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Full Research Article

The Common Agricultural Policy as a driver of water quality changes: the case of the Guadalquivir River Basin (southern Spain)

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Abstract. Several studies have analysed the effects of European environmental policies on water quality, but no detailed retrospective analysis of the impacts of the Common Agricultural Policy (CAP) reforms on observed water quality parameters has been carried out. This study evaluates the impact of the CAP and other drivers on the concentrations of nitrates and suspended solids in the Guadalquivir River Basin (southern Spain) over the 1999-2009 period. The most important drivers that are degrading both water quality indicators are exports from upland areas and agricultural intensification. Water quality conditions have improved in regions where there has been abandonment and/or deintensification. The decoupling process has reduced the concentration of nitrates and suspended solids in a number of subbasins. Although agricultural production and water efficiency in the basin have improved, high erosion rates have not yet been addressed.

Keywords. Common Agricultural Policy, freshwater quality, nitrates, suspended solids, panel data.

JEL Codes. Q18, Q25

1. Introduction

Water quality conditions play a critical role in present and forthcoming water sustainability. By 2015, the ecological status, as defined by the Water Framework Directive (WFD), of almost half of Europe's surface water bodies is likely to be poor, with pressure from diffuse agricultural pollution becoming a growing concern (EEA, 2012). Achieving more efficient and equitable water management objectives at catchment level, not only relates to the actual water resource itself, but is influenced by water related policies

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and the application of scientific knowledge, For instance, the Blueprint report (European Commission, 2012a) emphasizes the need for better implementation and deeper integration of water policy objectives with the Common Agricultural Policy (CAP).

During the period 1999 to 2009, two CAP reforms were implemented: Agenda 2000 and 2003 CAP. The objectives of Agenda 2000 were based primarily on the convergence of cereal support prices with world markets and the introduction of decoupled area payments for herbaceous crop growers (MAPA, 2002). The 2003 CAP reform introduced a more radical change with the establishment of the Single Payment Scheme (SPS) for all crops, decoupled from production and conditional upon cross-compliance. "Cross-compliance is a mechanism that links direct payments to compliance by farmers with basic standards concerning the environment, food safety, animal and plant health and animal welfare, as well as the requirement of maintaining land in good agricultural and environmental condition (GAEC)"[1]. Cross-compliance involves 18 statutory management requirements (e.g. the Nitrates Directive) and a number of measures for ensuring GAEC (e.g. control of soil erosion and soil organic matter content). Under both reforms, farmers could participate in agri-environmental measures (AEM) (e.g. organic farming, crop and farming extensification and set aside) as part of rural development programmes (OJEC, 1999) and some of these AEM became mandatory with the introduction of cross-compliance. Based on the changes introduced by the CAP reforms, we analyse their effects on existing water quality conditions.

Several differing results were reached with respect to the expected effects of the 2003 CAP reform on changes of nitrate pollution. On the one hand, no significant changes were found in water quality status when crop pattern changes after the reform do not differ significantly in nutrient requirements, as in the Midi-Pyrenees (France) (Belhouchette *et al.*, 2011). The concentration of nitrates in this case, would not decrease if there is not a big reduction in subsidies in the event of non-compliance (*ibid*). However, decoupled income support can lead to reductions in nitrate pollution as a result of more diversified production patterns and extensive management practices with the introduction of less nitrogen-intensive crops and the reduction of the cultivated area of the most nitrogen-intensive crops (Gallego-Ayala and Gómez-Limón, 2009; Cortignani and Severini, 2012). Volk *et al.* (2009) highlighted that the most effective way to mitigate nitrate pollution would be to pursue management practices that move away from conventional farming and towards eco-farming practices, and to convert arable land to pastureland. On the other hand, decoupled direct subsidies, together with higher agricultural prices, can stimulate intensification and crop development linked to trade liberalization (Martínez and Albiac, 2006; Sieber *et al.*, 2013). These potentially conflicting outcomes are an obstacle to the attribution of the observed environmental effects to policy reforms.

Another environmental concern is soil erosion. Some studies have identified this to be a consequence of intensification promoted by subsidies coupled to crop production (Boardman *et al.*, 2003), particularly for traditional olive orchards in Spain (de Graaff and Eppink, 1999). The SPS still favours more intensive olive orchards since the amount of this payment at farm level was, until 2014, dependent on historical production during the 1999-2002 period (de Graaff *et al.*, 2011). Nevertheless, there is no specific EU legal basis

[1] http://ec.europa.eu/agriculture/envir/cross-compliance/index_en.htm.

for soil protection in Europe, and the Soil Framework Directive proposal was withdrawn in 2014, although the European Commission is working on the Thematic Strategy for Soil Protection (European Commission, 2006), e.g. soil protection as an integral part of the GAEC (European Commission, 2012a).

In Europe, no ex-post evaluation of the implications of CAP implementation for observed surface water quality has yet been performed. Considering the failure to fulfil the goals of the WFD and the existing environmental concerns in Europe, this study proposes an approach for determining the implications of the CAP reforms for nitrates and suspended solids during the period 1999 to 2009, utilizing a specific case study in the Guadalquivir River Basin (southern Spain). Our approach aims at understanding the relationships between water quality parameters and factors such as agricultural production, natural environment and economic policy indicators.

2. Materials and methods

2.1 Site of study

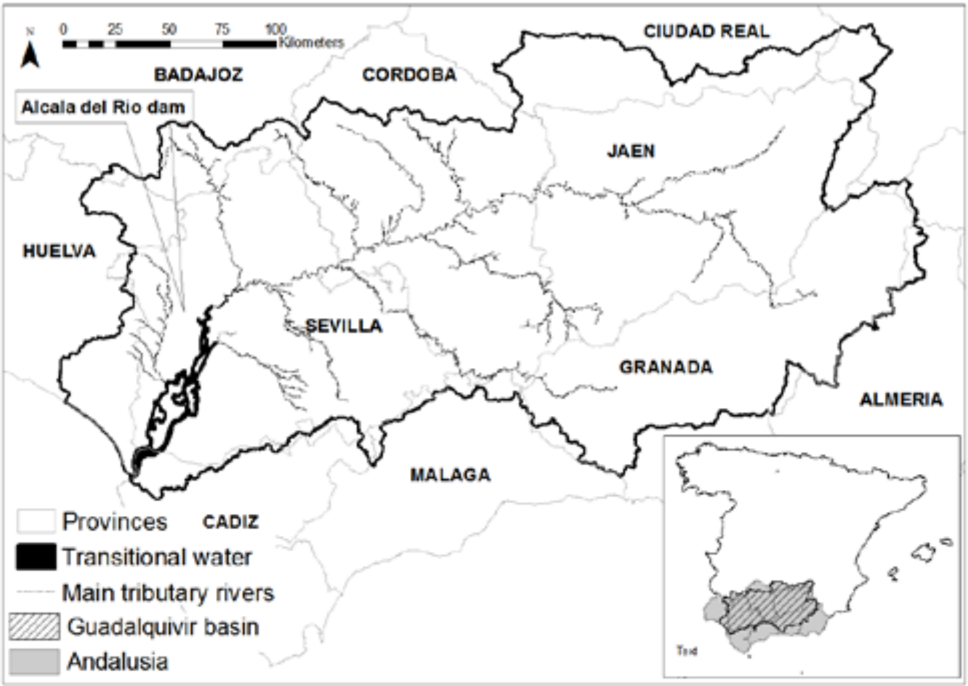
The Guadalquivir River Basin (GRB), located in southern Spain, was selected for the case study due to existing water quality concerns, as well as the importance of agricultural production in the basin. The main pressure on surface water bodies in the GRB is diffuse pollution (50%), followed by point pollution (37%) (GRBA, 2012). Emerging water quality problems caused by diffuse pollution are related to land and water use in the basin, i.e. agricultural production on hillsides aggravates water runoff and soil erosion (Blomquist *et al.*, 2005). There is a high risk of erosion in the basin as a result of olive orchard production (Gómez, 2008; Taguas *et al.*, 2011; Taguas *et al.*, 2013). Erosion rates are in excess of 50 t ha^{-1} $year^{-1}$ for 20% of the olive orchard extension in the upper and middle part of the GRB (RGA, 2008). In 2009, cropland accounted for approximately 2,650,780 ha (not including pastureland), with 31% of the area under irrigation. Olive orchards are the principal crop in the case study site and account for 56% of the total cropland area, followed by wheat (13%) and sunflower (9%) (MAGRAMA, 2012).

The GRB has a surface area of 57,530 km^2, and the climate in the basin is Mediterranean with precipitation ranging from 289 mm to 743 mm over the 1998-2009 period. The hydrographic network is configured around the axis of the Guadalquivir River, which is 655 km long and has a mean annual discharge of 7022 hm^3. Estuarine water bodies are to be found downstream, including the Doñana National Park, which is a high-biodiversity area and a wildlife shelter for migratory European and African birds (Fernández-Delgado, 2005) (Figure 1).

2.2 Subbasin delineation and water quality dataset

Based on a 100 × 100 m digital elevation model (DEM) (CNIG, 2011), the drainage area of each monitoring station is calculated. A total of 89 water quality monitoring stations are located within the GRB, which cover a total of 12,619 km^2. This accounts for 22% of the GRB. The drainage area identifies the area upstream of the monitoring station that is related to the existing water quality conditions. ArcGIS 9.3 (ESRI, 2009) is used

Figure 1. Guadalquivir River Basin.



to map out the subbasin boundaries that are delineated by means of a two-step process. First, if a monitoring station is located at the mouth of the subbasin, the delineated subbasins provided by GRBA (2012b) are selected. Second, if a monitoring station is situated inside a subbasin, the bottom boundaries are delineated using the DEM. The 89 delin eated subbasins are then classified depending on the dominant type of irrigated agriculture according to the criteria of geographic proximity, production orientation and economic importance (RGA, 2010a): olive orchards ('Olive'), mountainous areas ('Mountain'), intensive crops in coastal areas ('Coast') and semi-intensive crops ('Semi-intensive') (Figure 2).

A water quality dataset is created for each monitoring station, including the water quality indicator (nitrates and suspended solids) and the year and month for the period from September 1998 to August 2009 according to data available from the GRBA (2011), though the series are not complete for all stations (Table 1).

2.3 Water quality trends

In order to understand our water quality dataset, we identified the annual[2] water quality trends. In general, the water quality parameters in the GRB have remained stable dur-

[2] In our study, the annual scale refers to the period comprising the agricultural season from the beginning of September until end of August.

Figure 2. Study subbasins classified by dominant type of agriculture. Source: own elaboration based on monitoring station location (GRBA, 2011), digital elevation model (CNIG, 2011) and main irrigated regions (RGA, 2010a).

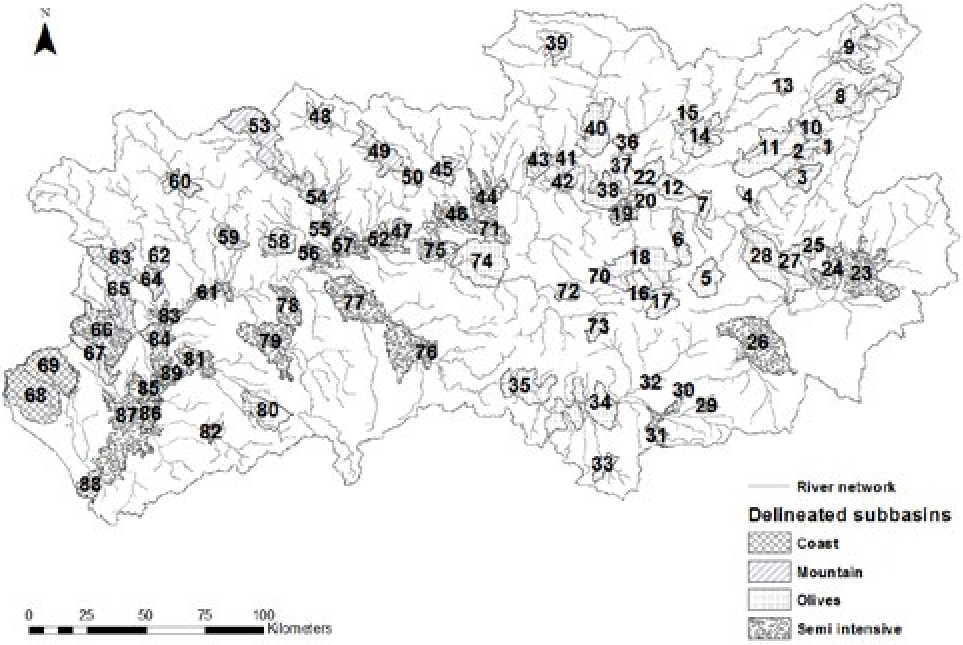


Table 1. Descriptive statistics for monthly observations at 89 sampling stations over the 9/1998-8/2009 period and by subbasin classified according to dominant type of irrigated agriculture.

Subbasin classification	No. Sampling stations	Nitrates				Suspended solids			
		No. Observations	p10 (mg L^{-1})	p50 (mg L^{-1})	p90 (mg L^{-1})	No. Observations	p10 (mg L^{-1})	p50 (mg L^{-1})	p90 (mg L^{-1})
Mountain	13	916	1	2	7	827	3	8	30
Olive	41	2201	1	5	21	2551	4	25	156
Coast	3	158	1	5	26	210	11	41	191
Semi-intensive	32	2307	2	12	33	2377	9	53	280

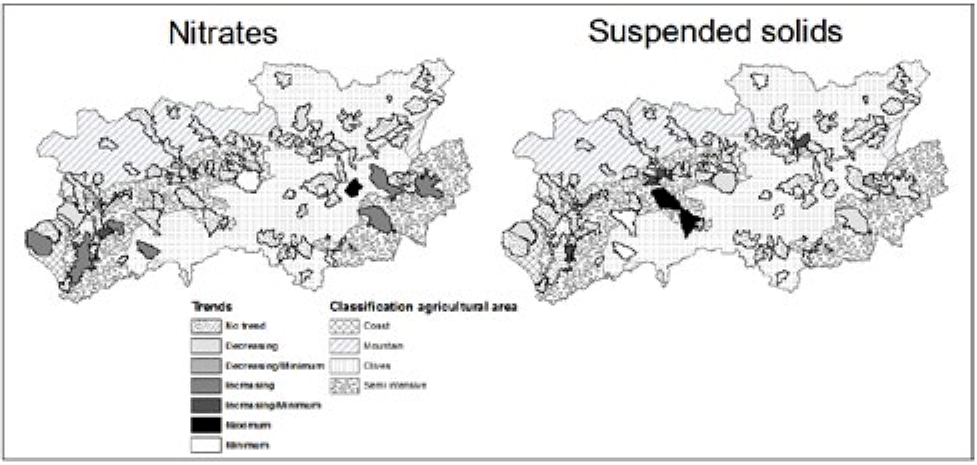
*p10: 10th percentile; p50: 50th percentile (median), p90: 90th percentile.

ing the study period. But most of the significant trends for both nitrates and suspended solids are indicative of a degradation of the water quality conditions. For nitrates, the prevalent existing annual trends indicate an increase (linear or minimum) in the concentration of nitrates (Figure 3). The group of nitrate-increasing trends occur mostly within

'Semi-intensive' areas in the south-eastern and mid-Guadalquivir valley and the river estuary, dominated by annual crops.

Regarding the behaviour of suspended solids, most trends show a quadratic component (15 out of 89 subbasins), and particularly a U-shaped curve (14 subbasins). Nevertheless, despite the acute concentrations of suspended solids in the basin, only five monitoring stations recorded reductions in the concentrations of suspended solids. The concentration of suspended solids is improving in the east, upper and middle part of the basin. The concentration of suspended solids (both linear and quadratic) is worsening across the subbasins classified as 'Olive' in particular and in 'Semi-intensive' areas along the Guadalquivir River axis (Figure 3).

Figure 3. Linear and quadratic trend regressions with significant coefficients ($p<0.05$) for median annual values of concentrations of nitrates and suspended solids. The linear trend provides the slope (*b)* of the time variable (*H)*, with $y_t = a + bH_t + \varepsilon_t$, where y_t is the concentration of either nitrates or suspended solids, H_t is the time trend ($t=1,...,T$) and ε_t is the error term. The quadratic trend equation $y_t = a + bH_t + cH_t^2 + \varepsilon_t$ is used to check for convex or concave trend behaviour. The classification of dominant agricultural areas across the basin is also shown.



2.4 Subbasin characterization

The main factors related to surface water quality conditions for each subbasin *i* and agricultural season *t*, characterized by 21 variables over the period September 1998 to August 2009 are as follows: 1) climatic and physical environmental characteristics (*Precipitation, Slope, Erosion, Soil permeability,* $Export_{NO3}$ *and* $Export_{SS}$), 2) urban point sources (*Population density*), 3) agriculture structure and productivity measures ($Biomass_{rainfed}$, $Biomass_{irrigated}$, *Shannon, % Drip* and N_{cons}), and 4) economic and policy indicators (*Agenda 2000 reform, 2003 CAP reform,* $Subsidies_{rainfed}$, $Subsidies_{irrigated}$, *% Coupling, VZ ratio, Crop price index,* $Price_N$ *and* $Price_{fuel}$) (see Table 2 for further details).

ArcGIS 9.3.1 (ESRI, 2009) was used to adapt geographical information from biophysical (i.e. basin) or administrative (i.e. municipal, province) level to subbasin level.

The mean annual precipitation, slope, erosion rates and soil permeability were calculated for each subbasin based on georeferenced information (CNIG, 2011; GRBA, 2012b; SGPYUSA, 2012). The categorical erosion and soil permeability variables were weighted by the area of each classification. Exports of nitrates ($Export_{NO3}$) and suspended solids ($Export_{SS}$) from upland areas were also considered as factors affecting the environmental quality. They were included as the concentration of the water quality indicator for the closest upstream subbasin. Although this study focuses on diffuse pollution, we also estimated point source pollution using population density based on the annual population of each municipality (INE, 2013).

For agricultural structure and productivity measures, total aboveground biomass in dry weight was used as an indicator of agricultural intensification since it can aggregate all agricultural biomass generated within a region. The study area includes 129 crop species. The total aboveground biomass was calculated separately for rainfed ($Biomass_{rainfed}$) and irrigated ($Biomass_{irrigated}$) systems by totalling the agricultural (t) and residual (t) production and dividing by the subbasin area (ha). Agricultural production comprises the economic or agricultural parts (grain, fibre, fruit or tuber). Residual production refers to the crop residues that remain in the field after the crop is harvested. Supplemental Material A details the calculation process for crop area and total aboveground biomass, distinguishing between rainfed and irrigated systems and annual and woody crops. Secondly, crop diversity was measured using the Shannon Index (*Shannon index*), which spatially and temporally characterizes crop area allocation to different species. A greater *Shannon index* value is indicative of more diversified agricultural areas. Thirdly, irrigation system modernization is included as the percentage of drip irrigation (*% Drip*) per type of agricultural classification ('Olive', 'Semi-intensive, 'Coast' and 'Mountain'), sourced from the Andalusian Regional Government (RGA 2010c). *% Drip* is interpolated and extrapolated between 1997 and 2008 to obtain the annual observations for the 1999-2009 period. Finally, since data on nitrogen fertiliser applications are not available at subbasin level, the average consumption of nitrogen (N_{cons}) at subbasin level was estimated by multiplying the average nitrogen rates in Spain (kg ha^{-1}) by the total cropland area per subbasin (ha) and dividing by the total area of each subbasin (ha).

For the economic and policy indicators, the study looks at the effects of the CAP, the EU Nitrates Directive and crop, fertiliser and fuel prices. Agricultural subsidies ($Subsidies_{rainfed}$ and $Subsidies_{irrigated}$ in € ha^{-1}) were used as the first CAP policy indicator for the 1999-2009 period, which considers Agenda 2000 (2000-2006) and the 2003 CAP reform (2007-2009) changes. A one-year lag for both variables ($L.Subsidies_{rainfed}$ and $L.Subsidies_{irrigated}$) was also considered, since farmers' behaviour might also be influenced by subsidies from the previous agricultural season. A total of 32 crops were eligible for subsidies. The second CAP policy indicator was the average percentage of coupled subsidy (*% Coupling*). Calculations for both CAP indicators are detailed in Supplemental Material B. Agenda 2000 introduced some voluntary agricultural AEMs for farmers (e.g. strip zones and organic farming) and the 2003 CAP reform considered GAEC as a requirement for cross-compliance. However, no information is available at subbasin level regarding the level of participation to AEMs during Agenda 2000 and GAEC during cross-compliance implementation. Since we cannot characterize the AEMs and GAEC in quantitative terms, a dummy variable characterizes agricultural policy implementation after the 2000/01 agri-

Table 2. Characterization of subbasins by agricultural season over the period from 9/1998 to 8/2009.

Variable classification	Variable	Units	Available data
Climate and physical environment	*Precipitation*	mm	- Monthly raster data (SGPYUSA, 2012)
	Slope	%	- Digital elevation model with a grid cell size of 100 m x 100 m (CNIG, 2011)
	Erosion	$t\ ha^{-1}$ Classification: 0-5, 5-12, 12-25, 25-50, 50-100, >200	- Erosion rates (GRBA, 2012b)
	Perm	Classification: 1: very low, 2: low, 3: medium, 4: high; 5: very high	- Soil permability (GRBA, 2012b)
	Exports of NO_3 and SS from upland areas ($Export_{NO3}$, $Export_{SS}$)	$mg\ NO_3\ L^{-1}$ $mg\ SS\ L^{-1}$	- Water quality indicator concentration for the nearest headwater subbasin (GRBA, 2011)
Urban point sources	*Population density*	population per km^{-2}	- Annual population by municipality (INE, 2013)
Agriculture structure and productivity measures	Total aboveground biomass for rainfed crops ($Biomass_{rainfed}$) and irrigated crops ($Biomass_{irrigated}$)[1]	$t\ ha^{-1}$	- Land use maps for years 1999, 2003 and 2007 (RGA, 2010; MARM, 2009). - Andalusian irrigated crop inventories for years 1996 and 2002 (RGA 1999; 2003) - Irrigated crop location in 2010[3] - Crop yields (MARM, 2012) - Harvest index (see Supplemental Material A Table A1) - Residue to product ratio (see Supplemental Material A Table A1)
	Crop diversity (*Shannon index*)		- Municipal crop area data (MAGRAMA, 2012a)
	Modernization of irrigation system (*% Drip*)	%	- Irrigation method (surface, sprinkler and drip) area by irrigated agriculture classification ('Olive', 'Semi-intensive, 'Coast' and 'Mountain') for years 1997 and 2008 in Andalusia (RGA, 2010b).
	Consumption of nitrogen (N_{cons})[1]	$kg\ ha^{-1}$	- Average national nitrogen consumption rates (MAGRAMA, 2012b)

Variable classification	Variable	Units	Available data
Economic and policy indicators	*Agenda 2000 reform*		- Dummy variable with value of 1 after 2000/01 agricultural season
	2003 CAP reform		- Dummy variable with value of 1 after 2006/07 agricultural season
	Agricultural subsidies for rainfed crops ($Subsidies_{rainfed}$) and irrigated crops ($Subsidies_{irrigated}$)[1,2]	€ ha^{-1}	- Agricultural subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) before 2006/07 agricultural season (see Supplemental Material B Table B1) - Percentage of decoupled payments, reference period and subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) after 2006/07 agricultural season (see Supplemental Material B Table B1) - Crop yields (MARM, 2012)
	Percentage of coupled subsidy (*% Coupling*)[1]	%	- Percentage of decoupled payments (see Supplemental Material B Table B1)
	Nitrate vulnerable zone ratio (*VZ ratio*)[1]		- Nitrate vulnerable zone and crop area affected by EU Nitrates Directive (BOJA 1999; BOJA 2001)
	Crop price index[1,2]		- National crop prices (MAGRAMA, 2012b)
	Fuel price ($Price_{fuel}$)	euro L^{-1}	- Annual fuel price (ASAJA, 2011) (UPA, 2012)
	Nitrogen element price ($Price_N$)	euro t^{-1} N	- Nitrogen element price. Annual fertiliser prices (MAGRAMA, 2012b)

[1] Municipal crop area data (MAGRAMA, 2012a) is also required for the calculation in order to determine crop area at subbasin level (see Supplemental Material A).
[2] A one-year lag was also considered as an explanatory variable.
[3] Provided by the Guadalquivir River Basin Authority.

cultural season (*Agenda 2000 reform*) and a second dummy variable accounts for agricultural policy changes after the 2006/07 agricultural season (*2003 CAP reform*).

The EU Nitrates Directive implementation is accounted for by the ratio of nitrate vulnerable zone per subbasin area (*VZ ratio*). The extension of the vulnerable zone was divided by the subbasin area and multiplied by the proportion of crop area affected by the EU Nitrates Directive within the vulnerable zone. A *Crop price index* was calculated based on 2000 current prices and weighted by crop production. A one-year lag for crop price index (*L.crop price index*) was also considered, since agricultural practices might also depend on prices from the previous agricultural season. The prices of the nitrogen element ($Price_N$) and fuel ($Price_{fuel}$) were also considered for the subbasin characterization. $Price_N$ was calculated as the weighted price of each fertiliser based on nitrogen element content.

2.5 Significant correlations between variables under study

A pairwise correlation analysis was performed between all independent and dependent variables under study in order to better understand their behaviour. Independent variables that present constant values between subbasins and significant correlations ($\rho>0.5$, $p<0.001$) with other explanatory variables were excluded from the subsequent analysis of panel data regressions.

2.6 Fitting panel data models

2.6.1 Model formulation

Panel data analysis for N units (subbasins) over T periods (agricultural seasons) is applied in order to explain the variation of the median (p50) of both physicochemical indicators taking into account the variables described in Section 2.4 that characterize the water quality status. Through panel data analysis we can model time series processes while accounting for heterogeneity across geographical units (i.e. subbasins characterized by monitoring stations). The general regression model for analysing panel data is formulated as follows:

$$y_{it} = z_i'\alpha + x'_{it}\beta + \varepsilon_{it} \qquad i= 1,..., N;\ t=1,..., T, \tag{1}$$

where X_{it} is the *i*th observation of each explanatory variable. Heterogeneity is controlled by the intercept $z_i'\alpha$, where z_i includes a constant term and a set of individual or group-specific variables (Greene, 2012). The error component model includes the unobservable unit effects (λ_i) and the remainder disturbance (u_{it}):

$$\varepsilon_{it} = \lambda_i + u_{it.} \tag{2}$$

The fixed effect (FE) model assumes λ_i to be fixed, independent of u_{it} and identically distributed (IID) $(0, \sigma^2_u)$. This model requires estimating N separate λ_i that, together with the intercept $z_i'\alpha$, comprise a dichotomous variable (ν_i) for each unit. FE only analyses the impact of variables that vary over time, since time-invariant variables are absorbed by ν_i. The random effect (RE) model assumes λ_i to be random, where λ_i ~ IID (0, σ^2_λ), u_{it}~ IID (0, σ^2_u) and λ_i to be independent of u_{it} (Baltagi, 2008).

However, FE or RE regression residuals often have attributes that ordinary least squares (OLS) cannot handle. Feasible generalized least squares (FGLS) or panel corrected standard errors (PCSE) can be used to deal with heteroskedasticity[3] (HET), contemporaneous cross-correlation[4] (CCC) and first-order autocorrelation[5] (AR (1)). Driscoll and Kraay standard errors (DKSE) is applied when the autocorrelation is a moving average type (MA) (Hoechle, 2007), which represents the average value of a variable over a given period of time. We opted for the PCSE or DKSE models, since our dataset does not always have the same number of observations per subbasin and FGLS requires rectangularized datasets.

2.6.2 Model selection

STATA 12 statistical software (StataCorp, 2011) was used for model fitting. We set up models on three different scales: 1) the whole basin including all subbasins ('Total'), 2) a group of subbasins selected according to the dominant type of irrigated agriculture: 'Olive', 'Semi-intensive', 'Coast' and 'Mountain', and 3) a group of subbasins selected according to actual water quality time trends: 'Increasing', 'Decreasing' and 'Minimum'. Regressions for the 'Maximum' classification were not carried out because there were fewer observations (less than 10). Since the analysis comprises two water quality parameters and different subbasin classifications, we used the following abbreviation: [water quality parameter] [subbasin classification].

Variables were log-transformed as *ln(variable+1)* and standardized to have a mean of zero and a standard deviation of one. Regressions were firstly run for pooled OLS, and we discarded the explanatory variables with a variance inflation factor greater than 4. Then panel RE and FE OLS were run. After running the RE model, we conducted the Breusch and Pagan Lagrange multiplier test (Breusch and Pagan, 1980) to verify the absence of RE with the null hypothesis that error variance across units is zero. After running FE regressions, we tested the null hypothesis that all dichotomous variables are zero (Ho: $\nu_1 = \nu_2 = \nu_i = 0$) by means of a F test (Snedecor and Cochran, 1983). The selection of RE or FE depends on whether the individual error component (u_{it}) and explanatory variables are correlated. This is identified using the Hausman test (Hausman, 1978).

We chose the PCSE model when disturbances were assumed to be heteroskedastic across panels or heteroskedastic and contemporaneously cross-correlated across panels with or without AR (1). Heteroskedasticity was detected in the FE residuals with the modified Wald statistic for groupwise heteroskedasticity (Greene, 2012). To test the cross-sectional correlation, we performed the Pesaran test (Pesaran, 2004) after running FE with the null hypothesis that the error term is independent across sections. Finally, autocorrelation was tested under the null hypothesis of no serial autocorrelation (Wooldridge, 2002).

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[3] The error variance of each panel is not constant.

[4] Observations of some panels are correlated with other panels during the same period of time. It refers to the error correlation of at least two or more panels.

[5] Autocorrelation occurs when errors are not independent with regard to time.

We opted for the DKSE model when heteroskedasticity, cross-correlation panels and MA were detected' MA was identified if the coefficient of determination presented larger values for the DKSE model than for PCSE. Explanatory variables with a pairwise correlation coefficient greater than 0.5 were excluded from the regressions. We only kept variables with significant coefficient estimators. The Wald test checked that the removed variables were not significant in the final panel model regression according to the null hypothesis that coefficients are zero.

3. Results and discussion

3.1 Main correlations between variables under study

We first analysed the correlations between all the examined independent and non-independent variables before performing the panel data regressions. The significant pairwise correlations that are reported in this section refer to p<0.001. Under farmers' control, larger fertilization rates (N_{cons}) are correlated to larger levels of nitrates and sus pended solids in rivers (ρ>0.5). Besides, when the price of nitrogen fertilisers ($Price_N$) and fuel ($Price_{fuel}$) increases, N_{cons} decreases (ρ>-0.1). $Price_N$ and $Price_{fuel}$ are larger after both the *Agenda 2000* (ρ>0.5) and *2003 CAP reforms* ($Price_N$: ρ>0.8, $\text{Price}_{\text{fuel}}$: ρ>0.6). Both $Price_N$ and $Price_{fuel}$ are highly correlated (ρ>0.9). The modernization of irrigation systems is also significant after both agricultural reforms (ρ>0.1) (see Supplementary Information C and Table C1). These results show that, aside from a few significant policy changes, there are other factors that occur in parallel, which may have an effect on farmers' behaviour, with consequences for water quality indicators, i.e. the increase of price of nitrogen fertilisers and fuel.

For the agricultural production systems, both larger values of agricultural intensification ($Biomass_{rainfed}$: ρ>0.2 and $Biomass_{irrigated:}$ ρ>0.5) and greater crop diversity (*Shannon index*: ρ>0.2) degrade the level of the two water quality indicators. As a result, we can expect agricultural regions with a greater variety of crops to be more intensified, as the Shannon index indicates with its positive correlation to $Biomass_{irrigated}$ (ρ>0.3). For erosion concerns, specific soil conservation practices should be considered with a greater variety of crops, e.g. vegetation filters, contour tilling. More frequent tillage practices or herbicide control of weeds reduce surface cover and roughness (Vanwalleghem *et al.*, 2011) and can lead to higher erosion risks.

In terms of the impacts of policy measures, we found larger concentrations of nitrates and suspended solids are positively correlated with a larger degree of coupling to production (*% Coupling*: ρ>0.4), agricultural subsidies ($\text{Subsidies}_{\text{rainfed}}$: ρ>0.1, $\text{Subsidies}_{\text{irrigated}}$: ρ>0.3), and implementation of nitrate vulnerable zones (*VZ ratio*: ρ>0.3). Furthermore, we found a greater probability of runoff or leakage for nitrates in the vulnerable zones that can also be explained by the greater soil permeability in these zones. Nevertheless, policy makers and farmers need to bear in mind that alongside controlling nitrogen pollution, additional measures need to be applied in vulnerable zones to restrict erosion rates, as the positive sign of the ratio of *VZ* to concentrations of suspended solids suggests. Concentrations of nitrates are higher after the *2003 CAP reform* (ρ>0.07), whereas the concentration of suspended solids decreases (ρ>-0.07) after the *Agenda 2000 reform*. The reduction of erosion rates after the *Agenda 2000 reform* in our study is consistent with Fleskens and de Graaff (2010). They also found positive environmental outcomes of cross-compliance and AEM policy instruments with respect to soil conservation.

3.2 Panel data analysis

Modelling results for the median annual measurements of nitrates (Table 5) and suspended solids (Table 6) indicate whether panel data analysis is required instead of pooled regression when there are relevant random effects or fixed effects between subbasins. Models fitted with fixed effects ('Minimum NO_3, 'Decreasing SS' and 'Increasing SS') require a dichotomous variable for each control entity for all time-invariant differences between subbasins, and only assess the net effect of predictors changing over time. $Price_N$ and $Price_{fuel}$ were not considered in the panel data regressions because of their high correlation ($\rho>0.5$, $p<0.001$) with the *Agenda 2000 reform* and the *2003 CAP reform*. Additionally, *erosion* was not included either in the analysis because of its correlation with *slope* ($\rho>0.6$, $p<0.001$) (see Supplementary Information C and Table C1).

For the total study area ('Total') and 'Olive' subbasin sample, the concentration of nitrates increases with nitrate export from upland areas ($Export_{NO3}$), biomass intensification ($Biomass_{rainfed}$ and $Biomass_{irrigated}$), population density (*population density*) and soil permeability (*Perm*). As a result, more agriculturally intensified regions are related to larger concentrations of nitrates, where, as in previous studies, export from upland areas can be a major contributor of excess nutrients (King and Balogh, 2011). Besides, there is a greater probability of nitrate leakage for more permeable soils, as the positive sign of *Perm* in 'Total' and 'Olive' (as well as in 'No trend') indicates.

In 'Mountain', concentrations of nitrates are larger with greater agricultural intensification, lower agricultural subsidies for irrigated areas ($Biomass_{irrigated}$ and $Subsidies_{irrigated}$) and before the *2003 CAP reform*. The negative sign of the *2003 CAP reform* might represent the effect of a shrinkage of agricultural areas by nearly 30% during the study period (from 41,675 ha to 29,130 ha), with this explanatory variable concealing drivers not directly related to CAP measures. Deintensification has occurred in 'Mountain' with a decrease of irrigated biomass, particularly with respect to industrial crops (from 7.7 to 4.9 t ha^{-1}) and cereals (from 12.2 to 10.4 to ha^{-1}). However, after the reform, there has been some intensification with respect to fodder (from 11.5 to 17.1 t ha^{-1}), as well as with the expansion of irrigated olive orchards (from 390 to 1,520 ha). Technical progress (i.e. irrigation of Spanish farms) makes it possible to maintain or even increase crop yields in areas with less propitious physical conditions (Bakker *et al.*, 2011). As Hatna and Bakker (2011) reported, processes of abandonment, intensification and expansion can be found at the same time and are more likely to occur in dry, warm and accessible areas.

This study also shows that the nitrate content of receiving water bodies can be reduced through irrigation system modernization (negative sign of *% Drip* in 'Increasing NO_3'). The concentration of nitrates drops with modernized systems because nitrogen losses in the irrigation return flows are reduced through the development of more efficient irrigation systems (Lecina *et al.*, 2010; Barros *et al.*, 2012). Similarly, the reduction in the area of vulnerable zones (negative sign of *VZ ratio* in 'Increasing NO_3' and 'Semi-intensive NO_3') is related to higher concentrations of nitrates. As a result, the whole alluvium area

Table 5. Panel data regressions for nitrates (NO_3) considering all the subbasins (Total), by type of agriculture (Olive, Coast, Semi-intensive and Mountain) and by existing time trend (Decreasing, Increasing, Minimum and No Trend). The regression models include significant variables ($p<0.05$) only.

Dependent variable NO_3	Subbasin classification								
Explanatory variables	Total	Olive	Coast	Semi-intensive	Mountain	Decreasing	Increasing	Minimum	No Trend
$Export_{NO3}$	0.21***	0.14**		0.27***			0.25**		0.23***
Precipitation						-0.19*			
Perm	0.10***	0.15*		-0.13***					0.11***
Population density	0.09***	0.15***	0.75***					39.65***	0.07***
$Biomass_{rainfed}$	0.32***	0.25***					0.35***		0.24***
$Biomass_{irrigated}$	0.37***	0.38***	-0.61*		2.85***				0.21***
Shannon				0.13*					0.09***
N_{cons}				0.71***					0.23***
% Drip				0.35**			-0.38***		
VZ ratio				-0.06***			-0.52***	3.69**	
Price index				0.09*					
L.Price index			-1.02*						
$Subsidies_{rainfed}$					-0.09*	0.30***			
$Subsidies_{irrigated}$		0.08*							
$L.Subsidies_{rainfed}$			-0.48*						
$L.Subsidies_{irrigated}$						0.17*			
% Coupling							0.76***	-0.98***	0.15*
Agenda 2000 reform	0.07*				-0.45***		0.62*		
2003 CAP reform		-0.11*	-1.71***	0.41***	1.62***	-0.27**	-0.95***	-19.21***	
Intercept	0.21***	0.14**		0.27***			0.25**		0.23***
No. Subbasins	89	41	3	32	13	5	3	14	71
No. Observations	873	334	30	260	137	35	70	51	717
R^2	0.49	0.46	0.76	0.49	0.28	0.53	0.54	0.70	0.57
Model	DKSE with RE HET MA CCC	PCSE with RE HET CCC	Pooled OLS	DKSE with RE HET MA CCC	Pooled OLS	Pooled OLS HET	Pooled OLS	PCSE with FE HET CCC	PCSE with RE HET CCC

* $p<0.05$, ** $p<0.01$, *** $p<0.001$. OLS: ordinary least squares, PCSE: panel corrected standard errors, DKSE: Driscoll-Kraay standard errors, FE: fixed effects, RE: random effects, AR(1): AR(1)-type autocorrelation, MA: autocorrelation with moving average, CCC: contemporaneous cross-correlation, HET: heteroskedasticity.

should be considered as a nitrate vulnerable zone when considering mitigation measures (Arauzo *et al.*, 2011; Arauzo and Valladolid, 2013). Besides, the observed negative trend with respect to the concentration of nitrates in 'Decreasing NO_3' is correlated with the implementation of the decoupling process (*% Coupling*), as well as with lower irrigated subsidies (*$Subsidies_{irrigated}$*). These results would be in line with the more extensive man-

Table 6. Panel data regressions for suspended solids (SS) for all subbasins (Total), by type of agriculture (Olive, Coast, Semi-intensive and Mountain) and by existing time trend (Decreasing, Increasing, Minimum and No Trend). The regression models include significant variables ($p<0.05$) only.

Dependent variable SS	Subbasin classification								
Explanatory variables	Total	Olive	Coast	Semi-intensive	Mountain	Decreasing	Increasing	Minimum	No trend
Export$_{SS}$	0.23***	0.17***	1.57***	0.34***		0.95***		0.29***	0.19***
Slope			-1.18**	-0.11***					
Population density	0.19***	0.28***		0.21***	0.13*	0.26*			0.20***
Biomass$_{rainfed}$	0.29***	0.45***							0.32***
Biomass$_{irrigated}$		0.10***							
Shannon								0.19***	
N_{cons}	0.08***								
% Drip		0.19***		-0.20***					
VZ ratio							0.16*		
L.Price index	0.09***	0.11***		0.30**					0.10***
Subsidies$_{rainfed}$						0.20*			
Subsidies$_{irrigated}$	-0.24***	-0.36***				-0.60***		-0.45***	-0.16***
% Coupling	0.23***	0.17***	1.57***	0.34***		0.95***		0.29***	0.19***
Agenda 2000 reform	-0.29***		4.77**	-0.46***	-0.51***			-0.44***	-0.26***
2003 CAP reform			-1.18**	-0.11***					
Intercept	0.19***	0.28***		0.21***	0.13*	0.26*			0.20***
No. Subbasins	89	41	3	32	13	5	3	14	71
No. Observations	803	330	31	291	133	39	28	134	638
R^2	0.60	0.52	0.86	0.62	0.49	0.86	0.89	0.74	0.58
Model	PCSE with RE HET CCC	PCSE with RE HET CCC	Pooled OLS HET	PCSE with RE HET	PCSE with RE HET AR(1)	PCSE with FE HET AR(1	Panel FE	PCSE with RE HET	PCSE with RE HET CCC

p<0.05, ** p<0.01, *** p<0.001. OLS: ordinary least squares, PCSE: panel corrected standard errors, FE: fixed effects, RE: random effects, AR(1): AR1-type autocorrelation, CCC: contemporaneous cross-correlation, HET: heteroskedasticity.

agement practices applied under decoupled income support (Piorr *et al.*, 2009; Cortignani and Severini, 2012). Consequently, in the light of our results, the new 2014 CAP reform should mitigate diffuse pollution thanks to total decoupling from production.

For the total basin (‘Total SS’), we find that erosion rates increase with export of sediments from upstream (*Export*$_{SS}$), biomass intensification (*Biomass*$_{rainfed}$ and *Biomass*$_{irrigated}$), agricultural subsidies in irrigated areas (*Subsidies*$_{irrigated}$), modernized irrigated systems (*% Drip*) and after the *2003 CAP reform*. *Export*$_{SS}$ is related to the negative sign of terrain slope (*Slope*) (as in ‘Semi-intensive SS’, ‘Mountain SS’, ‘Minimum SS’ and ‘No Trend SS’), since sediments are dragged from upstream and accumulated downstream (Gómez, 2008) (see also Table C1 in Supplementary Information C). More efficient irrigation systems are associated with higher concentrations of suspended solids, probably consistent with the expansion of new irrigated areas on steeper slopes in the basin (Gómez-Limón and Riesgo, 2012). In contrast, lower concentrations of suspended solids are found after the *Agenda 2000 reform* (as well as in 'Semi-intensive SS', 'Mountain SS', 'Minimum SS' and 'No Trend SS') perhaps because of the positive soil conservation effects of applied AEM.

Our study also highlights that a greater variety of crops (positive sign of *Shannon index* in *'Olive'* and 'No Trend SS') might be related to more frequent tillage practices or herbicide control of weeds. It is worth highlighting that soil management practices have been proven to be unsustainable in the Spanish olive sector worsened by frequent tillage, and the dependence on external sources of farm income (Gomez *et al.*, 2008; Junta de Andalucía, 2008; Xiloyannis *et al.*, 2008). As farmers do not notice the economic costs of inappropriate soil management much, they do not feel obliged to adopt soil conservation practices (Ibáñez *et al.*, 2014). Payments to farmers found in breach of not fulfilling the cross-compliance requirements under the new CAP (2015-2021) would be reduced immediately.

The decreasing trends of suspended solids ('Decreasing SS') in subbasins are explained by improving water quality conditions upstream ($Export_{SS}$), lower $Biomass_{rainfed}$, larger $Subsidies_{rainfed}$ and with the implementation of the decoupling process (*% Coupling*). As with the nitrate models, the decoupling process seems to be a useful tool for reducing erosion risks. This negative trend can also be explained by the reduction of the total agricultural area in this group of subbasins. By contrast, the 'Increasing SS' trend is supported by the *2003 CAP reform* and a higher price index from the previous season (*L.crop price index*). Higher crop prices may result in a greater production intensity (Kirchner and Schmid, 2013; Renwick *et al.*, 2013), which may lead farmers to make production decisions without evaluating the subsequent environmental consequences (Boardman *et al.*, 2003). Both fertiliser and fuel prices increased after the *2003 CAP reform* which incurred additional costs for farmers which probably resulted in the reduction of sustainable soil conservation practices, e.g. cover crops, contour tilling.

4. Conclusions

This study analyses diffuse pollution in the GRB resulting mainly from land use cover and agricultural practices, focusing on the effects of the CAP reforms from 1999 to 2009. The study identified significant correlations between all the examined independent and non-independent variables before performing the panel data regression. This enabled us to understand the existing relationships that later proved to be consistent with the results of the panel data regressions. The observed relationships of nitrates and suspended solids with the natural environment, agricultural sector characteristics, urban areas and economic policy factors should not be extrapolated because other basins will have different features. However, a better understanding of these correlations is useful for improving the coordination of policies related to water and land management.

It is not easy to discern the effects of point and diffuse sources on the water quality status. In general, exports of both nitrates and suspended solids from upland subbasins and the intensification of agricultural systems increase the concentration of both water quality indicators. In regions where agricultural abandonment and/or deintensification have taken place (i.e. 'Mountain NO_3' and 'Decreasing SS'), the water quality conditions have improved. The decoupling of agricultural subsidies through the CAP reform and the reduction of subsidies for irrigated land is also related to the improvement of both water quality indicators. Therefore, in the light of our results, the new 2014 CAP reform will perhaps bring about environmental benefits in terms of reduced diffuse pollution and erosion risks thanks to the decoupled support scheme. However, it is worth noting that there was a missed opportunity to create political synergies between the WFD and the CAP as the CAP 2014 GAEC did not establish measures to control irrigation, i.e. water abstraction permits, water meters and reporting on water use (European Court of Auditors, 2014).

Although some improvements in the concentration of suspended solids were observed in the basin, concentrations were found to increase in more productive areas with better water efficiency and larger subsidies for irrigated land after the 2006/07 agricultural season. The impact of intensification is particularly significant for erosion rates in 'Olive' and under irrigated conditions. Erosion rates were found to be larger in intensified agricultural regions because of poor soil management practices. There is a mismatch between the regulations concerning the WFD and soil protection, since the WFD does not provide any guidance on achieving good ecological status, specifically for sediment standards (Rickson, 2014) with many watercourses in Europe failing to meet the standard of 'good ecological status'.

Potential weaknesses and limitations due to the assumptions made in this study include the calculation of irrigation modernization and nitrogen consumption, since they underestimate variability across subbasins. Secondly, information regarding the level of participation in AEMs during Agenda 2000 and GAEC during cross-compliance implementation, would help to characterize this in quantitative terms. Improvements could be achieved based on those calculations. While our modelling framework included some of the most important income support measures and a key price index, post-2007 price volatility in international and European agricultural markets may interfere with our causation hypotheses. An intra-annual assessment would determine whether short-term commodity prices affect the physicochemical status of surface water bodies. Finally, low R^2 values suggest, particularly in the '*Mountain*' nitrates model, that additional explanatory variables, e.g. livestock load, are required to explain a larger proportion of the variance of the water quality parameters. Despite the technical and political difficulties that appeared during the negotiation of the post-2014 CAP, conditionality measures, greening components and rural development programmes in the 2014 CAP reform may well offer great opportunities for improving the water quality conditions encountered in the GRB.

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