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Effect of agri-environment schemes (2007–2014) on groundwater quality; spatial analysis in Bavaria, Germany

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ABSTRACT

Degradation of groundwater quality and contamination of drinking resources is one of the most widespread and harmful impacts of over-fertilisation in agriculture. As part of the Common Agricultural Policy (CAP), agri-environment schemes (AES) have as main objective the protection and management of the farm environment, including groundwater quality. In this article a spatial econometric model is applied to evaluate the impact of AES on groundwater nitrate concentrations. Bavaria, a federal state of Germany, is used as a case study, due to the findings of high nitrate concentrations in the groundwater. A significantly negative effect is found between AES expenditures focusing on grassland management and nitrogen concentrations, while AES focusing on crop management, organic farming and preservation of cultural landscape did not show a significant effect. The assessment of other factors such as cereals and forage showed a statistically positive effect on nitrate concentrations. However, loam soil texture, rainfall, and residential area were found to negatively affect nitrate concentrations.

1. Introduction

Agriculture is the dominant instigator of many environmental threats including climate change, biodiversity loss, soil erosion, and water pollution (Foley et al., 2011). Despite the global effort to balance food production with environmental protection, agriculture remains the largest global factor associated with health problems and degradation of resources such as water (Davey et al., 2020). Agriculture is accountable for 50–80% of the total nitrogen load observed in Europe's freshwater (including ground and surface water) (European Commission, 2007) and for around 55% of the nitrogen ending up in the seas (Bouraoui and Grizzetti, 2014).

Organic or mineral nitrogen fertilisers are vital for crop production but excessive use in agriculture can cause water pollution, but also the pollution of air and soils. Degradation of groundwater quality and contamination of drinking resources is one of the most widespread and harmful impacts of over-fertilisation in agriculture (Lord et al., 2002; Schröder et al., 2004). In addition to fertiliser volume, agricultural practices such as manure management, crop cultivation (Lord et al., 2002; Rankinen et al., 2007), soil texture (De Ruijter et al., 2007), and water balance (Boumans et al., 2001; Elmi et al., 2002; Salo and Turtola, 2006) were found to influence the leaching of agricultural nitrogen as nitrate (Wick et al., 2012).

In Germany for example, a recent assessment of the chemical status of groundwater indicates that 34.8% of all groundwater bodies have a poor chemical status due to diffuse pollution with nitrate (27.1% of groundwater bodies exceed the quality standard) and pesticides (2.8% of groundwater bodies exceed the quality standard) from agriculture (Arle et al., 2017). Increased nitrate levels can be harmful to human health. An upper nitrate threshold of 50 mg per litre (mg/l) has therefore been set for groundwater and drinking water across the EU while concentrations lower than 25 mg/l are considered harmless. In 2018, the nitrate concentrations threshold (50 mg/l) in groundwater in Bavaria (the largest federal state in Germany) was exceeded in approximately 7% of the total monitoring stations (Lfu, 2018).

Regarding water quality, the EU Water Framework Directive (2000/60/EC) has been the most important EU regulation aiming at achieving a "good ecological status" for all water bodies in Europe. It is supported by the EU Nitrate Directive (91/676/EEC), which aims to prevent pollution of surface waters and groundwater from agricultural sources and to improve the quality of water, through the implementation of mandatory regulations.

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Both directives are interlinked with the EU's Common Agricultural Policy (CAP). Since its implementation in 1962, under the Treaty of Rome, CAP has undergone multiple reforms and has gradually integrated instruments to support the environment (Pe'er et al., 2019). The CAP amendment in 1987 (EU Regulation 1760/87) covered up to 50% of the costs on environmentally sensitive areas and in 1992 agri-environment schemes (AES) became mandatory for all EU Member States (EU Regulation 2078/92) (Batáry et al., 2015). Following the "subsidiarity" principle, each Member State was free to design its own schemes. Therefore a range of initiatives were developed in the EU Member States during the 1990s, including measures like the conservation of grasslands, the reduction of the use of fertilisers and/or the restoration of wetlands (Buller et al., 2000; Latacz-Lohmann and Hodge, 2003).

In the late 1990s, approximately 20% of the EU agricultural land was under some type of AES (Herzog, 2005), while in 2005 the farmland under AES increased up to 25% (Kleijn et al., 2006). Following the Agenda 2000 reform, non-productive activities were introduced to CAP, such as cultural services or landscape maintenance. In 2010 the CAP integrated environmental concerns in its regulations, in order to tackle food security, environment and climate change in a balanced territorial development approach (Navarro and López-Bao, 2018).

Over the period 2007–2013, the EU financed AES with nearly 20 billion € (European Commission, 2014). Since 1993 the EU budget allocated to AES has increased exponentially, which in 2010 reached 3.026 million € and in 2013 approximately 5.035 million € (Pavlis et al., 2016). The measures included under the 2007–2013 EU rural development program were grouped into 12 categories (e.g. organic farming, management of landscape, pastures and high nature value farmlands, integrated production, and other extensification of farming systems) (Pavlis et al., 2016).

Water pollution reduction, soil protection, access to countryside, protection of cultural landscapes and heritage as well as biodiversity protection have been clear objectives for many AES. Supporting organic farming has been common in almost all countries, assuming that organic farming has a positive impact on the environment (Tuck et al., 2014; Auerswald et al., 2003; Sanders and Heβ, 2019).

AES are voluntary contracts under which farmers receive payments for adopting specific environmental management practises or for meeting a certain environmental goal. A large part of literature has analysed the impacts of AES on economic outcomes. For example, a number of studies have analysed the effect of AES on farm income (Arata and Sckokai, 2016), farm productivity (Lansink et al., 2002; Mary, 2013; Mennig and Sauer, 2020; Salhofer and Streicher, 2005), and farm employment (Arata and Sckokai, 2016; Unay-Gailhard and Bojnec, 2019). Although the beneficial effects of AES on the environment have been questioned due to poor targeting and a lack of site-specific payments. There are only few studies (Auerswald et al., 2018) that have analysed the environmental impacts of AES mostly due to the lack of availability of environmental monitoring data.

The early 2000s was the period when the first well-designed studies were published on the ecological effects of AES (Batáry et al., 2015). For example, Kleijn and Sutherland (2003) conducted literature review and examined the effect of AES with biodiversity targets on biodiversity. Since that review, a number of studies assessed the impact of AES on biodiversity (Breeuwer et al., 2009; Feehan et al., 2005; Fuentes-Montemayor et al., 2011; Kaligarič et al., 2019; Kleijn et al., 2006; Carvell et al., 2007; Kleijn and Sutherland, 2003), on soil (Marconi et al., 2015; Marriott et al., 2005), environmental quality (Bartolini et al., 2020; Villanueva et al., 2017; Zabel, 2019) and water quality (Jones et al., 2017; Poole et al., 2013; Reinhard and Linderhof, 2015; Slabe-Erker et al., 2017). The findings from these studies have been quite divergent and at times contradictory in relation to the effect that AES have on the environment.

Reducing emissions of pollutants to water is a major policy challenge, although the literature focusing on the assessment of this policy on water quality is limited. For example, Poole et al. (2013) assessed the ecological effectiveness of AES in an English lowland river basin, using aquatic macroinvertebrates as indicators of river health. Results showed that high proportions of AES river options within the same distance were correlated with higher proportions of sediment-sensitive macroinvertebrates (Poole et al., 2013). Jones et al. (2017) evaluated the impact of Welsh AES on water quality and freshwater ecosystem condition through a combined monitoring and modelling framework. They found that the Wales AES can reduce diffuse pollution from agriculture, but these benefits were not evenly distributed across the landscape. Both studies did not use long-term water chemistry data to evaluate water quality but they rather estimated biological response to pollutants. Früh-Müller et al. (2019), analysed the spatial distribution of AES and their correlation with selected environmental pressure and land-use indicators in Germany. Their main findings showed that the uptake of AES tended to be low in regions with high ammonium deposition or high shares of utilised agricultural area on organic soils. Finally, they concluded that AES are not very attractive to land managers in productive agricultural regions. An integrated landscape-scale design of programs could help to account for regional differences in environmental and economic conditions to increase the success of agri-environmental policies.

Slabe-Erker et al. (2017) assessed the effect of AES on groundwater nitrate and pesticide levels, using a spatial econometric model for the period 2007–2013. Based on their findings, AES did not have any statistically significant impact on nitrate levels in groundwater, while they were associated with reduced pesticide levels. Reinhard and Linderhof (2015) evaluated whether the development of nitrogen surplus in a number of EU member states was significantly linked to the spending on AES. Using a spatial econometric model on a panel dataset they found a significant decrease of N surplus with increasing agri-environmental expenditures.

The aim of this analysis is to assess whether AES applied in Bavaria have improved the groundwater quality in terms of reducing nitrate concentrations for the period 2007–2014. In addition to the expenditure of the AES payments, the effect of other factors such as land cover, weather and soil characteristics were also examined. In order to achieve that, a spatial panel data econometric model was developed.

1.1. Agri-environment schemes in Bavaria (KULAP)

The large diversity of Bavarian agricultural systems and landscapes has been incorporated into AES of the Bavarian Rural Development Programme 2007–2013. The Bavarian Rural Development Programme includes two AES programmes, the Nature Conservation Programme, which translates to Vertragsnaturschutzprogramm, and the Bavarian Cultural Landscape Programme, which translates to Bayerisches Kulturlandschaftsprogramm (KULAP). The present study does not consider the first programme, which is limited to farms in nature conservation areas but rather considers KULAP as the core funding instrument of Bavarian agri-environmental policy.

The aims of KULAP in the past have been the reduction of fertiliser input and chemical pest control to improve the quality of water and soil bodies, landscape conservation, the decrease in soil erosion and maintenance of biodiversity and habitat protection (Mayer et al., 2008). KULAP subsidises a number of eligible measures focusing on the preservation of biodiversity and landscape structure, such as organic farming, crop rotation, mulch sowing, extensive grassland with limited fertiliser use. Most of them are correlated with soil nutrient retention and reduced or no application of nitrogen mineral fertilisers and thus should alleviate the impact of nitrogen mineral fertilisers on the environment.

The KULAP of the funding period 2007–2013 essentially pursued three main goals; the promotion of environmentally friendly land management; the alleviation of the agricultural impact on the environment, such as reducing the input of substances into the soil, water, and air; and the promotion of sustainable, area-wide land management to maintain a typical regional cultural landscape (ART, 2016).

KULAP applies five-year contracts, which farmers stipulate voluntarily. Farmers who participate in these schemes receive payments which are estimated as a means of compensation of any income forgone or additional costs associated with the contracted farming practices. Organic farming was offered as a whole farm measure with a premium of 200 €/ha. Measures related to all grassland areas of a farm offered payments ranging between 50 and 600 €/ha depending on the extent of the management restrictions. Measures related to arable farming (e.g. extensive or diverse crop rotation, winter greening, mulch sowing, conversion of arable land into grassland, buffer strips for water and soil protection) ranged between 21 and 920 €/ha (Art, 2016).

KULAP is widely adopted in Bavaria where 55% of the agricultural land is financially linked to some KULAP schemes (Mayer et al., 2008). For the period 2007–2013 around 950 million € in public funds have been available for KULAP. Up to 1.23 million hectares of arable land and grassland (including areas from the previous funding period) have been financially supported during the programme period 2007–2013, out of which 38% was grassland, 30% arable land, and 25% organic farming (ART, 2016).

2. Material and methods

2.1. Study area

Bavaria is located in the southeast part of Germany, accounting for around 20% of the German territory (70,550 km^2) (Fig. 1). It is the largest Federal State in Germany as well as a major contributor to the German agricultural sector. In 2015, Bavaria accounted for 20% of the German gross value added in the sectors agriculture, forestry, and fishery. Approximately one-third of the German farms are located in Bavaria with an average size of 29.5 ha in 2015 which is smaller than the average national farm size (Bstmelf, 2016).

According to Wiesmeier et al. (2012), up to 35% of the total land area in Bavaria is forest area, another 35% is cropland and 16% is grassland. Land used for crop production is mainly clustered in the middle and northwest parts of the federal state (Fig. 2). Grassland and forest areas span from north to east and are aggregated in the south, with small clusters located in the northwest borders and southern Alpine and pre-Alpine regions. Varying natural conditions contribute to significant diversity in agricultural production. For example, the Alpine region favours dairy farming, while the loessial landscape along the river Danube experiences intensive arable use and hog production (Bayerisches Staatsministerium für Landwirtschaft und Forsten, 2002). The fertile soils of Southern Bavarian except for the Alpine region as well as some north-western parts of Bavaria, allow intensive crop farming, while Eastern Bavaria is more suitable for farms specialised in pig fattening or breeding and poultry keeping.

2.2. Data sources and descriptive statistics

A balanced spatial panel data aggregated at municipality-level was constructed for the years 2007–2014. Nitrate groundwater data were obtained from the Bavarian state office for the environment (Lfu, 2018). Under the guidelines of the European Water Framework Directive (WFD), the LFU assesses the groundwater quality in terms of pollutants such as nitrate, pesticides, and their breakdown products as well as heavy metals. Pesticide concentrations in groundwater in Bavaria are far below the limit, hence the current study is focusing only on nitrate concentrations.

The water samples at the chosen monitoring sites were collected

Fig. 1. Map displaying the location of the study area Bavaria within Germany and Europe.

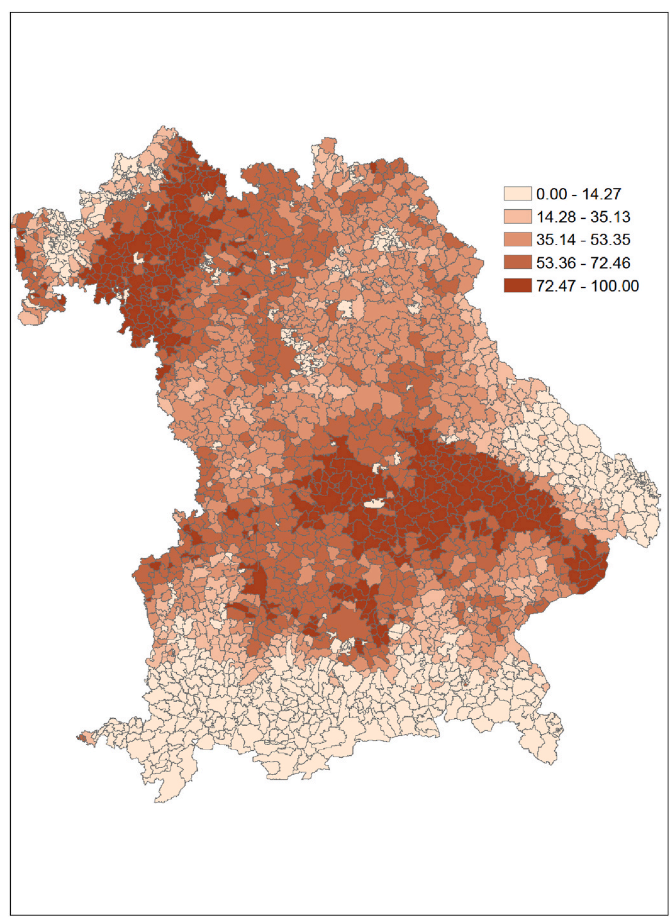

Fig. 2. Cropland and grassland (100 - cropland) share (in %) of Bavarian agricultural area on the level of municipalities (delineated by lines).

from the groundwater aquifer and close to the surface aquifer. These sites are representative of the land use distribution (human settlements, forest, grassland, arable land, and special crops). The monitoring network density in Germany is around 3.5 monitoring sites per 1000 km^2 (Arle et al., 2017); in Bavaria, the monitoring network density is twice as high with around 500 measuring points across 59 groundwater bodies.

The annual average number of nitrate sampling points for the observed period was 389. Geostatistical interpolation was used to create a map and to determine the mean nitrate concentration for all 2052 municipalities and 183 territories including forests and lakes. The reason for aggregating nitrate concentrations at municipality level was that land use data are only available on municipality level or higher.

KULAP data on the Bavarian AES were obtained from the Bavarian State Research Centre for Agriculture (LfL). AES were classified into four groups; whole-farm measures applicable for organic farming (AES I), grassland measures such as prohibiting mineral fertilisers and regulating manure application (AES II), arable land measures including crop management such as extensive and diverse crop rotation (AES III), and measures for special forms of cultivation to preserve cultural landscapes (AES IV). All AES payments were expressed per hectare of the total municipality area.

In an attempt to capture the amount of mineral fertilisers applied, the total farm-level costs of fertilisers were used as a proxy and averaged per municipality. Farm information on fertiliser costs were provided by the LfL as a farm accountancy sample. Fertiliser costs were deflated with the corresponding price index using 2015 as the base year to obtain implicit quantities. These values were aggregated at the municipality level and were expressed per utilised agricultural area (UAA). Detailed agricultural information on crop cultivation (in hectares) and the number of animals per farm was provided by the LfL. Data on crops were available on a farm level and an annual basis. Approximately 70 crops were aggregated into four groups: (i) oilseed and protein crops, (ii) cereal and maize, (iii) row crops and vegetables, and (iv) forage crops. All groups were aggregated at the municipality level and expressed as a share of the total UAA. The animal numbers were also available on a farm level. They were converted into livestock units, aggregated at the municipality level and expressed per total UAA.

In order to control for potential non-agricultural sources of nitrogen, the forest land and residential area were included in the analysis. Both variables were expressed as a share of the total municipality area. Physical boundary conditions such as soil characteristics and rainfall were considered as significant factors that affect groundwater pollution. Soil texture was taken from the topsoil physical properties for Europe developed by Ballabio et al. (2016). Based on this study, silt is the most common soil texture (71%) in Bavaria followed by sandy loam and loam (Appendix, Figure i). Soil quality was captured by *WE2* (called Ackerzahl) (Hangen and Förster, 2013; Rossiter et al., 2018) which closely relates to yield (or plant available water) capacity. This variable was obtained by the Bavarian tax office (Bayerisches Landesamt für Steuern, 2019). Data on annual rainfall (in millimetres per year) between 2007 and 2014 were provided by the German Meteorological Service (DWD), LfL and LfU. The rainfall observations stem from 106 weather stations throughout Bavaria, which were again interpolated and then aggregated to mean values at the municipality level.

All interpolations employed the empirical Bayesian kriging method (see section 2.3). The interpolation analysis and all the mappings were performed in ArcGIS software. The detailed descriptive statistics of all variables in the analysis are presented in Table 1. Maps of all variables (OIL, FG, CL, VEG, LU, FR, RE, WE2, SOIL, RAIN) are included in Appendix.

2.3. Econometric model

Spatial econometrics is a set of methods suitable for dealing with spatial autocorrelation as well as spatial heterogeneity when performing regression analysis for both cross-sectional and panel data (Anselin, 2003). In general, spatial econometric models are regression models that consider the spatial factor when identifying relationships between dependent and independent variables (Grekousis, 2020). Therefore, due to spatial autocorrelation and spatial heterogeneity the ordinary least squares (OLS) is inefficient because the independence of residuals is violated, errors are not normally distributed and a single model cannot describe the existing relationships (Murray, 2010).

Lagrange Multiplier (LM) tests have been widely used to test for random effects and serial or cross-sectional correlation in panel data models (Breusch and Pagan, 1980). In the present study, the joint Lagrange multiplier (LM) test (Baltagi et al., 2003) was used to test for random effects and spatial autocorrelation. The joint LM test for the hypothesis of no random effects and no spatial autocorrelation (H0) was found significant, hence the null hypothesis was rejected (Croissant and Millo, 2019). The rejection of the null hypothesis for no spatial correlation in the residuals indicates that OLS is inefficient for this study.

The spatial error model is one of the main specifications in literature (Slabe-Erker, 2017; Croissant and Millo, 2019), which is appropriate when the "innovation" relative to the observation is expected to have an effect on the outcomes of neighbouring ones. In the case of the current analysis, a drastic change to a given region will influence the relevant dependent variable in that region and propagate – with distance-decaying intensity – toward nearby ones (Croissant and Millo, 2019). Another interpretation is that there is some explanatory variable that was not included in the model and is causing the residual to be high.

Table 1
Descriptive statistics of nitrate concentration in groundwater and potential influencing factors; all values are averages on the municipality level.

Variables	Definition	Units	Mean	Std. Dev.	Min	Max
NO_3	Nitrate concentration	mg/l	20.53	10.03	3.17	60.97
AES I	AES on organic farming	€/ha	11.80	17.9	0.00	198.70
AES II	AES focusing on grassland	€/ha	14.20	23.44	0.00	532.20
AES III	AES focusing on crop production	€/ha	9.92	13.06	0.00	304.90
AES IV	AES focusing on preserving cultural landscape	€/ha	0.64	3.49	0.00	150.43
FC	Fertiliser cost per UAA	€/ha	151.52	71.46	0.00	487.58
OIL	Oilseed and protein crops as a share of UAA	%	5.37	5.20	0.00	72.55
FG	Forage as a share of UAA	%	14.10	10.12	0.00	60.87
CL	Cereal crops as a share of UAA	%	31.56	21.00	0.00	91.00
VEG	Vegetables as a share of UAA	%	3.32	7.42	0.00	100.00
LU	Livestock units per UAA	LU/ha	1.37	2.20	0.00	88.06
FR	Forest area as a share of municipality territory	%	36.15	23.30	0.00	99.00
RE	Residential area as a share of municipality territory	%	3.01	3.50	0.00	50.90
WE2	Soil quality	Interval-scaled	39.30	10.98	0.00	69.30
LOAM	Loam	Dummy	0.02	0.15	0.00	1.00
SILT	Silt	Dummy	0.71	0.44	0.00	1.00
RAIN	Rainfall	mm/yr	861.23	174.87	559.24	1753.75

If there is a high residual in one area that would ripple through the whole area.

A spatial error model was applied to estimate the effect of AES on nitrate concentrations in groundwater. In a random effects specification, the unobserved individual effects are assumed uncorrelated with the other explanatory variables in the model and can therefore be safely treated as components of the error term (Croissant and Millo, 2019).

The formulation of the spatial model built for the analysis was derived from the work of Anselin (1988):

$$y_{it} = X_{it}\beta + a_i + u_{it} \tag{1}$$

$$u_{it} = \lambda \sum_{i \neq j} W_{ij} u_{jt} + \varepsilon_{it} \quad \text{with } u_{jt} \sim \text{IID}\aleph\left(0,\ \sigma^2\right), \tag{2}$$

where y_{it} is the value of the dependent variable y for municipality i in time t; X_{it} is the value of the explanatory variable under focus for municipality i in time t; β is the model coefficient for X_{it}; α_i is the constant, W_{ij} is the spatial weighting matrix Wn between municipalities i and j of dimension (nxn) in which neighbourhood relationships between sample individuals are defined. For example, the neighbourhood is expressed as zero-one and takes the value equal to 1 for a pair of regions that have a common border, while 0 otherwise. The weights are row standardized and are equal for all neighbours, regardless of the length of the border. That is, in a region surrounded by n neighbours, each weight will be 1/n (Kopczewska, 2020). The term $\lambda \sum_{i \neq j} W_{ij} u_{jt}$ captures spatial interaction through the spatial autoregressive specification of the error term, and λ is the spatial autoregressive parameter (Bouayad-Agha et al., 2018).

The estimated equation to examine the effect of AES and other factors on groundwater nitrate concentrations was the following:

$$NO_{3it} = AESI_{it-1} + AESII_{it-1} + AESIII_{it-1} + AESIV_{it-1} + NO_{3it-1} + FC_{it} + OIL_{it} + FG_{it} + CL_{it} + VEG_{it} + LU_{it} + FR_{it} + RE_{it} + WE2_{it} + LOAM_{it} + SILT_{it} + RAIN_{it} \tag{3}$$

The estimation of the spatial error model was performed in R software, using the R package *splm* (Millo and Piras, 2012).

Land cover consisting of different crop types is expected to contribute positively to increasing nitrate concentrations in groundwater (Schröder et al., 2004), due to high fertilisation rates. On the contrary, forests and residential areas are expected to have a negative effect on nitrate concentrations.

The most fertile soils in Bavaria are associated with high fertilisation rates, therefore, soil fertility is expected to show a positive effect on nitrate concentrations in groundwater. Soil texture was deemed important as it influences groundwater recharge and adsorption, microbial transformation, and travelling velocities of agrochemicals. Loam and silt soils are known to retain more water than sandy soils, therefore, they are expected to show a negative relationship with nitrate concentrations.

There is evidence that rainfall plays a significant role in explaining variations in groundwater nitrate concentration (Schweigert et al., 2004). Schweigert et al. (2004) concluded that precipitation in autumn could lead to nitrate leaching, due to full soil reservoirs. Many studies found a positive relationship between nitrate concentrations in groundwater and precipitation through nitrate leaching due to full soil reservoirs (Korsaeth and Eltun, 2000; Rankinen et al., 2007). In contrast, high average precipitation could also favour the uptake of nitrogen by crops, hence, lead to decreasing nitrate concentrations in groundwater (Schweigert et al., 2004; Sieling and Kage, 2006) or cause a dilution of nitrate and thus lower nitrate concentrations (Hofreither and Pardeller, 1996).

There are various kriging methods that can be utilised for generating continuous surface of values (LILLY, 2016). In the present study the empirical Bayesian kriging method was used. The empirical Bayesian kriging is a method where the construction of a semivariogram that appropriately represents a dataset is automated (LILLY, 2016). As opposed to other kriging methods that use a single semivariogram during estimation, the empirical Bayesian kriging utilises more than one semivivariograms when generating a surface (Krivoruchko, 2012).

3. Results

AES patterns indicated that the reception of AES I and AES II were highly concentrated in areas rich in grasslands, while AES III flowed to areas rich in croplands (compare Figs. 2 and 3). AES IV contributed least to AES payments while AES I and II were four to five times higher (Table 1). The AES III and nitrate spatial overlap (Fig. 3) suggest that AES III were concentrated in areas with groundwater high in nitrate concentrations, which were also cropland dominated areas.

3.1. Spatial and temporal nitrate pattern

Nitrate concentrations showed a pronounced spatial pattern (Fig. 4), which was similar to the pattern of cropland (Fig. 2). The mean annual nitrate concentrations observed at the beginning of the implementation period of AES in 2007 ranged between approximately 3-67 mg/l (Fig. 4; left map) and between 3 and 71 mg/l at the end in 2014 (Fig. 4; right map). No significant differences were observed between the mean nitrate concentrations in 2007 and 2014. The highest nitrate concentrations were observed mostly in the north-western part (the loess-dominated landscape in lower Franconia) and in the south-eastern centre (loess-dominated landscape along the river Danube where specialised arable farms are dominant).

The nitrate concentration at state level, when aggregated over all municipalities, showed a descending trend from 2007 to 2013 (Fig. 5). In 2013 concentration spiked upwards again and reached the same level as in the beginning of the study period.

3.2. Results of the spatial error model

In this section, the results of the spatial error model presented in Table 2, are analysed. Table 2 (column 2) illustrates the results of Equation (3), through which the effect of AES, farm, and environmental characteristics on groundwater quality have been estimated.

Surprisingly, $AESI_{t-1}$ focusing on organic farming failed to show any significant effect on groundwater nitrate concentrations. It was expected that $AESI_{t-1}$ would have a negative effect on nitrate concentrations considering that the application of chemical fertilisers is forbidden by farmers under these measures and given that a recent review study showed an effect of organic farming on reducing nitrate contamination (Sanders and Heβ, 2019). An increase in $AESII_{t-1}$ by 1 € per ha would lead to a decrease in next year's nitrate levels by 0.005 mg/l. The significant negative relationship between AES II and nitrate concentrations, while holding all other variables constant, points to the effectiveness of grassland measures to improve groundwater quality. The result for AES III $_{t-1}$ failed to show any significant effect, despite the relatively high payment rates and the extent of their application especially by farms in croplands (Fig. 3). It needs to be noted, though, that the individual AES III schemes aimed mostly at biodiversity protection and soil erosion reduction. AES IV_{t-1} focusing on the preservation of

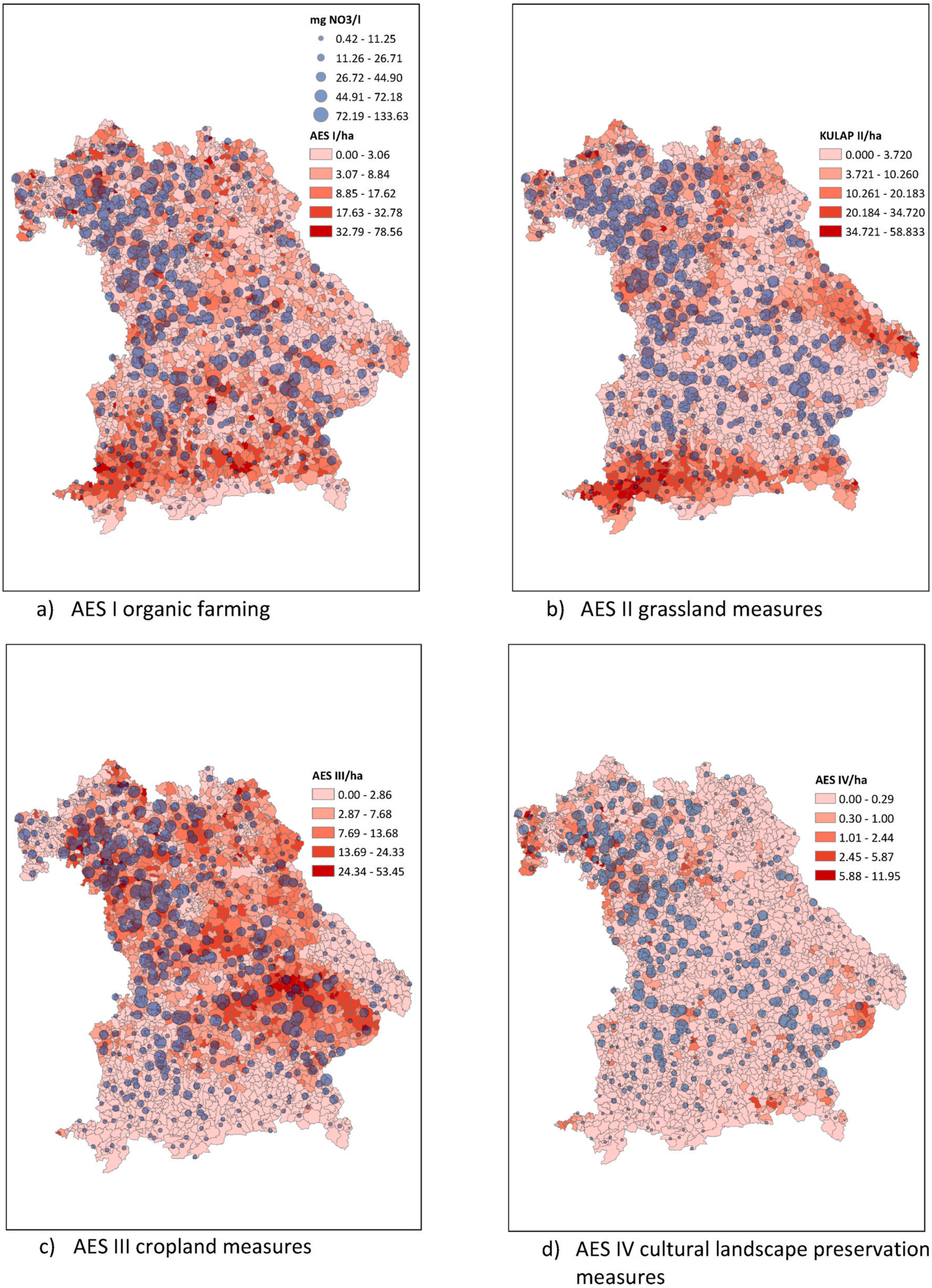

Fig. 3. Spatial overlap between mean AES (2007–2014) per ha and mean nitrate levels in groundwater for the same period.

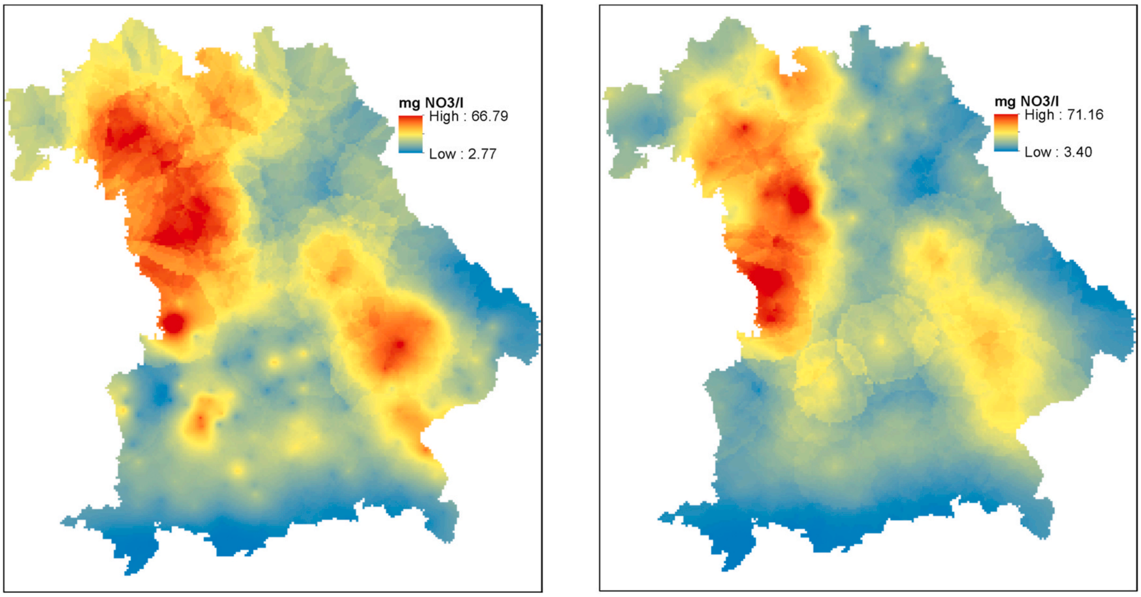

Fig. 4. Mean nitrate levels (mg NO_3/l) in groundwater for 2007 and 2014.

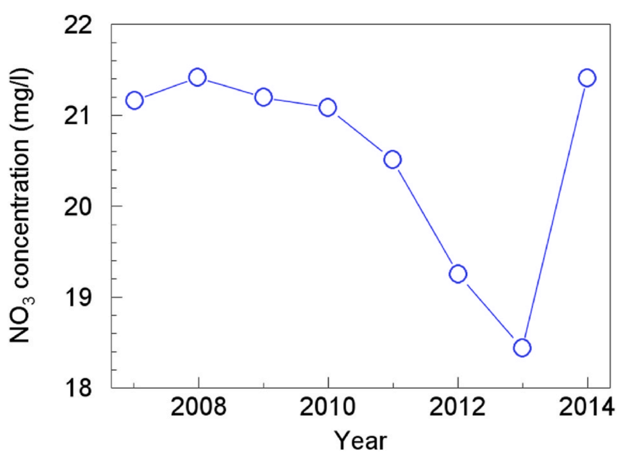

Fig. 5. Nitrate concentration trend between 2007 and 2014.

Table 2
Results of the spatial error model for nitrate concentration in groundwater (for explanation of variables see Table 1).

Variable	Coefficient	P-value
Constant	7.89***	0.000
$AESI_{t-1}$	−0.000	0.642
AESII $_{t-1}$	−0.002*	0.030
AESIII $_{t-1}$	−0.002	0.174
AESIV $_{t-1}$	0.002	0.621
NO_{3t-1}	0.838***	0.000
FC	−0.000	0.303
OIL	−0.010	0.074
FG	0.007**	0.007
CL	0.006**	0.002
VEG	0.006	0.097
LU	0.007	0.390
FR	−0.002	0.102
RE	−0.026***	0.000
WE2	0.004	0.113
SOIL3	−0.449**	0.001
SOIL9	0.047	0.459
RAIN	−0.005***	0.000
n	17,880	
Phi	0.000	0.373
rho	0.809***	0.000
Joint LM Test	19,065***	
R^2	0.84	

cultural landscape did not show any significant relationship with nitrate concentrations. This outcome was expected considering their goal, low payment rates and their limited application in comparison with the other three groups (Fig. 3). The lagged dependent variable NO_{3t-1} showed a significant positive effect, indicating that past nitrate levels affect current levels.

The results on fertiliser costs did not show any significant effect, although a positive relationship between fertiliser costs and groundwater contamination with nitrate was anticipated. Comparing the mean nitrate levels (Fig. 4) with the spatial pattern of croplands (Fig. 2), it is evident the regions high in nitrate loadings overlap with the regions where croplands are situated.

Investigating the relationship between different crop types and nitrate concentration in groundwater, it was found that forage and cereals showed a statistically significant positive effect, while oilseeds and vegetables did not show any significant effect. Livestock units also did not show any significant effect on nitrate concentration, which either indicates that the amounts of organic fertiliser are well dealt with or that the range of livestock density is reasonable. Results for the effect of the residential area on nitrate concentration met the initial expectations. The share of residential area was expected to show a significant negative effect, assuming that residential areas are not contributing to groundwater contamination with nitrate, unlike cropland. Forest coverage failed to show significant effect, despite the assumption that it negatively affects nitrate concentrations in groundwater.

With respect to physical characteristics, the results for the effect of soil and rainfall on nitrate concentration in groundwater were anticipated. High rainfall significantly negatively influences nitrate levels in groundwater because of the dilution effect. In addition, regions with high precipitation are often dominated by grasslands, which additionally reduce the input of nutrients into groundwater. Loamy soils are known to retain more water than sandy soils hence, a negative relation was anticipated. Silt soils did not show any significant effect, though. Soil quality (WE2) was positively related to nitrate levels mostly because these soils usually reflect an excessive application of fertilisers due to intensive crop production.

4. Discussion

Nitrogen is a vital input in agricultural production, while its extensive use can put environmental strain on (ground)water, soil and air. In the present study, the impact of CAP AES on Bavaria's groundwater quality was investigated for the AES period 2007–2014, through a spatial panel model at the municipality level.

In contrast to the findings by Slabe-Erker et al. (2017) who suggested that all AES groups had no statistically significant influence on groundwater pollution with nitrate, the current study shows that AES focusing on grassland measures can be effective in reducing nitrate levels. A significant negative relationship between AES payments and the level of nitrate surplus was corroborated by Reinhard and Linderhof (2015). The non-significant effect of the three AES groups (AES I, AES III, AES IV) on groundwater quality points to limitations in scheme targeting. For example, the schemes' (2007–2014) main focus is on improving biodiversity, soil quality or the preservation of cultural landscape. Furthermore, the non-spatially targeted nature of agri-environment schemes could explain their inadequacy to improve water quality. The results of the current study indicate that AES can be effective in reducing nitrate levels in groundwater, however, no reduction effects were found in the regions with the highest nitrate loadings. Hence, spatially targeted AES might have the potential to improve the environmental performance of farms through improving water quality (Cullen et al., 2018). Früh-Müller et al. (2019), also concluded that to successfully address site-specific conditions, measures should include a regional component, which should be based on the guidance of environmental experts (Feindt et al., 2017).

Identifying the limitations of the schemes allows policymakers to develop AES that better target water and soil pollution. Specifically, the schemes (2007–2014) with high adoption rates and a focus on arable land include only one measure (a scheme promoting the planting of catch crops) that could directly affect nitrate leaching. Large-scale schemes limiting fertiliser use on arable land for example did not exist. Apart from CAP payments, further factors play a significant role in water pollution. Therefore, it is indispensable that AES capture also the physical properties of the areas where farms are located, such as soil texture, rainfall, and soil fertility. A bottom-up approach of AES could contribute to better targeting of environmental pollution, although expert costs and bureaucratic procedures could be a significant drawback. AES have been a considerably important policy tool to reduce negative externalities from agriculture, however, there is still room for further improvements. The attractiveness of measures for special forms of cultivation to preserve cultural landscapes might be increased by higher premiums through the inclusion of an incentive component.

The most recent CAP reform post-2020 proposes a new set of objectives upon which member states will develop their national strategic plans. Granting more flexibility to member states is an important step towards more regionally targeted AES. Enhanced conditionality which sets binding minimum good environmental and agricultural condition standards for all payments has the potential to increase the effectiveness of the payments. As part of an effort to tackle the water contamination issue in Europe, a new standard is set which includes the compulsory use of the new Farm Sustainability Tool for Nutrients. The examination of the last CAP reform measures in the future is required in order to examine their effectiveness. However, data collection of environmental indicators is imperative for the development of these studies.

In addition to the influence of AES on groundwater quality, the influence of environmental characteristics was also investigated. The negative relationship between groundwater contamination with nitrate and rainfall is in accordance with previous studies (Ernstsen et al., 2015; Kawagoshi et al., 2019; Outram et al., 2016; Wang et al., 2016; Zereg et al., 2018). For example, Kawagoshi et al. (2019) showed a significantly negative relationship between nitrate concentrations and heavy rain in combination with a highly permeable subsurface. In line with Wick et al. (2012), a variety of crops were found to have a positive impact on groundwater contamination. This finding suggests that crop farms have the potential to mitigate groundwater contamination if they adopt strategies that circulate farm nutrients, such as crop diversification including legumes in the crop rotations, use cover crops, etc. Measures that promote reduction in the application of mineral fertilisers through precision measuring tools are key to improving groundwater quality. Compulsory soil testing if incorporated in farmer's decision-making process can possibly reduce over fertilisation which also result in farm economic losses. Schemes for farmer training support could educate farmers on up-to-date tools and strategies that are applicable to their region depending on the pedoclimatic conditions of the region.

Some limitations of this approach are worth mentioning. Data before the AES implementation in 2007, would have possibly shown better the actual impact of AES, assuming that no other related policy was in place before 2007. These data, however, were not available and the assumption of no incentives before 2007 does also not hold true given the long history of the nitrate problem. Therefore, the lack of long-term data might not show the big picture of the long-term existence of CAP measures. The fact that AES failed to show any significant effect in the period 2007–2014 does not capture the overall long-term effect that AES might have had since their first implementation. As such, it is possible that in the absence of AES the water contamination might have been worse. Acknowledging the political importance of these schemes, the results of this study do not intend to diminish the overall and long-term effectiveness of AES.

Another limitation of this study is that we had to neglect the travel time between the soil and groundwater, which – among other factors –depends on the depth to groundwater. The depth to groundwater could be controlled for in a future similar research. In order to capture the complex interaction among rain, land use, soil, groundwater depth and the amount of groundwater recharge, the combination of a hydrological model and spatial model is required.

5. Conclusions

The current study provides the first spatially explicit analysis of the effect of agri-environmental payments on groundwater quality in Bavaria, taking into account interlinkages between environmental and physical characteristics and nitrate concentrations in groundwater. The results show a negative relationship between the allocation of grassland measures and nitrate concentrations. Other schemes were not successful in improving groundwater quality. Agri-environment measures targeting croplands, organic farming, and other sensitive areas would probably benefit from a stronger link between scheme requirements and groundwater protection potential and an integrative land-scale design, accounting for regional, soil and weather differences. Improved targeting would also result in less windfall effects.

Acknowledgements

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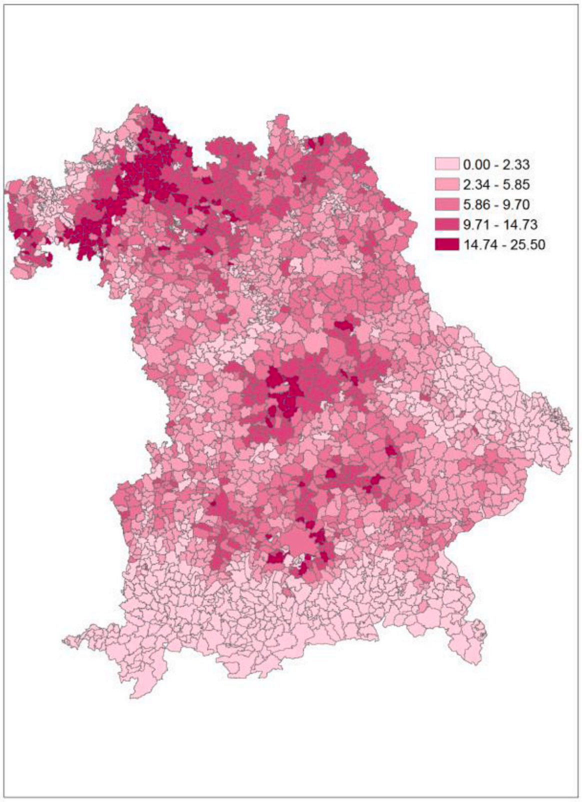
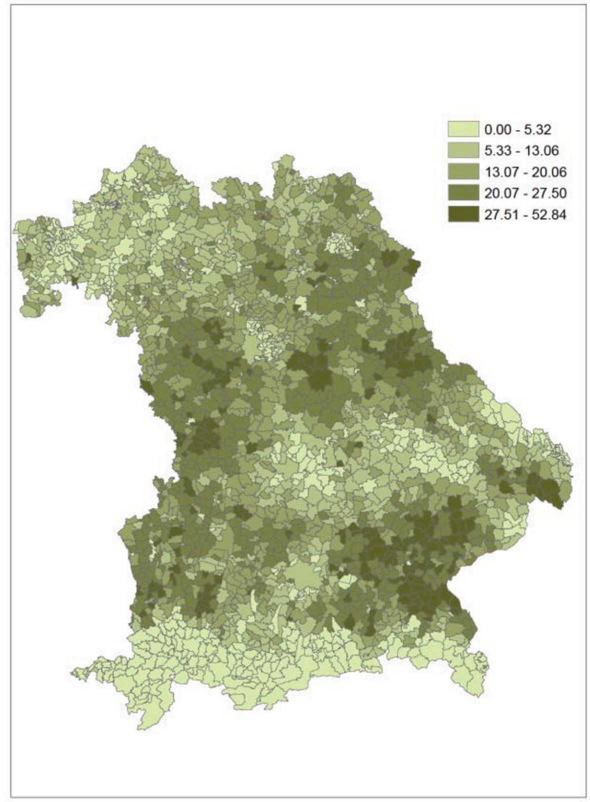
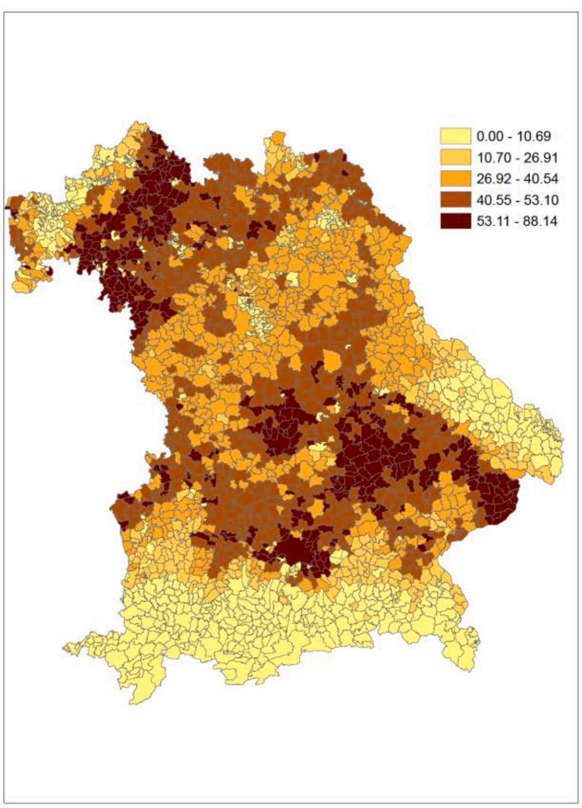
Appendix A



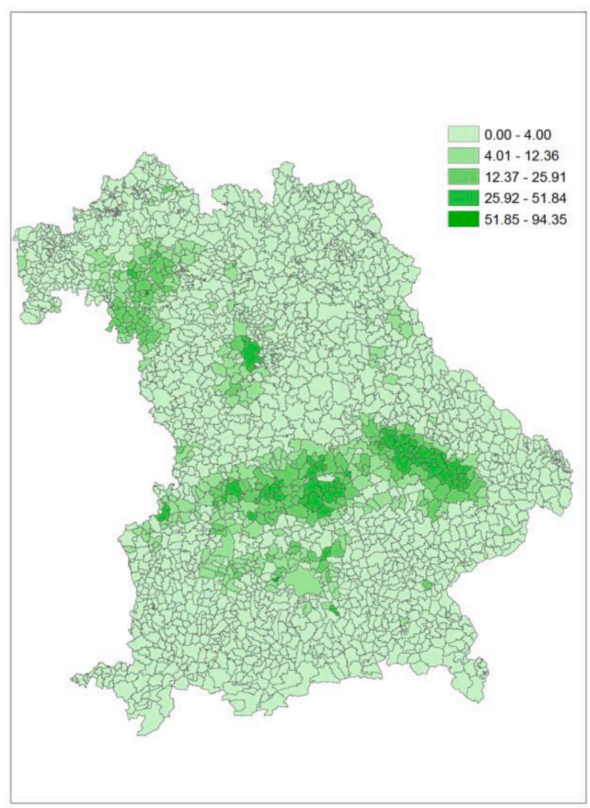
a) Oilseed and protein crops as a share of UAA (%)



b) Forage as a share of UAA (%)



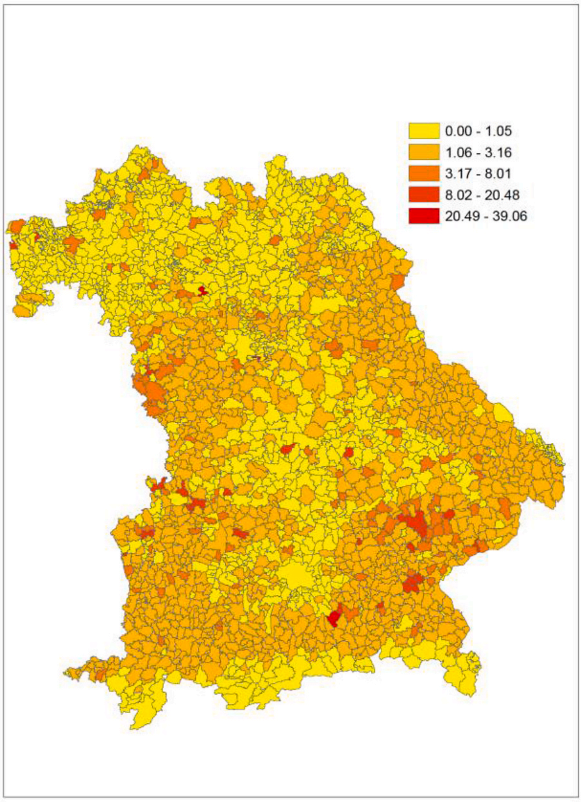
c) Cereal crops as a share of UAA (%)



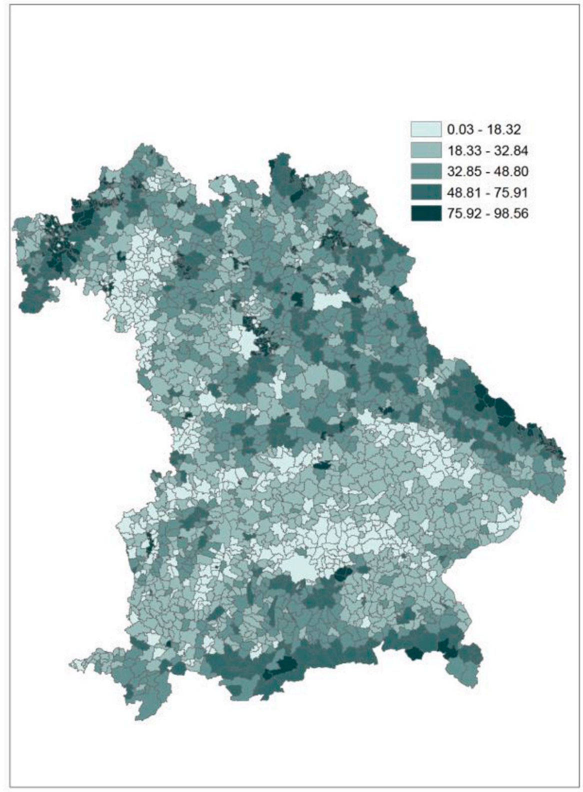
d) Vegetables as a share of UAA (%)

(*continued on next page*)

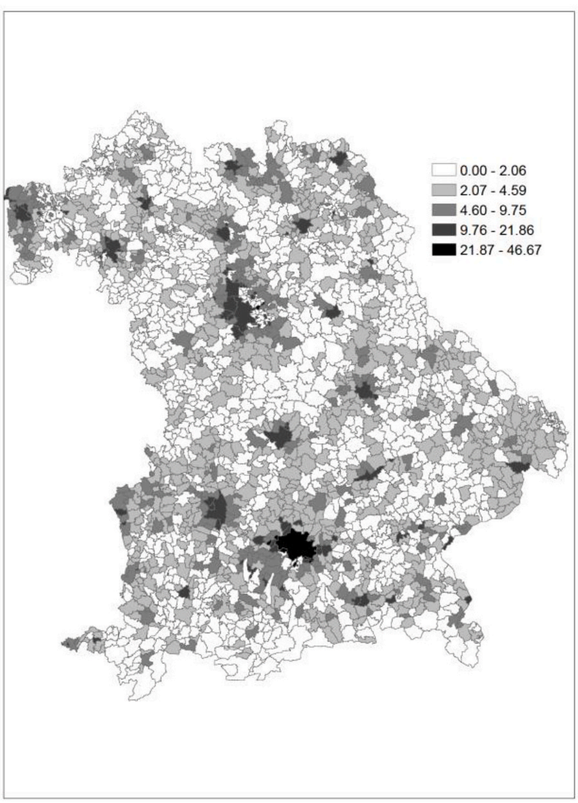
(*continued*)



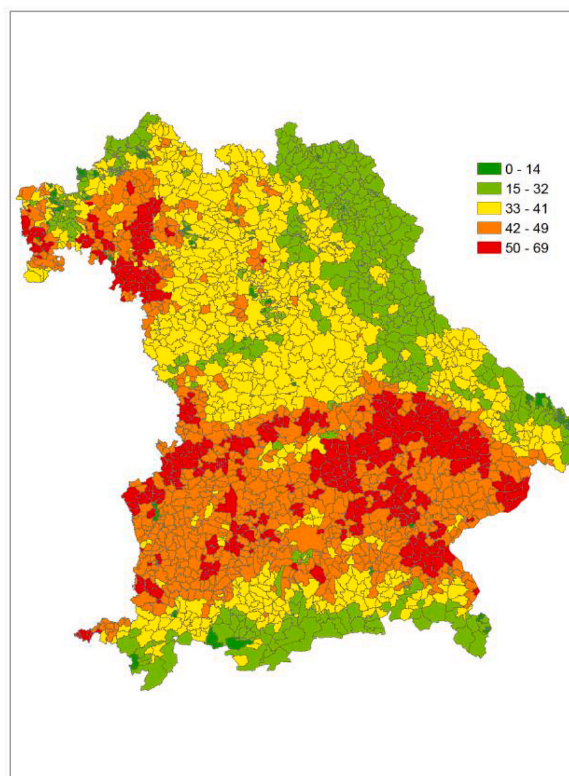
e) LU per UAA



f) Forest as a share of municipality territory (%)



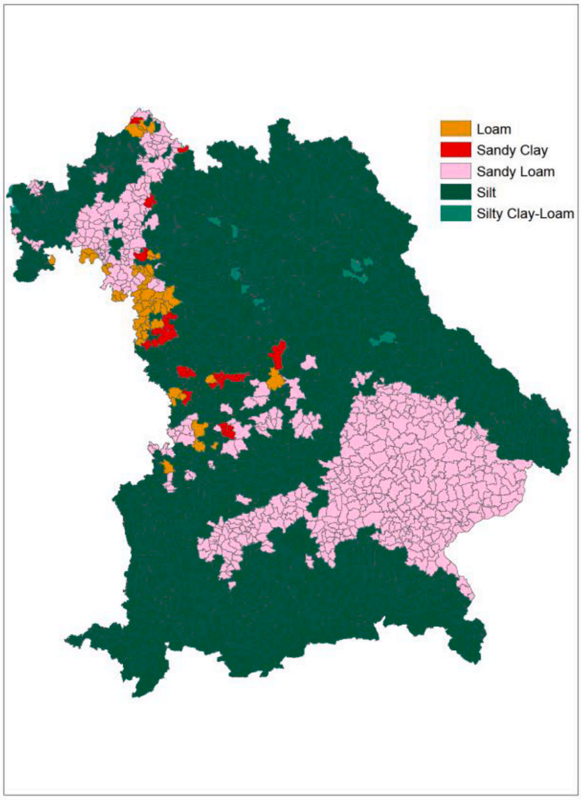
g) Residential area as a share of municipality territory (%)



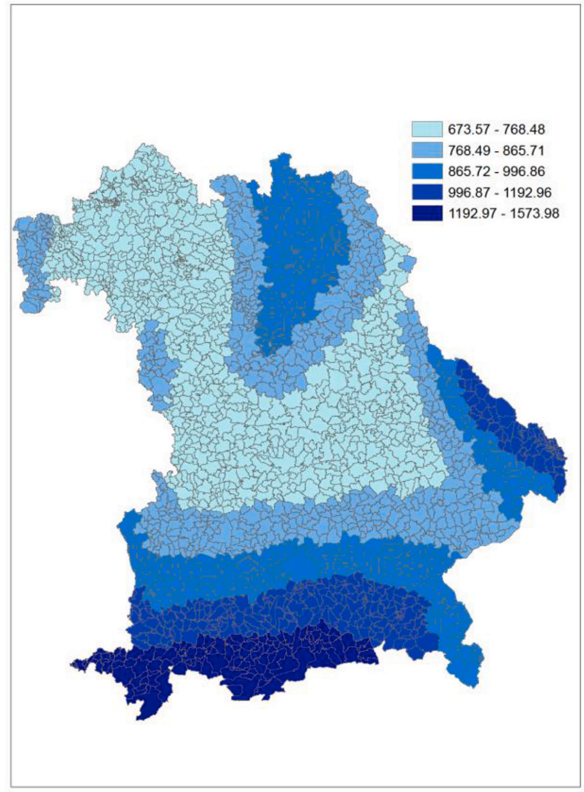
h) Soil quality

(*continued on next page*)

(continued)



i) Soil texture



j) Mean rainfall (mm/yr)

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