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Estimating the impact of agri-environmental payments on nutrient runoff using a unique combination of data

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ABSTRACT

This study is the first to estimate the effect of Agri-Environmental Schemes (AES) on nutrient runoff 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 runoff. However, uptake of *Grassed buffer zones, Pastures and meadows* and *Organic production* was associated with increased nutrient runoff.

1. Introduction

The Baltic Sea has the largest hypoxic zones caused by nutrient overloads in the world and an estimated 40% of nitrogen (N) and 24% of phosphorus (P) runoff to Swedish freshwaters and the Baltic Sea are from agricultural land (Brandt and Ejhed, 2002). 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 runoff is described as a non-point source emission and, in contrast to point source emissions, difficult to measure (Horan and Ribaudo et al., 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; Shortle and Horan, 2013; Kling et al., 2016). This study uses water quality data to trace the nutrient runoff from agricultural land and evaluates the impact of paying farmers for nutrient abatement measures on nutrient runoff.

Using monitoring technology to transform the agricultural nutrient runoff from a non-point source emission to a point source emission is costly (Millock et al., 2002; Xepapadeas, 2011) and has been considered infeasible. For example, Shortle and Horan, (2013) states that “non-point instruments cannot be based on actual runoff“. Instead, the typical approaches for evaluating abatement measures are structural modelling and (small scale) field trials. However, using actual runoff in a reduced-form framework is becoming common and the advantage is that the biophysical processes do not have to be modelled explicitly (see Kling et al., 2016 for an overview).

Agri-Environmental Schemes (AES), part of the second Pillar of the EU Common Agricultural Policy (CAP), are a targeted tool for reducing nutrient runoff. One fourth of the agricultural area in EU is registered in AES (Laukkanen and Nauges, 2014). Like most other EU countries, Sweden has a wide range of AES (e.g. payments for establishing grassed buffer zones or wetlands). This study uses a comprehensive panel containing micro-level data on farms and water quality to estimate the effect of AES payments on nutrient runoff for Sweden. Closest to us is Keiser and Shapiro (2017) who merge US water quality data and municipal sources of water pollution. With this data they analyse the impact of the Clean Water Act and the effects of water pollution regulation on home values. Other studies to use water quality data are Sigman (2002, 2005), Lipscomb and Mobarak (2016) and Smith and Wolloh, (2012). These studies merge water samples with socioeconomic data (e.g. GDP, per capita income or the unemployment rate) at the country or state level. But to our knowledge, no previous study has linked water quality data and abatement measures at the farm level.

We use a unique combination of data sources – and almost a full population of watersheds (covering 91% of the Swedish land area and AES payments to 83% of Swedish farms) – to estimate an average AES effect on treated watersheds. Our approach merges information over a period of seventeen years (1997–2013) on the concentrations of N and P in water samples from 2376 lakes and watercourses in Sweden with information on watersheds, retention rates and agri-environmental subsidies paid to about 37,000 farms in the vicinity of these waters. The

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key to our research design is linking water quality sampling sites to upstream farms. We use GIS maps of Sweden's about 50,000 sub-watersheds, which describe the upstream-downstream relationships between all watersheds. Using coordinates, we match all sampling sites and farms to their respective sub-watershed, 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 runoff is the variation in the amount of AES-payments *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.

Despite agricultural field trials and model simulations, the impact of individual AES is still uncertain[1]. The problem is that: i) field trials have low external validity because impacts are heterogeneous and depends on watershed characteristics and land use (Khanna et al., 2003; Rabotyagov et al., 2010); and ii) model simulations rely heavily on theoretical assumptions, e.g., functional form and biophysical processes (Kling et al., 2016). To handle the complexity field trials prefer environments predisposed to nutrient runoff; but in a "laboratory" environment the average impact is likely to be overestimated.[2]

In addition, investigating textbook implementation on representative farm types – which field trials often do – mainly capture the second step in the relationship between AES payments and nutrient runoff, i.e. the biophysical impact (Balana et al., 2011). The first step is the relationship between AES payments and farmers' behaviour, and payments have both real effects and windfall effects on behaviour. With positive self-selection into a programme – farmers with low costs of complying with AES requirements are more likely to enter a programme – the windfall effects risk being large (Chabé-Ferret and Subervie, 2013). Insufficiencies in implementation and regulation should also be accounted for in the analysis.

In the present study we incorporate not only the hydrological link between an action and nutrient runoff, but also the effect on farmers' implementation of measures and whether the AES target high-impact watersheds. The estimated impact may be small or non-existent, however efficient the measure may be under ideal conditions. Moreover, an AES may have an impact – a co-benefit or unintended impact – on nutrient runoff even if the target objective is e.g. biodiversity (Balana et al., 2011). Thus, in this study we consider all large scale AES – also AES where reduced nutrient runoff is not a specified objective.

2. 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.

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 from about 4300 sites spread across Sweden. In addition to a range of water quality indicators, the date and coordinates of each water sample are typically observed. The average number of water samples per year is 9.6. For some sites the sampling period does not cover all years from 1997 to 2013.[4]

The key to our research design is how we link water quality sampling sites to upstream farms, using GIS data provided by the Swedish Meteorological and Hydrological Institute (SMHI). SMHI maintains maps separating Sweden's surface area into about 50,000 sub-watersheds, including a map describing the upstream-downstream relations between all sub-watersheds. With an average size of just over 10 km^2, the sub-watersheds is the smallest geographical unit of observation commonly used in Swedish hydrological research[5]. After matching all sampling sites and farms to their respective sub-watershed, we are able to identify all farms located upstream from a given sampling site.

Fig. 1 illustrates the research design. The figure shows three sampling sites located in three different sub-watersheds. The downstream sub-watershed is part of a larger watershed which also includes the two upstream sub-watersheds. Hence, all farms (and their received AES payments) in the figure affect the nutrients at the downstream sampling site, whereas the nutrients at the upstream sampling sites are only affected by the farms located in their respective upstream sub-watersheds. Moreover, by modelling the retention, AES payments going to farms in upstream sub-watersheds have a smaller impact than payments going to farms in downstream sub-watersheds (the retention is further explained in Section 4).

We end up using 33,706 sub-watersheds – where 2376 sub-watersheds contains at least one sampling site – covering about 91% of Sweden's total area[6]. 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, a sampling site has about 300 farms upstream.

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 change in nutrient concentration (μg N or P per litre) in water samples over time is shown in Figs. 2 and 3 . As can be seen, N and P concentrations have decreased since the early 1990s, by around 3000 μg N/L and 5 μg P/L.[7]

3. The AES

The second Pillar of the CAP contains a wide variety of AES,[8] a handful of which aim at reducing nutrient runoff. The application of AES are compulsory for Member States and their design should be adapted to the national or regional farming systems and environmental conditions. The AES are voluntary and participating farms generally sign five-year contracts in which they agree to follow mandatory abatement measures. In this study we analyse the following AES with a nutrient runoff 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)),

[1] In the discussion, we compare our AES effects with evaluation results from the non-economic literature. In Grenestam and Nordin (2015) a broader survey of the non-economic literature is provided.

[2] For example, when comparing the reduction in N for seven wetlands in a field trial with simulated reductions for 2,400 randomly chosen wetlands, the results differed considerably and were much smaller for the randomly chosen wetlands (Strand and Weisner, 2013).

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

[4] We show later that our results are robust to a balanced panel. 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 errors).

[5] For sub-watersheds with an active monitoring station, the median size is about 15 km^2.

[6] The excluded sub-watersheds are often located in coastal areas.

[7] Trend analysis by biologists show 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).

[8] During the period there has been a couple of other AES; but they are all very small in terms of covered hectares and none has a clear nutrient runoff aim.

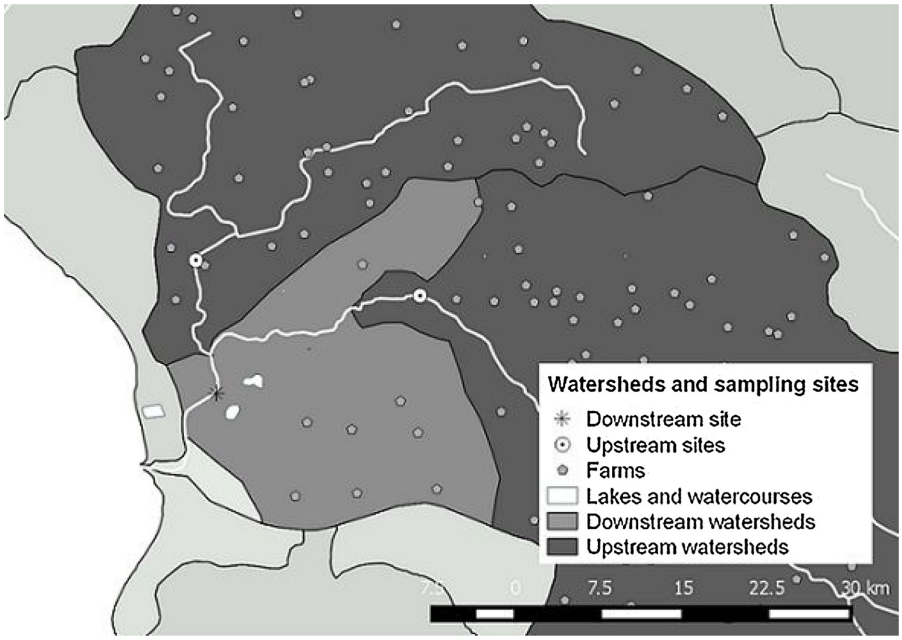
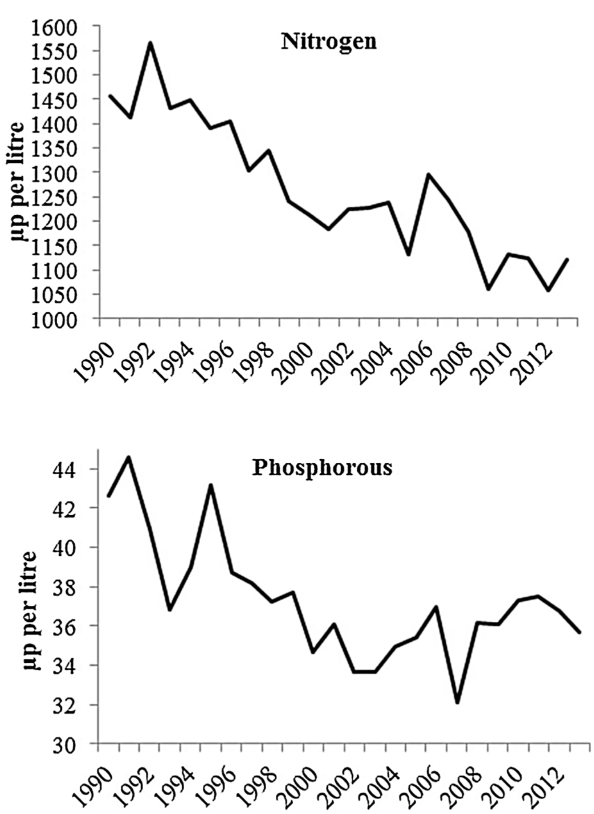


Fig. 1. Illustrating the research design.

Cultivated grassland (1997–2002 and 2007–2013) and *Open and varied landscape* (1997-2013).

We also evaluate two subsidies without an expressed nutrient control aim: *Culturally significant landscape elements* (1997–2013) and *Pastures and meadows* (1997–2013). These provide large payments and may potentially have an impact on nutrient runoff, although their specified aim is to facilitate conservation of a varied agricultural landscape and biodiversity.



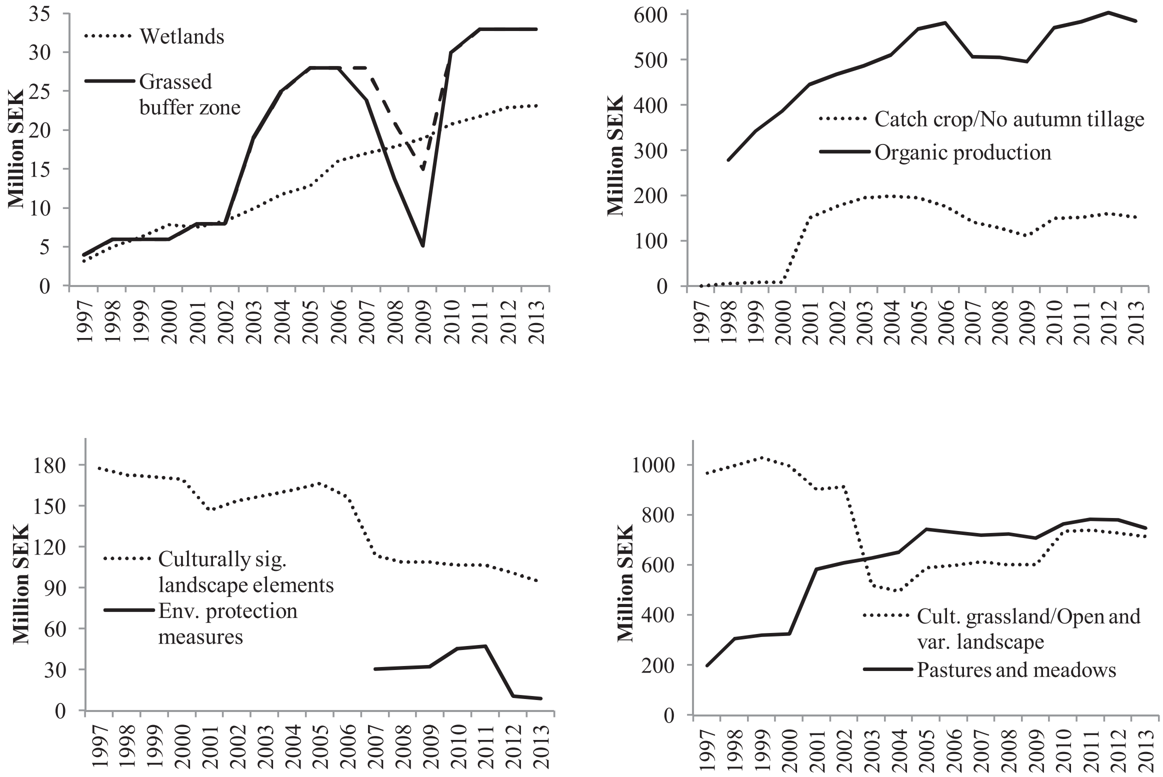
Figs. 2 and 3. Average nitrogen and phosphorus concentrations (µg/l) in water samples from swedish watersheds, 1997–2013.

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.

Yearly total payments in the period 1997–2013 for each AES studied here are shown separately in Figs. 4–7 (but discussed in detail later). The total payments vary widely over the period, partly as a result of changes in eligibility requirements and payment schemes. The schemes are politically determined and not related to e.g. yields or crop prises. 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 have had an impact on nutrient runoff. However, 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, the large changes in payments are good for identification and lower the risk of spurious relationships. The main problem is, rather, the discrepancy between payments and hectares covered. To have a measure of hectares covered is preferred, and using payments as a proxy for hectares covered implies a type of measurement error in the independent variable.[9] Another source of measurement error is the use of simulated retention rates (see Section 4.1). Knowing that random measurement error in the independent variable gives rise to an attenuation bias, it implies an underestimated subsidy effects. However, with systematic measurement errors the sign of the bias is unknown.

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

[9] This is why we do not deflate the payments. Deflated payments is more of a monetary measure and we prefer a proxy of the covered hectares of land (with deflated payments it would seem as if less hectares where covered over time). Also, the payments have not been adjusted for inflation.



Figs. 4–7. Total yearly payments for each agri-environmental scheme (AES) in Sweden, 1997–2013.

Table 1
Descriptive statistics.

		Mean	Standard deviation			Zeros
			Overall	Between	Within	
Nitrogen (micrograms per litre)	N = 20,914	1210.61	1414.58	1392.64	467.68	
Phosphorus (micrograms per litre)	N = 21,275	36.24	45.39	45.01	23.29	
Wetlands (SEK × 1000)		15.39	73.52	63.13	33.81	70,61%
Organic production (SEK × 1000)		1119.27	4973.16	4094.04	1744.39	14,72%
Catch crop/No autumn tillage (SEK × 1000)		269.69	2142.69	1573.17	1105.00	60,43%
Grassed buffer zone (SEK × 1000)		40.99	242.48	184.88	125.72	60,63%
Env. protection measures (SEK × 1000)		23.52	200.48	134.18	160.41	86,30%
Pastures and meadows (SEK × 1000)		1082.44	3584.77	2957.37	1183.91	8,33%
Cultivated Grassland/Open and varied landscape (SEK × 1000)		1608.25	5707.63	5037.4	1329.98	3,27%
Culturally significant landscape elements (SEK × 1000)		278.59	919.08	791.47	209.05	20,98%

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. We come back to this after having presented the econometric model.

Descriptive statistics for the nutrients and the AES are presented in Table 1. 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. Also, after removing within-watershed variation with watershed linear time trends (see Section 5.1) around 80% of the within variation in payments remain. The least within-watershed variation is found for AES that target most of the watersheds (i.e almost all watersheds received subsidies): *Organic Production*, *Pastures and meadows*, *Cultivated grassland/Open and varied landscape* and *Culturally significant landscape elements*.

4. A model between AES payments and nutrient runoff

The relationship between agricultural production and nutrient runoff 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 surface runoff from a field also interplays with land use at other locations in the watershed, and the reduction in nutrient runoff from a certain field can therefore depend on overall conservation measures within a watershed. Thus, due to endogenous 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 runoff 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 between farm and sampling site. 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 sub-watershed 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:

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

With k types of subsidies, β is a $k \times 1$vector of parameters and S_{it} is a $k \times 1$ vector of weighted sums of farm payments. To remove seasonal variation in nutrient concentrations caused by variation in water flows, we calculate the nutrient concentration at a sampling site as the deseasonalised yearly sample average.[10] A watershed fixed effect, α_i, determines the average nutrient contribution from time-invariant unobserved watershed characteristics in watershed *i*. 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. National variation in year-to-year weather and other national changes in nutrients are captured by a set of yearly time dummies, μ_t. However, the subsidy effects, β, may still be biased due to time variant factors at the local level. In an effort to deal with this concern we include in a sensitivity test, linear watershed-specific time trends to remove additional noise in the water samples caused by linear trends in watershed characteristics. In Section 5.1 we further discuss this concern and do additional sensitivity tests.

4.1. Modelling nutrient retention

As previously mentioned, a watercourse may contain multiple water sampling sites. Here we allow the nutrient runoff 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. In Fig. 1, for example, farms located in a upstream sampling site are 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:

$$S_{it} = \sum\nolimits_{j=1}^{n} w_{ij} S_{jt} \tag{2}$$

where the k-vector S_{jt} denotes the subsidies received by farm j and the contribution of farm j to S_{it} is weighted by a factor 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 sub-watershed. 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 sub-watershed level.[11] 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. 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 watershed 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 that 70% of gross N added in a watershed arrives at the mouth of the watershed. Our weights are calculated by simply taking the product of the retention coefficients (w_l) for the r sub-watersheds forming the downstream water path between farm j and the sampling site. The r watersheds include the watershed of origin and that of the sampling site:

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

Because $0 \le w_{lij} \le 1$, the calculated weight w_{ij} diminishes in value for each additional sub-watershed 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.[12]

4.2. Modelling the AES payments

We merge the *Cultivated grassland* and *Open and varied landscape* payments, because they basically preserve the same type of land (grass leys on arable land). 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*. The total payments for these two AES are shown in Fig. 7 . It is likely that removing and merging these subsidies have an impact on nutrient runoff, but exploring this merits a study of its own.

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[13]. A decrease in payments in 2007 was caused by a reduction in the rate from SEK 3000 to SEK 1000 per hectare, a reduction that was removed in 2010. It is difficult to use payments and the payment schemes 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 Fig. 4).

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,*

[10] We estimate a model with monthly dummies and use the yearly average of the residual variation.

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

[12] Since distance and watershed size is closely related, the retention rates also capture differences in watershed size.

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

Table 2
Effects of SEK 1000 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*** (0.337)	−1.169*** (0.178)	−0.459* (0.237)	−0.0292*** (0.0111)	−0.0213*** (0.00709)	−0.00690 (0.0069)
Organic production	0.0320*** (0.0112)	0.0050* (0.0028)	0.0121 (0.0087)	0.0015*** (0.0004)	0.0004*** (0.0001)	0.0007 (0.0005)
Catch crop/No autumn tillage	−0.0308*** (0.0111)	−0.0052 (0.0032)	−0.0122 (0.0084)	−0.00146** (0.0006)	−0.0003** (0.0001)	−0.0013** (0.0006)
Grassed buffer zone	0.162* (0.0859)	0.0659** (0.0306)	0.0355 (0.0731)	0.00346 (0.0034)	0.00115 (0.00126)	−0.0013 (0.0033)
Env. protection measures	−0.587*** (0.104)	−0.107*** (0.0257)	−0.339*** (0.0638)	−0.0115*** (0.0034)	−0.0023*** (0.0008)	−0.00568* (0.0034)
Pastures and meadows	0.0577*** (0.0154)	0.0079** (0.0032)	0.0359** (0.0141)	0.00011 (0.0006)	−0.0001 (0.0001)	0.000474 (0.00056)
Cult. grassland/Open and var. landscape	0.0032 (0.00346)	−0.0008 (0.0011)	0.0018 (0.0031)	−0.0000 (0.0002)	−0.0001 (0.0001)	−0.0000 (0.000)
Culturally sig. landscape elements	−0.446*** (0.0771)	−0.147*** (0.0281)	−0.285*** (0.0635)	−0.00795** (0.00321)	−0.00265** (0.00112)	−0.00373 (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

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. ***$p < 0.01$, **$p < 0.05$, *$p < 0.1$.

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.

5. Results

We begin by estimating the retention-weighted model above and compare it to the unweighted model. Next, we perform a range of sensitivity tests. Finally, we assess the size of the AES effects.

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

Columns (1) and (4) in Table 2 show the results of the retention-weighted model for N and P, respectively.[14] Columns (2) and (5) show the same results using unweighted sums of payments. The coefficients show the impact of SEK 1000 on the yearly amount of nutrients flowing through a watershed. The size of the effects is assessed in a later section. 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.

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 runoff is insignificant in the unweighted model.

5.2. Sensitivity tests

Time-variant watershed characteristics that covary with the subsidy variation may bias the AES effects. For example, 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 linear changes in nutrients at the watershed level. However, a problem is that this approach 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 conservative estimate of the AES effect. To maintain a reasonably parsimonious model, we introduced the linear time trends at the major watershed level[15]. 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*.

We also control 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, including these variables does generally not affect our estimates (see Table A1, columns (1)-(2)).

[14] When comparing the models we have to consider that the payments are retention weighted. 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 payments are scaled down. 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.

[15] SMHI formally defines a major watershed 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 watersheds and a number of "residual" coastal areas, which can be smaller than 200 km^2. Our watersheds cover 104 main watersheds and 48 coastal areas, i.e. we include 152 linear time trends.

Differences in precipitation is another reason for concern. Timing and rate of precipitation are critical factors affecting runoff. We have analyzed the precipitation and found that it varies *between* regions[16], but *within* regions precipitation is fairly similar. In a model with three sets of time dummies (instead of the national time dummies), capturing the fluctuations in precipitation between- and within each region (Southern Sweden, Mid Sweden and Northern Sweden) we have tested if precipitation is a problem. The model (Table A1, columns (3)-(4)) shows that the subsidy effects are almost identical to the subsidy effects in Table 2, indicating that precipitation is not a problem.

We have also included one year lagged payments to analyze the dynamics. The effects are, in general, not delayed (Table A1, columns (5)-(8)). However, since the current effects for phosphorus decrease when including lagged effects it indicates that the impact of AES on the runoff of phosphorus should be modelled dynamically. For nitrogen, the impact of wetlands may also benefit from being modelled dynamically. A more ambitious dynamic modeling of the AES-nutrient relationships demands that the hydrological system is considered, but this warrants future research.

The data includes water samples from both lakes and watercourses. In a sensitivity analysis (Table A1, columns (9)-(10)) where we remove samples from lakes we find that the overall results remain. However, since lake water samples are more common in certain regions this is partly a regional sensitivity test.

Another concern is that our watersheds are not nationally representative. If monitoring efforts are concentrated to areas with extensive nutrient leakage, our sample is skewed which could put the external validity of our study into question. Since monitoring is administrated mostly at the county and municipal level, we examine differences in monitoring effort by county. While there is some variation in samples per hectare of farmland[17] across counties, this does not seem to be positively correlated with N or P concentrations. Excluding the three counties with the largest number of samples per hectare of farmland does not affect our results (note Table A2, columns (1)-(2)). Finally, we have tried using water sample for different parts of the year (Table A2, columns (3)-(8)), and find that there is seasonal variation in impact (generally, a larger impact in spring). However, a seasonal analysis merits a study on its own.

5.3. Accounting for cross-sectional correlation

By construction, our data is characterized by overlap between watersheds. Overlapping watersheds introduce cross-sectional correlation, which may lead bias standard errors unless accounted for. As a robustness check, we verify that the significance of our baseline results is not influenced by 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). In our SEM specification, the error term consists of a temporally and a spatially correlated component.[18] The spatial component is jointly dependent on the error term of all N watersheds, subject to a spatial correlation coefficient ρ and a watershed-specific weighting factor w_{ik} which is an element of the known weight matrix W. 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. For details on our SEM estimation procedure, we refer to our working paper (Grenestam and Nordin, 2016).

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 spatial error model (SEM). 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. We conclude that cross-sectional correlation, while likely present, does not invalidate the results from our baseline specification.

5.4. Assessing the size of the AES effects

For each AES, column (1) in Table 4 shows the impact of SEK 1000 on the yearly amount of nutrients flowing through a watershed. To obtain this number, we multiply the AES effects (μg N/L 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.[19] 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 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 1000 SEK spent, with N removal of between 270 kg (with time trends) and 870 kg (without time trends). Because the support is around SEK 3000 per ha of wetland, the impact per ha of wetland is between 810 and 2610 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. For P, the removal in our study is between 12 and 51 kg per ha wetland, which is lower than the 100 kg P per ha wetland reported by Weisner et al. (2015).

As a further example, our estimated effect of *Catch crops/No autumn tillage* is similar to the effect found in a field experiment examining a combination of the two measures: 9.2–23.1 kg N per ha in our study,[20] and 16 N per ha in Hansen and Djurhuus (1997). For the other AES, field trial estimates are not available. 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.

6. Discussion

In agreement with field trials *Wetland* and *Catch crop/No autumn tillage* reduce nutrient runoff (see e.g. Braskerud et al., 2005; Strand and Weisner, 2013; Johannesson et al., 2015, for *Wetland* and Hansen and Djurhuus, 1997; Askegaard et al., 2011, for *Catch crop/No autumn*

[16] We have analyzed precipitation data for 68 precipitation stations in Sweden and found that the yearly fluctuations in precipitation is similar within regions (i.e. within Southern Sweden (Götaland) and within Mid Sweden (Svealand)). In Northern Sweden (Norrland) the within variation is larger, but since there is little farmland in Northern Sweden (8% of total farm land in Sweden) this problem is minor.

[17] The average number of samples per hectare of farmland is 0.18 (sd = 0.1).

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

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

[20] 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.

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

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 (kg) brought about by different agri-environmental schemes (AES), expressed per SEK 1,000.

	Nitrogen		Phosphorus	
	Model without time trends included	*Model with time trends included*	*Model without time trends included*	*Model with time trends included*
Wetlands	−866.9 (195.2)	−265.8 (137.2)	−16.9 (6.4)	−4.0 (4.0)
Organic production	18.5 (6.5)	7.0 (5.0)	0.9 (0.3)	0.4 (0.3)
Catch crop/No autumn tillage	−17.8 (6.4)	−7.1 (4.8)	-0.8 (0.3)	−0.7 (0.3)
Grassed buffer zone	93.8 (49.7)	20.6 (42.3)	2.0 (2.0)	−0.8 (1.9)
Env. protection measures	−339.9 (60.2)	−196.3 (36.9)	−6.7 (2.2)	−3.3 (1.9)
Pastures and meadows	33.4 (8.9)	20.8 (8.2)	0.1 (0.3)	0.3 (0.3)
Cult. grassland/Open and var. landscape	1.9 (2.0)	1.0 (1.8)	0.0 (0.1)	0.0 (0.1)
Culturally sig. landscape elements	−258.3 (44.6)	−165 (36.8)	−4.6 (1.9)	−2.2 (1.6)
Mean yearly amount of nutrients flowing through a watershed (kg)	697161		12251	

The numbers are based on the retention-weighted models 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. Robust clustered standard errors in brackets.

tillage). Additionally, *Environmental protection measures* – a type of advisory measure – seems to reduce nutrient runoff. This finding is in agreement with a recent evaluation of extension services in Sweden, (involving a similar set of measures) who found that the service reduced the N surplus on farms (Höjgård and Nordin, 2016).

However, for *Grassed buffer zones* and *Organic production* the results are not in agreement with the expressed aim of reducing nutrient runoff. It is important to be cautious when interpreting these results and remember that the estimates may be biased due to unobserved factors. On the other hand, there are plausible explanations to the increased nutrient runoff for *Grassed buffer zones* and *Organic production*. In essence, what distinguishes organic production from conventional production in Sweden is that mineral fertilisers (and chemical pesticides) are not permitted. Thus, it is probably the use of manure that causes the increased runoff, an assumption supported by other studies (Torstensson et al., 2006; Aronsson, et al., 2011). It should also be noted that several abatement measures may target the same land and in areas where the uptake of *Organic production* is high the uptake of other AES is also high.[21] This affects the translation of the effects and for *Organic production* it implies that the positive impact is conditional on uptake of other AES.

For countries with a similar climate to Sweden, we only found buffer zone field experiments in neighbouring countries. In Finland, buffer zones reduce P runoff from clay soil (Uusi-Kämppä, 2005, 2008) and in Norway they reduce both N and P runoff (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 fully understood (Correll et al., 1997; Mayer et al., 2007). Moreover, although the construction of a buffer zone may decrease nutrient runoff, the removal and subsequent ploughing up of buffer zones may increase nutrient runoff substantially.[22] Thus, since our estimates include windfall and side effects of a scheme – like the continual construction and removal of buffer zones over time – discrepancies between our study and field trials are expected.

Importantly, we find that *Pastures and meadows* increased nutrient runoff and, since this subsidy covers a substantial amount of land, the total impact is potentially large. The support is conditional on grazing the land, and grazing is known to increase nutrient runoff (see e.g. Dougherty et al., 2004; Monaghan et al., 2007; Parvage et al., 2011). Alternatively, the subsidy may prevent pastures and meadows from achieving the leaching decreasing effect of becoming overgrown. Hence, when evaluating the total environmental impact of *Pastures and*

[21] Organic production is the most correlated subsidy to the other subsidies indicated by a high variance inflation factor.

[22] Also, Swedish regulations allow grazing of buffer zones. Grazing implies a trampling damage to the buffer zones, and if cattle congregate on buffer zones around watercourses, this results in direct inflow of manure and urine into the waters.

meadows, the benefit of a varied agricultural landscape and biodiversity has to be weighted with the adverse impact on nutrient runoff.

Cultivated grassland/Open and varied landscape is assumed to reduce nutrient runoff, but we did not find a significant impact. Compared with schemes with a clearly defined aim of reducing nutrient runoff (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 nutrient runoff. Moreover, 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 runoff, but if the subsidy increases fertilised grassland at the expense of land in fallow the impact is the opposite. To investigate this, we 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, the large impact of *Culturally significant landscape elements* on nutrient runoff is a surprise, because many of the managed landscape element are not supposed to impact nutrient runoff. However, the effect is possibly caused by management of certain landscape element which resemble wetlands, e.g. small waters and open ditches. 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). Here it is also important to point out that the effects in this study are identified from the observed variation in payments and for some elements there is probably little variation in payments. For example, 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.

Can our retention weighted watershed fixed-effect model using merged farm level data and water samples provides plausible effects of abatement measures on nutrient runoff? The estimated effects are expected in terms of size, and for most AES they have the expected sign, which partially validates our approach. However, as is the case with most applied research in a non-experimental setting, the effects may be biased due to unobserved factors. The key to estimating a causal effect is a plausibly exogenous change in AES. Moreover, the analysis is marginal which means that the effects are identified from variation in the data, and the effects may be heterogeneous – features shared with most applied research. The main strength of our method is that our estimates incorporate local idiosyncrasies in farmers' implementation, ecology and AES take-up that are hard to replicate in a small-scale field experiment. We believe that our approach works for evaluating abatement measures, but it shares the general concerns common to applied research.

7. Conclusion

To reduce the nutrient overload in the Baltic Sea, the nutrient runoff 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 runoff, with wetlands having the largest impact. It also finds that *Grassed buffer zones*, *Organic production* and *Pastures and meadows* increase nutrient runoff. These adverse effects are associated with extensive livestock farming. Yet, since *Grassed buffer zones* and *Organic production* aim to reduce nutrient runoff we have to interpret these results with caution.

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. However, since abatement measures are likely to target high-impact watersheds and be selective in uptake external validity is difficult to achieve. The impact is also conditional on Swedish watershed characteristics and water regulation, e.g. EUs Nitrates Directive (91/676/EC).[23]

The main contribution of the study is that it uses actual water quality data to investigate the effects of agricultural abatement measures. It is a first step and the subsidy effects may be biased due to time varying unobserved factors. To gain better understanding of the mechanisms and to rule out spurious results an in-depth analysis of each AES is necessary; a suggestion is to use exogenous variation caused by changes in subsidy schemes. Future research should also try to consider the problem with endogenous diffusion coefficients by modelling interactions between different AES and different watershed characteristics. The impact of an abatement measure may vary with the combination of AES and is likely to vary with watershed characteristics.

Another step is a cost-benefit analysis of the various AES, but at this stage it is difficult. Willingness to pay studies (SwEPA, 2009) calculate the societal value in Sweden of a 1 kg reduction in nitrogen reaching the Baltic Sea to be in the interval between SEK 4–70 per kg. The wide interval makes a cost-benefit exercise meaningless.

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Appendix A

[23] The 1991 Nitrates Directive (91/676/EC) aims to protect water quality across Europe. It requires EU 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. However, the Nitrate Directive was implemented before the study period and should therefore not have a direct impact on nutrient runoff.

Table A1
Sensitivity tests.

	Model with covariates		Model with regional time dummies		Model with current and lagged AES				Without water stations in lakes	
					Current	Lag (-1)	Current	Lag (-1)		
	N	*P*	*N*	*P*	*N*		*P*		*N*	*P*
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)
Wetlands	−1.329*** (.312)	−.0291** (.0128)	−1.505*** (.338)	−.032** (.0129)	−.960** (.386)	−.601* (.341)	−.0193 (.0176)	−.0165 (.0161)	−1.390*** (.414)	−.0359** (.0160)
Organic production	.0350*** (.0111)	.0014*** (.0004)	.0315*** (.0115)	.0014*** (.0004)	.0227 (.0192)	.0126 (.0218)	−.0001 (.0007)	.0012* (.0007)	.0316* (.0172)	.0017*** (.0006)
Catch crop/No aut. tillage	−.0137 (.0099)	−.0009* (.0005)	−.0307*** (.0113)	−.0012** (.0005)	−.0413** (.0161)	.0118 (.0166)	−.0015 (.0010)	.0002 (.0008)	−.0568*** (.0183)	−.0021** (.0009)
Grassed buffer zone	.0212 (.0939)	.0009 (.0043)	.159* (.0867)	.0040 (.0036)	.188** (.0827)	.126 (.111)	.0057 (.0049)	−.0010 (.0039)	.467** (.207)	.0089 (.0088)
Env. protection measures	−.607*** (.103)	−.0125*** (.0042)	−.591*** (.105)	−.0123*** (.0043)	−.536*** (.126)	-.0781 (.0820)	−.0063** (.0032)	−.0036 (.0035)	−.771*** (.156)	−.0145** (.0057)
Pastures and meadows	.0993*** (.0200)	.0010 (.0009)	.0587*** (.0156)	.0005 (.0008)	.0451** (.0201)	−.0280* (.0156)	.0013 (.0011)	−.0010 (.0008)	.0790*** (.0199)	.00039 (.0010)
Cult. Grass./Open landsc.	−.0068* (.0036)	−.0003* (.0002)	.0031 (.0034)	−.0001 (.0001)	.0021 (.0038)	.0065* (.0036)	.0001 (.0002)	−.0001 (.0002)	.0092 (.0061)	−.0001 (.0003)
Cult. sig. landscape elem.	−.447*** (.0756)	−.0100** (.0040)	−.444*** (.0775)	−.0103** (.0042)	−.449*** (.110)	.118 (.0971)	−.0033 (.0041)	−.0047 (.0050)	−.416*** (.0990)	−.0089* (.0049)
Observations	23,507	23,924	23,507	23,924	22,249		22,665		17,816	18,071
R-squared	0.036	0.008	0.037	0.010	0.035		0.007		0.019	0.010
Number	2,376	2,426	2,376	2,426	2,343		2,393		1,871	1,921

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. Robust clustered standard errors in brackets. ***p < 0.01, **p < 0.05, *p < 0.1.

Table A2
Sensitivity tests.

	Excluding counties with most samples per hectare		Spring (march-may)		Winter (nov-march)		Summer (april-oct)	
	N (1)	P (2)	N (3)	P (4)	N (5)	P (6)	N (7)	P (8)
Wetlands	−1.560*** (0.346)	−0.0348** (0.0137)	−1.926*** (0.332)	−0.0217** (0.00849)	−1.961*** (0.520)	−0.0400*** (0.0127)	−1.246*** (0.276)	−0.0251** (0.0126)
Organic production	0.0300*** (0.0112)	0.0014*** (0.0004)	0.054*** (0.0158)	0.0005 (0.0003)	0.0143 (0.0195)	0.0014*** (0.0005)	0.0320*** (0.0102)	0.0014*** (0.0005)
Catch crop/No aut. tillage	−0.0294** (0.0116)	−0.0012** (0.0005)	−0.0612*** (0.0171)	−0.0011** (0.0005)	0.00064 (0.0175)	−0.0011** (0.0005)	−0.0421*** (0.0121)	−0.0012** (0.0006)
Grassed buffer zone	0.126 (0.0847)	0.0035 (0.0037)	0.143 (0.0910)	0.0039 (0.0031)	0.0461 (0.101)	−0.0067 (0.0059)	0.182** (0.0901)	0.0089** (0.0039)
Env. protection measures	−0.600*** (0.106)	−0.0130*** (0.0044)	−0.869*** (0.146)	−0.0086*** (0.0028)	−0.760*** (0.151)	−0.0083* (0.0048)	−0.476*** (0.0939)	−0.0132*** (0.0043)
Pastures and meadows	0.0746*** (0.0172)	0.0004 (0.0008)	0.105*** (0.0202)	0.0015*** (0.0005)	0.0811*** (0.0204)	0.0013** (0.0006)	0.0557*** (0.0150)	−0.0005 (0.0010)
Cult. Grassl./Open landsc.	0.00304 (0.00360)	−0.0001 (0.0002)	0.0010 (0.005)	0.0001 (0.0001)	−0.0030 (0.0060)	−0.0002 (0.0003)	0.0071* (0.0041)	−0,0000 (0.0002)
Cult. sig. landscape elem.	−0.485*** (0.0808)	−0.0099** (0.0043)	−0.596*** (0.0995)	−0.0064** (0.0026)	−0.607*** (0.0955)	−0.0086 (0.0054)	−0.359*** (0.0787)	−0.0097** (0.0040)
Observations	18,799	19,060	19,923	20,27	20,237	20,561	23,159	23,598
R-squared	0.049	0.008	0.029	0.021	0.043	0.007	0.025	0.009
Number	1,997	2,337	1,963	2,001	1,987	2,022	2,266	2,316

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. Robust clustered standard errors in brackets. ***p < 0.01, **p < 0.05, *p < 0.1.

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