	5m- grass/herbaceous vegetation buffer; Effiency removal rates for glyphosate, fenpropimorph and propiconazole respectively.
Vianello et al. (2005)	Pe	~86-98	6m wide buffer composed of trees, shrubs and grass
Liu et al. (2008)	S	45-100	Review study over 80 scientific papers
Hansen et al. (2010)	B (*)	35-95 m	Moderation of stream temperature
		35-95 m	Provision of food and resources
		40-100 m	Improvement of in-stream biodiversity
		100-200 m	Improvement of terrestrial biodiversity

(*) Quantification of benefits accruing to biodiversity thanks to buffer zones are still of semi- qualitative type. According to Hansen (2010) efficiency can be expressed as a width of the buffer which deploys or augments a certain ecological function (i.e. connectivity and terrestrial habitat for fauna, inputs for aquatic fauna, riparian vegetation extent and shading)

(**) Efficiency was mainly considered accordingly to the variable width

2.2 The economics of buffer zones

Water pollution, caused by the intensification in the use of fertilisers and pesticides, is a current policy issue in many countries. In the recent European environmental policy discussions around the reorientation of the Common Agricultural Policy (CAP)[9], water associations are demanding as well as the implementation of riparian buffer strips, which are considered as a potentially refundable non-market service (Sieber et al., 2010). The main problem with such approach is the difficulty in determining benefits and costs associated with buffer zones. Consistent with most analyses of the costs and benefits of natural resources management alternatives, the social marginal costs of setting a buffer strip (i.e. income foregone, costs of management), might appear easier to quantify than social marginal benefits due to the fact that the latter are seen as non-market values – e.g. water quality , species diversity and valuable fish species – and thence more difficult to evaluate (Anbumozhi et al., 2005). Nevertheless, attempts to take such environmental positive externalities into account have been performed. The Danish Economic Council, for instance, evaluated the effect of pesticide-free buffer zones on biodiversity and water quality and found out that benefits exceeded the social costs of lower agricultural productivity, even with the inner methodological problems associated with the hypothetical valuation methods (Danish Economic Council, Autumn 2004).

An easy method for eventually consider these potential benefits is thence to provide a financial support to farmers for changing land use pattern and dedicate part of their fields to buffer zones. The amount of the compensation is usually quantified in relation to the losses accruing to the farmer for not cultivating that portion of land any longer (or at least for the time the buffer zone is in place). Farmers can then remain on income levels comparable to the previous situation without internalising external costs (Hediger and Lehmann, 2007). However, experience has demonstrated that sometimes compensation is not adequate enough and farmers may be not willing to participate to such support schemes especially when agriculture is high productive at field margins and land prices are high (as it occurred, for instance, in the Netherlands with the implementation of buffer zones, Dworak et al. 2009). Besides, investments in nature conservation projects typically demand a long lead time before benefits start to appear. According to the classic economic view of time value, and since benefits only accrue in the distant future, evaluation of such projects through discounted measures of project worth often turns out to be rather unfavourable, with direct costs (capital investment) at establishment of the programme/project weighing heavily (and negatively) in the analysis (Ebregt and Greve, 2000). From a more strict economic perspective, if market prices for environmental services prevail, they will eventually present an imperfect reflection of value, i.e. market failure works to the detriment of nature conservation efforts. It gets often problematic to integrate the direct and indirect benefits of nature conservation in an analytical framework, and the incremental costs and benefits of establishing buffer zones are even more difficult to assess. It is therefore difficult to establish the feasibility and sustainability of buffer zones from a purely economic point of view, using the available methodologies, even though it is acknowledged that such zones have important indirect (secondary, non-use) benefits (Ebregt and Greve, 2000).

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[9] The Common Agricultural Policy (CAP) is the main EU tool supporting agriculture, accounting for 34 % of the EU budget in the period 2007–2013. A “health check” of the Common Agricultural Policy was performed and eventually issued on November 2008. Presently, public debates and strategies are being implemented for further amendments that are set to take place by 2013 (EEA, 2011).

2.3 Agri-environmental support programs

Despite concerns on the full economic viability, nowadays buffer zones presently account for around 70% of the most commonly applied measures adopted for tackling environmental pressures of agriculture[10] (Dworak et al., 2010) and have been included in financial support schemes[11]. In recent years, agricultural support measures have been integrated into the so called Rural Development Programs (RDPs). The Rural Development policy for the period 2007-2013 is mainly built upon three themes such as improving agricultural competitiveness, improving the environment and supporting land management and improving the quality of life and diversifying the economy in rural areas. Within this program, each country can decide on the extent of the remuneration and the design of agri-environmental policy measures that address national environmental problems such as e.g. nutrient emissions (Elofsson, 2010). Nowadays, the current approach is for buffer zones to be set up through agri-environmental payments, or, alongside the application of article 38 of the Rural Development Regulation[12] (as, for instance, in the situation in which buffer strips or zones are created with significant impact on farming activity). According to Dworak et al. (2009), two main payment schemes are available: a one-time payment (which is project based and includes the costs for buying the land and for support investments) and a continuous one (to maintain the area of the field converted into buffer zone). The RDPs with EU co-financing scheme offer compensation to farmers in the form of continuous support of up to 100% of the total costs, based exclusively on income foregone and additional costs (payments for environmental benefits are not contemplated). Under the River Basin Management Plants (RBMPs) scheme, time limited compensation payments are also possible for mandatory measures which go beyond legal requirements. Overcompensation is also a possibility envisaged within the Rural Development Programmes. Along normal regulations for establishing payments, overcompensation is allowed when sufficient justification is stated (i.e. multiple benefits), low acceptance of farmers has to be increased or overcompensation already took place in the previous programming period. While usually national sources are used to support this measure, the EU also participates in co-financing (Dworak et al., 2009).
Alongside the River Development Programmes, few other referential European pieces of legislation are worth it to be mentioned such as the Water Framework Directive (WFD) and the Nitrates and

[10] Diffuse pollution by nitrate (92%) or by phosphorus (90%), pesticides (74%), morphological deteriorations (50%), water abstraction for irrigation (28%) (Assessment of agricultural measures in draft River Basin Management Plans, September 2009)

[11] The Europe-wide directive 99/1257/EEC included a first scheme of measures for promoting the implementation of riparian strips.

[12] Article 38, Council Regulation (EC) No 1698/2005: "Natura 2000 payments and payments linked to Directive 2000/60/EC. 1. Support provided for in Article 36(a)(iii), shall be granted annually and per hectare of UAA to farmers in order to compensate for costs incurred and income foregone resulting from disadvantages in the areas concerned related to the implementation of Directives 79/409/EEC, 92/43/EEC and 2000/60/EC. 2. Support shall be limited to the maximum amount laid down in the Annex. For payments linked to Directive 2000/60/EC, detailed rules, including the maximum amount of support, shall be fixed in accordance with the procedure referred to in Article 90(2)."

Pesticides Directives. The Water Framework Directive (WFD)[13] entered in force in December 2000. It has the ambitious target of achieving a good status[14] in all water bodies in the European Union by 2015. Exemptions from not fulfilling the target, such as time derogation or lowered objectives to be granted, are also included in the text. The directive addresses inland surface waters, transitional waters, coastal waters and groundwater under natural geographical and hydrological units (river basin) and establishes innovative principles for water management, including public participation in planning and new economic approaches. As for the latter, it is clearly stated in the Directive that MS shall analyse the costs of water provision as well as if the consumers bear the full costs. Additionally, cost-efficiency and cost-benefit analysis are referred to as central economic methods in order to achieve environmental targets at the lowest costs and to evaluate if some of them would eventually be too costly (Jacobsen, 2007). With regard to buffer zones, to date, no stated obligation to their use and/or implementation is mentioned in the Directive.

As for the Nitrates Directive (91/676/EEC), buffer strips are cited as a concrete requirement for farmers inside National Vulnerable Zones[15] (NVZs) defined by Member States and when they are included within National Action plans. Outside of the NVZs, buffer zones can be included in the codes of good practice to be defined by the Member States, but in such cases cross-compliance is not applied (except if established on a voluntary basis under rural development measures).

The Framework Directive on the Sustainable Use of Pesticides (Directive 2009/128/EC) focuses on the setting up of national action plans (NAPs) with concrete measures to reduce use, risks and to monitor the use of pesticides. The first NAPs have to be finalised by 2012. In Article 11 of 2009/128/EC[16] it is stated that Member States have to ensure that appropriate measures to protect the aquatic environment and drinking water supplies from the impact of pesticides are taken, including buffer zones adjacent to water courses and areas for abstraction of drinking water. The buffer zones where pesticides must not be used or stored shall be appropriately-sized, though this remains a highly debated matter.

With the latest reform of the CAP (regulation n°73/2009) the "*establishment of buffer strips along water courses*" has become an obligatory GAEC (Good Agricultural and Environmental Condition) standard and thus part of the cross-compliance obligations[17]. Such mandatory establishment would then compensate for the loss of compulsory set-aside[18]. Between January 1st 2010 and January 1st 2012 at the latest, buffer strips will have to be implemented at national level (Art. 6, Council Regulation (EC) No 73/2009)[19]. Thus, measures that were voluntary with compensation payments

[13] Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy

[14] Both ecological and chemical status

[15] Areas of land which drain into polluted or threatened waters and which contribute to N pollution

[16] Specific measures to protect the aquatic environment and drinking water

[17] The term 'Cross-compliance' links direct payments to farmers to their respect of environmental and other requirements set both at EU and national levels. Cross compliance was introduced by the CAP reform in 2003 (http://ec.europa.eu/agriculture/capreform/infosheets/crocom_en.pdf)

[18] Measure introduced in 1992 that required farmers to take part of the area under arable crops out of production in order to curb sur-plus production of certain crops.

[19] Unless a Member State has already defined, a minimum requirement for the good agricultural and environmental condition before 1 January 2009 and/or there are already in force national rules addressing that specific standard (Art. 6, paragraph 1 a) and b))

(such as buffer zones) could then become mandatory without payment (WssTP Workshop “Buffer Strips and Buffer Zones – Targeting Water Quality”, Bruxelles, February 2010). Nowadays, one of the main concerns throughout Europe, especially when referring to the GAEC obligations under the cross-compliance theme, remains on how to consider and integrate the different farming systems along different watercourses. The following sections will investigate the latest approaches in the Scandinavian countries of Sweden, Denmark and Norway.

2.3.1 Sweden

The agri-environmental support schemes are part of the Rural Development Programme that, for Sweden, spans a time of six years, from 2007 to 2013. After 2010, the financial resources allocated to improve water quality account for over 80 EUR million per year. Such amount has been calculated for targeting reduction of both nitrogen and phosphorous load (in the amount of 1900 ton N per year and of 30 ton P per year) to the sea[20]. Nevertheless, the achieving of this objective will contribute at the same time to the fulfilling of the not less paramount targets such as the ones stated in the Water Management Ordinance and Zero Eutrophication Programme (Swedish Environmental Protection Agency, Report 5989 • Sweden’s Commitment under the Baltic Sea Action Plan, Socio-economic impact assessments, 2009). Among the measures to be implemented it is contemplated the adoption and strengthening of buffer strips alongside waterways. The stated priority area for assistance comprises all arable lands in central and southern Sweden. Farmers who still voluntarily adopt such measure can benefit of a financial compensation depending on if the requirements are being fulfilled, such as:

- To sow the riparian strip no later than the spring of the first year of installation and
- To keep a width which ranges from a minimum of 6 to a maximum of 20 meters.

The application of fertilizers is also forbidden and the strip shall not be modified in its nature until the end of the commitment period that is valid for 5 years. The payment scheme in place guarantees SEK 3000/hectare (335,7 €/ha) for buffers in agricultural areas. Of this amount, 2000 SEK are granted as areal support and the remaining 1000 SEK as extra compensation for the conversion of the land. A higher compensation of SEK 4000/hectare (447,7 €/ha) is provided instead when best crop-growing conditions apply. The level of payment is calculated on the basis of loss of income when no crop can be planted on the buffer and costs for sowing the buffer strip. The main reason behind the establishment of buffer zones in Sweden is to counteract the losses of phosphorous, manure and pesticides from arable land to inland waters through surface runoff. Besides, especially in the south of the country, where the NVZs policy applies, they also help reducing the load of nitrogen (given the amount of buffer zones in 2005, they had an efficiency of almost 4 kg N/ha). As for phosphorous, the estimated impact of buffer zones in Sweden accounted for 6.5 tonnes of less P leached into watercourses in 2008. Given the surface of land dedicated to buffer zones of 6 984 hectares, the retention efficiency corresponds to 0,93 kg P/ha. (SLU Sveriges lantbruksuniversitet Axel 2 – utvärdering av åtgärder för att förbättra miljön och landskapet, 2010). Set in relation to the payment scheme of 3000 SEK/ha, the cost per reducing a kg of P equals to 3 250 SEK.

[20] In 2005, the mean leaching rates for agricultural swedish land have been quantified in 18 kg N/ha and 0.52 kg P/ha (NATURVÅRDSVERKET, Rapport 5823 • Läckage av näringsämnen från svensk åkermark)

2.3.2 Denmark

In the Action Plan for the Aquatic Programme III (2004), it was stated the introduction of 30 000 hectares of 10-meter wide buffer zones[21] (which went integrating and expanding the previous mandatory width of 2 meters) along natural watercourses before 2009 and a further 20 000 hectares by 2015. The approach was voluntary and the support payment, within the EU compensation framework, accounted for 600-1200 DKK/ha given by the potential income lost (such amount have always varied from 0 (sandy soil with no irrigation) to 2000/3000 DKK/ha for clay soil, and eventually a rate in between has been identified). Nevertheless, already in 2006, a decrease in the uncultivated[22] buffer zones area of around 4 000 ha was acknowledged (Carl Bro, 2008) and a further investigation (Jacobsen, 2006) found out that around 30-50 000 hectares of the total buffer zone area were cultivated. The permanent cessation of the mandatory set-aside scheme had been regarded as one of the possible causes of this trend. In the initial financing plan, 113 DKK million were dedicated to the implementation of buffer zones, which are considered a necessary requirement especially in Denmark where all land is regarded as Nitrate Vulnerable Zone (NVZ).
The Green Growth agreement[23] (2009), which implements the Danish Rural Development Programme (RDP) for 2010 – 2013 and the provisions of the Baltic Sea Action Plan, had initially the aim of reducing losses of nitrogen and phosphorous in the amount of 19 000 tonnes and 210 tonnes respectively, between 2010 and 2015 (Report on Denmark´s implementation of the HELCOM BSAP, 2010). With regard to emissions of phosphorous, the latest estimates, also in connection to the evaluation of the Action Plan for the Aquatic Programme III (Notat: Effekt på fosforudledning af 10 m brede randzoner, Aahrus University, January 2011), predict that a 10-meter buffer zone along all watercourses and lakes larger than 100m^2 will favour a reduction of 6-38 tonnes of P per year provided annual harvesting and removal of plant material[24]. The total area with buffer strip connotation extends for around 53 000 hectares. As for nitrogen emissions, the latest estimates on the expected retention efficiency have been quantified between 26-66 kg N/ha per year (whether it is clay or sandy soil)[25]. The payment per hectare of land appointed to buffer zones would be set at 2 600 DKK per year (348,7 €/ha).
Current debates are mainly on whether having energy crops, extensive grazing or no crop at all in the buffer strip is a cost effective implementation of the buffer zone scheme itself, especially in the view of the natural environment constraints (in low land areas for instance, no more than 5% of the field area can be converted to buffer).

21 In the Danish context the term “buffer zones” is mainly used in relation to ammonia deposition, whereas “randzoner” is mainly used in the view of water protection.

22 A number of extensively cultivated land of fallow and some pasture fell also in this class.

23 Long-term plan defining environment and nature policies and the agriculture industry’s growth conditions in the Danish context. A total of DKK 13.5 billion (1.8 b€) is to be invested in Green Growth until 2015, which is about a 50% increase in investments compared to previous initiatives.

24 Effekt på fosforudledning af 10 m brede randzoner, DANMARKS MILJØUNDERSØGELSER OG DET JORDBRUGSVIDENSKABELIGE FAKULTET, AARHUS UNIVERSITET, 2011

25 Sammenfattende notat om mulighederne for iværksættelse af yderligere virkemidler til opnåelse af målene om randzoner I VMP III aftalen.

2.3.3 Norway

How much financial support the farmers get, varies depending on which county they live in, and to the priority of the areas. Priority areas can be areas close to a drinking water source or in areas with high erosion risk. The financial compensation for having established buffer zones varies from 666 to 1333 NOK/ha (from 84,8 to 169,7 €/ha). The requirement for the width of the buffer is between 5 to 6 meters. In priority areas the subsidy can double in its amount, but the buffer zones must vary accordingly, having a minimum width of 12 meters. The established buffer zones must last for at least five years and consist of grass (permanent buffer zones can also include some scattered trees). Other requirements to be fulfilled in order to get financial support are that the areas should have been previously used to grain, potatoes or vegetables production, and that the purpose and the goals for the measures are to reduce erosion and nutrient runoff from agricultural areas. In most areas it is required that the buffer zones are harvested.

Table 1 Current requirements for buffer strips in Denmark, Sweden and Norway (adapted from Dworak et al., "*International review on payment schemes for wet buffer strips and other types of wet zones along privately owned land, Ecological Institute*", Berlin, 2009).

Country	Width	Approach	Compensation per hectare of buffer strip installed[26]	Restrictions
Sweden	At least 6 meters broad but maximum 20 meters	Voluntary	447,7 €/ha (in areas with best crop-growing conditions) 335,7 €/ha (in the remaining areas)	The use of fertilizers/pesticides is prohibited 5 years of minimum commitment
Denmark	10 meters (along all open watercourses and lakes in excess of 100 m^2)	Mandatory	348,7 €/ha	Under discussion *
Norway	From 5 to 6 meters (in priority areas minimum width of 12 meters)	Voluntary Mandatory (2 meters to get the full size production subsidy from the State)	84,8 €/ha (169,7 €/ha in priority areas)	Harvesting mandatory in some areas Life span of minimum 5 years Grass (or scattered trees) No fertilizers allowed

* (see text)

[26] Currency conversion rates utilized (1 EUR = 8.93465 SEK, 1 EUR = 7.45656 DKK, 1 EUR = 7.85038 NOK, at 20th May 2011)

3. The efficiency of buffer zone width: a Swedish case study

As previously stated, among the design parameters of a buffer strip, the width of the buffer is an important one when to determine both costs of installation and efficiency in pollutants retention. However, the relationship between the width and the impacts is difficult to assess due to the variations in site conditions and pollutant types (Dosskey, 2008). A model is an alternative approach. Drawing on the study by Dosskey et al. (2008)[27], which exploited the possibility of using a model for establishing a relationship between width and trapping efficiency for sediment and water as well as being a prompt tool for the design of the most effective buffer strip, a similar approach has been tested for the evaluation of the potential performance of buffer zones in retaining phosphorous under Swedish referential conditions.

The choice of focusing on phosphorous is due to the fact that in the Scandinavian countries, the control of P transport from agricultural fields is strictly merged to the control of P at the source: in an environmental context where 90% of phosphorous losses can occur from only 10% of the total catchement area and during just 1% of the time, buffer zones can exert a strong impact along with other prevention strategies (Bergström et al., 2007).

Two Swedish regions (one in the South-West, one in Central Sweden) have been selected due to their specific climatic conditions (see Appendix A) and their land and agricultural use. Site and design features such as slope, soil type, cultivation type and width have been simultaneously modelled in order to produce as more realistic outputs as possible as well as to investigate the variation of the effectiveness with respect to such changing variables.

3.1 The model: ICECREAM DB

The ICECREAM DB model is a field scale model that is used for simulating nutrient losses from agricultural lands through processes of runoff and leaching (Liu, 2010). It is based on the models CREAMS[28] though adjusted for Nordic climate conditions, and it is currently utilized for calculating normalized phosphorous losses from arable land in Sweden at both regional and national scale. As specified in the Naturvårdsverket Rapport 5823 (Sweden, 2008), in setting up the model, Sweden has been divided in 22 leaching regions with different climate conditions, agricultural practices, fertilization regimes and production rates. Leaching coefficients are calculated upon combination of 13 types of crops, 10 types of soil-texture classes, 2 different fertilization regimes, 3 slope classes and 3 soil-phosphorous classes. Such leaching coefficients represent simulated losses over one year based on normalized climate and crop yields. For P, estimated losses include both root-zone leaching and losses through surface runoff. The model also includes an option for evaluating the conditions of different field segments that can differ in crop, slope and management within each single simulation run, hence being an appropriate function for including and examining the buffer strip effects.

[27] "A design aid for determining width of filter strips"

[28] The Chemicals, Runoff, and Erosion from Agricultural Management Systems model (Knisel, 1980). For thorough insights on the ICECREAM model refer to Naturvårdsverket Rapport 5823 (Sweden, 2008).

3.2 Simulations

Trapping efficiencies[29] for both particulate and dissolved P were calculated for 32 different combinations out of 5 set of site variables such as climate region, slope, field length, soil and cultivation type. Each set consisted of two values which represented a range of agricultural conditions (Table 3). Eventually, for each combination, simulations were run for several buffer widths: 0, 4, 12, 20 and 30m when field length was 200m and 0, 8, 24, 40 and 60m when field length was 400m providing buffer area/field area ratios of 0.02, 0.06, 0.10 and 0.15 The choice of such values reflects the one adopted by Dosskey et al. (2008). For all simulations, a 20-year meteorological series (1985-2005) has been used. Estimations of the total amount of P (given by the sum of particulate P and solute P) leached have been calculated for every year and then averaged on the 20-year time span. The buffer strip was represented by integrating a second field segment with ley having same slope and soil texture as the adjacent field. Uniform conditions of traversing run-off have been assumed.

Table 3 Variables utilized in the simulations in comparison with ones adopted by Dosskey et al. (2008)

Variable	ICECREAM DB	Dosskey et al. (2008)
Climate (*)	Regions 1b and 6 20-year metereological data series	Single rainfall event (61mm in 1 hour)
Slope	2 and 10%	2 and 10%
Field length	200 and 400 meters	200 and 400 meters
Soil type (*)	Sandy loam (S03) Silty clay loam (S08)	Fine sandy loam Silty clay loam
Cultivation type	Spring barley Winter wheat	USLE C factor (0,15 and 0,50)
Buffer width	4,12, 20, 30 (200m field) 8, 24, 40, 60 (400m field)	4,12, 20, 30 (200m field) 8, 24, 40, 60 (400m field)
Buffer strip	Grass ley (Manning's n= 0,15/plant stage)	Well-established grass (Manning's n=0,40)

(*) A more detailed description is provided in the Appendix A

3.3 Simulation results and discussion

With regard to the estimated losses of phosphorous, large differences have been identified among the several combinations (appendix B). With slope of 10%, estimations of P leaching appear fairly larger than the Swedish medium value per year. For instance, losses of total P under the scenario "01b_10_400_scl_ww" accounted for more than 120 kg per year from a 4 hectare large field[30].

[29] Calculated as the difference between the total amount of P leached and the amount retained thanks to the introduction of the buffer strip.

[30] Though ICECREAM DB returns losses in kg ha^{-1} $year^{-1}$, the total loss must be thought over an area given by the product of the field length (variable) and the buffer extension along the water receiving body (default

Sensitiveness of the model to the slope and field length parameters as well as the combination of the other values, might be adduced as reason for such estimations. Generally, highest losses of total P have been identified from longer fields and steeper soils as well as from silty clay loam soils other than sandy loam ones. On the other hand, trapping efficiencies showed a pretty regular trend whose distribution being mainly influenced by the width of the buffer strip instead of other site conditions variables. A summarising graph has been created out of the all simulations representing the range of efficiency variations. Five indicative curves out of thirty-two scenarios considered have been highlighted (Figure 1) (Table 4). A non linear regression was conducted. A logarithmic function was chosen because of the good fitness (high r^2 coefficients) showed for the relationships investigated (for both field lengths). However, the performance of the curves dropped for buffer widths less than 4 meters along with an increase in the difference between the values predicted by the model and the ones obtained by the logarithmic function. Hence, estimations outside the range of simulation must be critically evaluated. Nevertheless, the shape of the curves (getting steeper and more skewed at lower widths whereas flatter when width increases) seem to reflect theoretical assumptions, being the efficiency thought to be higher in the first meters of a buffer and decreasing at larger widths. The efficiency rates appear thence pretty reasonable especially if we consider a shorter interval such as between 4 and 20 meters (which include the Swedish recommended width range for buffers).

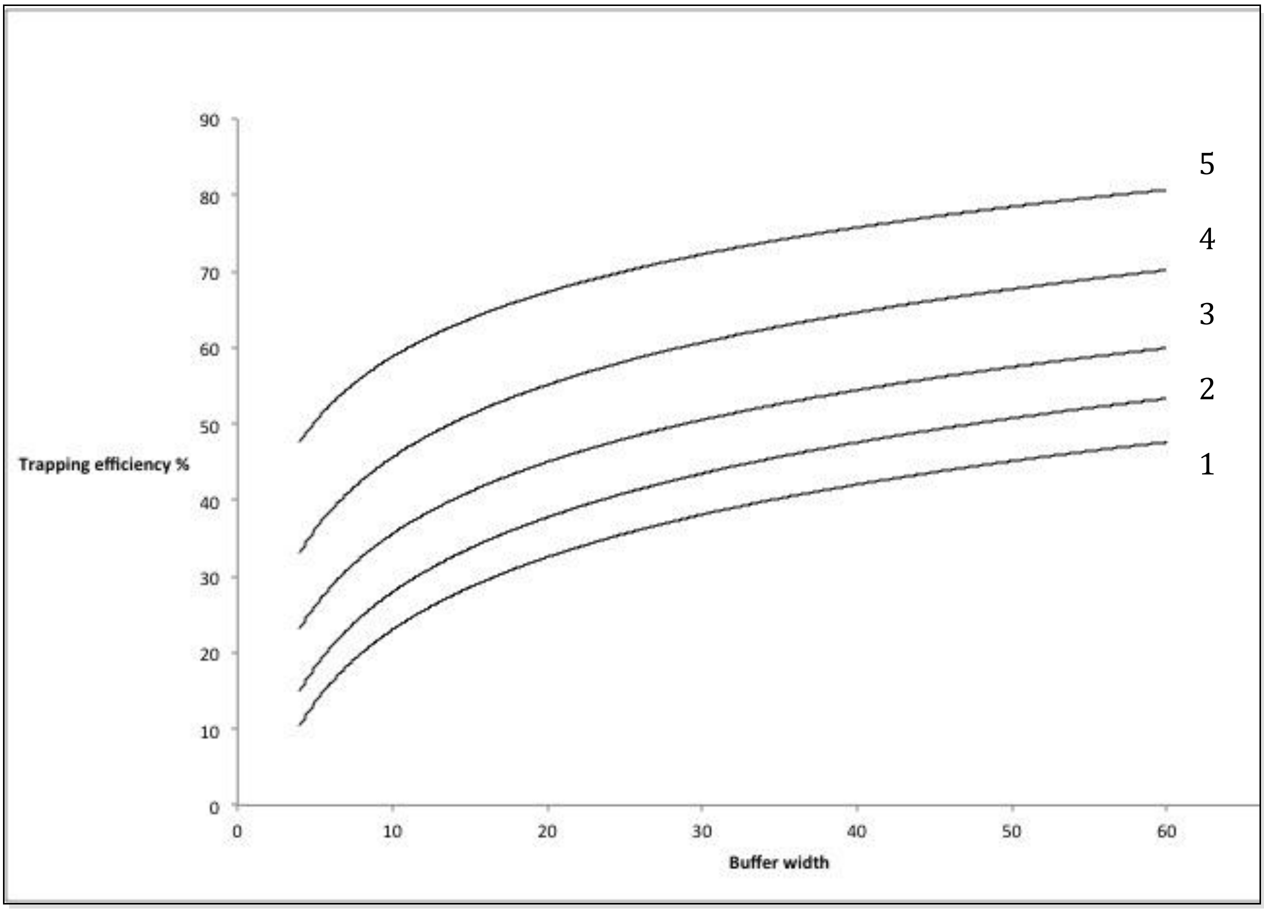

Figure 1 Relationships between trapping efficiency and buffer width for five different scenarios out of the total set of simulations

value in the model i.e. 100 meters). Hence, in all the simulations, the load of total P has to be adjusted to the area investigated.

Table 4 Parameters adopted for simulation curves in Figure 1

Curve number	Climate region	Slope (%)	Field length (m)	Soil length	Crop
5	06	10	200	Sandly loam	Winter wheat
4	06	2	200	Silty clay loam	Winter wheat
3	01b	2	200	Silty clay loam	Spring barley
2	01b	2	200	Sandy loam	Spring barley
1	01b	2	400	Sandy loam	Spring barley

Differently from the crop/vegetation and management factor (i.e. C factor, Universal Soil Loss Equation) used in the reference study of Dosskey (2008), two cultivation parameters have been utilised instead, namely spring barley and winter wheat. With regard to less steep field, a higher efficiency in retention of total P was detected for winter wheat other than spring barley. A minor difference was spotted for steeper fields (Figure 2 and 3).

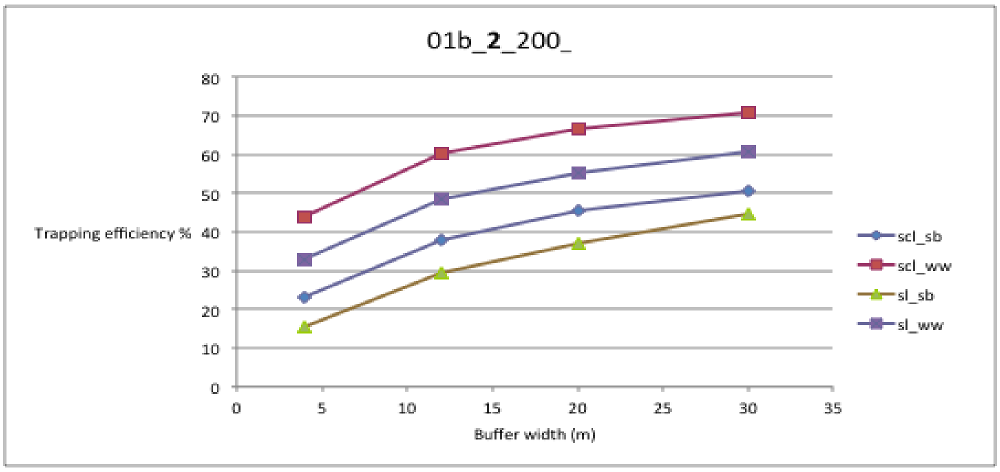

Figure 2 Trapping efficiency as a function of slope and cultivation type

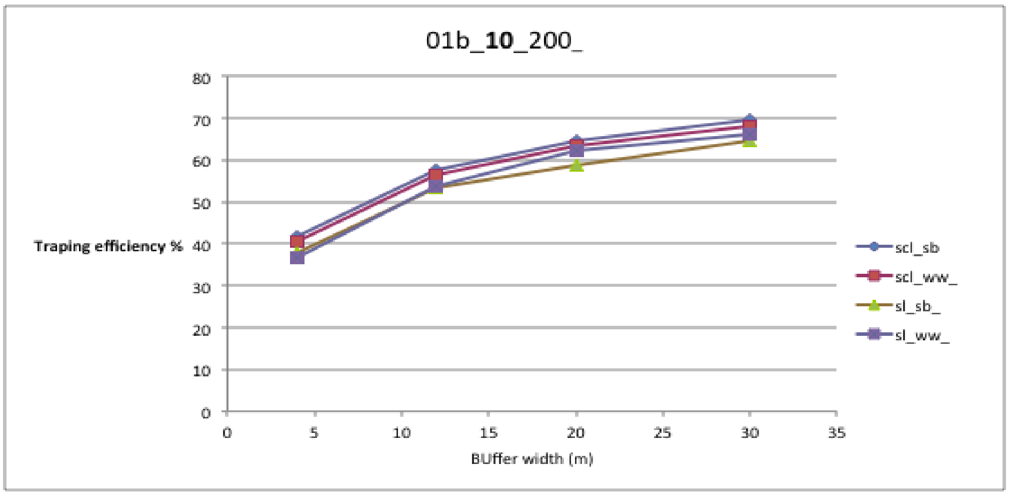

Figure 3 Trapping efficiency as a function of slope and cultivation type

A similar pattern can be observed with regard to the soil effect on the efficiency. For less step fields, and under similar site conditions, silty clay loam soils generally returned better results. When it comes to steeper fields, the efficiency was rather similar between the two typologies of soil considered. However, the potential presence of preferential flow in well-structured clay soils can highly increase the leaching of P if compared to the flows in a sandy soil and hence hamper and decrease the efficiency. When looking at the effects of slope and field length, keeping the other variables constant, the efficiency proved to be generally higher for steeper and shorter fields and lowest for flatter fields (Figure 4). With regard to regions, efficiency proved to be higher in region 01b other than in region 06 (the latter having both lower average annual runoff and precipitation).

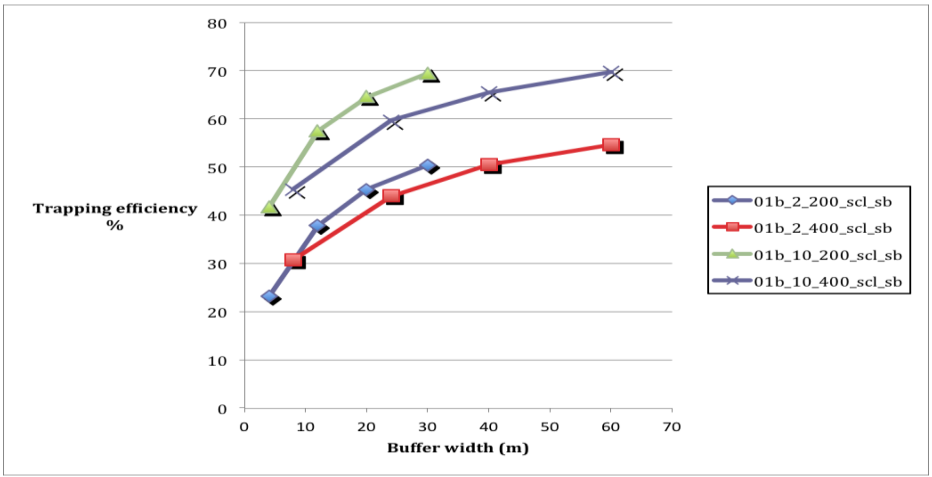

Figure 4 Example of trapping efficiency in relation to different slopes and field lengths, keeping other variables fixed

Generally, buffer zones are thought to better retain particulate other than dissolved forms of it. Nevertheless, what emerged from the simulations was that in some years the cumulative losses of dissolved P were higher than the particulate P. Thus, climate events and its variation among years can be believed to be quite influential for such outcomes. The vegetation reference for the buffer strip used in the modelling exercise was ley with very low roughness coefficient (0.15 at its mid-season stage versus the 0.40 Manning´s n coefficient utilised by Dosskey). Such value reflects the conditions of a standard buffer strip and its management practices most likely for the Swedish context, hence the lack of data in the model settings relative to other types of vegetation, made not possible investigate other effects.

3.4 Cost effectiveness of buffer zone width

Having estimated the amount of P lost from the land under the set of chosen variables, and knowing the efficiency rates for each scenario, considerations on the cost per reducing a kg of P have been drawn, also in accordance to the Swedish national agri-environmental support, i.e. 3000 SEK/ha (Table 5). In order to make more realistic deductions, only scenarios with 2% slope have been considered (the 10% slope is considered distant from the average steepness of soils in the regions investigated.

Table 5 Example of estimations of costs per reducing a kg of P in region 01b for width between 4 and 20 meters

Witdh	4	6	8	10	12	14	16	18	20
Scenarios	trapping efficiency %								
01b_2_200_scl_sb	23,13	28,64	32,55	35,58	38,06	40,15	41,97	43,57	45,00
01b_2_200_scl_ww	44,48	49,94	53,80	56,80	59,26	61,33	63,12	64,71	66,12
01b_2_200_sl_sb	14,94	20,68	24,76	27,92	30,51	32,69	34,58	36,25	37,75
01b_2_200_sl_ww	33,07	38,62	42,56	45,62	48,12	50,23	52,05	53,67	55,11
01b_2_400_scl_sb	22,43	27,31	30,77	33,46	35,65	37,51	39,11	40,53	41,80
01b_2_400_scl_ww	42,23	46,81	50,07	52,59	54,65	56,39	57,90	59,24	60,43
01b_2_400_sl_sb	10,42	15,99	19,94	23,01	25,51	27,63	29,46	31,08	32,53
01b_2_400_sl_ww	31,70	36,77	40,36	43,15	45,43	47,36	49,03	50,50	51,82
	kg of P reduced								
01b_2_200_scl_sb	0,12	0,16	0,20	0,22	0,24	0,26	0,27	0,29	0,30
01b_2_200_scl_ww	2,60	2,92	3,14	3,32	3,46	3,58	3,69	3,78	3,86
01b_2_200_sl_sb	0,09	0,13	0,15	0,17	0,18	0,20	0,21	0,22	0,23
01b_2_200_sl_ww	0,30	0,35	0,38	0,41	0,43	0,45	0,47	0,48	0,49
01b_2_400_scl_sb	1,18	1,43	1,61	1,76	1,87	1,97	2,05	2,13	2,19
01b_2_400_scl_ww	5,92	6,56	7,02	7,37	7,66	7,90	8,11	8,30	8,47
01b_2_400_sl_sb	0,13	0,20	0,25	0,29	0,32	0,34	0,37	0,39	0,41
01b_2_400_sl_ww	0,66	0,76	0,84	0,89	0,94	0,98	1,02	1,05	1,07
	cost per kg of P reduced (SEK)								
01b_2_200_scl_sb	107,82	130,63	153,26	175,26	196,62	217,43	237,75	257,65	277,17
01b_2_200_scl_ww	46,17	61,70	76,35	90,39	103,98	117,22	130,15	142,84	155,31
01b_2_200_sl_sb	1325,35	1435,76	1599,15	1772,56	1946,92	2119,63	2289,89	2457,51	2622,56
01b_2_200_sl_ww	404,87	520,01	629,19	733,81	834,89	933,10	1028,95	1122,78	1214,88
01b_2_400_scl_sb	101,95	125,61	148,64	170,89	192,45	213,42	233,89	253,93	273,59
01b_2_400_scl_ww	20,28	27,44	34,21	40,71	47,01	53,15	59,16	65,06	70,86
01b_2_400_sl_sb	924,92	904,01	966,44	1047,07	1133,12	1220,64	1308,16	1395,06	1481,09
01b_2_400_sl_ww	182,60	236,14	286,80	335,33	382,21	427,77	472,24	515,78	558,52

The amount of kg of P reduced was calculated for each type of buffer, whose extension was given by the width multiplied by a fixed length value of 100 meters. However, if looking at the acreage of buffer zone and the amount of load reduction within each region, differences, which eventually return different costs per kg of P reduced, emerge (Table 6).

Table 6 Comparison between region 1b and 6 regarding buffer zones extension, load P reduction and cost for reducing a kg of P (SEK)[31]

	Areal 2008 (ha)	Load reduction 2008 (ton P)	Reduction of P per area (kg P/ha)	Cost for reducing a kg of P (SEK)
Region 1b	140	0,66	4,7	640
Region 6	3 348	2,06	0,61	4920
Total Sweden	6 984	6,51	0,93	3250

In order to suggest the most cost-effective implementation scheme, an analysis of marginal costs variation throughout the all range of scenarios was performed. Charts have been produced for every single parameter indicative of site conditions (i.e. region, field length, soil type, cultivation type) to also highlight possible clusters of more or less influential variables. For comparison's sake, costs per reducing a kg of P have been averaged between consecutive buffer widths, thus creating new width ranges of 2 meters size each, covering the 4-20 meters referential range. Besides, all values

[31]Load reductions have been estimated considering a buffer strip with a fixed width of 10 meters.

have been sorted in increasing order from the lowest to the highest costly scenario. With regard to the first variable “region”, data are thus summarized (Table 7) and presented in charts (Figure 5 and 6).

Table 7 Average of costs per kg of P reduced within each segment in relation to regions (01b and 06)

	regions							
width range	**4-6**	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
scenarios	**average of cost per kg of P reduced within each segment**							
01b_2_400_scl_ww	23,86	30,83	37,46	43,86	50,08	56,16	62,11	67,96
01b_2_200_scl_ww	53,93	69,02	83,37	97,19	110,60	123,68	136,49	149,07
01b_2_400_scl_sb	113,78	137,13	159,77	181,67	202,94	223,66	243,91	263,76
01b_2_200_scl_sb	119,23	141,95	164,26	185,94	207,03	227,59	247,70	267,41
01b_2_400_sl_ww	209,37	261,47	311,07	358,77	404,99	450,00	494,01	537,15
01b_2_200_sl_ww	462,44	574,60	681,50	784,35	884,00	981,03	1075,87	1168,83
01b_2_400_sl_sb	914,46	935,22	1006,76	1090,10	1176,88	1264,40	1351,61	1438,08
01b_2_200_sl_sb	1380,56	1517,45	1685,86	1859,74	2033,28	2204,76	2373,70	2540,04
	4-6	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
	average of cost per kg of P reduced within each segment							
06_2_400_scl_ww	176,49	227,54	276,15	323,02	368,56	413,02	456,58	499,39
06_2_200_scl_ww	492,50	621,95	744,99	863,35	978,07	1089,86	1199,20	1306,46
06_2_400_sl_ww	965,32	1222,57	1467,06	1702,28	1930,32	2152,58	2370,02	2583,35
06_2_400_scl_sb	1078,22	1129,97	1228,83	1338,41	1450,67	1563,01	1674,49	1784,74
06_2_200_scl_sb	1612,73	1792,11	2001,35	2214,80	2426,78	2635,74	2841,32	3043,54
06_2_200_sl_ww	2003,57	2512,37	2996,32	3461,78	3912,81	4352,12	4781,66	5202,87
06_2_400_sl_sb	3384,82	3258,29	3427,36	3663,30	3921,42	4187,35	4455,38	4722,95
06_2_200_sl_sb	4552,06	4911,98	5410,83	5938,31	6469,57	6996,95	7517,87	8031,58

To better spot potential dissimilarities, the same vertical axis proportion has been applied and a line indicating the national overall cost per reducing a kg of P (3250 SEK/kg of P reduced) have been included in the charts.

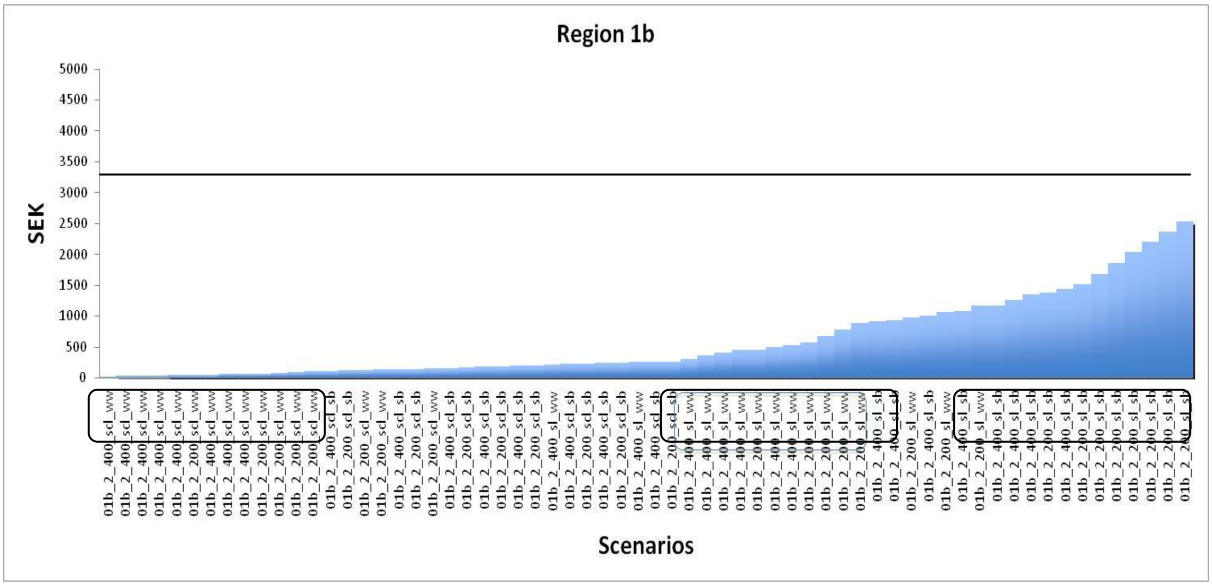

Figure 5 Marginal cost increase region 1b

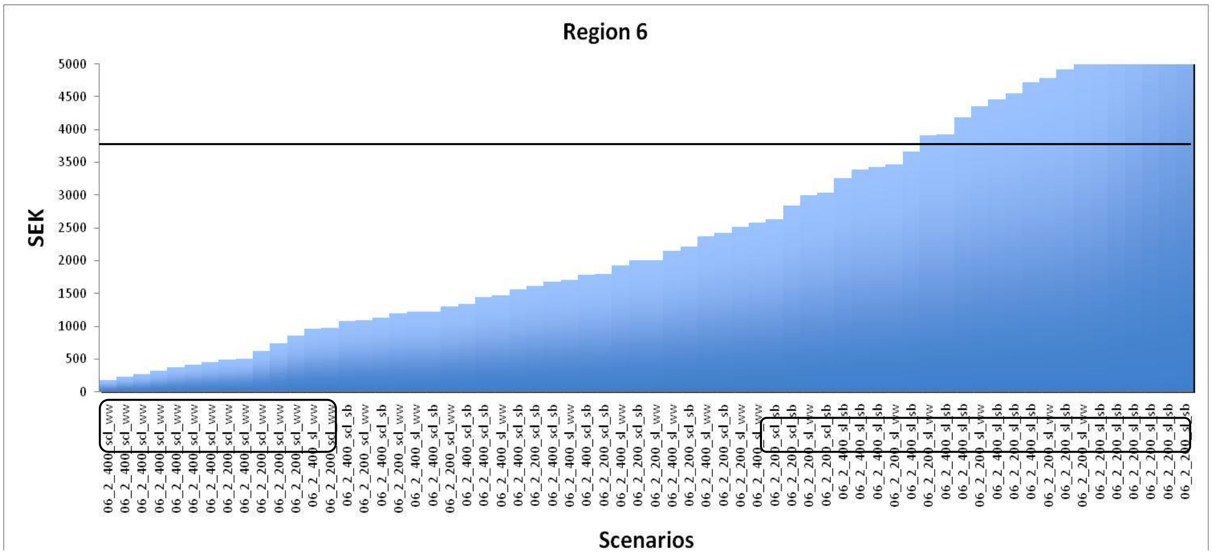

Figure 6 Marginal cost increase region 6

The same approach has been adopted for the other variables: field length (Table 8, Figure 7 and 8), soil type (Table 9, Figure 9 and 10) and cultivation (Table 10, Figure 11 and 12).

Table 8 Average of costs per kg of P reduced within each segment in relation to field lengths (200 and 400)

field length								
width range	**4-6**	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
scenarios	**average of cost per kg of P reduced within each segment**							
01b_2_200_scl_ww	53,93	69,02	83,37	97,19	110,60	123,68	136,49	149,07
01b_2_200_scl_sb	119,23	141,95	164,26	185,94	207,03	227,59	247,70	267,41
01b_2_200_sl_ww	462,44	574,60	681,50	784,35	884,00	981,03	1075,87	1168,83
06_2_200_scl_ww	492,50	621,95	744,99	863,35	978,07	1089,86	1199,20	1306,46
01b_2_200_sl_sb	1380,56	1517,45	1685,86	1859,74	2033,28	2204,76	2373,70	2540,04
06_2_200_scl_sb	1612,73	1792,11	2001,35	2214,80	2426,78	2635,74	2841,32	3043,54
06_2_200_sl_ww	2003,57	2512,37	2996,32	3461,78	3912,81	4352,12	4781,66	5202,87
06_2_200_sl_sb	4552,06	4911,98	5410,83	5938,31	6469,57	6996,95	7517,87	8031,58
	4-6	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
	average of cost per kg of P reduced within each segment							
01b_2_400_scl_ww	23,86	30,83	37,46	43,86	50,08	56,16	62,11	67,96
01b_2_400_scl_sb	113,78	137,13	159,77	181,67	202,94	223,66	243,91	263,76
06_2_400_scl_ww	176,49	227,54	276,15	323,02	368,56	413,02	456,58	499,39
01b_2_400_sl_ww	209,37	261,47	311,07	358,77	404,99	450,00	494,01	537,15
01b_2_400_sl_sb	914,46	935,22	1006,76	1090,10	1176,88	1264,40	1351,61	1438,08
06_2_400_sl_ww	965,32	1222,57	1467,06	1702,28	1930,32	2152,58	2370,02	2583,35
06_2_400_scl_sb	1078,22	1129,97	1228,83	1338,41	1450,67	1563,01	1674,49	1784,74
06_2_400_sl_sb	3384,82	3258,29	3427,36	3663,30	3921,42	4187,35	4455,38	4722,95

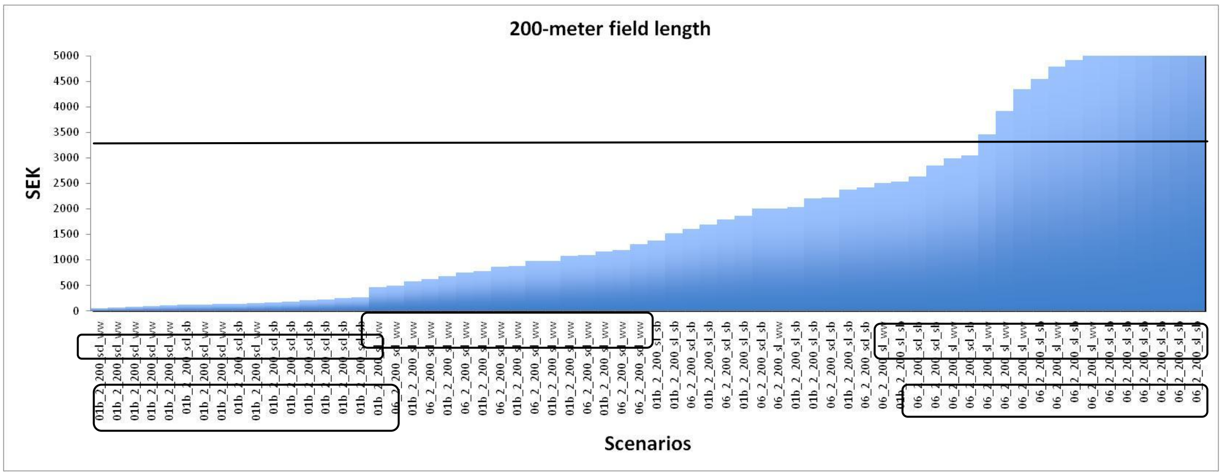

Figure 7 Marginal cost increase 200-meter field length

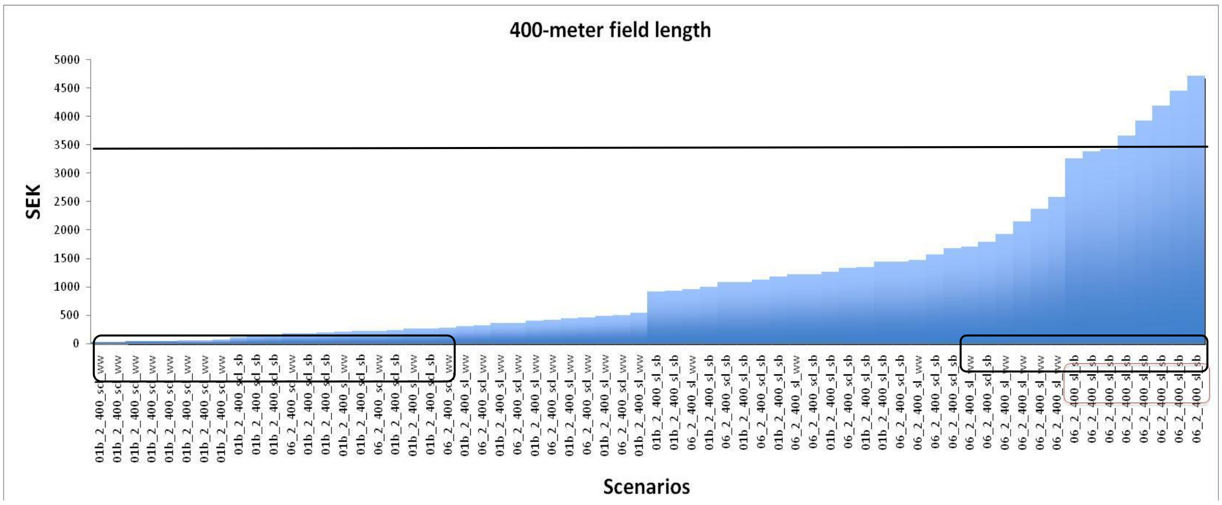

Figure 8 Marginal cost increase 400-meter field length

	soil type							
width range	**4-6**	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
scenarios	**average of cost per kg of P reduced within each segment**							
01b_2_400_scl_ww	23,86	30,83	37,46	43,86	50,08	56,16	62,11	67,96
01b_2_200_scl_ww	53,93	69,02	83,37	97,19	110,60	123,68	136,49	149,07
01b_2_400_scl_sb	113,78	137,13	159,77	181,67	202,94	223,66	243,91	263,76
01b_2_200_scl_sb	119,23	141,95	164,26	185,94	207,03	227,59	247,70	267,41
06_2_400_scl_ww	176,49	227,54	276,15	323,02	368,56	413,02	456,58	499,39
06_2_200_scl_ww	492,50	621,95	744,99	863,35	978,07	1089,86	1199,20	1306,46
06_2_400_scl_sb	1078,22	1129,97	1228,83	1338,41	1450,67	1563,01	1674,49	1784,74
06_2_200_scl_sb	1612,73	1792,11	2001,35	2214,80	2426,78	2635,74	2841,32	3043,54
	4-6	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
	average of cost per kg of P reduced within each segment							
01b_2_400_sl_ww	209,37	261,47	311,07	358,77	404,99	450,00	494,01	537,15
01b_2_200_sl_ww	462,44	574,60	681,50	784,35	884,00	981,03	1075,87	1168,83
01b_2_400_sl_sb	914,46	935,22	1006,76	1090,10	1176,88	1264,40	1351,61	1438,08
06_2_400_sl_ww	965,32	1222,57	1467,06	1702,28	1930,32	2152,58	2370,02	2583,35
01b_2_200_sl_sb	1380,56	1517,45	1685,86	1859,74	2033,28	2204,76	2373,70	2540,04
06_2_200_sl_ww	2003,57	2512,37	2996,32	3461,78	3912,81	4352,12	4781,66	5202,87
06_2_400_sl_sb	3384,82	3258,29	3427,36	3663,30	3921,42	4187,35	4455,38	4722,95
06_2_200_sl_sb	4552,06	4911,98	5410,83	5938,31	6469,57	6996,95	7517,87	8031,58

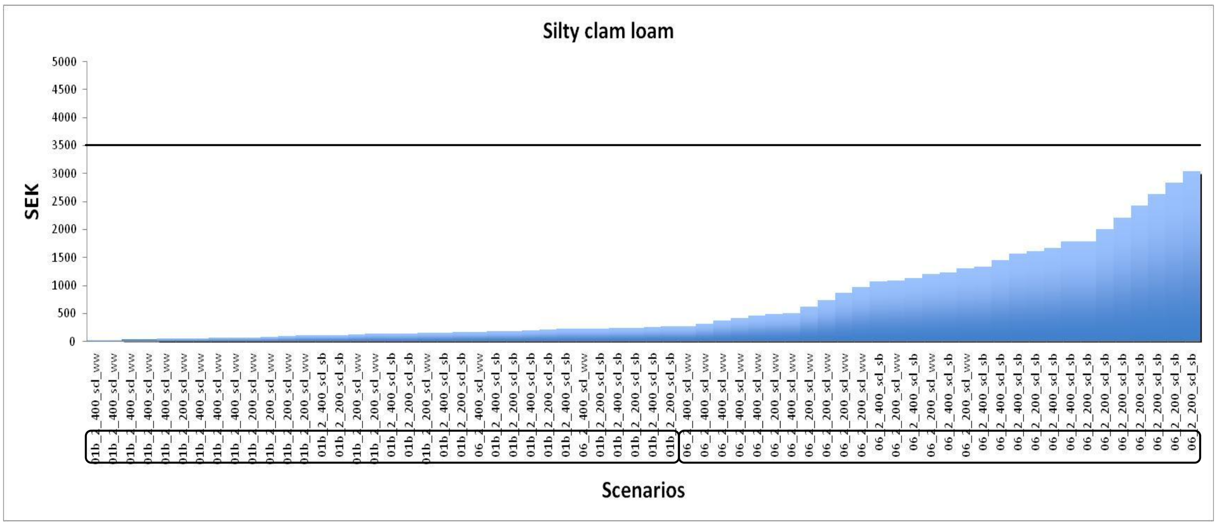

Figure 10 Marginal cost increase silty clay loam

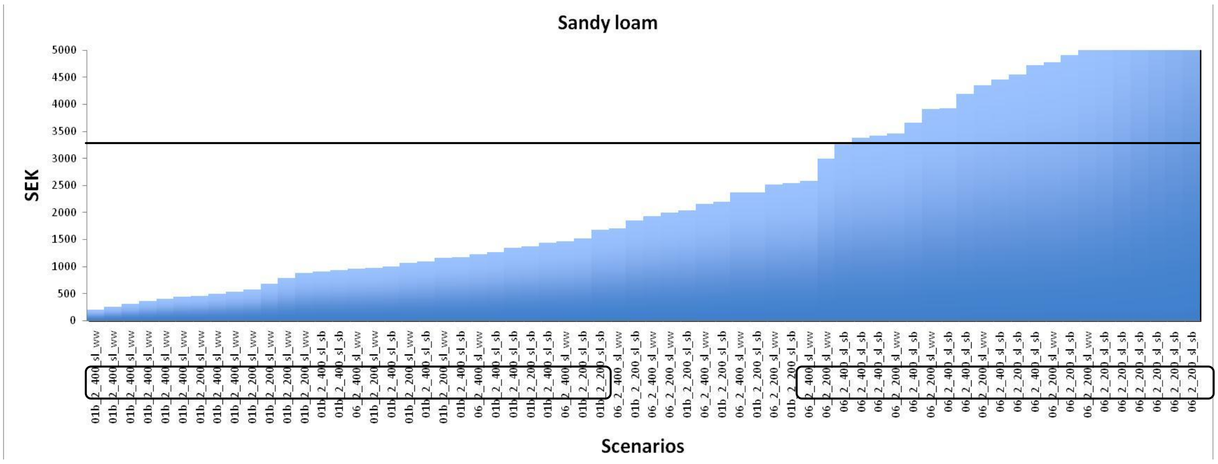

Figure 11 Marginal cost increase sandy loam

Table 10 Average of costs per kg of P reduced within each segment in relation to cultivation (ww and sb)

	cultivation							
width range	**4-6**	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
scenarios	**average of cost per kg of P reduced within each segment**							
01b_2_400_scl_ww	23,86	30,83	37,46	43,86	50,08	56,16	62,11	67,96
01b_2_200_scl_ww	53,93	69,02	83,37	97,19	110,60	123,68	136,49	149,07
06_2_400_scl_ww	176,49	227,54	276,15	323,02	368,56	413,02	456,58	499,39
01b_2_400_sl_ww	209,37	261,47	311,07	358,77	404,99	450,00	494,01	537,15
01b_2_200_sl_ww	462,44	574,60	681,50	784,35	884,00	981,03	1075,87	1168,83
06_2_200_scl_ww	492,50	621,95	744,99	863,35	978,07	1089,86	1199,20	1306,46
06_2_400_sl_ww	965,32	1222,57	1467,06	1702,28	1930,32	2152,58	2370,02	2583,35
06_2_200_sl_ww	2003,57	2512,37	2996,32	3461,78	3912,81	4352,12	4781,66	5202,87
	4-6	**6-8**	**8-10**	**10-12**	**12-14**	**14-16**	**16-18**	**18-20**
	average of cost per kg of P reduced within each segment							
01b_2_400_scl_sb	113,78	137,13	159,77	181,67	202,94	223,66	243,91	263,76
01b_2_200_scl_sb	119,23	141,95	164,26	185,94	207,03	227,59	247,70	267,41
01b_2_400_sl_sb	914,46	935,22	1006,76	1090,10	1176,88	1264,40	1351,61	1438,08
06_2_400_scl_sb	1078,22	1129,97	1228,83	1338,41	1450,67	1563,01	1674,49	1784,74
01b_2_200_sl_sb	1380,56	1517,45	1685,86	1859,74	2033,28	2204,76	2373,70	2540,04
06_2_200_scl_sb	1612,73	1792,11	2001,35	2214,80	2426,78	2635,74	2841,32	3043,54
06_2_400_sl_sb	3384,82	3258,29	3427,36	3663,30	3921,42	4187,35	4455,38	4722,95
06_2_200_sl_sb	4552,06	4911,98	5410,83	5938,31	6469,57	6996,95	7517,87	8031,58

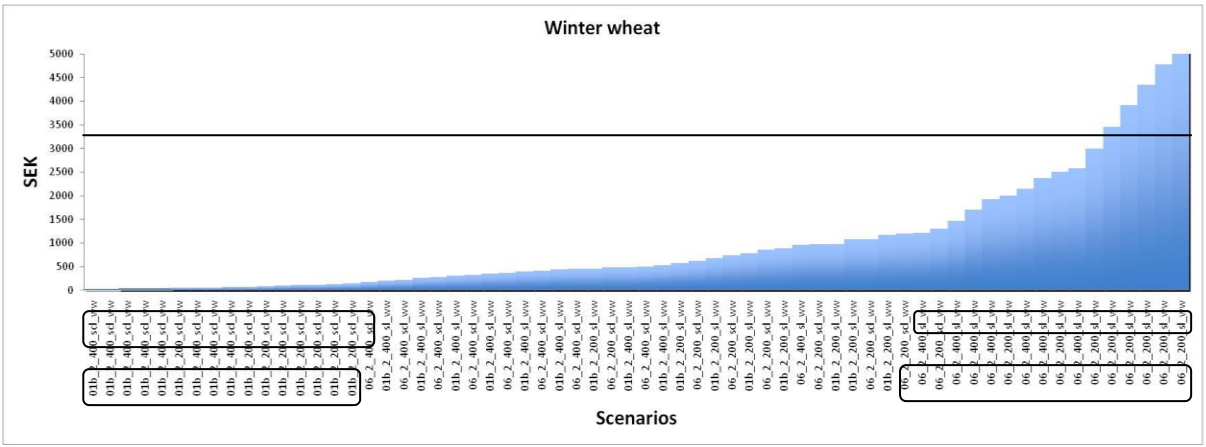

Figure 12 Marginal cost increase winter wheat

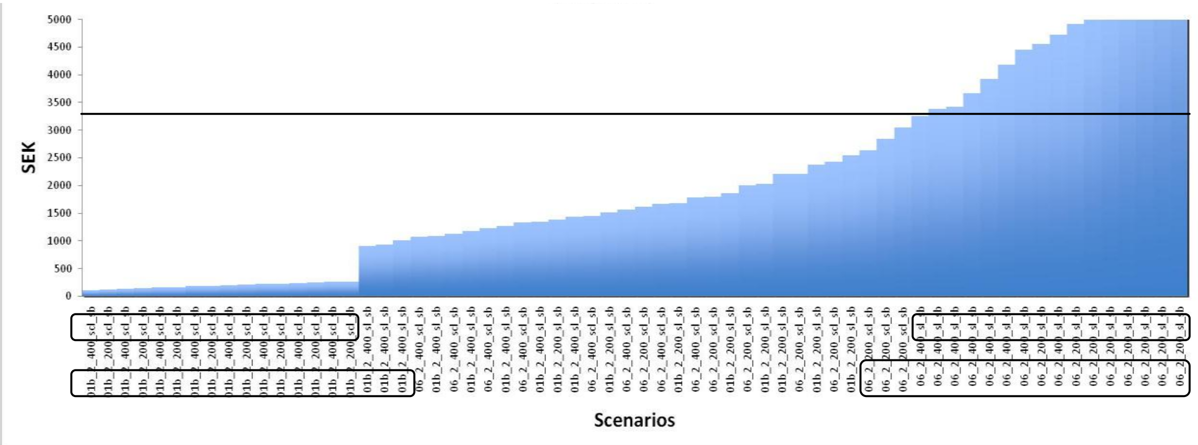

Figure 13 Marginal cost increase spring barley

What appears from the marginal cost charts interpretation is an evident tendency for costs to be lower in region 01b, for 400-meter fields, silty clay loamy soils and soils cultivated with winter wheat when compared to their respective paired site variable.

In general, the variable that most of all returned lowest costs is region 01b. However, when looking more closely to each chart, groupings of parameters could be detected. The trend that emerges indicates that, regardless the variable kept fixed in the analysis, parameters which returned always lowest costs where “region 01b”, “silty clay loam soil” and “winter wheat crop”. Field length did not appear to be a significant parameter.

One other consideration shall be put forward about the width ranges. Across all the simulations, it appeared that narrower widths did not necessarily return lower costs.

4. Conclusions

One conclusive consideration can thence be that the width of a buffer might not be that influential as other site conditions when looking at the cost-efficiency in reducing the load of phosphorous from agricultural fields. Besides, when thinking of under which condition is to prefer investing in the implementation of buffer zones, priority shall be directed where “region 01b”, “silty clay loam soils” and “winter wheat cultivation” conditions apply.

With regard to steep soils (with high slope levels), making suggestions gets problematic. They are generally less productive than flatter soils and, consequently, losses of income are lower. Consequently, the payment for installing buffers shall be lower according to the actual approach but it is not. Hence, considerations about the slope of the fields must be attentively taken into account and this is the main reason why only 2%-slope fields have been analyzed.

Moreover, regarding eventual suggestions on the most effective width, general assumptions are hard to be made. The Swedish national approach of considering a range of buffer strip width from a minimum of 6 to a maximum of 20 meters, seems cautionary. Neverthless, problems do arise when farmers apply and eventually benefit from the agri-environmental support programme for buffer zones even though their fields might not produce that high level of P losses (requiring thence investments in buffer zones) as well as when their income from cultivation is lower than the support payment. Under certain circumstances, it makes then participating in the programme more profitable for farmers but very little or not at all for the reducing P-losses target. One could adduce that the payment provided in Sweden for reducing a kg of P (3250 SEK) is quite averaged over the different regions. Similar consideration can be drawn also on the amount of money farmers get in the form of the total subsidy (335 to 447 €) for participating in the buffer zone scheme. This approach looks similar to the one adopted in Denmark (348 €), though less if compared to Norway (85 to 170 €). However, as the study has identified, local differences in costs and efficiencies are likely to appear and be significant.

Insofar, a more down-scaled and differentiated payment scheme for buffer zones based on more localized and easy-to-establish parameters (such as climate and soil type, but also the load of P and reduction targets) would produce a more cost-effective approach. For instance, in some cases, it would be even more appropriate and cost-effective investing in wide buffer strips (e.g. 01b_2_200_scl_sb, 277.17 SEK per kg of P reduced, 20-meter buffer) whereas saving money when high efficiency can be achieved with short buffers (e.g. 01b_2_200_scl_ww, 46.17 SEK per kg of P reduced, 4-meter buffer). Similar considerations have been pointed out in Denmark as well: a spatially differentiated design of buffer zone width, which varies from the referential 10 meters in relation to local load potential, would be a more cost-effective instrument than having a standard width (Notat: Effekt på fosforudledning af 10 m brede randzoner, Aahrus University, January 2011).

Obviously, all the results presented in the study are determined from a model and thus, they shall be critically evaluated. The ICECREAM DB model proved to have a pretty good balance between accuracy and ease of use, as well as a good history of implementation and supporting background studies (e.g. Liu, 2010). Nevertheless, the model itself has own assumptions that eventually influence the outcomes. For instance, the type of flow set as default does not consider possible natural depressions in the topography of the area, the vegetation in the buffer can only be set as grass, with even low roughness coefficient. The possibility of having different type of vegetation could be included and exploited in order to maximize efficiency (as suggested already by some studies cited in the literature). One advantage of this study was to include continuous metereological data for a long time series (20-years). As a matter of fact, hydrology of the area is a paramount parameter when dealing with losses through surface runoff especially at high latitudes with pretty cold climate and freezing/thawing cycles. This can be considered as an improvement if compared to the study of Dosskey et al. (2008) and other studies cited in the literature review (when experiments were carried out under simulated runoff conditions or under limited time span).

Though potential for buffer zones for targeting multiple environmental pressure (such as nitrogen retention and biodiversity) is acknowledged, the cost-efficiency analysis has been performed only for phosphorous losses and under specific parameters. Nevertheless, the afore-mentioned benefits do accrue (though with variable significance) whenever a buffer zone is set in place. These must be seen as positive externalities which weigh in favour of the setting of a buffer itself and eventually increase its cost-effectiveness. Further and in-depth research on the cost effectiveness of nitrogen retention and biodiversity enhancement in a buffer would definitely integrate and complete the present study and help understanding the large potential of such landscape designs.

With specif regard to phosphorous, further research is also needed for better quantifying its leaching reduction rates and its relations with the soil phosphorous status. Studies on a larger scale (i.e. catchment size) are also required.

Acknowledgments

This project would not have been possible without the support of several people. I wish to express my gratitude to my supervisor, prof. Dennis Collentine who was constantly and truly helpful and supportive throughout these months of work. Special thanks to my co-supervisor from Copenhagen University, Brian H. Jacobsen, for his valuable assistance, guidance and contribution. Sincere gratitude also to the people who helped, to larger or lesser extents, to the completion of this project, namely Frank Schmieder, Martin Futter and Anne-Grete B. Blankenberg.

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Appendices

APPENDIX A. Values utilised in the ICECREAM simulations

Climate regions

1b	Produktionsområde (bas PO18)	Skåne- och Hallands slättbygd, Hallandsdelen
	Climate station	Halmstad
	Annual average runoff	538 (mm)
	Annual average precipitation	1140 (mm)
	Annual average temperature	8°C
6	Produktionsområde (bas PO18)	Mälar- och Hjälmarbygden.
	Climate station	Stockholm
	Annual average runoff	200 (mm)
	Annual average precipitation	831 (mm)
	Annual average temperature	7°C

Soil texture classes

Sandy loam (S03)	Texture class porosity	0,453 (m^3 m^{-3})
	Wilting point (pF 4.2)	0,095 (m^3 m^{-3})
	Saturated hydraulic conductivity	2,59 (12) cm min^{-1}
Silty clay loam (S08)	Texture class porosity	0,471(m^3 m^{-3})
	Wilting point (pF 4.2)	0,208 (m^3 m^{-3})
	Saturated hydraulic conductivity	0,15 (12) cm min^{-1}

APPENDIX B. Summary of simulation runs

01b_2_200_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	1,202680538	0
4	0,924154	23,15881311
12	0,747416	37,85415359
20	0,65729681	45,34734797
30	0,597213319	50,34314599

01b_2_400_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	1,311733419	0
8	0,9092632	30,68231801
24	0,734546548	44,00184237
40	0,648513557	50,56056759
60	0,594592733	54,67122171

01b_2_200_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	2,921298971	0
4	1,637553	43,94435434
12	1,160212	60,28437995
20	0,979275	66,47809726
30	0,855120552	70,72807129

01b_2_400_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	3,502966367	0
8	1,770408814	49,45972559
24	1,27002731	63,74423341
40	1,105390814	68,44414994
60	0,980157929	72,01920241

01b_2_200_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,30306549	0
4	0,256176	15,4717353
12	0,213359	29,59970478
20	0,190502	37,14163902
30	0,168298	44,46810828

01b_2_400_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,31132949	0
8	0,247583	20,47557087
24	0,205000419	34,15322823
40	0,182482295	41,38611959
60	0,159912724	48,63553608

01b_2_200_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,448085714	0
4	0,300337476	32,97320878
12	0,231218	48,39871198
20	0,201776	54,96932985
30	0,176454	60,62048078

01b_2_400_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,518291619	0
8	0,309566	40,27184916
24	0,236111	54,44437237
40	0,206477	60,16200293
60	0,178192695	65,61922117

01b_10_200_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	14,3070874	0
4	8,3298703	41,77801486
12	6,073201362	57,55109903
20	5,066338729	64,58860852
30	4,367823024	69,47091395

01b_10_400_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	15,1888127	0
8	8,300274619	45,35270939
24	6,136203324	59,60050701
40	5,248566257	65,44452579
60	4,604659267	69,68387618

01b_10_200_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	31,87267961	0
4	19,00565747	40,37006709
12	13,9151857	56,34133725
20	11,6696117	63,38678816
30	10,15746718	68,13111635

01b_10_400_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	32,89563365	0
8	18,31944639	44,31040123
24	13,4543917	59,09976428
40	11,59361212	64,75638
60	10,26755673	68,78747847

01b_10_200_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	3,431559081	0
4	2,132716305	37,84993193
12	1,606252029	53,19177113
20	1,413218129	58,8170247
30	1,212199148	64,67497371

01b_10_400_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	3,642674319	0
8	2,202766095	39,52887625
24	1,679038314	53,90643886
40	1,464621605	59,79268316
60	1,2494476	65,69971701

01b_10_200_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	9,564041862	0
4	6,063494643	36,60112816
12	4,432778452	53,65162014
20	3,596139324	62,399377
30	3,234170238	66,18406439

01b_10_400_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	10,78877815	0
8	5,748053043	46,7219275
24	4,36588291	59,5331107
40	3,930663533	63,56711131
60	3,554483543	67,05388234

06_2_200_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,2124684	0
4	0,171262095	19,39408626
12	0,139780271	34,21126557
20	0,116713743	45,06771696
30	0,100464267	52,71566658

06_2_400_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,216416871	0
8	0,163417481	24,4894911
24	0,133251548	38,42829963
40	0,111430414	48,51121655
60	0,096204129	55,54684441

06_2_200_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,367878724	0
4	0,22627669	38,49149847
12	0,174439133	52,5824349
20	0,14656979	60,1581225
30	0,126622667	65,58032349

06_2_400_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,499111557	0
8	0,25960099	47,98738143
24	0,204183981	59,09051233
40	0,174255519	65,08685953
60	0,147768467	70,39369965

06_2_200_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,092545881	0
4	0,077947157	15,77457976
12	0,065429867	29,3000769
20	0,05728749	38,09828175
30	0,048843429	47,22247163

06_2_400_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,094620733	0
8	0,075540248	20,16522705
24	0,062994552	33,4241554
40	0,054915805	41,96218648
60	0,046711276	50,63314926

06_2_200_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,115475238	0
4	0,080375324	30,39605275
12	0,06781441	41,27363525
20	0,060784881	47,36111226
30	0,053546071	53,62982375

06_2_400_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,121918429	0
8	0,076620495	37,15429559
24	0,065268443	46,46548219
40	0,058340919	52,14757955
60	0,051321814	57,90479349

06_10_200_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	3,086650329	0
4	1,72670479	44,05894395
12	1,260857138	59,15128039
20	1,087781024	64,75852759
30	0,911727176	70,46224615

06_10_200_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	7,458627971	0
4	4,210273214	43,55163938
12	3,206330386	57,01179362
20	2,797915952	62,48752501
30	2,423669629	67,5051546

06_10_200_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	0,898056681	0
4	0,520154162	42,08002981
12	0,383603581	57,28514813
20	0,303970652	66,15239786
30	0,261534919	70,87768238

06_10_200_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	4,599848143	0
4	2,385720762	48,13479298
12	1,833603619	60,13773581
20	1,538178257	66,56023831
30	1,229637229	73,26787341

06_10_400_scl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	3,332674467	0
8	1,77363391	46,78046334
24	1,329035676	60,12104724
40	1,172009729	64,83275699
60	0,980908767	70,56691926

06_10_400_scl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	7,850726171	0
8	4,138548067	47,28451896
24	3,249855505	58,60439616
40	2,867672014	63,47252532
60	2,480145181	68,40871625

06_10_400_sl_sb

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	1,212901124	0
8	0,615093186	49,28744201
24	0,484778862	60,03146074
40	0,406706638	66,46827758
60	0,329805071	72,80857731

06_10_400_sl_ww

buffer_width	Tot_P_losses (kg/ha/y)	% trapping efficiency
0	5,350610824	*
8	2,661428743	
24	1,159579648	
40	1,813032105	
60	1,461356505	

*Efficiency.rates.in.the.last.simulation.have.not.been.calculated.due.to.unexplained.model.behaviour.with.regard.to.buffer.width.of.24m