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

Grenestam Erik[a], Nordin Martin[b,*]

[a] Department of Economics, Lund University, PO Box 730, SE-220 07, Lund, Sweden
[b] AgriFood Economics Centre and Department of Economics, Lund University. PO Box 730, SE-220 07, Lund, Sweden

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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

* Corresponding author.
E-mail address: martin.nordin@agrifood.lu.se (M. Nordin).

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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.