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AgriFood economics centre

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Abstract

This study is the first to estimate the effect of Agri-Environmental Schemes (AES) on nutrient leaching using abatement data and water samples on a large scale. This unique combination of data sources identifies all farms located upstream from a given water sampling site. By using watersheds that cover 91% of the Swedish land area and AES payments to 83% of Swedish farms, the study is almost a full population evaluation. A watershed fixed-effect model estimates whether within-watershed variation in AES payments affects nitrogen and phosphorus concentrations in water samples. For the period 1997-2013, the study finds that higher uptake of the AES *Wetland*, *Catch crop/No autumn tillage*, *Environmental protection measures* and *Culturally significant landscape elements* was associated with reduced nutrient leaching. However, uptake of *Grassed buffer zones*, *Pastures and meadows* and *Organic production* was associated with increased nutrient leaching.

JEL classification: Q53, Q58, C1

Key words: nutrient leaching, agri-environmental schemes, evaluation

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1. Introduction

The largest hypoxic zones world-wide caused by nutrient overloading are in the Baltic Sea (the hypoxic zone in the Gulf of Mexico is the world's second-largest) and an estimated 40% of nitrogen (N) and 24% of phosphorus (P) leaching to Swedish freshwaters and the Baltic Sea are from agricultural land (Brandt and Ejhed, 2003). Many different abatement measures are in use to reduce the nutrient load and it is important to assess the impact of each measure. However, agricultural nutrient leachate is a non-point source emission and, in contrast to point source emissions, difficult to measure (Horan and Ribaudo, 1999). Problems in tracing the precise source of nutrients from agricultural land mean that it is difficult to determine which measures are effective (Primdahl et al., 2003; Balana et al., 2011; Kling, 2011). Thus, there is a need for new techniques to identify the impact of environmental measures on nutrient leaching.

Agri-Environmental Schemes (AES), part of the second Pillar of the EU Common Agricultural Policy (CAP), are a targeted tool for reducing nutrient leaching. This study estimates the effect of AES on nutrient leaching using a unique combination of data sources. Our approach merges information over a period of seventeen years (1997-2013) on the concentrations of N and P in water samples from about 4,300 lakes and watercourses in Sweden with information on watersheds, retention rates and agri-environmental subsidies paid to about 39,000 farms in the vicinity of these waters. In the modelling literature, some studies have evaluated the impact of different land use scenarios on water quality (Kling et al., 2014), but to our knowledge no previous study has tried to merge abatement data and water samples on a large scale.

The key to our research design is linking water quality sampling sites to upstream farms. We use GIS maps of Sweden's about 50,000 minor basins (sub-watersheds), which describe the upstream-downstream relationships between all basins. Using coordinates, we match all sampling sites and farms to their respective minor basin, allowing us to identify all farms located upstream from a given sampling site. We then use a watershed fixed-effect model to control for differences in nutrient concentrations due to time-invariant watershed characteristics, e.g. hydrology, soil and vegetation. Consequently, the variation used to identify the effect of the respective subsidies on nutrient leaching is the variation in the amount of subsidies paid *within* a given watershed. To model the absorption of nutrients (through natural and artificial biochemical processes) along the way to the sampling site, we weight the subsidies by simulated nutrient retention rates.

Like most other EU countries, Sweden has a wide range of AES (e.g. payments for establishing grassed buffer zones or wetlands). However, despite agricultural field trials (see e.g. Aronsson et al., 2011), trend analysis of agricultural rivers (Kyllmar et al., 2006; Ulén and Fölster, 2007; Fölster et al., 2012) and model simulations (see e.g. Torstensson and Aronsson, 2000; Johnsson et al., 2008; Liu et al., 2012), the impact of individual AES is still unknown. The problem is that since nutrient leaching is a complex process: i) field trials have low external validity because controlled environments remove much of the complexity; and ii) model simulations are associated with large uncertainties, e.g. due to poor input data and failure to incorporate recent findings in the models (Bergström et al., 2007).

The literature also fails to evaluate the first stage of the relationship between AES and nutrient leaching, which is the relationship between AES and farmers' implementation of measures. Moreover, studies that investigate the effect of AES on agricultural practices rarely investigate the chief outcome: the environmental impact. However, the entire chain is important and, from an evaluation point of view, omitting the first stage introduces a bias. Investigating textbook implementation on representative farm types – which field trials often do – fails to evaluate the overall link between AES and nutrient leaching (Balana et al., 2011). For example, the environmental impact of a relevant measure (on which subsidy payments are conditioned) may be low because of poor implementation. Hence, it is important to evaluate the average impact of an AES and insufficiencies in implementation should be encompassed in the analysis. The average impact also depends on whether the AES targets areas where the reduction in nutrient leaching is potentially significant. Therefore, in the present study we incorporates not only the hydrological link between an action and nutrient leaching, but also the effect on farmers' implementation of measures and whether the AES target high-impact watersheds. Moreover, an AES may have an impact – a co-benefit or unintended impact – on nutrient leaching even if the target objective is e.g. biodiversity (Balana et al., 2011). Thus, in this study we also consider AES where reduced nutrient leaching is not a specified objective.

2. CAP and the environmental action programme

The 1991 Nitrates Directive (91/676/EC) aims to protect water quality across Europe and requires member countries to designate Nitrate Vulnerable Zones (NVZ) and set up compulsory and voluntary Action Programmes, e.g. measures limiting fertiliser application and setting a minimum storage capacity for livestock manure. In NVZ, all Action Programmes are compulsory. In Sweden, the major agricultural regions are included in the NVZ.[1]

Besides the regulatory Action Programmes, the second Pillar of the CAP contains a wide variety of AES, a handful of which aim at reducing nutrient leaching. Some AES have multiple aims, e.g. the support for cultivated grassland aims at reducing nutrient leaching and, through a varied agricultural landscape, preserving biodiversity. Two other subsidies without an expressed nutrient control aim are that for (semi-natural) pastures and meadows and that for culturally significant landscape elements, which provide large payments and may potentially have an impact on nutrient leaching, although their specified aim is to facilitate

[1]This study is not an evaluation of the Nitrate Directive, which was implemented before the study period and therefore did not affect the results in this study.

conservation of a varied agricultural landscape and biodiversity. Thus, in this study we analyse the following AES:

Subsidies with a nutrient leaching aim

- Catch crops (1997-2013)
- No autumn tillage (2001-2013)
- Grassed buffer zones (1997-2013)
- Wetlands (1997-2013)
- Organic production (1997-2013)
- Environmental protection measures (including a wide variety of measures, e.g. soil mapping, having a crop production plan and calculating nutrient balances (2007-2013)

Subsidies with dual aims or without an expressed nutrient aim

- Cultivated grassland (with a nutrient aim) (1997-2002 and 2007-2013)
- Open and varied landscape (no nutrient aim) (1997-2016)
- Culturally significant landscape elements (no nutrient aim) (1997-2013)
- Pastures and meadows (no nutrient aim) (1997-2013).

During the study period (1997-2013), three different agri-environmental programmes (up until 2000, 2001-2006 and 2007-2013) were in place. Although the programmes were similar, individual AES requirements differed and may (as will be discussed later) have had an impact on nutrient leaching, but it is outside the scope of this study to explore the different programmes and changes in AES requirements.

Most subsidies were available for the entire period and only *No autumn tillage* and *Environmental protection measures* were introduced later, in 2001 and 2007, respectively. In the data from The Swedish Board of Agriculture the payments for *No autumn tillage* and *Catch crops* are merged, since although implementation of both practices is not required, the combination is assumed to be best practice and therefore we could not analyse them separately. The subsidies *Cultivated grassland* and *Open and varied landscape* target and preserve the same land (grass leys on arable land) and before 2003 both subsidies were available. However, between 2003 and 2006, *Cultivated grassland* was removed, and in 2007 *Cultivated grassland* was reintroduced and replaced the subsidy *Open and varied landscape*. In this study we merged the subsidies, and treat them as payments aimed at preserving the same land.

In the following section, we explore the anticipated impact of each AES on nutrient leaching and present findings from previous research on the impact of abatement measures on nutrient leaching.

3. Previous research

Because nutrient leaching is strongly weather- and soil-dependent, in our review of the literature we focus mainly on Swedish and other Scandinavian studies.

The area closest to the present study is the trend analysis literature. Using small samples (12 to 65 rivers or watersheds), trend analysis has shown that N and P concentrations in Swedish watercourses have decreased since the 1990s (see e.g. Kyllmar et al., 2006; Ulén and Fölster, 2007; Fölster et al., 2012). This decrease has been attributed to greater areas of catch crops and grassed buffer zones, less autumn tillage and fewer livestock (Ulén and Fölster, 2007; Fölster et al., 2012). However, these studies use only small samples and model the relationship between land use and nutrients inadequately; the panel dimension of the data is not utilised and the impact of different measures on nutrient leaching is not modelled simultaneously.

Another problem is that the watersheds analysed are often a selective sample of watersheds in high leaching environments. For example, Kyllmar et al. (2006) analysed 27 small and homogeneous watersheds in leaching-sensitive regions. These watersheds were chosen by the Swedish Environmental Protection Agency (Swedish EPA) and are monitored under the EU Nitrate Directive.[2] Ulén and Fölster (2007) analysed 12 rivers in watersheds with at least 30% agricultural land. The largest sample of watersheds was used in Fölster et al. (2012), but it was also the most selective sample; from 3,600 watersheds they chose the 65 with the highest leaching potential from agricultural land. However, any overall evaluation of measures should use a random sample of watersheds, as otherwise the results cannot be generalised to watersheds outside the sample.

Another relevant area is the field experiment literature. Field experiments have higher precision, but even lower external validity, than trend analysis. Catch or cover crops are fast-growing crops used between seasons to capture N in soil and reduced autumn tillage is assumed to reduce leaching over the winter. Substantial evidence from field trials shows that catch crops and abandoning autumn soil tillage can reduce N leaching (e.g. Hansen and Djurhuus, 1997; Askegaard et al., 2011).

For countries with a similar climate to Sweden, we only found buffer zone field experiments in neighbouring countries. In Finland, buffer zones seem to reduce P leaching from clay soil (Uusi-Kämppä, 2005, 2008) and in Norway they reduce both N and P leaching (Sövik et al., 2012). The lack of Swedish studies is problematic because the effectiveness of buffer zones varies widely (Mayer et al., 2007) and the mechanisms responsible for removing nitrate within buffers are not thoroughly understood (Correll, 1997; Mayer et al., 2007).

In cold temperate regions (Nordic countries, Switzerland and the state of Illinois, US), constructed wetlands reduce P losses, but the impact varies between wetlands (Braskerud et al., 2005). For seven constructed wetlands in southern Sweden, water samples showed a large reduction in N concentration (Strand and Weisner, 2013) and retention of P (Johannesson et al., 2015). However, these seven wetlands were not randomly chosen and, on comparing the reduction in N with simulated reductions for 2,400 randomly chosen wetlands, the results

[2]The Nitrate Directive requires member states to analyse nitrate concentrations in their waters.

differed considerably and were much smaller for the randomly chosen wetlands (Strand and Weisner, 2013). Thus, field experiments may provide biased results of the general impact of measures.

Compared with schemes with a clearly defined aim of reducing nutrient leaching (buffer zones, wetlands and catch crops), the impact of the cultivated grassland scheme depends on the counterfactual land use. If the alternative to cultivated grasslands is annual crops, grassland leaches less (Gustafson, 1987; Ulén, 1988), but if the support increases the area of fertilised grassland (intensive farming) at the expense of land in fallow (extensive grassland), it is likely to increase leaching.

The support for pastures and meadows may increase nutrient leaching because the payment is conditional on grazing the land. Research has found that grazing increases nutrient leaching (see e.g. Dougherty et al., 2004; Monaghan, 2007; Parvage et al., 2011), and for livestock farming urine is the main cause of N leaching (Monaghan, 2007). On the other hand, if pastures and meadows are overgrown, nutrient leaching may decrease.

In essence, what distinguishes organic production from conventional production in Sweden is that mineral fertilisers (and chemical pesticides) are not permitted. A Swedish experimental study comparing leaching in cropping systems with and without mineral fertilisers found similar levels of N and P leaching. On the other hand, yields were lower in the organic system without mineral fertilisers, which resulted in higher leaching per unit of harvested crop (Aronsson et al., 2007).

The impact of preserving culturally significant landscape elements is uncertain, but since this AES compensates farmers for the management of e.g. small ponds and open ditches, it may reduce nutrient leaching because small ponds and open ditches resemble wetlands. Open ditches are the most common landscape element in the AES, e.g. in 2006 almost 35,000 km of open ditches were managed (SLU, 2010).

Finally, the environmental protection AES includes a wide variety of measures, e.g. soil mapping, having a crop production plan and calculating nutrient balances. It is uncertain whether these measures have an impact on nutrient leaching, but a recent evaluation of the Swedish advisory service *Greppa näringen* (*Capture nutrients*), which involves a similar set of measures, found that the service reduced the N surplus on farms (Höjgård and Nordin, 2015).

To sum up, the literature shows that the impact of abatement measures depends on watershed characteristics and the alternative land use, and therefore the impact varies between watersheds, creating heterogeneous effects. Thus, when evaluating a practice it is not sufficient to consider only a leaching friendly environment, because it is the average effect of measures that is important. Accordingly, if a large share of a subsidy is misdirected, or imperfectly implemented, the average effect may be small or non-existent, whereas if targeted and implemented correctly the subsidy effect would be large.

4. Data

The panel data used in this study consist of two main components; AES payments to farms and water quality data. The Swedish Board of Agriculture holds data on all agricultural and environmental policy payments to all Swedish farms from 1997 to 2013. By merging these data with the Statistical Business Register (provided by Statistics Sweden), we identify the location of each farm. Excluding farms with multiple farmstead locations and farms not receiving AES payments leaves us with a sample of about 44,300 farms.

Data on water quality is provided by the Swedish University of Agricultural Sciences (SLU). The set consists of data collected within the national environmental monitoring programmes and in other initiatives, and is based on around 239,000[3] water quality samples

[3]The full sample contains about 275,000 observations. After dropping sampling sites without upstream farms, the sample was reduced to 239,000.

from about 4,300 sites spread across Sweden. In addition to a range of water quality indicators, the date and coordinates of each water sample are typically observed. Sampling frequency varies and for some smaller sites the sampling period does not cover all years from 1997 to 2013.[4]

Our area of interest is the nutrient concentration in water samples, specifically the total N and total P concentration. Total N and total P include compounds typically found in mineral fertilisers, such as ammonium, nitrate and phosphate, as well as organic forms found in manure and other fertilisers commonly used in organic farming.

The key to our research design is how we link water quality sampling sites to upstream farms, using GIS data in the form of shape files provided by the Swedish Meteorological and Hydrological Institute (SMHI). SMHI maintains maps separating Sweden's surface area into about 50,000 minor basins, including a map describing the upstream-downstream relations between all minor basins. With an average size of just over 10 km^2, the minor basin is the smallest geographical unit of observation commonly used in Swedish hydrological research. After matching all sampling sites and farms to their respective minor basin, we are able to identify all farms located upstream from a given sampling site. We end up using 33,706 minor basins, covering about 91% of Sweden's total area[5]. About 83.4% (36,910 out of 44,269) of the Swedish farms in our sample can be matched to a downstream sampling site. On average, the surface runoff from a farm passes through 14 sampling sites before flowing into the ocean or across a land border and a sampling site has about 300 farms upstream.

To remove seasonal variation in nutrient concentrations caused by variation in water flows, we calculate the nutrient concentration at a sampling site as the de-seasonalised yearly sample average. In the case of multiple sampling sites within a single minor basin, we

[4]Having an unbalanced panel is not a problem for the analysis. When using only sampling sites where we have water samples for at least 14 out of 17 years, the results in this study are unchanged (although a smaller sample implies larger standard error).
[5]The excluded minor basins are often located in coastal areas.

collapse the data at the minor basin level by taking the yearly average of all samples within the basin. This gives us water quality data from 2,376 minor basins.

The change in nutrient concentration (µg N or P per litre) in water samples over time is shown in Figures 1 and 2. As can be seen, N concentrations have decreased since the 1990s, by around 250 µg N/L. On the other hand, P concentrations are as high today as during the late 1990s.

Figures 1 and 2 about here

5. Modelling the relationship between payments and nutrient leaching

The relationship between agricultural production and nutrient leaching is a complex process and determined by the particular watershed characteristics. Important characteristics are e.g. hydrology, soil, distance, vegetation and land use (Ribaudo et al., 1999). The leaching from a certain field also interplays with land use at other locations within the watershed, and the reduction in nutrient leaching from a certain field can therefore depend on overall conservation measures within a watershed. Thus, due to non-constant diffusion coefficients, field-by-field evaluation of measures is not recommended (Khanna et al., 2003; Rabotyagov et al., 2010) and the analysis should preferably be scaled up to the watershed level. The effect of different abatement measures may also be cumulative and nonlinear (Kling et al., 2014).

Measuring differences in nutrient leaching between watercourses is indeed complex. Nonetheless, by using a watershed fixed-effect model, which relates the change in environmental subsidies going to a particular watershed to the change in nutrients in a downstream water sample, much of the complexity is accounted for. Watershed effects remove fixed nutrient differences in the water samples that depend on factors such as hydrology, soil and distance. Thus, the variation used to identify the subsidy effect on nutrients is the *within*-watershed variation, and not the *between*-watershed variation.

In our model, the upstream runoff area from a minor basin where one or more sampling sites are located constitutes a watershed. On average, our watersheds have an area of about 1200 km^2. We start by assuming that the nutrient concentration in a water sample at a certain sampling site *i* at time *t* can be expressed as:

$$Nutrient_{it} = \alpha_i + \mu_t + S_{it}'\beta + \varepsilon_{it} \tag{1}$$

As explained in the previous section, nutrient concentrations are de-seasonalised to remove seasonal variation in leaching and water flow rates. A watershed fixed effect, α_i, determines the average nutrient contribution from time-invariant unobserved watershed characteristics in watershed *i*. National variation in year-to-year weather and other national changes in nutrients are captured by a set of yearly time dummies, μ_t. In a sensitivity test, linear watershed-specific time trends are included to remove additional noise in the water samples caused by linear trends in watershed characteristics. As recommended, we analyse the subsidy impact at the watershed level and calculate retention-weighted (see below) sum of payments going to farms in watershed *i* at time *t*. With *k* types of subsidies, β is a $k * 1$ vector of parameters and S_{it} is a $k \times 1$ vector of weighted sums of farm payments.

Modelling nutrient retention

As mentioned above, a watercourse may contain multiple water sampling sites. Here we allow the nutrient emissions from farmland to have an impact on the observed nutrient concentration at all downstream sampling sites. We accomplish this by allowing a farm to be included in several overlapping watersheds. For example, for a farm located upstream from two sampling sites (located in different minor basins), the farm is included in the watershed formed by the upstream sampling site and the larger watershed formed by the downstream sampling site (where the upstream watershed is part of the larger watershed). Because natural and artificial biochemical processes (e.g. denitrification) serve to absorb nutrients along the way (a process known as retention), the impact at the sampling site will diminish depending on distance, land use and a range of other factors. We assume here that the relevant subsidy measure affecting the nutrient concentration at a sampling site is a weighted sum of subsidies received by all n upstream farms:

$$S_{it} = \sum_{j=1}^{n} w_{ij}\, S_{jt} \qquad (2)$$

where the k-vector $\mathrm{S_{jt}}$ denotes the subsidies received by farm j and the contribution of farm j to S_{it} is weighted by a factor $\mathrm{w_{ij}}$. These weighting factors are specific to each farm sampling site pair and capture the reduction in nutrients originating from a farm by the time the runoff reaches the sampling site in question. The weights are based on simulated retention rate, i.e. the rate at which nutrients (specifically N and P) are absorbed within a minor basin. Retention rates are simulated in the HYPE hydrological model developed by SMHI (Lindström et al, 2010). HYPE is a processed-based model capable of simulating nutrient flows at the minor basin level.[6] Our retention rates are output from a HYPE model calibrated for Sweden 1999-2011. The calibration period does not perfectly overlap with our period of interest (1997-2013), but as climate, land use and other forcing factors change at a very slow pace, any error due to this discrepancy is assumed to be negligible.

HYPE dynamics are generated by observed weather conditions, but the model has advanced static routines for simulating the flow of N in soil, lakes and rivers. As for fertiliser use, HYPE incorporates factors such as crop-specific fertiliser and irrigation regimes and multiple combinations of soil type and land use in a minor basin. Simulated retention rates should be interpreted as averages during the calibration period.

We define the retention coefficient as the ratio between net amount of nutrients remaining at the mouth of the basin and gross amount added. By construction, the coefficient can take on a value between 0 and 1. As an example, an N retention coefficient of 0.7 means

[6]For simulation purposes, some basins are merged to form larger units; we assume that these basins have identical retention coefficients.

that 70% of gross N added in a basin arrives at the mouth of the basin. Our weights are calculated by simply taking the product of the retention coefficients ($\mathrm{w_l}$) for the r minor basins forming the downstream water path between farm j and the sampling site. The r basins include the basin of origin and that of the sampling site:

$$\mathrm{w_{ij}} = \prod_{\mathrm{l=1}}^{\mathrm{r}} \mathrm{w_{lij}} \tag{3}$$

Because $0 \leq \mathrm{w_{lij}} \leq 1$, the calculated weight $\mathrm{w_{ij}}$ diminishes in value for each additional minor basin through which the runoff passes. The idea behind these weights is straight-forward: If the impact at the sampling site of nutrients originating from a given farm is presumed to be low, the contribution of the farm's AES payments to the weighted watershed sum will be low.

6. Descriptive statistics and the AES

Yearly total payments in the period 1997-2013 for each AES studied here are shown separately in Figures 3 to 6. The total payments vary widely over the period, partly as a result of changes in eligibility requirements and payment schemes. However, as already mentioned, it is outside the scope of this study to perform an in-depth analysis of each AES and to explore whether changes in the subsidy schemes affected their impact. Nevertheless, many of these changes resemble natural experiments and cause exogenous variations in payments. From that perspective, large exogenous variations in data are good for identification and lower the risk of spurious relationships arising.

The main problem is, rather, the discrepancy between payments and hectares covered. Most changes in payments represent a change in hectares covered, but it is not uncommon to have changes in payments per hectare (or for the rules for calculating the payments to change). It is preferable to have a measure of hectares covered, since a discrepancy between payments and hectares covered implies a type of measurement error in the independent variable. Knowing that random measurement error in the independent variable gives rise to an attenuation bias, this means that the subsidy effects will be underestimated.

Figures 3 to 6 about here

However, to rule out the possibility of changes in AES driving the results, an overall exploration of the subsidy schemes is necessary. The subsidy schemes generally change when a new programme is implemented (in 2001 and 2007). Since our subsidy data are reported separately for each programme, this means that we can estimate programme-specific subsidy effects. We thus modelled the main changes in payments and AES requirements and found that they did not have a major impact on the results, i.e. merging the subsidies over time or modelling them as programme-specific subsidy effects has a relatively small impact on the results. With respect to sign and significance, the estimated effects were the same for all programme periods.

However, even if the subsidy effects are robust in the programme periods, it is likely that the different subsidy schemes have some impact on the size of the effect. It is, of course, interesting and of great importance to know if different regulations and payment schemes affect the size of the effect, but to do this accurately requires specific econometric modelling (including specific sensitivity tests) of each AES. While a detailed exploration of each AES is for future research to consider, some additional comments about the individual subsidy schemes and the modelling of the subsidies in this study are necessary.

As already mentioned, we merge the *Cultivated grassland* and *Open and varied landscape* payments, because they basically preserve the same type of land. The total payments for these two AES are shown in Figure 4. It is likely that removing and merging these subsidies (as described earlier) have an impact on nutrient leaching,[7] but exploring this merits a study of its own.

[7]Changes in ASE requirements over time, and between subsidies, could also have had an impact on nutrient leaching.

The payments for *Grassed buffer zone* increased greatly in 2001 because the maximum width of zones went from 6 to 20 m, but payment per hectare was the same.[8] A decrease in payments in 2007 was caused by a reduction in the rate from SEK 3,000 to SEK 1,000 per hectare, a reduction that was removed in 2010. It is generally too complicated to use payments and the payment schemes to try to calculate hectares covered, but here we do an exception and adjust the payments. During the period 2004-2009, we multiply the payments by a factor of 3 so that the *Grassed buffer zone* payments represent only the decrease in hectares and not the change in payment per hectare (see the dotted line in Figure 3).

By reducing the payments to non-certified organic production in 2007, total payments decreased. Between 1997 and 2000, some semi-natural pastures were eligible for the subsidy *Open and varied landscape,* and therefore the payment for *Open and varied landscape* (*Pastures and meadows*) was higher (lower) during this period. In 2001, when the subsidy *No autumn tillage* was introduced, there was a large increase in payments for *Catch crops and no autumn tillage*. The new programme in 2007 involved regulatory changes in *Culturally significant landscape elements* (e.g. open ditches had to be surrounded by agricultural land on both sides, the payment for farm roads was decreased and the element “rows of trees and bushes” was removed), which decreased the support. Finally, for *Environmental protection measures* no new commitments were approved after 2011. However, since the *Environmental protection measures* are partly an advisory type of service with a possible long-lasting impact, we model the payments as remaining after 2011 even through the payment thereafter was zero. This appears to be a correct assumption, as modelled this way the effect is almost twice as large as for modelling the impact as zero when the commitment ended.

Descriptive statistics for the nutrients and the AES are presented in Table 1. Since the retention-weighted payments are weighted downwards, the mean weighted payments are

[8]Buffer zone width has been found to have an impact on nutrient removal (Syversen, 2005).

always smaller than the mean unweighted payments. The exception is *Environmental protection measures*, for which the within-watershed variation is always smaller than the between-watershed variation. Notably, without a sufficient amount of within-watershed variation, it may be difficult to identify AES effects. However, since the within-watershed variation corresponds, on average, to almost 50% of the overall variation, a lack of within-watershed variation should not be a problem. The least within-watershed variation is found for AES that target most of the watershed (few zeros): *Organic Production*, *Pastures and meadows, Cultivated grassland/Open and varied landscape* and *Culturally significant landscape elements*.

Table 1 about here

Results

We begin by estimating the retention-weighted model above and compare it to the unweighted model. If the retention-weighted model gives larger subsidy effects than the unweighted model it indicates that our empirical approach identifies true effects of the AES.[9] If the findings were spurious, the weighting should not have an impact on the results. To remove nutrient variations in the water samples caused by varying watershed characteristics, we include watershed-specific time trends. We apply a spatial error model to try to control for spatially correlated error terms between overlapping watersheds. Finally, we assess the size of the AES effects and whether the effects are linear.[10]

Main results - comparing the retention-weighted model with an unweighted model

[9]When comparing the models we have to consider that the payments are retentionweighted. We therefore rescale the retention-weighted payments with the average retention rate in the sample (i.e. divide the payments by 0.638), as otherwise the effects are smaller in the weighted model because of different scales of payments. This transformation means that the marginal effect corresponds to a SEK 1,000 change in real payments instead of a SEK 1,000 change in retention-weighted payments.

[10]We also analyse whether the impact of the AES occurs with a lag and whether the effects are linear (not reported). We did not find any delayed effects, indicating that the effects are, in general, linear.

Columns (1) and (4) in Table 2 show the results of the retention-weighted model for N and P, respectively. Columns (2) and (5) show the same results using unweighted sums of payments. In the retention-weighted model, the following AES have a negative effect on both nutrients: *Wetlands*, *Catch crops/No autumn tillage*, *Environmental protection measures* and *Culturally significant landscape elements*. The largest effect is for *Wetlands*. *Organic production* has a positive impact on both nutrients, and *Grassed buffer zones* and *Pastures and meadows* have a positive effect on N. For *Cultivated grassland/Open and varied landscape* the effect is insignificant for both nutrients. The size of the effects is assessed in a later section.

Table 2 about here

When using the unweighted model, the significant effects decrease by about 70% and 45% for N and P, respectively. Moreover, the negative effect of *Catch crops/No autumn tillage* on nitrogen leaching is insignificant in the unweighted model.

Controlling for watershed-specific time trends

Time-variant watershed characteristics that covary with the subsidy variation may bias the estimates of AES effects. However, random changes in watershed characteristics are unlikely to be correlated to payments, e.g. local weather conditions affecting nutrient leaching are not likely to vary systematically with the payments. On the other hand, the AES may target regions where deforestation and investments in sewerage systems and wastewater treatment facilities are common. Thus, particularly for the AES that increase linearly (*Wetlands*, *Organic production*, *Pastures and meadows*), there is a risk of the AES picking up a linear trend in e.g. deforestation. The AES effects may also be biased if the AES covaries with regional changes in farming systems or land use, e.g. the general decrease in cattle is likely to be larger in some areas than others. A standard approach for removing common trends is to add regional time trends. In our case, watershed linear time trends remove nutrient variations caused by time-variant watershed characteristics. However, a problem with this approach is that it is econometrically demanding and likely to remove part of the true effect if the AES effect is identified from a linear increase in AES. Hence, this test provides a lower bound of the causal effect. To maintain a reasonably parsimonious model, we introduced the linear time trends at the major basin level. SMHI formally defines a major basin as a watershed that is not part of a larger downstream watershed and has an area larger than 200 km^2 at the sea mouth. SMHI separates Sweden into 119 major basins and a number of "residual" coastal areas, which can be smaller than 200 km^2. Our watersheds cover 104 main basins and 48 coastal areas, i.e. we include 152 linear time trends. Columns (3) and (6) in Table 2 show the results with these linear time trends included. All estimates decrease and for N the effects remain significant for: *Wetlands*, *Environmental protection measures*, *Pastures and meadows* and *Culturally significant landscape elements*. For P the effects remain significant for: *Catch crops/No autumn tillage* and *Environmental protection measures*.

Thus, this test suggests that most AES effects on nutrient leaching are causal in direction and that the effects lie in the span between the effects in columns (1) and (3) for N and (4) and (6) for P. Although we cannot decide if the effects are closer to the lower or higher bound, it should be noted that the lower bounds are probably underestimated when considering the attenuation bias.

We also controlled for the following farm covariates: production value, investment costs, number of cattle, hectares of arable and grassland and other agricultural subsidies (e.g. firm subsidies and direct payment). However, since these variables are endogenous (the decision to apply for AES and these outcomes may be jointly determined), we do not include them in our preferred models. Nevertheless, inclusion of these variables is generally not affecting the AES effects.[11] Moreover, the variables cattle and hectares of arable and

[11]The size of the AES effects is robust to the inclusion of the covariates, but certain combinations of these variables may reduce the significance level of some of the AES effects.

grassland are taken from the *Farm Register* (provided by Swedish Board of Agriculture) and these data are only collected for the years 1999, 2003, 2005, 2007 and 2010. Another problem with the *Farm Register* is that it includes only farms with at least 2 hectares of arable land, which means that it reports aggregate changes incorrectly. For example, according to the *Farm Register* the number of cattle increased during the study period. However, due to a structural change that decreased the number of small farms with cattle, total number of cattle actually decreased nationally during the period. As an alternative way to control for changes in livestock, we included the decoupled animal payments for the period 1998-2004 as a proxy for changes in livestock (after 2004 the decoupled payments were replaced by the Single Farm Payments) and found robust result for the AES effects. Finally, the decoupling of the direct payments have no impact on the results (analysed by including the direct payments for the coupled period (-2004) and the decoupled period (2005-) separately).

Estimating a fixed-effects spatial error model

By construction, there is overlap between watersheds. Overlapping watersheds introduce cross-sectional correlations in the panel, which may lead to biased standard errors. As a robustness check, we verify that the significance of our baseline results is not influenced by a correlation between watersheds. We opt to use a fixed-effects spatial error model (SEM) with a spatial weighting matrix that capture the location of watersheds and the assumed magnitude of correlation between any pair of watersheds. Our implementation of the SEM draws on the work of Elhorst (2014), Lee and Yu (2010) and Piras and Millo (2012).

The SEM is a type of feasible generalised least squares (FGLS) estimation technique, utilising an estimate of a generalised variance matrix. The structure imposed on the variance matrix is based on a new composite error term, extending specification (1):

$$Nutrient_{it} = \alpha_i + \mu_t + S_{it}'\beta + u_{it} \tag{4}$$

$$u_{it} = \rho \sum_{k=1}^{N} w_{ik} u_{kt} + v_{it} \quad (5)$$

$$v_{it} = \psi v_{it-1} + \epsilon_{it} \quad (6)$$

where $\epsilon_{it} \sim IID(0, \sigma^2)$ is an idiosyncratic error term specific to each watershed and year. The error term u_{it} consists of a serially and a spatially correlated component.[12] The spatial component is jointly dependent on the error term of all N watersheds, subject to a spatial correlation coefficient ρ[13] and a watershed-specific weighting factor w which is an element of the known weight matrix W.

The SEM is commonly estimated using maximum likelihood,[14] where the error variance components ψ and ρ are parameters to be estimated. The weight matrix W is an $N \times N$ non-negative matrix where an element, w_{ik}, is equal to the accumulated retention coefficient between sampling site i and k for upstream-downstream watershed neighbours and zero otherwise, i.e. the weights describe how much of the nutrients present at sampling site i remain at sampling site k.

A caveat is that the SEM estimation routine requires a balanced panel. We balance our panel using a combination of list-wise deletion and imputation of missing water samples. The imputation procedure is simple. If a sampling site is missing a sample for year t, the missing value is replaced by the average of the sample concentration for periods t-1 and t+1 when applicable. This increases our sample size by about 5%.

The results of this test are shown in Table 3. Columns (1) and (3) show results for our standard fixed effects specification, but when using the balanced panel. Columns (2) and (4) show results for the fixed-effects spatial error model (SEM). The results when using the balanced panel, but the standard specification, resemble the main results presented in Table 2

[12]SEM with a serially correlated error component is sometimes referred to as SEMSR.

[13] Given a well-behaved weight matrix, ρ will be within $(1/\omega_{min}, 1)$ where ω_{min} is the smallest real characteristic root of W.

[14] We use the maximum likelihood estimation routines developed by Piras and Millo (2012).

(columns (1) and (4)). However, the smaller sample means that we lose significance for some AES.

Table 3 about here

Comparing the models for N (columns (1) and (2)), it can be seen that the AES effects on N concentrations are reduced by 20-30% when spatial correlations are taken into account, but that the significance levels are very similar to our standard fixed effects specification. For P (comparing columns (1) and (2)), there is generally a similar reduction in the AES effects; an exception is *Organic production*, where the reduction is larger and the effect becomes insignificant.

Our estimate of ρ indicates a significant positive correlation between watersheds of about 0.3, which is expected given the overlap between watersheds. Further interpretation of ρ is difficult due to the spatial and temporal feedback loops following an innovation in ϵ_i , but for our purposes it is sufficient to treat it as a 'nuisance' parameter. We conclude that the discrepancies between the results in Table 3 and our baseline results in Table 2 are mainly driven by the reduced sample size, i.e. the overlap between watersheds is not a problem for the analysis.

Assessing the size of the AES effects

For each AES, column (1) in Table 4 shows the impact of SEK 1,000 on the yearly amount of nutrients flowing through a watershed. To obtain this number, we multiply the AES effects (µg N or P/L) by the yearly water flow passing through a watershed. For our watercourses the estimated water flow is, on average, 18.36 m^3 per second.[15] Columns (1) and (2) show the yearly AES effects in kg N per year, while columns (3) and (4) show the corresponding values for P. In columns (1) and (3), we use the AES effects without controlling for watershed time

[15]This figure is a based on a HYPE simulation, adjusted using measured flow rates from about 330 sampling sites all around Sweden.

trends (columns (1) and (4) from Table 2) to compute a higher bound of the effect. In columns (2) and (4), we use the AES effects with watershed time trends included (columns (3) and (6) in Table 2) to compute a lower bound. These numbers help us assess the size of the AES effects.

Wetlands have the largest impact per 1,000 SEK spent, with N removal of between 870 kg (without time trends) and 270 kg (with time trends). Because the support is around SEK 3,000 per ha of wetland, the impact per ha of wetland is between 810 and 2,610 kg. Nitrogen removal of at least 1000 kg per ha wetland has been documented in field experiments in Sweden (Strand and Weisner, 2013; Weisner et al., 2015). Thus, for wetland we can conclude that removal of N calculated using our approach is similar to the removal rate found in small-scale studies, although the effect without time trends may be overestimated. For P, the removal in our study is between 51 and 12 kg per ha wetland, which is lower than the 100 kg P per ha wetland reported by Weisner et al. (2015).

Table 4 about here

Moreover, comparing our effect of *Catch crops/No autumn tillage* to the effect in a field experiment examining combination of the two measures reveal similar effects: 9.2-23.1 kg N per ha in our study,[16] and 16 N per ha in Hansen and Djurhuus (1997). For the other AES, similar comparisons are not possible. Nevertheless, since the assessment show that the *Wetland* and *Catch crops/No autumn tillage* effects are of a plausible size, it indicates that the other AES effects are credible too.

Discussion

As expected, *Wetland*, *Catch crop/No autumn tillage* and *Environmental protection measures* reduce nutrient leaching. However, for *Grassed buffer zones* and *Organic production* the

[16]The support is SEK 1,300 per ha when both measures are implemented, i.e. the effect has to be multiplied by 1.3 to get the effect per ha.

results are not in agreement with the expressed aim of reducing leaching. In *Organic production* it is perhaps the use of manure that causes the increased leaching, an assumption supported by other studies (Torstensson et al., 2006; Aronsson et al., 2007). However, the finding that *Grassed buffer zones* increases or has no impact on nutrient leaching is a surprise finding and is not supported by findings elsewhere. One then has to ask the question of whether our finding could be spurious. Our answer is no – the impact may be zero − but there is no reason to believe that buffer zones, overall, decreased nutrient leaching during the period 1997-2013. With much exogenous variation in the data (a large increase in hectares covered in 2003 and a decrease in 2007 followed by an increase in 2010), it is highly unlikely that the effect is spurious.

On the other hand, since previous studies in other Nordic countries have found that *Grassed buffer zones* decrease nutrient leaching, the non-negative effect in our study is likely to be caused by some hitherto unacknowledged feature. There are two possible explanations for the non-negative effect of buffer zones in the present study. First, the Swedish regulations allow grazing of buffer zones, but to maintain the nutrient reduction it is recommended elsewhere that buffer zones should be mowed and the material removed (Uusi-Kämppä, 2005**)**, and they should preferably not be grazed because of cattle trampling damage to the buffer zones. Moreover, if cattle congregate on buffer zones around watercourses, this results in direct inflow of manure and urine into the waters. Second, although the construction of a buffer zone may decrease nutrient leaching, the removal and ploughing up of these buffer zones may increase nutrient leaching substantially, i.e. the effect of buffer zones in this study may be due to continual construction and removal of buffer zones over time.

Importantly, we find that *Pastures and meadows* increased nutrient leaching and, since this subsidy covers a substantial amount of land, the total impact is large. The effect is probably due to the adverse effects of grazing the land, which is a requirement of the subsidy.

Alternatively, the subsidy may prevent pastures and meadows from achieving the leaching decreasing effect of becoming overgrown.

However, the area of grazed pastures and meadows is not increasing because of an increasing number of grazing livestock. Instead, grazing livestock decreased during the study period and without *Pastures and meadows*, which subsidises extensive farming, the number of grazing livestock would probably decrease further. Hence, when evaluating the total environmental impact of *Pastures and meadows*, the benefit of a varied agricultural landscape and biodiversity has to be weighted with the adverse impact on nutrient leaching.

Cultivated grassland/Open and varied landscape is assumed to reduce nutrient leaching, but we did not find a significant impact. However, since we evaluate the effect of two different AES preserving the same arable land, the specific effect of each AES is uncertain. The impact may also vary over time and between regions, depending on the counterfactual land use – if the subsidy mainly prevents land from being used for annual crops it probably reduces nutrient leaching, but if the subsidy increases fertilised grassland at the expense of land in fallow the impact is the opposite. To investigate this, we would need data on different types of land, i.e. we could then differentiate the impact depending on whether the subsidies increased cropland, or not.

Finally, *Culturally significant landscape elements* was found to have a large impact on nutrient leaching, but the effect is possibly caused by the management of certain landscape element, e.g. small waters and open ditches. Plausibly, for landscape elements with a small impact on nutrient leaching, there is a lack of variation in payments during the period. Much of the payment goes to landscape elements that are protected by regulations and, since they cannot be removed, there is little to no variation in payment for these elements. Conservation of a varied agricultural landscape has a positive side-effect on nutrient leaching. However, in the new programme starting in 2015, *Culturally significant landscape elements* has been removed.

Conclusions

To reduce the nutrient overload in the Baltic Sea, leaching from agricultural land has to decrease. The second Pillar of the CAP contains a wide variety of AES, a few of which are nutrient abatement measures. This study evaluates the impact of AES payments to Swedish farmers on nutrient concentrations in downstream water samples. By also evaluating AES without an explicit aim of controlling nutrients, the study brings novel knowledge to the research field.

The watershed fixed-effect model used here find that *Wetland*, *Catch crop/No autumn tillage*, *Environmental protection measures* and *Culturally significant landscape elements* reduce nutrient leaching, with wetlands being the most effective measure. It also find that *Grassed buffer zones*, *Organic production* and *Pastures and meadows* increase nutrient leaching. These adverse effects seem to be associated with extensive livestock farming.

However, the main contribution of the study is that it proposes a new approach for investigating the effects of abatement measures. With the watershed fixed-effect model, it is possible to remove fixed nutrient differences in water samples that depend on factors such as hydrology, soil and distance. The remaining nutrient variation is then linked to the sum of AES paid to farmers located upstream in the watershed of the water sampling site.

Moreover, in contrast to agricultural field trials, which explore textbook implementation of practices, our approach incorporates insufficiencies in implementation and unintended impacts in the analysis. Adverse effects of some AES and the positive effect of *Culturally significant landscape elements* plausibly fall into this category and are not likely to be captured by other approaches.

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Tables and Figures

Table 1. Descriptive statistics for the nutrients (nitrogen, phosphorus) and Agri-Environmental Schemes (AES)

		Mean	Standard deviation			Zeros
			Overall	Between	Within	
Nitrogen (µg/L)	N=20,914	1210.61	1414.58	1392.64	467.68	
Phosphorus (µg/L)	N=21,275	36.24	45.39	45.01	23.29	
Wetlands (SEK×1000)	Unweighted	15.39	73.52	63.13	33.81	70.61%
	Weighted	10.12	40.92	28.16	23.92	
Organic production (SEK×1000)	Unweighted	1119.27	4973.16	4094.04	1744.39	14.72%
	Weighted	609.39	2095.86	1538.73	825.10	
Catch crop/No autumn tillage (SEK×1000)	Unweighted	269.69	2142.69	1573.17	1105.00	60.43%
	Weighted	202.53	1169.63	793.28	597.12	
Grassed buffer zone (SEK×1000)	Unweighted	40.99	242.48	184.88	125.72	60.63%
	Weighted	27.04	114.56	82.05	61.49	
Env. protection measures (SEK×1000)	Unweighted	23.52	200.48	134.18	160.41	86.30%
	Weighted	21.17	132.56	72.58	103.96	
Pastures and meadows (SEK×1000)	Unweighted	1082.44	3584.77	2957.37	1183.91	8.33%
	Weighted	601.54	1417.44	1099.88	479.18	
Cultivated grassland/Open and varied landscape (SEK×1000)	Unweighted	863.57	3489.88	2680.3	1812.08	3.27%
	Weighted	1251.93	3002.67	2358.69	879.28	
Culturally significant landscape elements (SEK×1000)	Unweighted	278.59	919.08	791.47	209.05	20.98%
	Weighted	166.39	412.80	332.79	95.15	

Table 2. Effects of SEK 1,000 provided within Agri-Environmental Schemes (AES) on nitrogen and phosphorus concentrations in water samples (µg/L)

	Nitrogen			*Phosphorus*		
	(1)	(2)	(3)	(4)	(5)	(6)
Wetlands	-1.497***	-1.169***	-0.459*	-0.0292***	-0.0213***	-0.00690
	(0.337)	(0.178)	(0.237)	(0.0111)	(0.00709)	(0.0069)
Organic production	0.0320***	0.0050*	0.0121	0.0015***	0.0004***	0.0007
	(0.0112)	(0.0028)	(0.0087)	(0.0004)	(0.0001)	(0.0005)
Catch crop/No autumn tillage	-0.0308***	-0.0052	-0.0122	-0.00146**	-0.0003**	-0.0013**
	(0.0111)	(0.0032)	(0.0084)	(0.0006)	(0.0001)	(0.0006)
Grassed buffer zone	0.162*	0.0659**	0.0355	0.00346	0.00115	-0.0013
	(0.0859)	(0.0306)	(0.0731)	(0.0034)	(0.00126)	(0.0033)
Env. protection measures	-0.587***	-0.107***	-0.339***	-0.0115***	-0.0023***	-0.00568*
	(0.104)	(0.0257)	(0.0638)	(0.0034)	(0.0008)	(0.0034)
Pastures and meadows	0.0577***	0.0079**	0.0359**	0.00011	-0.0001	0.000474
	(0.0154)	(0.0032)	(0.0141)	(0.0006)	(0.0001)	(0.00056)
Cult. grassland/Open and var. landscape	0.0032	-0.0008	0.0018	-0.0000	-0.0001	-0.0000
	(0.00346)	(0.0011)	(0.0031)	(0.0002)	(0.0001)	(0.000)
Culturally sig. landscape elements	-0.446***	-0.147***	-0.285***	-0.00795**	-0.00265**	-0.00373
	(0.0771)	(0.0281)	(0.0635)	(0.00321)	(0.00112)	(0.00281)
Watershed specific linear time trends	no	no	yes	no	no	yes
Observations	23,507	23,507	23,507	23,924	23,924	23,924
Number of watersheds	2,376	2,376	2,376	2,426	2,426	2,426
R-squared	0.034	0.025	0.075	0.007	0.005	0.028

Note: The dependent variable is the nutrient content in the water samples (µg/L). The AES payments are measured in thousand SEK. Watershed and time fixed effects are included in every specification. In columns (1), (3), (4) and (6), the payments are weighted with retention rates. Columns (3) and (6) show watershed-specific linear time trends. Robust clustered standard errors in brackets. ***p<0.01, **p<0.05, *p<0.1.

Table 3. Effects of SEK 1,000 provided within Agri-Environmental Schemes (AES) on nitrogen and phosphorus concentrations in water samples (µg/L), based on a model with spatially correlated errors

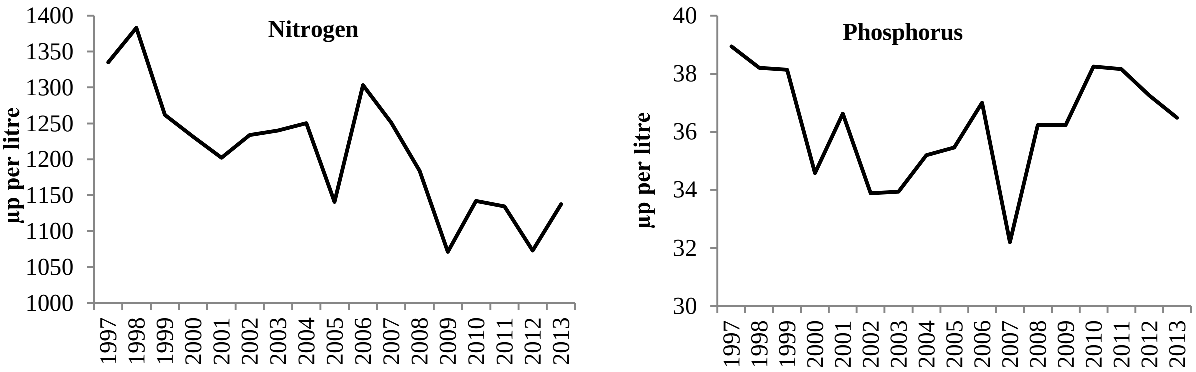
	Nitrogen		*Phosphorus*	
	(1)	(2)	(3)	(4)
Wetlands	-1.797***	-1.248***	-0.0300**	-0.0207*
	(0.468)	(0.192)	(0.0127)	(0.0068)
Organic production	0.0281**	0.0161**	0.00130***	0.00037
	(0.0123)	(0.0080)	(0.000456)	(0.000376)
Catch crop/No autumn tillage	-0.00764	-0.00517	-0.000367	-0.000141
	(0.0109)	(0.0094)	(0.000387)	(0.000473)
Grassed buffer zone	0.0918	0.0610	0.00127	0.00033
	(0.119)	(0.0790)	(0.00294)	(0.00286)
Env. protection measures	-0.573***	-0.403***	-0.0119***	-0.0092***
	(0.132)	(0.063)	(0.00373)	(0.00227)
Pastures and meadows	0.0529***	0.0415***	-0.000391	0.0001
	(0.0200)	(0.0130)	(0.000648)	(0.00491)
Cult. grassland/Open and var. landscape	-0.000507	-0.00015	-0.000259	-0.000247
	(0.00323)	(0.00429)	(0.000250)	(0.00026)
Culturally sig. landscape elements	-0.623***	-0.459***	-0.0102***	-0.0074***
	(0.131)	(0.067)	(0.00377)	(0.00208)
Rho		0.3232***		0.2951***
		(0.0121)		(0.0130)
Psi		0.2677***		0.1146***
		(0.0102)		(0.0105)
Observations	9,486	9,486	10,455	10,455
Number of watersheds	558	558	615	615

Note: The dependent variable is the nutrient content in the water samples (µg/L). The AES payments are measured in thousand SEK and weighted with retention rates. Watershed and time fixed effects are included in every specification. Columns (1) and (3) represent our baseline fixed effects model with standard errors clustered at the watershed level. Columns (2) and (4) are the results from the spatially correlated error model described by eqs. 4 to 6. ***p<0.01, **p<0.05, *p<0.1.

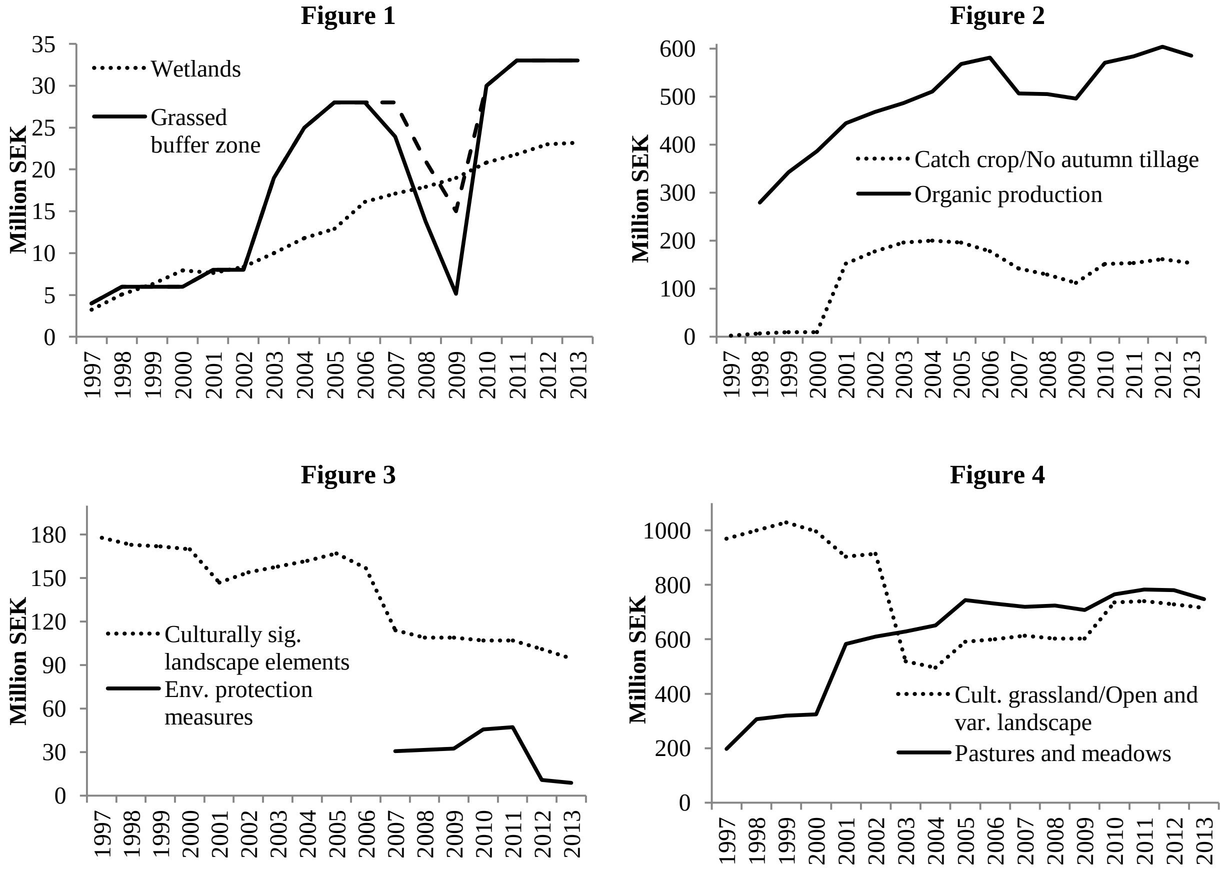
Table 4. Calculated yearly reduction in nitrogen and phosphorus concentrations brought about by different Agri-Environmental Schemes (AES), expressed per SEK 1,000

	Nitrogen		*Phosphorus*	
	Higher bound (*without time trends included)*	Lower bound (*with time trends included)*	Higher bound (*without time trends included)*	Lower bound (*with time trends included)*
Wetlands	-866.9	-265.8	-16.9	-4.0
Organic production	18.5	7.0	0.9	0.4
Catch crop/No autumn tillage	-17.8	-7.1	-0.8	-0.7
Grassed buffer zone	93.8	20.6	2.0	-0.8
Env. protection measures	-339.9	-196.3	-6.7	-3.3
Pastures and meadows	33.4	20.8	0.1	0.3
Cult. grassland/Open and var. landscape	1.9	1.0	0.0	0.0
Culturally sig. landscape elements	-258.3	-165	-4.6	-2.2

Note: The numbers are based on the estimates from Table 2, columns (1), (3), (4) and (6). The AES effects are multiplied by the average water flow (18.26 ton/s) and transformed into yearly kg nutrients per SEK 1,000.



Figures 1 and 2. Average nitrogen and phosphorus concentrations (µg/L) in water samples from Swedish watersheds, 1997-2013.



Figures 3 to 6. Total yearly payments for each Agri-Environmental Scheme (AES) in Sweden, 1997-2013.