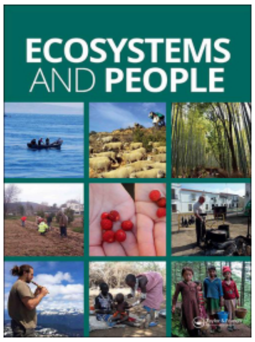

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Quantification and mapping of the nutrient regulation ecosystem service demand on a local scale

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ABSTRACT

In this study, the nutrient regulation ecosystem service (ES) demand was quantified and mapped in an agriculturally-dominated landscape in the federal German state of North Rhine-Westphalia. The demand was assessed in a case study area on an individual field scale. As an indicator for the nutrient regulation ecosystem service demand, nitrogen budgets were calculated. The assessment includes a comparison of an *agriculturally* calculated nitrogen budget to an *ecologically* calculated nitrogen budget. The agricultural calculation is based on legal regulations and considers volatile nitrogen losses from fertilizers, whereas the ecological calculation incorporates the total amount of nitrogen and includes also the atmospheric nitrogen deposition. Furthermore, the positive effects of additional agricultural practices on the nutrient regulation ES demand were identified. The spatial distribution of the nutrient regulation ES demand was compared to the distribution of the nitrate leaching and denitrification potential in order to analyse the relative vulnerability of individual fields to nutrient oversupply. The findings of this study, which highlight in particular the suitability of enlarged crop rotation systems, can be used to support sustainable agricultural practices and land management strategies on the local sale.

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1. Introduction

The excess of nutrients in agricultural systems leads to high nutrient losses, which pose a serious threat to the environment. Lost nitrogen and phosphate degrade ground and surface water quality and threaten biodiversity as well as the climate (Dise et al. 2011; Erisman et al. 2013; Sutton et al. 2013; Leip et al. 2015; Kuhn 2017; Taube 2018). The enrichment of nutrients in water bodies leads to eutrophication (Welte and Timmermann 1985; Fu et al. 2012; Chislock et al. 2013; Dominati 2013; Jónsson et al. 2016; Jónsson and Davídsdóttir 2016). Through eutrophication processes, the ecological status of water bodies is endangered, for instance through the excessive growth of phytoplankton and macroalgae, harmful algal blooms and especially the formation of hypoxic zones (Selman and Greenhalgh 2009; Dise et al. 2011). In order to reduce these effects, the German Fertilizer Ordinance (dt.: Düngeverordnung, DüV 2017) implements the EU Nitrates Directive 91/676/EEC into national regulations, aiming to reduce nitrate (NO3-) emissions from the agricultural sector into water bodies (The European Council 1991; DüV 2017; Kuhn 2017). Initiated by infringement proceedings of the European Commission against Germany concerning the national nutrient situation, the DüV (2017) as well as the German Fertilizer Law (2017) have recently been revised. The modified DüV includes measures that limit the amount of applied fertilizers, and that concern management and technical practices as well as sanctions for violating these regulations (DüV 2017; Kuhn 2017; Bundesanstalt für Landwirtschaft und Ernährung 2018).

The concept of ecosystem services (ES) has been developed in order to increase the understanding of the interrelations between human activities and the environment. Ecosystem services are defined by Burkhard et al. (2012a) as '[…] the contributions of ecosystem structure and function – in combination with other inputs – to human well-being'. ES analyses aim, amongst others, to assess the capacity of ecosystems to provide desired benefits. Thus, they are convenient to support sustainable land management (Smith et al. 2012; Bachmann-Vargas 2013). This is highlighted by the adoption of the ES concept in policies addressing ecological threats. Within the Biodiversity Strategy to 2020, the European Union has asked its member states to map and assess the states of their ecosystems and the services they provide (Maes et al. 2012). Thus, this study was executed in a highly relevant framework integrating contemporary research and assessment approaches as well as political requirements.

Generally, ES can be divided into three main categories; provisioning, regulating and cultural ES (Kandziora et al. 2013; Burkhard et al. 2014; Sohel

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et al. 2015; Stoll et al. 2015; Haines-Young and Potschin 2017; Schneiders and Müller 2017). In the context of this study, the regulating ES nutrient regulation was assessed (for definition see Section 2.2). Within the ES concept, the ES potential has been defined to refer to the hypothetical maximum yield of selected ES (Burkhard et al. 2014), whereas the ES flow describes ES that are actually used in a specific area and time, driven by a demand for ES (Villamagna et al. 2013; Syrbe et al. 2017). The potential and demand for the ES nutrient regulation have been assessed in previous studies (Barrios 2007; de Bello et al. 2010; Bicking et al. 2018, 2019). Thereby, the ES concept has been proven to be a suitable approach for assessing the nutrient situation within the agricultural context (Power 2010; Willemen et al. 2017). The suitability of the ES concept is supported in particular by the strong dependence of agricultural systems on ES on the one hand and the provision of ES, e.g.in the form of crop production, on the other hand (Power 2010; Gutierrez-Arellano and Mulligan 2018).

In this study, the nutrient regulation ES demand was quantified and mapped on the scale of individual fields in the case study area Eversen. Based upon literature research (Leip et al. 2011; Eurostat 2013; Özbek and Leip 2015; DüV 2017; Bicking et al. 2018), two different methodologies were developed for calculating nutrient budgets as indicator for the nutrient regulation demand; a budget from the agricultural and a budget from the ecological perspective (for more information see Section 2.3).

Nitrogen and phosphorus are the two nutrients which mostly limit production in both natural and agricultural systems (Vitousek et al. 2002). Therefore, they are commonly applied to agricultural grounds in vast amounts (Vitousek et al. 1997; Aulakh and Malhi 2005; Power 2010). Based upon its relevance for crop production (Bruns 2012) and the contemporary discussions and the infringement proceedings of the European Commission against Germany concerning the national nutrient situation, the calculation was executed exemplarily for the nutrient nitrogen.

The study includes an evaluation of the effects of different agricultural practices on the nutrient regulation ES demand. Different crop rotation systems, as well as other management options, were considered in the local assessment and their influence on the nutrient regulation ES was evaluated. Subsequently, aiming for a more holistic perspective, the spatial distribution of the nutrient regulation ES demand was compared and linked to the nitrate leaching and denitrification potential in order to perform a vulnerability analysis. A vulnerability analysis evaluates the weakness of a system with consideration to a potential threat (Wisner et al. 2004; Weißhuhn et al. 2018). In this study, the vulnerability of the system in regard to nutrient surplus was assessed. The assessment aimed to analyse the spatial distribution of the damage potential of excess nutrient loads based upon the environmental and management constellations and could operate as an early warning system and identify hotspot areas (Weißhuhn et al. 2018). Thereby, the study integrates the ecosystem service demand with aspects from ecosystem condition.

The overall target of this study was to generate knowledge on the agricultural nutrient situation on the local scale in the study area. Above all, the study aimed to deliver findings which can support the development of sustainable strategies for local land management and agricultural practices. The study was guided by the following three research objectives:

(1) Identification of differences between the ecological and agricultural nitrogen budget.
(2) Assessment of the role of agricultural management, in particular crop rotation, on the nutrient regulation demand.
(3) Evaluation of the system's vulnerability in regard to nutrient surpluses using soil processes.

In the following section, background information on the study area, the ES nutrient regulation, the investigated crop rotation is given and the materials and methods used in the study are introduced. Thereafter, a description of the results is given. The findings of the study are discussed and finally, conclusions are drawn with regard to the research objectives outlined above.

2. Materials and methods

Subsequent to the information on the research objectives of the study, the study area, the ES nutrient regulation, as well as the potential influences of crop rotation, are introduced in the following section which provides insights into the applied methods and underlying datasets. The differences between the agricultural and ecological nitrogen budgets are outlined in detail with regard to the considered parameters and their calculations.

2.1. Study area

The study area is located in the county Hoexter in the federal state of North Rhine-Westphalia (NRW), Germany (Figure 1). The study area is part of the natural regions Lippe Uplands (dt.: Lipper Bergland), Oberwälderland, Warburger Börde and Egge region (dt.: Egge). The annual mean temperature and precipitation in the area are 9.9°C and 918 mm (Deutscher Wetterdienst 2019), respectively. The agricultural grounds of the cooperating farmer are located close to the village Eversen. Due to its proximity to the village, the study area is called Eversen throughout this paper. The study area is situated between 135 and 231 m above sea level. The soil types Luvisol and Cambisol dominate the landscape, which is primarily used as agricultural land (Geologischer Dienst NRW 2016a, 2016b). Structural elements, such as hedgerows

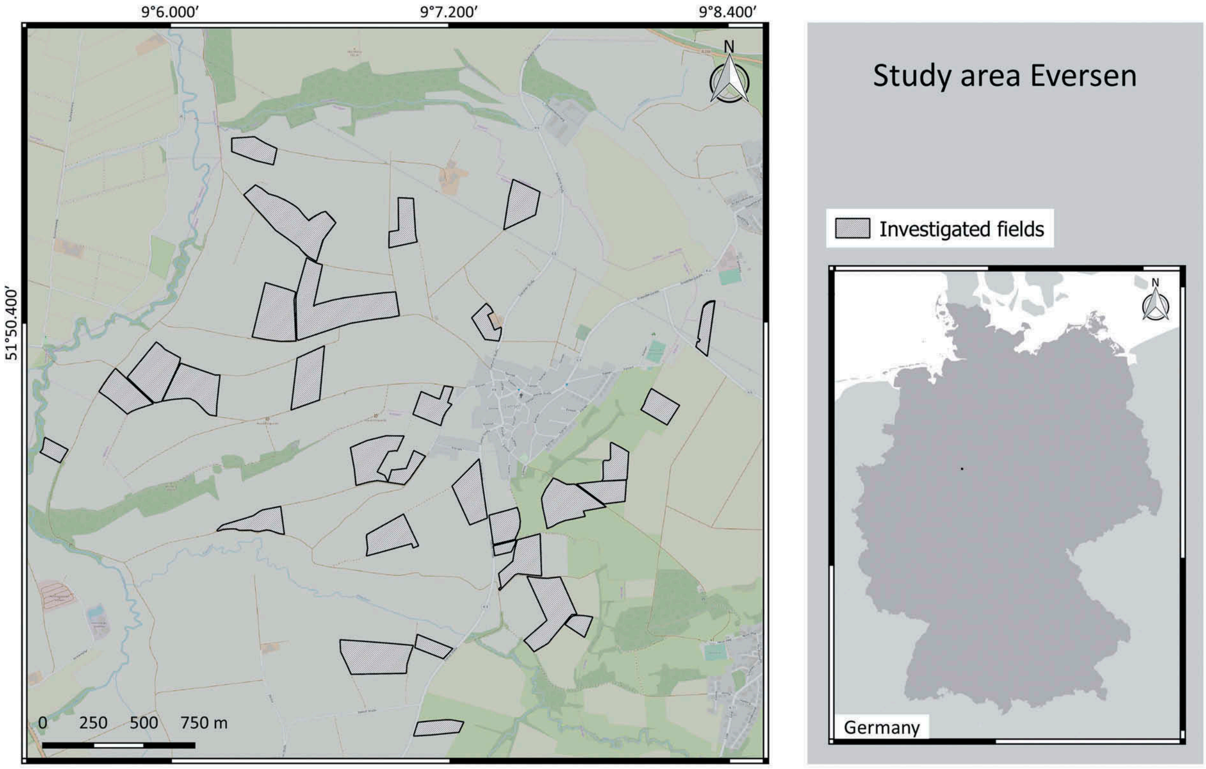

Figure 1. Location of the study areas Eversen incl. investigated fields (base maps: OpenStreetMap, 2019 and GeoBasis-DE/BKG, 2019).

and trees, divide the agricultural fields and increase the diversity and heterogeneity of the landscape (UIH Ingenieur- und Planungsbüro 2016). Geologically, the area contains lower and middle Keuper materials from the Triassic period (Geologischer Dienst NRW 2016a). The focal study area around Eversen is located at the southern border of the Lippe Uplands, a part of the German Central Uplands (Andres 1989; von Zezschwitz 2001; Bundesamt für Naturschutz 2012). The case study area is situated within the Lippe Uplands, in the Steinheimer Börde, a fertile loess region (Andres 1989; UIH Ingenieur- und Planungsbüro 2016).

2.2. Untangling nutrient regulation in the ecosystem service framework

Due to intensive agricultural practices, natural nutrient cycles have been altered (Vitousek et al. 1997). For instance, through the application of vast amounts of fertilizers, the naturally closed nutrient cycles have opened up (Tivy 1987; Chapin et al. 2002). This means that in and outputs of nutrients are getting out of balance. As a result, areas might suffer from either nutrient deficiency or nutrient oversupply (Sutton et al. 2011a, 2011b, 2013; Özbek and Leip 2015). The oversupply of nutrients leads to high nutrient loses which pose a serious threat to the environment. Nitrogen and phosphate degrade the ground and surface water quality and threaten biodiversity and the climate (Sutton et al. 2013; Kuhn 2017; Taube 2018). Enrichment of nutrients in water bodies leads to eutrophication (Welte and Timmermann 1985; Fu et al. 2012; Chislock et al. 2013; Dominati 2013; Jónsson et al. 2016; Jónsson and Davídsdóttir 2016). Besides, high groundwater nitrate concentration can pose a significant risk to human health (Follett and Follett 2001; Townsend et al. 2003; Galloway et al. 2004; Umweltbundesamt 2018a).

Therefore, the ecosystem service nutrient regulation, the ability and magnitude of an ecosystem to recycle nutrients (Burkhard et al. 2014), is a major concern when it comes to agricultural practices and land management as it ensures a functioning and sustainable nutrient cycle (Tivy 1987). The ecosystem service nutrient regulation has triggered quite some discussion within the ecosystem service research domain. In the context of nutrient regulation, studies also refer to the capacity of ecosystems to provide filtering, absorption and retention of nutrients (Dominati 2013; Jónsson et al. 2016; Jónsson and Davídsdóttir 2016). Within the Common International Classification of Ecosystem Services (CICES) nutrient regulation is not specifically mentioned (Haines-Young and Potschin 2018). Nevertheless, it can be classified under class 2.2.4.2 decomposition and fixing processes and their effect on soil quality (Haines-Young and Potschin 2018). Some of the ecosystem service classification schemes defined nutrient regulation as a supporting service (Millenium Ecosystem Assessment 2005; Wilcke et al. 2013; Ghaley et al. 2014; Brockerhoff et al. 2017). Arguments for this classification have mainly been based upon the general understanding of supporting ecosystem services. Supporting services have been understood as services necessary for the provision of all other ecosystem services, e.g. biomass production, soil formation and water cycling (UNEP 2008).

Looking into most recent literature from the field of ecosystem service research, it is striking that a conceptual shift is taking place. There seems to be more consent that ecosystem services can be subdivided into only three main categories (Kandziora et al. 2013; Sohel et al. 2015; Stoll et al. 2015; Brockerhoff et al. 2017; Haines-Young and Potschin 2017; Schneiders and Müller 2017), as outlined in the introduction. Within these frameworks, the categories provisioning, regulating and cultural services cover all individual ecosystem services. Furthermore, an agreement has been reached on the relevance of ecosystem properties and conditions. In lieu of the concept of supporting ecosystem services, they are understood as the functional base for an ecosystem to deliver certain ES (Müller and Kroll 2011; Müller and Burkhard 2012; Brockerhoff et al. 2017; Syrbe et al. 2017; Maes et al. 2018). Ecosystem properties correspond to biophysical structures and processes (Maes et al. 2014; Burkhard and Maes 2017; Syrbe et al. 2017), whereas the ecosystem condition represents an ecosystem's general functionality (Müller and Burkhard 2012; Schneiders and Müller 2017). It is determined by the physical, chemical and biological characteristics, structures and quality features of an ecosystem (Maes et al. 2014, 2018; Erhard et al. 2017). Abiotic factors, e.g. soil texture, as well as biotic soil components, such as fauna and micro-flora, are responsible for certain ecosystem functions and in turn influence ecosystem service potentials through e.g. the supply of nutrients, biological control and maintenance of the soil structure (Gupta et al. 2010; Sandhu et al. 2010; Ghaley et al. 2014). Ecosystem functions are also defined as the ecological mechanisms supporting ecosystem condition and maintaining ecosystems (Brockerhoff et al. 2017), resulting from the interactions of ecosystem properties, thus between ecosystem structures and processes (Truner et al. 2000; Ansink and Hasund 2008; Banerjee et al. 2013; Brockerhoff et al. 2017). Therefore, the evaluation of ecosystem's properties and conditions are highly relevant for a comprehensive ES assessment (Müller and Kroll 2011; Müller and Burkhard 2012; Syrbe et al. 2017).

In Figure 2, the ecosystem service concept including ecosystem's properties and conditions is untangled focussing on nutrient regulation. The conceptional scheme illustrates the interdependencies between the individual aspects and outlines the role of this study in the task of developing a comprehensive and holistic understanding of the ecosystem service nutrient regulation.

Within the holistic ecosystem service concept, the ecosystem service nutrient regulation can be subdivided into ecosystem service potential, flow and demand, whereby the potential is directly based upon an ecosystem's properties and conditions (Figure 2). Thus, the potential of ecosystems to provide nutrient regulation differs. Ecosystem properties which are highly relevant for the ecosystem service nutrient regulation involve ecosystem processes with regard to nutrient flows and pools (Schneiders and Müller 2017). The deduced

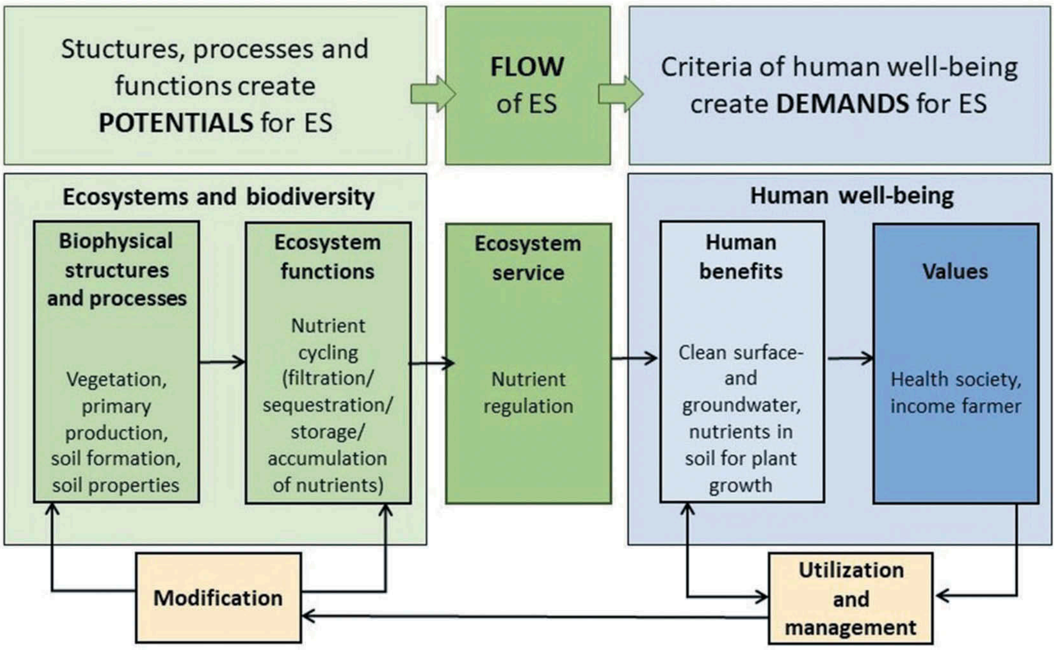

Figure 2. Diagram showing the relations between ecosystem structures, processes, functions and services with relation to the ecosystem service nutrient regulation (based upon Haines-Young and Potschin (2010), Bachmann-Vargas (2013) and Schneiders and Müller (2017)).

ecosystem condition refers to functions such as nutrient cycling, which are determined by soil processes such as nitrate leaching and denitrification. These directly influence the potential of ecosystems to provide nutrient regulation. More natural ecosystems have in general, based for instance on higher biodiversity, more stable structures and functions and therefore higher potentials to provide multiple regulating ES, including nutrient regulation (Fu et al. 2012; Burkhard et al. 2014).

Policy and the society as such, striving for a clean environment and compliance with regulations, can be defined as beneficiaries with a certain demand for the ecosystem service (Villamagna et al. 2013; Bicking et al. 2018). Power (2010) mentions a different perspective and states that the agroecosystem provides and consumes ES, simultaneously. In order to ensure for instance a stable supply of the provisioning ES crop production, agroecosystems are strongly dependent on other (mainly regulating) ES, such as nutrient regulation (Power 2010; Burkhard et al. 2012b; Gutierrez-Arellano and Mulligan 2018). In that sense, the excess amount of nutrients entering the environment has been defined as the indicator for the ecosystem service nutrient regulation. When the nutrient surplus entering the environment is lowered by means of certain land management practices, such as reduced fertilization or adapted crop rotation, the demand for the ecosystem service is minimized. The flow of the ecosystem service nutrient regulation describes the actual nutrient regulation which is taking place (Burkhard et al. 2014; Syrbe et al. 2017). The flow is driven by the demand, thus driven by the surplus of nutrients which enter the environment (Syrbe et al. 2017).

Another aspect worth mentioning is the fact that livestock farmers need to dispose the manure and slurry produced on their farms. Therefore, the application of organic fertilizer onto agricultural fields often has two targets, disposing manure and slurry on the one hand and supplying nutrients to the agricultural grounds on the other hand. Usually, other intensive agricultural practices are performed alongside the application of nutrients, e.g. monoculture or short-term crop rotations, intensive tillage and application of plant protectants (Power 2010). All of these measures aim to increase one single ES, crop production. However, these practices can degrade the environment and decrease the condition of the ecosystem and thereby the potential supply of other ES (Baulcombe et al. 2009; Power 2010; Bruns 2012). Additionally, these circumstances also decrease the future potential of an ecosystem to provide the ES crop production (Power 2010). Thus, even though the application of vast amounts of nutrients can increase a single ES in the short run, the long term provision of multiple ES is diminished (Power 2010). In order to guarantee for long term food security and thus human health, it is of great importance to ensure sustainable nutrient conditions, where in and outputs of nutrients are in balance (Baulcombe 2009; Vitousek et al. 2009). The ES nutrient regulation supports these targets, securing sustainable nutrient cycles (Tivy 1987). As outlined in the introduction, the federal republic of Germany has more nutrient surplus conditions rather than nutrient deficiencies. The spatial distribution of the nutrient surplus in Germany exhibits regional differences with the highest nitrogen surpluses in the Northwest and Southeast of the country (Klement and Bach 2017).

As a consequence of the vast applications of both organic and mineral fertilizer onto the agricultural grounds in Germany, the nutrient concentration of a large share of the ground and surface waters is dramatically high. According to the Umweltbundesamt (2018b), 27,1% of the 1200 groundwater bodies in Germany were in a bad chemical condition with reference to nitrate (>50 mg N/l) in 2017. Also, the groundwater body located in our study area was defined to be in a bad chemical state in regard to nitrate concentrations in 2017 (Umweltbundesamt 2018b). Therefore, there is a high demand for the ES nutrient regulation.

Concurrent to the biogeochemical properties of different nutrients, the corresponding processes in the environment differ. In that respect, it needs to be considered that this study only deals with the nutrient nitrogen as one of the most important nutrients in our environment.

2.3. Nitrogen budget

The method utilized to calculate the nitrogen budget was guided by the DüV (2017) (fertilizer ordinance). As stated in the introduction, the DüV (2017) is the German legislation regulating fertilisation in the agricultural sector, implementing regulations specified in the EU Nitrates Directive (The European Council 1991). In the following section, relevant aspects of the DüV (2017) are summarized, focussing on information on fertilizer application planning and the calculation of the nutrient balance that emphasize all matters related to nitrogen.

In regard to the obligatory fertilizer application planning, the DüV (2017) defines default nutrient requirements for agricultural plants. In this context, also certain corrective values related to the yield are specified (DüV 2017; Kuhn 2017; Bundesanstalt für Landwirtschaft und Ernährung 2018). Considering the N requirements, thresholds for mineral fertilizer applications are dependent on N delivery from the soil and from organic fertilizers. For the N delivery from organic fertilizer, only a specified share of the original N application has to be considered. Certain circumstances, such as specific weather conditions, may allow for a correction of the calculated limit (DüV 2017; Kuhn 2017; Taube 2018). Nutrient contents of manure may be measured or default values from the DüV (2017) may be used for fertilizer planning. The Nitrates Directive limits the application of manure N to 170 kg per hectare and year. The DüV (2017) maintains the threshold but considers additional (plant-based) nutrient sources, such as digestate from biogas plants and compost, and lowers default loss factors considering potential losses in the stable and during application. However, the threshold is calculated on farm level and does not need to be met on each individual plot (The European Council 1991; DüV 2017; Kuhn 2017; Bundesanstalt für Landwirtschaft und Ernährung 2018; Taube 2018).

The DüV (2017) specifies regulations concerning the calculation of nutrient balances, covering the methodological approach, surplus thresholds and sanctions. The nutrient balance is calculated as a surface balance for nitrogen and phosphorus, whereby the nutrient removal through harvest is subtracted from the nutrient inputs via organic and/or mineral fertilizers (DüV 2017; Kuhn 2017; Bundesanstalt für Landwirtschaft und Ernährung 2018; Taube 2018). For nutrient inputs via manure, default loss rates referring to NH3 losses in the stable and during storage and application are subtracted. Again, default values for nutrient contents for manure and harvested products are specified in the DüV (2017). The calculated multi-year average surplus needs to comply with a certain threshold. Farmers may calculate the nutrient surplus at the farm-level or aggregate the calculation from an individual plot-level to farm level. Thus, the nutrient surplus refers to the average of the farm and not to single plots. Besides, only farms with a specific size and intensity characteristics need to calculate nutrient balances (DüV 2017; Kuhn 2017). The DüV (2017) specifies sanctions for violating the regulations concerning the nutrient surpluses. In addition, the DüV (2017) contains regulations concerning fertilizer blocking periods, respective manure storage capacities and manure application techniques (DüV 2017; Kuhn 2017; Bundesanstalt für Landwirtschaft und Ernährung 2018). The DüV (2017) authorises the federal states to require farmers to submit their nutrient balance to the responsible institutions and the DüV (2017) specifies additional measures to be considered in so-called pollution hotspot areas.

Consistent with the research objective, a method to calculate spatially explicit nitrogen budgets was developed. Next to the DüV (2017), additional literature (Leip et al. 2011; Bach et al. 2014; Özbek and Leip 2015; Taube et al. 2015; Bicking et al. 2018) was used for the development of the calculation methods. The nitrogen budgets were calculated on the scale of individual fields in the study area.

According to Taube (2018), the share of organic fertilizer which is accounted for in the nutrient balances according to the DüV (2017) is too low, considering the technological development of the last 20 years aiming to increase the nutrient utilization rate. The DüV (2017) specifies the remaining share that can be considered to be lost, whereas Taube (2018) claims, the values are well below the technical recommendations of the federal state authorities and also lower compared to the regulations in other countries (e.g. Denmark). He argues that a necessary and technically meaningful differentiation of the values according to crop types or time of application was omitted and that altogether the DüV (2017) has no effect with regard to better utilization of organic fertilizers and thus a reduction of nutrient losses into the environment. Another amendment of the DüV is currently in negotiation and is expected to enter into force in spring 2020. The amendment is to be included amongst others field specific nutrient surplus limits and reduced fertilizer requirements (Bundesministerium für Ernährung und Landwirtschaft 2019; Landwirtschaftskammer Schleswig-Holstein 2019).

Beside the above criticism, from the ecological perspective, the priority of the nitrogen budgets should be the generation of knowledge on the genuine nutrient situation and not to maximize crop production which should only be a secondary target. Only then, can the nutrient budgets support sustainable agricultural practices. Therefore, two different calculations have been performed; an agricultural nitrogen budget, which considers the loss factors for the application of fertilizers, in line with the specifications in the DüV (2017), and an ecological nitrogen budget which considers the total amount of fertilizer applied (Leip et al. 2011). In addition to that, the ecological nitrogen budget also considers atmospheric N-deposition as a nitrogen input parameter. Table 1 gives an overview of the parameters considered in the two different approaches.

Table 2 summarizes the different parameters which have been considered for the calculation of the nitrogen budgets, the respective methods and specific data sources that have been used. Most information was provided by the farmer, either from his field record system or through personal communication. The crop type related agricultural practices considered in the

Table 1. Parameters considered for the agricultural and ecological nitrogen budget calculation.

Parameter[a]	Agricultural nitrogen budget	Ecological nitrogen budget
Mineral fertilizer	x	x
Organic fertilizer from livestock	x	x
Digestate from biogas plants	x	x
Biological nitrogen fixation	x	x
Atmospheric N-deposition		x
Yield	x	x
Fertilizer loss rates	x	

[a]No compost and sewage sludge have been applied onto the investigated fields in the considered timeframe.

Table 2. Overview of methodologies and data sources used for the parameters considered for the calculation of the nitrogen budgets.

Parameter	Indicator with quantification unit	Quantification methods and data sources
Nutrient input	Mineral fertilizer (kg N/(ha*year))	Data on application from field record system (von Ruschkowski 2018)
	Organic fertilizer from livestock (kg N/(ha*year))	Data on application from field record system (von Ruschkowski 2018); data on N-content of organic fertilizer from field record system (von Ruschkowski 2018) and from laboratory reports by Landwirtschaftliche Untersuchungs- und Forschungsanstalt NRW (Landwirtschaftliche Untersuchungs- und Forschungsanstalt 2018)
	Digestate from biogas plants (kg N/(ha*year))	Data on application from field record system (von Ruschkowski 2018)
	Biological nitrogen fixation (kg N/(ha*year))	Data on cultivation (grassland and legumes) from field record system (von Ruschkowski 2018); data on specific efficiency of nitrogen fixation derived from Landwirtschaftskammer Niedersachsen (2016) and other sources (a.o. Loges et al. 1998; Sächsische Landesanstalt für Landwirtschaft 2007; Loges 2013)
	N-deposition (wet, dry and occult) (kg N/(ha*year))	Data on nitrogen deposition in Germany in 2009 for different land use types from the Umweltbundesamt (UBA 2009; Stickstoffdeposition. PINETI-2 (Pollutant INput and EcosysTem Impact). Personal communication)
Nutrient output	Yield (crop and grassland) (kg N/(ha*year))	Yield estimated by the farmer; data on average N-content of grass/crop type from DüV (2017)
	Stable, storage and application loss rates (%)	Specific deduction rates based upon specification in field record system (von Ruschkowski 2018) according to DüV (2017)

nitrogen budgets on the field scale are summarized in crop type specific profiles (Appendix A).

2.3.1. Crop rotation

Crop rotation refers to the sequence of specific crops planted on one field, whereby the succeeding crop belongs to a different family than the previous crop (PAN Germany 2010). Crop rotation is crucial in supporting the long-term productivity of crop cultivation and thereby the provisioning ecosystem service crop production. In general, crop rotation affects different highly relevant environmental issues, such as species diversity, biodiversity at the landscape level, soil fertility and soil health, water and climate change (European Commission – DG ENV 2010; Bruns 2012). Proper crop rotation influences even more ecosystem services, as it reduces weed and disease pressure, preserves soil fertility and biodiversity and at the same time secures economic viability (Bruns 2012; Bayerische Landesanstalt für Landwirtschaft 2016). The additional advantages which can be gained using crop rotation (European Commission – DG ENV 2010; PAN Germany 2010; VALERIE 2017) can be linked to the following provisioning and regulating ecosystem services: drinking water, groundwater recharge/water flow, local climate regulation, global climate regulation, air quality regulation, erosion regulation, nutrient regulation, water purification, pest and disease control and pollination. Of course cultural ecosystem services, such as landscape aesthetics and recreation will be also influenced.

As crop rotation is such an integral part of sustainable land management, it is addressed in different policies at EU level. The 2003 Common Agricultural Policy (CAP, European Commission 2019) requires farmers to maintain their fields in a Good Agricultural and Environmental Condition (GAEC). One of the defined standards of the GAEC deals with crop rotation. The Agri-Environmental Measures (AEM) aim to encourage farmers by offering financial means to apply agricultural production methods, which sustain and/or enhance environmental conditions (European Commission Agri-environment measures 2005; The European Council 2005). The financed AEMs may include measures on crop rotation. In addition to that, the European policy on organic farming (The European Council 2007) considers crop rotation as a keystone for holistic sustainable farming management. Crop rotation was included within the Nitrate Directive (The European Council 1991) in the form of enhanced crop management (e.g. intercropping, soil covers) and crop-specific guidelines. The significance of crop rotation within EU policies underlines the high relevance of this agricultural practice. Therefore, management options with regard to crop rotation have been considered in this study. In order to identify the effects of different crop rotation systems on the nutrient regulation demand, the nitrogen budgets were calculated for different crop rotation systems. Table 3 gives an overview of the crop rotation systems, which have been considered in this study. The 4-year cycle crop rotation systems listed in the table have formerly been employed in the case study area. Nowadays, the farmer changed to the 5-year crop rotation systems, including grass/clover cultivation.

2.4. Nitrate leaching and denitrification potential

The denitrification potential, represented by the denitrification rate in kg NO3-N/ha/a, was evaluated and made available by the Geologischer Dienst (Geological Service) NRW (2017a). The Geologischer Dienst NRW (2017a) based the evaluation on information on a broad set of soil parameters including soil texture, soil-water condition and soil organic carbon. The dataset is provided as part of the soil map of North Rhine Westphalia on the scale 1:50′000 (Geologischer Dienst NRW 2017b). The nitrate

Table 3. Crop rotation systems considered in the calculation of the nitrogen budgets.

Crop rotation cycle	Crop types	Note
4 years	Rapeseed/Silage maize/Field bean Wheat Triticale Barley	Former systems
5 years	Rapeseed/Silage maize/Field bean Grass/clover Wheat Triticale Barley	Current systems

leaching potential is represented by the soil water exchange rate in %/a. For the quantification of the soil water exchange rate, a method by Müller and Waldeck (2011) was applied, considering soil depth, texture, plant available water, yearly evapotranspiration and precipitation as well as the groundwater level. Soil data was obtained from the soil map of North Rhine Westphalia, climate data was provided by the Climate Data Center of the German Meteorological Service (DWD Climate Data Center 2018a, 2018b).

Data of the individual spatial components were mapped using the GIS software ArcGIS and QGIS. Besides, a statistical comparison of the maps was performed using R. For that matter, the map comparison statistic was adopted (Hagen-Zanker 2006; Schulp et al. 2014). The map comparison statistic summarises the relative differences of the compared maps (Schulp et al. 2014) and is calculated for each pair of maps based upon the following formula:

$$MCS = \frac{\sum_{n=1}^{N}\left(\frac{|a-b|}{\max(a,b)}\right)}{N} \quad (1)$$

Where MCS is the map comparison statistic, and a and b correspond to the normalized values of the assessed indicators (nitrogen budget, nitrate leaching potential and denitrification potential). The statistical analysis aims to identify the average difference between each pair of compared datasets (Hagen-Zanker 2006; Schulp et al. 2014; Ma et al. 2019).

2.4.1. Vulnerability analysis

Subsequently, the datasets were used for performing an ecosystem vulnerability analysis. As outlined in the introduction, vulnerability analyses aim to assess the weaknesses of a certain system in regard to a potentially harmful threat (Wisner et al. 2004; Weißhuhn et al. 2018). In our case, the potentially harmful threat is the surplus of nutrients, causing eutrophication. Vulnerability is commonly outlined as a function of exposure, sensitivity and adaptive capacity (Turner II et al. 2003; Füssel 2007; Frazier et al. 2014; Weißhuhn et al. 2018). Thereby, the exposure stands for the probability of the specific hazard, the sensitivity describes the susceptibility to the hazard and the adaptive capacity measures the ability of the system to deal with the hazard (Weißhuhn et al. 2018). Translated to the datasets of this study, the nitrogen budget (nutrient regulation ES demand) was used as exposure to the hazard of nutrient oversupply. The nitrate leaching potential measures the sensitivity of an ecosystem and the denitrification potential assesses the adaptive capacity of an ecosystem to cope with the potential nutrient surplus. The spatial representation of an ecosystem's vulnerability allows for the identification of hotspot areas. Specific land management plans with regard to for instance protection can be developed for and implemented in these hotspots areas (Zurlini et al. 1999; Aretano et al. 2015). The relative vulnerability of the individual fields was estimated based on a GIS analysis. Therefore, the datasets of the nutrient regulation ES demand, nitrate leaching and denitrification potential were normalized and combined, weighted evenly (Appendix B). Both the nutrient regulation ES demand and the nitrate leaching potential increase the relative vulnerability whereas the denitrification potential reduces the relative vulnerability (Formula 2).

$$\begin{aligned} \textit{Relative vulnerability} = {} & \textit{nutrient regulation ES demand} \\ & + \textit{nitrate leaching potential} \\ & - \textit{denitrification potential} \end{aligned} \quad (2)$$

3. Results

The nitrogen budgets were assessed on the scale of individual fields. Both the agricultural nitrogen budget and the ecological nitrogen budget were calculated. On average, the ecologically calculated annual nitrogen budget was higher by 49 kg N per hectare than the agriculturally calculated annual nitrogen budget. The assessment includes an evaluation of different crop rotation systems (see Table 4). Both nitrogen budgets were calculated for the different crop rotation systems. The average agricultural practices of the season 2017/2018 in terms of fertilizer application were used for the calculation.

Table 4 presents the calculated average annual nitrogen budgets for the assessed crop rotation systems. Generally, the agricultural budgets delivered lower values. The findings demonstrate the differences in the two calculation methods. According to the agricultural nitrogen budgets, for both the 4- and

Table 4. Annual average nitrogen budget for the 4-year and 5-year crop rotation system. The green shading highlights the lowest budget within the respective crop rotation system.

	Crop rotation (ø 2017/2018)					
	4-year cycle			5-year cycle		
	Rapeseed Wheat Triticale Barley	Silage maize Wheat Triticale Barley	Field bean Wheat Triticale Barley	Rapeseed Grass/clover Wheat Triticale Barley	Silage maize Grass/clover Wheat Triticale Barley	Field bean Grass/clover Wheat Triticale Barley
ø budget (agri) in kg N/(ha*a)	−5	−27	−11	−4	−21	−8
ø budget (eco) in kg N/(ha*a)	53	34	26	45	30	24

5-year cycle, the crop rotation systems including maize deliver the lowest budgets. According to the ecological budget calculation, the respective crop rotation systems including field beans result in the lowest annual nitrogen budgets. Crop rotation including rapeseed delivers the highest nitrogen budgets for all considered rotation systems and calculation methods. Generally, the 5-year crop rotation system results in lower annual nitrogen budgets compared to the 4-year crop rotation system. On average, these nitrogen budgets are 10% lower compared to the 4-year crop rotation system. In addition to the fertilizer application and the crop rotation system, additional management options have been implemented to prevent soil erosion and surface runoff and to support nutrient regulation. Since 2017, some field margins have been left fallow and flower strips have been cultivated. The implementation of these measures decreased the annual average nitrogen budget by 3%.

As outlined in Section 2.3, the share of nutrients which should be considered in the budget according to the DüV (2017) is under criticism. According to Taube (2018), these specifications and regulations have no effect in terms of improving the use of organic fertilizers and thus reducing nutrient losses to the environment. Therefore, the ecological nitrogen budget has been defined as the most appropriate indicator for the nutrient regulation ES demand. The nutrient regulation ES demand has been quantified and mapped for the individual fields in the case study area Eversen (Figure 3).

The findings indicate a spatial variation of the nutrient regulation ES demand. Generally, the annual average nitrogen budget of the individual fields ranged between 14 and 52 kg N per hectare, with an overall average of around 32 kg N per hectare. As explained in the introduction, in order to guarantee a more holistic ES assessment, underlying relevant ecosystem processes were assessed. We examined two soil processes, which are highly relevant for the ES nutrient regulation, namely the nitrate leaching and denitrification potential (Figure 4).

As outlined in Section 2.4, the nitrate leaching and denitrification potential are based on different soil properties and the climatic conditions. Thus, the differentiation in the study area originates from the spatial distribution of these environmental characteristics. Therefore, the underlying spatial patterns of the two potentials resemble each other. Nevertheless, both potentials have a distinct spatial distribution (Figure 4). Overall, the study area is characterized by rather low nitrate leaching potentials. However, scattered patches with higher nitrate leaching potentials are spread throughout the whole study area. Additionally, these patches feature low denitrification potentials (denitrification rates: < 10 kg NO3-N/(ha*a)). These areas correspond to the soils with the lowest field capacities in the study area. The largest part of the study area is characterized by medium denitrification potentials (denitrification rates: 10–30 kg NO3-N/(ha*a)). Areas with higher than average denitrification rates rarely coincide with the location of the fields which have been investigated. A visual comparison of the spatial distribution of the nitrate leaching and denitrification potential to the nutrient regulation ES demand (Figures 3 and 4) showed no distinct correlation. These findings are supported by the map comparison statistics which deliver the following results (Table 5).

The map comparison statistics indicate a moderate randomness between each pair of maps. This supports the assumption that each dataset contributes auxiliary information to the assessment. Taking this into consideration, a vulnerability analysis was performed which combines the spatial distribution of the nutrient regulation ES demand with the respective datasets of the nitrate leaching and denitrification potential. Thereby, the ecosystem's vulnerability to the oversupply of nutrients was assessed. The nutrient regulation ES demand, in the form of the ecological nitrogen budget, has been defined as the exposure to the hazard of nutrient oversupply. The nitrate leaching and denitrification potential served as sensitivity and adaptive capacity of the ecosystems, respectively.

The qualitative vulnerability assessment identified hot and cold-spots of relative vulnerability (Figure 5) in the case study area Eversen. A large number of the fields located in the cross section from Northwest to Southeast was characterized by a very high vulnerability. In addition, three fields in the Northwest fall into the same category. In these areas, the soil processes and agricultural practices led to the most unfortunate combination. Another aspect, which should be considered with

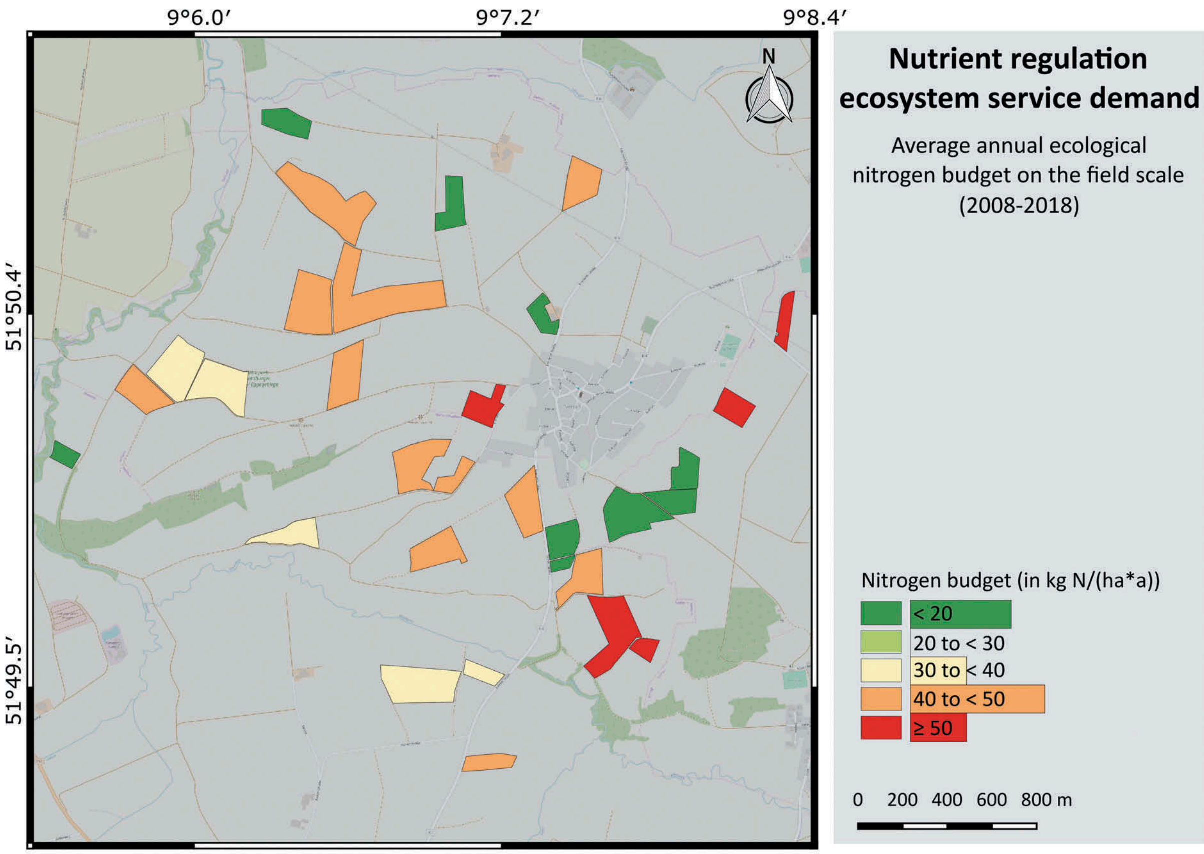

Figure 3. Nutrient regulation ES demand on the individual fields in the case study area Eversen based on the average annual ecological nitrogen budget from 2008 to 2018. The frequency distribution refers to the number of fields in the respective categories (base map: OpenStreetMap, 2019).

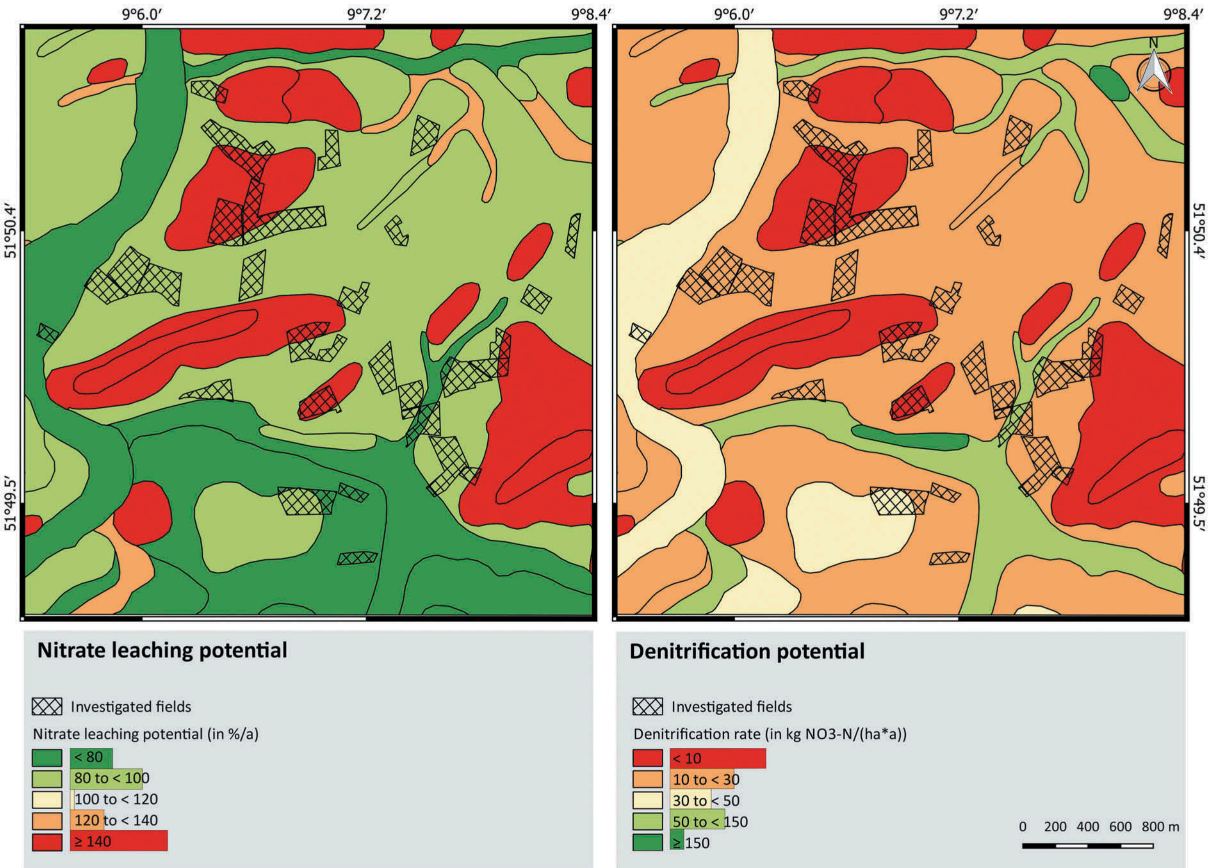

Figure 4. Nitrate leaching potential and denitrification potential (Geologischer Dienst NRW 2017a) in the case study area Eversen (please consider the respective color schemes of the two potentials). The frequency distribution refers to the number of polygons in the respective categories.

Table 5. Map comparison statistics (MCS) of nutrient regulation ES demand, nitrate leaching and denitrification potential. Identical maps result in a MCS of zero, a MCS of 0.5 indicates that the pair of maps are random and a opposing pair of maps produce a MCS of 1.

Relevant datasets	Nutrient regulation ES demand	Nitrate leaching potential	Denitrification potential
Nutrient regulation ES demand	(0)	0.47	0.47
Nitrate leaching potential		(0)	0.56
Denitrification potential			(0)

respect to the vulnerability of the area, is the proximity of the fields to water bodies and nature protection areas (Figure 5). The case study area is surrounded by both. This proximity should be perceived as an additional incentive to implement a sustainable nutrient management concept.

4. Discussion

4.1. The discrepancy between ecology and agriculture

The ecosystem service demand assessment was primarily based on the farmer's field record system, which includes data for the years 2008–2018 (von Ruschkowski 2018). The annual average nitrogen budgets (in kg N/(ha*a)) were calculated using two different approaches. Firstly, an agricultural nitrogen budget was calculated which has been developed based on the DüV (2017) and other agricultural literature (a.o. Leip et al. 2011; Bach et al. 2014; Özbek and Leip 2015; Taube et al. 2015). It considers nitrogen deduction rates for fertilizer application. Secondly, an ecological nitrogen budget was calculated which considers the whole nitrogen cycle. Therefore, no deductions were applied and additionally the atmospheric N-deposition was considered as an input parameter. Expectedly, the agricultural nitrogen budget was lower than the ecological nitrogen budget. The investigated fields delivered ecological annual nitrogen budgets up to 52 kg N per hectare.

4.2. Plant championship

The fertilization practices were compared to the defined N requirements according to the fertilization planning of the DüV (2017). For each agricultural crop type, crop profiles were compiled which refer to agricultural practices related to fertilization and residual management and the estimated yield (Appendix A). The comparison refers to the general N requirements of the agricultural crop (DüV 2017 Appendix 4). Exceptions of the N requirements, due to, for example, specific N soil contents, were not considered. However, the comparison indicates that the fertilization practices were well

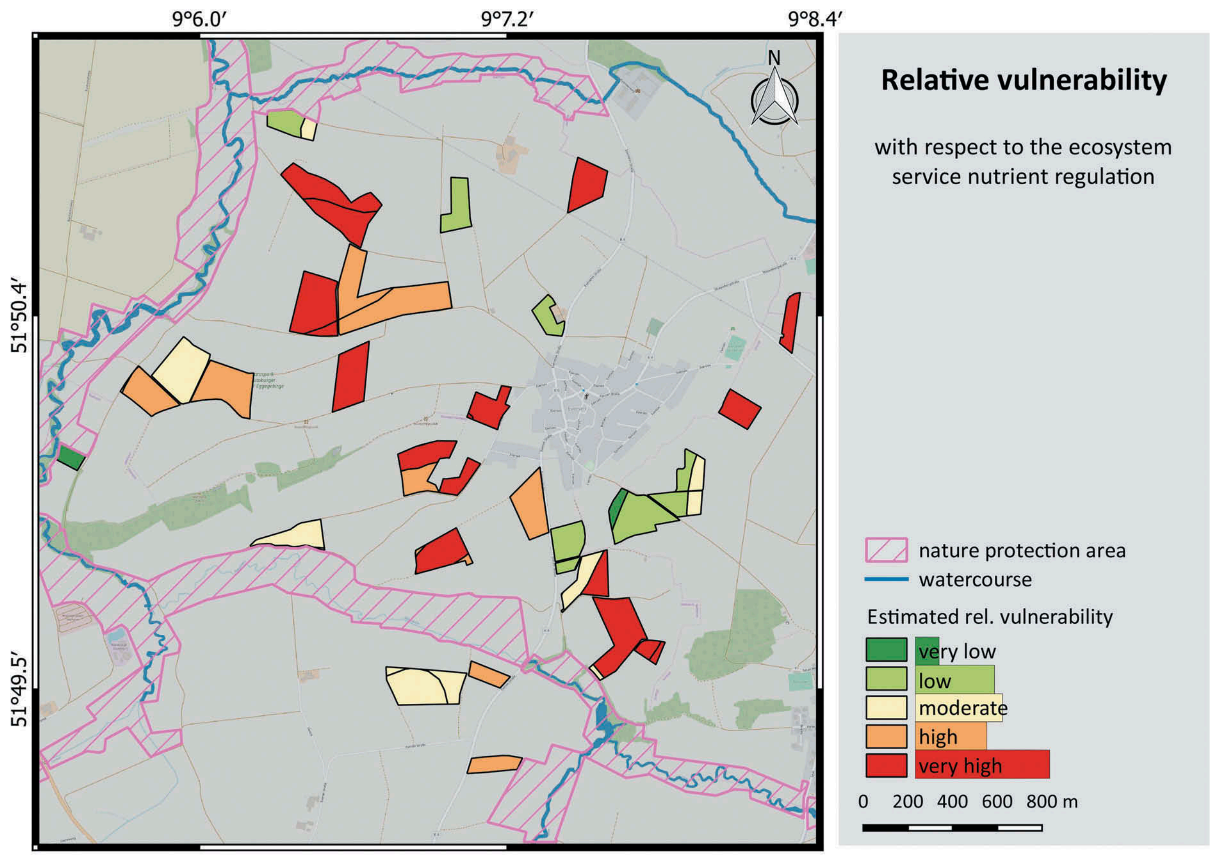

Figure 5. Location of water courses, nature protection areas and estimated relative vulnerabilities with respect to the ES nutrient regulation on the individual fields in the case study area Eversen. The relative vulnerability accounts for the nutrient regulation ES demand (ecological), the nitrate leaching and the denitrification potential. The frequency distribution refers to the number of polygons in the respective categories (base map: OpenStreetMap, 2019).

below the defined N requirements according to the DüV (2017). The fertilization practices for rapeseed were closest to the N requirements defined by the DüV (2017). The comparison of the different crop rotation systems for both the annual agricultural budget and the annual ecological budget corresponds to our expectations. The annual average nitrogen budget is generally lower in the 5-year crop rotation system, compared to the 4-year crop rotation system. These findings are in line with the literature (European Commission – DG ENV 2010; PAN Germany 2010; Bruns 2012; VALERIE 2017). Furthermore, the comparison also supports the above outlined findings on the differences between the agricultural and ecological budgets. The agricultural calculation identifies the lowest nitrogen budgets for the crop rotation system including maize. However, according to the ecological calculation, the crop rotation system including field beans performs best. As the agricultural budget allows for nitrogen deduction and only the ecological budget delivers integral insights into the total amount of nitrogen introduced into the environment, the crop rotation system including field beans can be considered as a best practice method. These findings support the conception of the different calculation methods (agricultural and ecological) outlined above. The crop rotation including rapeseed delivers highest nitrogen budgets for all considered crop rotation systems. Next to the relatively high fertilizer application rates for rapeseed, the high product-to-residue ratio, which implies relatively high amounts of nitrogen remaining on the field after harvest, explains these findings.

According to the agricultural calculation, even negative nitrogen budgets occurred in the case study area, which might cause nitrogen deficiency in the long run. Nevertheless, as discussed in Section 2.3, the loss values considered in the calculation of the agricultural nitrogen budget are subject to criticism (Taube 2018). According to Taube (2018), the nutrient share which is considered in the budgets is too low from an ecological and also technical perspective. Thus, it is most likely that the calculated agricultural nitrogen budgets deliver an incorrect impression of the nutrient situation. Therefore, the ecological nitrogen budget has been selected as the indicator for the nutrient regulation ES demand in the case study area. Considering the management implications, in particular the extended crop rotation system including grass/clover should be highlighted as an efficient measure for reducing the nitrogen budgets and therefore the demand for nutrient regulation.

4.3. Putting ES demand into perspective

By embedding the analysis in the ES framework a comprehensive assessment of the nutrient situation was possible. The major focus of this study was the calculation of nitrogen budgets as an indicator of the ES demand. As proposed by recent literature, the ecosystem properties and conditions are fundamental for an ecosystem's ES supply capacity (Power 2010; Müller and Kroll 2011; Müller and Burkhard 2012; Syrbe et al. 2017; Maes et al. 2018). Therefore, the study included an assessment of some fundamental ecosystem processes with regard to the ES nutrient regulation: Nitrate leaching and denitrification potential. Both of these soil processes influence the nutrient cycle within the ecosystem and thereby account for the ES potential of nutrient regulation. However, more information is necessary to ensure an integral assessment of the nutrient regulation ES supply. The available data allowed performing a spatially explicit vulnerability assessment. The findings of the nutrient regulation ES demand were compared to the spatial distribution of the nitrate leaching and denitrification potential. All information was combined in order to assess the relative vulnerability of the area to nutrient oversupply. The consideration of adjacent nature protection areas and watercourses next to the agricultural fields increased the significance of the evaluation. Of course, land management and agricultural practices of the entire corresponding catchment area are of relevance for the quality of water bodies. Nevertheless, the impact factor in particular with reference to the lag time increases with increasing proximity. In addition to surface water, also groundwater bodies should be considered in an integrated assessment. As the spatial representation of an ecosystem's vulnerability allows for the identification of hotspot areas (Weißhuhn et al. 2018), the evaluation can serve as a foundation to detect areas where special attention needs to be paid with respect to land management strategies or agricultural practices (Zurlini et al. 1999; Aretano et al. 2015). The spatially explicit vulnerability evaluation can serve in particular as a base for designating additional environmental conservation measures. Measures, such as the establishment of flower strips, fallow land or the cultivation of green infrastructure such as hedgerows, can be implemented in areas which are identified to exhibit high or very high vulnerability to nutrient oversupply. In the case study area, the fields located in the West and in the cross-section from Northwest to Southeast were characterized by (very) high vulnerability. In particular, these fields should be given special attention in regard to balanced nitrogen in- and outputs.

4.4. Reality check

The calculations of the nitrogen budgets were primarily based on data and information that were obtained directly from the farmer. Information such as the application of fertilizers on the fields was recorded in his field record system (2018). The information was available and specified with regard to the quantity as well as to the date of application. In regard to the average yield per hectare, the information was of a different quality. The yield of the different crop types was not weighted and recorded accordingly. The farmer had estimated the average yield for the different crop types. Next to the general uncertainty of this approach, the information on the average yield was static and no annual variation was included.

Crop rotation was considered in this study with regard to the calculation of the nitrogen budgets. The calculation was based on a summation of the in and outputs of the different crop types during the considered period of time (4 and 5 years). This summation included aspects related to biological nitrogen fixation through legumes. However, it did not include other crop type-specific aspects such as vegetation period (including winter soil coverage), root mass and tillage practices, which influence the nutrient cycle (Jacobs et al. 2009; Bruns 2012; Jie et al. 2013; Busari et al. 2015). A more complex modelling approach, which considers these aspects, would most likely result in even more significant differences between crop rotation systems (Power 2010). Another limitation of the assessment is that all calculations were based on total nitrogen quantities and no differentiations were made between organic and inorganic nitrogen (Smith et al. 2013). The mobility and plant availability of organic and inorganic nitrogen (Smith and Hadley 1989; Smith et al. 2013) were not considered. Thus, no differences were made between biologically-fixed nitrogen (e.g. through the cultivation of field beans), organic and mineral fertilizer application.

In addition to the specific uncertainties and limitations, which originate from the methodological approach, also general uncertainties of ES research need to be considered (Hou et al. 2013). General uncertainties come from the complexities of ecosystems and human-environmental interactions. Therefore, the ES concept is of multi, inter and transdisciplinary nature (Burkhard 2017; Maes 2017), incorporating, in particular, the ecological and socio-economic research domains. Besides, challenges arise from the terminology within the ES concept. For instance, the differentiation between ES potential, flow and demand need to be taken into account (Burkhard et al. 2012b, 2014; Schröter et al. 2014; Dunford et al. 2017; Bicking et al. 2018). Additionally, for the interpretation of results, input data, applied methodologies and in particular, the employment of proxies need to be considered. In this study, the nitrogen budgets were used as a proxy for the nutrient regulation ES demand. The careful consideration of all these issues is essential for the interpretation of ES assessments in order to safeguard the comprehension of the results (Dunford et al. 2017).

4.5. The road ahead is golden

A promising approach for further assessments is the development of an elaborated model for the calculation of nitrogen budgets on the field scale. This model should be developed for a case study area in order to enable the inclusion of further temporal and spatial explicit variations. The highly elaborated model package RAUMIS-GROWA-DENUZ-WEKU (Heidecke et al. 2014; Ackermann et al. 2015; Wendland et al. 2015) could serve as an example for a methodological approach which allows for integral assessments on local scales. Just as the package RAUMIS-GROWA-DENUZ-WEKU, this model should integrate an assessment of further relevant nutrients, in particular phosphorous, and contain a comprehensive hydrological model. In particular, general aspects concerning seasonality and other agricultural practices such as tillage should be included. The model should also allow for temporal-explicit specifications with regard to fertilizer application and other agricultural practices. The model should generate profound knowledge on the processes and functions which play crucial roles in agro-ecosystems. Eventually, the insights can be used in order to increase the integral understanding of ecosystems as parts of complex social-ecological systems.

5. Conclusions

This study contributes to the contemporary debate on nutrient management, surplus and nitrate groundwater concentrations. The nitrogen budget was defined as an indicator for the nutrient regulation demand, which was quantified and mapped. Furthermore, the assessment included an evaluation of other agricultural practices and a vulnerability assessment incorporating information on soil processes. Summing up the results obtained in the study, the following can be stated in regard to the research objectives:

(1) **Identification of differences between the ecological and agricultural nitrogen budget.**
The assessment of the nutrient regulation ES demand shows that high nitrogen budgets are common practices. A strong variation was discovered between the agricultural and ecological budget. The agricultural budget even showed nutrient deficiency situations. However, according to recent literature, the share of nutrients which are considered in the agricultural budget is too low and not in line with current scientific and technological development. The ecological budget, on the other hand, considered all nutrient inputs, even atmospheric deposition. From the ecological perspective, this budget has the greater significance.

(2) **Assessment of the role of agricultural management, in particular crop rotation, on the nutrient regulation demand.**
The assessment identified the effects of different agricultural practices on the nutrient regulation ES demand. The comparison of different crop rotation systems revealed that the assessed 5-year crop rotation systems result in lower

average annual nitrogen budgets than the 4-year crop rotation system. Generally, the crop rotation including field beans leads to lowest nitrogen budgets within their respective system. Besides, the effect of additional agricultural measurements aiming to reduce surface runoff, prevent erosion and increase nutrient regulation on the nutrient regulation ES demand was identified through the calculation of the nitrogen budgets.

(3) **Evaluation of the system's vulnerability in regard to nutrient surpluses using soil processes.**
The ES concept indicates that the capacity of an ecosystem to provide ES is based on the ecosystem properties and conditions, corresponding to structures, processes and functions of an ecosystem. One aspect of ecosystem condition, which is highly relevant for the capacity of an ecosystem to provide the ES nutrient regulation, is nutrient cycling. In this study, two soil processes (nitrate leaching and denitrification), affecting an ecosystem's nutrient cycling, have been assessed. Thus, even though there were no direct measurements on the actual nutrient regulation ES potential and ES flow available for this study, the incorporation of the nitrate leaching and denitrification potential already increased the scope of the ES assessment. The combination of the nutrient regulation ES demand with the nitrate leaching and denitrification potential was used in order to estimate the relative vulnerability of the case study area in the context of the ES nutrient regulation. Commonly, a vulnerability assessment incorporates information on the exposure, sensitivity and adaptive capacity with regard to a specific hazard. In this study, the nitrogen budget, nitrate leaching and denitrification potential were used as indicators for these parameters. The assessment identified hot and cold spots of vulnerability in the study area and can, therefore, be used to support sustainable land management policies and agricultural practices concerning fertilization strategies. For instance, the information on the spatial distribution of the relative variability on the field scale can support a farmer to make decisions on crop rotation and fertilization plans. Besides, a farmer can commission soil nutrient analysis on fields, which are identified to have a high nutrient regulation demand and/or install conservation measures aiming to prevent unsustainable nutrient conditions, safeguarding the environment and concurrently securing his/her own future prospects

Generally, this study investigated the contemporary nutrient issue which originates from conventional agricultural practices and intensive land management. The study identified the reducing effects that agricultural practices such as specific crop rotation systems have on the nitrogen budget. Next to these findings, also the spatial assessment of the nutrient regulation ES demand and the spatial vulnerability assessment delivered valuable results. The study area is an agriculturally dominated landscape. Therefore, sustainable agricultural practices and land management strategies are essential in order to safeguard the production of food and fodder on the one hand and to prevent environmental pollution and consequential degradation of soils and water bodies on the other hand. The findings of this study can serve as a foundation for the development of land management policies and agricultural practice plans aiming to decrease nitrogen budgets and thereby reducing the nutrient regulation ES demand.

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Data availability statement

The authors confirm that most data supporting the findings of this study are available in the article and its supplementary materials. The remaining data that support the findings of this study are available on request from the corresponding author, SB. The data are not publicly available due to data privacy law.

References

Ackermann A, Heidecke C, Hirt U, Kreins P, Kuhr P, Kunkel R, Mahnkopf J, Schott M, Tetzlaff B, Venohr M, et al. 2015. Der

Modellverbund AGRUM als Instrument zum landesweiten Nährstoffmanagement in Niedersachsen. Braunschweig: Johann Heinrich vonThünen-Institut; p. 314.
Andres W. 1989. The central german uplands. In: Ahnert F, editor. Landforms and landform evolution in West Germany. Stuttgart, Germany: CATENA. p. 347.
Ansink E, Hein L, Hasund K. 2008. To value functions or services? An analysis of ecosystem valuation approaches. Environmental Values. 17(4):489–503.
Aretano R, Semeraro T, Petrosillo I, De Marco A, Pasimeni MR, Zurlini G. 2015. Mapping ecological vulnerability to fire for effective conservation management of natural protected areas. Ecol Modell. 295:163–175. doi:10.1016/j.ecolmodel.2014.09.017
Aulakh MS, Malhi SS. 2005. Interactions of nitrogen with other nutrients and water: effect on crop yield and quality, nutrient use efficiency, carbon sequestration, and environmental pollution. Adv Agron. 86:341–409.
Bach M, Hillebrecht B, Hunsager EA, Stein M. 2014. Berechnung von Stickstoff-Flächenbilanzen für die Bundesländer - Jahre 2003 bis 2011. doi: 10.13140/RG.2.1.1809.5608
Bachmann-Vargas PL. 2013. Ecosystem services modeling as a tool for ecosystem assessment and support for decision making process in Aysén region, Chile (Northern Patagonia).
Banerjee O, Crossman ND, De Groot R. 2013. Ecological processes, functions and ecosystem services. In: Wratten S, Sandhu H, Cullen R, Costanza R, editors. Ecosystem services in agricultural and urban landscapes. 1st ed. Chichester: John Wiley & Sons, Ltd. p. 218.
Barrios E. 2007. Soil biota, ecosystem services and land productivity. Ecol Econ. doi:10.1016/j.ecolecon.2007.03.004.
Baulcombe D. 2009. Reaping the benefits: science and the sustainable intensification of global agriculture. London: The Royal Society.
Baulcombe D, Crute I, Davies B, Dunwell J, Gale M, Jones J, Pretty J, Sutherland W, Toulmin C. 2009. Reaping the benefits: science and the sustainable intensification of global agriculture. London: The Royal Society. Report.
Bayerische Landesanstalt für Landwirtschaft. 2016. Fruchtfolgen im ökologischen Landbau. https://www.lfl.bayern.de/schwerpunkte/oekolandbau/106961/index.php.
Bicking S, Burkhard B, Kruse M, Müller F. 2018. Mapping of nutrient regulating ecosystem service supply and demand on different scales in Schleswig-Holstein, Germany. One Ecosyst. 3:e22509. doi:10.3897/oneeco.3.e22509
Bicking S, Burkhard B, Müller F. 2019. Bayesian belief network-based assessment of nutrient regulating ecosystem services in Northern Germany. PLoS One. doi:10.1371/journal.pone.0216053.
Brockerhoff EG, Barbaro L, Castagneyrol B, Forrester DI, Gardiner B, Lyver POB, Meurisse N, Oxbrough A. 2017. Forest biodiversity, ecosystem functioning and the provision of ecosystem services. Biodivers Conserv. 3005–3035. doi:10.1007/s10531-017-1453-2.
Bruns HA. 2012. Concepts in crop rotations. In: Aflakpui G, editor. Agricultural science. Rijeka: InTech; p. 252.
Bundesamt für Naturschutz. 2012. Landschaftssteckbrief - 36401 Lipper Bergland. [accessed 2019 Feb 15]. https://www.bfn.de/landschaften/steckbriefe/landschaft/show/36401.html.
Bundesanstalt für Landwirtschaft und Ernährung. 2018. Die neue Düngeverordnung. Ostbevern.
Bundesministerium für Ernährung und Landwirtschaft. 2019. Düngung. [accessed 2020 Jan 1]. https://www.bmel.de/DE/Landwirtschaft/Pflanzenbau/Ackerbau/_Texte/Duengung.html.
Burkhard B. 2017. Integrative approaches. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. 374.
Burkhard B, De Groot R, Costanza R, Seppelt R, Jørgensen SE, Potschin M. 2012a. Solutions for sustaining natural capital and ecosystem services. Ecol Indic. 21:1–6. doi:10.1016/j.ecolind.2012.03.008
Burkhard B, Kandziora M, Hou Y, Müller F. 2014. Ecosystem service potentials, flows and demands-concepts for spatial localisation, indication and quantification. Landscape Online. 34:1–32. doi:10.3097/LO.201434
Burkhard B, Kroll F, Nedkov S, Müller F. 2012b. Mapping ecosystem service supply, demand and budgets. Ecol Indic. 21:17–29. doi:10.1016/j.ecolind.2011.06.019
Burkhard B, Maes J. 2017. Mapping ecosystem services. Sofia: Pensoft Publishers.
Busari MA, Kukal SS, Kaur A, Bhatt R, Dulazi AA. 2015. Conservation tillage impacts on soil, crop and the environment. Int Soil Water Conserv Res. doi:10.1016/j.iswcr.2015.05.002.
Chapin FS, Matson PA, Mooney HA. 2002. Principles of terrestrial ecosystem ecology. New-York (USA): Springer Verlag.
Chislock MF, Doster E, Zitomer RA, Wilson AE. 2013. Eutrophication: causes, consequences, and conttrols in aquatic ecosystems. Nat Educ. doi:10.1017/S0958344007000237.
de Bello F, Lavorel S, Díaz S, Harrington R, Cornelissen JHC, Bardgett RD, Berg MP, Cipriotti P, Feld CK, Hering D, et al. 2010. Towards an assessment of multiple ecosystem processes and services via functional traits. Biodivers Conserv. doi:10.1007/s10531-010-9850-9.
Deutscher Wetterdienst. 2019. Climate data center. [accessed 2019 Feb 15]. https://www.dwd.de/DE/leistungen/cdcftp/cdcftp.html?nn=17626.
Dise NB, Ashmore M, Belyazid S, Bleeker A, Bobbink R, de Vries W, Erisman JW, Spranger T, Stevens CJ, van den Berg L. 2011. Nitrogen as a threat to European terrestrial biodiversity. In: Sutton MA, Howard CM, Erisman JW, Billen G, Bleeker A, Grennfelt P, van Grinsven H, Grizzetti B, editors. The European nitrogen assessment. Cambridge: Cambridge University Press.
Dominati EJ. 2013. Natural capital and ecosystem services of soils. Ecosyst Serv N Z. 132–142. doi:10.1007/s00787-005-0495-2.
Dunford RW, Harrison PA, Bagstad KJ. 2017. Computer modelling for ecosystem service assessment. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. p 374.
Düngegesetz. 2017. Düngegesetz vom 9. Januar 2009 (BGBl. I S. 54, 136), das zuletzt durch Artikel 1 des Gesetzes vom 5. Mai 2017 (BGBl. I S. 1068) geändert worden ist
DüV. 2017. Verordnung über die Anwendung von Kultursubstraten und Pflanzenhilfsmitteln nach den Grundsätzen der guten fachlichen Praxis beim Düngen (Düngeverordnung - DüV).
DWD Climate Data Center. 2018a. REGNIE-Raster der täglichen Niederschlagshöhe für Deutschland.
DWD Climate Data Center. 2018b. Tägliche Raster der potentiellen Evapotranspiration über Gras: version 0.x.
Erhard M, Banko G, Malak DA, Martin FS. 2017. Mapping ecosystem types and conditions. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. 374.

Erisman JW, Galloway JN, Seitzinger S, Bleeker A, Dise NB, Roxana Petrescu AM, Leach AM, de Vries W. 2013. Consequences of human modification of the global nitrogen cycle. Philos Trans R Soc B. doi:10.1098/rstb.2013.0116.
European Commission. 2019. The common agricultural policy at a glance. https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/cap-glance_en.
European Commission – DG ENV. 2010. Environmental impacts of different crop rotations in the European Union. Contract N° 07.0307/2009/SI2.541589/ETU/B1 - Final Report.
European Commission Agri-environment measures. 2005. https://ec.europa.eu/agriculture/envir/measures_en.
Eurostat. 2013. Methodology and handbook eurostat/OECD nutrient budgets.
Follett JR, Follett RF. 2001. Chapter 4 - utilization and metabolism of nitrogen by humans. In: Follett RF, Hatfield JL, editors. Nitrogen in the environment: sources, problems and management. Amsterdam: Elsevier Science; p. 65–92.
Frazier TG, Thompson CM, Dezzani RJ. 2014. A framework for the development of the SERV model: a spatially explicit resilience-vulnerability model. Appl Geogr. 51:158–172. doi:10.1016/j.apgeog.2014.04.004
Fu B, Wang YK, Xu P, Yan K. 2012. Modelling nutrient retention function of ecosystem – a case study in Baoxing County, China. Procedia Environ Sci. 13:111–121. doi:10.1016/j.proenv.2012.01.011
Füssel H-M. 2007. Vulnerability: A generally applicable conceptual framework for climate change research. Global Environ Change. 17:155–167. doi:10.1016/j.gloenvcha.2006.05.002
Galloway JN, Dentener FJ, Capone DG, Boyer EW, Howarth RW, Seitzinger SP, Asner GP, Cleveland CC, Green PA, Holland EA, et al. 2004. Nitrogen cycles: past, present, and future. Biogeochemistry. doi:10.1007/s10533-004-0370-0.
Geologischer Dienst NRW. 2016a. Geologie und Boden in Nordrhein-Westfalen.
Geologischer Dienst NRW. 2016b. Boden in Nordrhein-Westfalen. https://www.gd.nrw.de/zip/broschuer_boden.pdf.
Geologischer Dienst NRW. 2017a. Denitrifikationspotenzial. Krefeld.
Geologischer Dienst NRW. 2017b. IS BK 50 Bodenkarte von NRW 1 : 50.000. https://open.nrw/dataset/ab0f265c-dabe-4bcd-869a-d97dc328c141bkg.
Ghaley BB, Porter JR, Sandhu HS. 2014. Soil-based ecosystem services: A synthesis of nutrient cycling and carbon sequestration assessment methods. Int J Biodivers Sci Ecosyst Serv Manage. 10:177–186. doi:10.1080/21513732.2014.926990
Gupta V, Rovira A, Roget D. 2010. Principles and management of soil biological factors for sustainable rainfed farming systems. In: Tow P, Cooper I, Partridge I, Birch C, editors. Rainfed farming systems. Berlin: Springer. p. 1336.
Gutierrez-Arellano C, Mulligan M. 2018. A review of regulation ecosystem services and disservices from faunal populations and potential impacts of agriculturalisation on their provision, globally. Nat Conserv. 30:1–39. doi:10.3897/natureconservation.30.26989
Hagen-Zanker A. 2006. Comparing continuous valued raster data: a cross disciplinary literature.
Haines-Young R, Potschin M. 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli DG, Frid CLJ, editors. Ecosystem ecology: a new synthesis. Cambridge University Press. p. 174.
Haines-Young R, Potschin M. 2017. Categorisation systems: the classification challenge. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. p 374.
Haines-Young R, Potschin MB. 2018. Common international classification of ecosystem services (CICES) V5.1 and guidance on the application of the revised structure. Eur Environ Agency. 53:19.
Heidecke C, Kreins P, Wendland F, Keller L, Kuhr P, Jülich BT, Heidecke C, Kreins P, Wagner A, Trepel M. 2014. Stickstoffeinträge ins Grundwasser und die Räumlich differenzierte Quantifizierung der Stickstoffeinträge ins Grundwasser und die Oberflächen- gewässer Schleswig-Holsteins. doi:10.3243/kwe2014.06.001.
Hou Y, Burkhard B, Müller F. 2013. Uncertainties in landscape analysis and ecosystem service assessment. J Environ Manage. 127:S117–S131. doi:10.1016/J.JENVMAN.2012.12.002
Jacobs A, Rauber R, Ludwig B. 2009. Impact of reduced tillage on carbon and nitrogen storage of two Haplic Luvisols after 40 years. Soil Tillage Res. doi:10.1016/j.still.2008.08.012.
Jie Y, Haijin Z, Xiaoan C, Le S. 2013. Effects of tillage practices on nutrient loss and soybean growth in red soil slope farmland. Int Soil Water Conserv Res. 1:49–55.
Jónsson JÖG, Davídsdóttir B. 2016. Classification and valuation of soil ecosystem services. Agric Syst. 145:24–38. doi:10.1016/j.agsy.2016.02.010
Jónsson JÖG, Davíðsdóttir B, Nikolaidis NP. 2016. Valuation of soil ecosystem services. Adv Agron. 142:353–384. doi:10.1016/bs.agron.2016.10.011
Kandziora M, Burkhard B, Müller F. 2013. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators: A theoretical matrix exercise. Ecol Indic. 28:54–78. doi:10.1016/j.ecolind.2012.09.006
Klement L, Bach M. 2017. Darstellung hochaufgelöster Stickstoff-Überschüsse aus der Landwirtschaft. Zeitschrift für amtliche Statistik Berlin Brandenburg. 2:50–51.
Kuhn T. 2017. The revision of the German fertiliser ordinance in 2017. Food and Resource Economics, Discussion Paper.
Landwirtschaftliche Untersuchungs- und Forschungsanstalt. 2018. Prüfbericht.
Landwirtschaftskammer Schleswig-Holstein. 2019. Diskussion Düngeverordnung 2020. [accessed 2020 Jan 1]. https://www.lksh.de/landwirtschaft/duengung/diskussion-duengeverordnung-2020/.
Leip A, Billen G, Garnier J, Grizzetti B, Lassaletta L, Reis S, Simpson D, Sutton MA, De Vries W, Weiss F, et al. 2015. Impacts of European livestock production: nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. Environ Res Lett. doi:10.1088/1748-9326/10/11/115004.
Leip A, Britz W, Weiss F, de Vries W. 2011. Farm, land, and soil nitrogen budgets for agriculture in Europe calculated with CAPRI. Environ Pollut. 159:3243–3253. doi:10.1016/j.envpol.2011.01.040
Loges R. 2013. Leguminosen im Futterbau: aktuelle und zukünftige Bedeutung sowie Forschungsbedarf Einleitung und Problemstellung. Lfl. p. 9–20.
Loges R, Kornher A, Taube F. 1998. Ertrag, Futterqualität und N2 -Fixierungsleistung von Rotklee und Rotklee/Gras. Mitteilungen der Gesellschaft für Pflanzenbauwissenschaften. p. 139–142
Ma L, Bicking S, Müller F. 2019. Mapping and comparing ecosystem service indicators of global climate regulation in Schleswig-Holstein, Northern Germany. Sci Total Environ. 648:1582–1597. doi:10.1016/j.scitotenv.2018.08.274

Maes J. 2017. Specific challenges of mapping ecosystem services. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. p 374.
Maes J, Egoh B, Willemen L, Liquete C, Vihervaara P, Schägner JP, Grizzetti B, Drakou EG, La Notte A, Zulian G, et al. 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst Serv. 1:31–39. doi:10.1016/j.ecoser.2012.06.004
Maes J, Teller A, Erhard M, Grizzetti B, Barredo J, Paracchini M, Condé S, Somma F, Orgiazzi A, Jones A, et al. 2018. Mapping and assessment of ecosystems and their services: an analytical framework for mapping and assessment of ecosystem condition in EU.
Maes J, Teller A, Erhard M, Murphy P, Paracchini ML, Barredo JI, Grizzetti B, Cardoso A, Somma F, Petersen J-E, et al. 2014. Mapping and assessment of ecosystems and their services - indicators for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020.
Millenium Ecosystem Assessment. 2005. Ecosystems and human well-being: synthesis. Washington (DC): Island Press.
Özbek FŞ, Leip A. 2015. Agriculture, ecosystems and environment estimating the gross nitrogen budget under soil nitrogen stock changes : A case study for Turkey. Agr Ecosyst Environ. 205:48–56. DOI:10.1016/j.agee.2015.03.008.
Müller F, Burkhard B. 2012. The indicator side of ecosystem services. Ecosyst Serv. 1:26–30. doi:10.1016/j.ecoser.2012.06.001
Müller F, Kroll F. 2011. Integrating ecosystem theories - gradients and orientors as outcomes of self-organized processes. Int J Des Nat Ecodyn. 6:318–341. doi:10.2495/DNE-V6-N4-318-341
Müller U, Waldeck A. 2011. Auswertungsmethoden im Bodenschutz Niedersachsen. Hannover: Landesamt für Bergbau, Energie und Geologie.
PAN Germany. 2010. Crop rotation. [accessed 2019 Jan 15]. http://www.oisat.org/control_methods/cultural__practices/crop_rotation.html.
Power AG. 2010. Ecosystem services and agriculture: tradeoffs and synergies. Philos Trans R Soc B. 365:2959–2971. doi:10.1098/rstb.2010.0143
Sächsische Landesanstalt für Landwirtschaft. 2007. Umsetzung der Düngeverordnung - Hinweise und Richtwerte für die Praxis.
Sandhu HS, Wratten SD, Cullen R. 2010. The role of supporting ecosystem services in conventional and organic arable farmland. Ecol Complexity. 7:302–310. doi:10.1016/j.ecocom.2010.04.006
Schneiders A, Müller F. 2017. A natural base for ecosystem services. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. p 374.
Schröter M, Barton DN, Remme RP, Hein L. 2014. Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway. Ecol Indic. 36:539–551. doi:10.1016/j.ecolind.2013.09.018
Schulp CJE, Burkhard B, Maes J, Van Vliet J, Verburg PH. 2014. Uncertainties in ecosystem service maps: A comparison on the European scale. PLoS One. 9. doi:10.1371/journal.pone.0109643.
Selman M, Greenhalgh S. 2009. Eutrophication: policies, actions, and strategies to address nutrient pollution.
Smith P, Ashmore MR, Black HIJ, Burgess PJ, Evans CD, Quine TA, Thomson AM, Hicks K, Orr HG. 2013. REVIEW: the role of ecosystems and their management in regulating climate, and soil, water and air quality. J Appl Ecol. 50:812–829. doi:10.1111/1365-2664.12016
Smith R, Dick J, Trench H, van Oijen M. 2012. Extending a Bayesian belief network for ecosystem evaluation. Conference of the Human Dimensions of Global Environmental Change on Evidence for Sustainable Development. p. 12.
Smith SR, Hadley P. 1989. A comparison of organic and inorganic nitrogen fertilizers: their nitrate-N and ammonium-N release characteristics and effects on the growth response of lettuce (Lactuca sativa L. cv. Fortune). Plant Soil. doi:10.1007/BF02220704.
Sohel MSI, Ahmed Mukul S, Burkhard B. 2015. Landscape's capacities to supply ecosystem services in Bangladesh: A mapping assessment for Lawachara National Park. Ecosyst Serv. 12:128–135. doi:10.1016/j.ecoser.2014.11.015
Stoll S, Frenzel M, Burkhard B, Adamescu M, Augustaitis A, Baeßler C, Bonet FJ, Carranza ML, Cazacu C, Cosor GL, et al. 2015. Assessment of ecosystem integrity and service gradients across Europe using the LTER Europe network. Ecol Modell. 295:75–87. doi:10.1016/j.ecolmodel.2014.06.019
Sutton MA, Bleeker A, Howard CM, Bekunda M, Grizzetti B, de Vries W, Van Grinsven HJM, Abrol YP, Adhya TK, Billen G, et al. 2013. Our nutrient world.
Sutton MA, Howard CM, Erisman JW, Billen G, Bleeker A, Grennfelt P, van Grinsven HGrizzetti B. 2011a. The European nitrogen assessment: Sources, effects and policy perspectives. Cambridge: Cambridge University Press.
Sutton MA, Oenema O, Erisman JW, Leip A, van Grinsven H, Winiwarter W. 2011b. Too much of a good thing. Nature. 472:159–161. doi:10.1038/472159a
Syrbe RU, Schröter M, Grunewald K, Walz U, Burkhard B. 2017. What to map? In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. 374.
Taube F. 2018. Expertise zur Bewertung des neuen Düngerechts (DüG, DüV, StoffBilV) von 2017 in Deutschland im Hinblick auf den Gewässerschutz. p. 25
Taube F, Henning C, Albrecht E, Kluß TRC, Taube F, Henning C, Albrecht E, Kluß TRC. 2015. Nährstoffbericht des Landes Schleswig-Holstein. MELUR - Ministerium für Energiewende, Landwirtschaft, Umwelt und ländliche Räume Schleswig-Holstein. doi:10.13140/RG.2.1.2965.0005.
The European Council. 1991. Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources.
The European Council. 2005. COUNCIL REGULATION (EC) No 1698/2005 of 20 September 2005 on support for rural development by the European Agricultural Fund for Rural Development (EAFRD).
The European Council. 2007. Council regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91.
Tivy J. 1987. Nutrient cycling in agro-ecosystems. Appl Geogr. doi:10.1016/0143-6228(87)90044-0.
Townsend AR, Howarth RW, Bazzaz FA, Booth MS, Cleveland CC, Collinge SK, Dobson AP, Epstein PR, Holland EA, Keeney DR, et al. 2003. Human health effects of a changing global nitrogen cycle. Front Ecol Environ. doi:10.1890/1540-9295(2003)001[0240:HHEOAC]2.0.CO;2.
Truner R, Van Den Bergh J, Söderqvist T, Barengregt A, Van der Straaten J, Maltby E, Van Ierland E. 2000. Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecol Econ. 35:8074–8079.
Turner II BL, Kasperson RE, Matson PA, Mccarthy JJ, Corell RW, Christensen L, Eckley N, Kasperson JX,

Luers A, Martello ML, et al. 2003. A framework for vulnerability analysis in sustainability science. PNAS:100:8074–8079.
UIH Ingenieur- und Planungsbüro. 2016. Bewertung des Schutzgutes "Landschaftsbild und Landschaftserleben" im Kreis Höxter.
Umweltbundesamt. 2018a. Indicator: nitrate in groundwater. https://www.umweltbundesamt.de/en/indicator-nitrate-in-groundwater#textpart-1.
Umweltbundesamt. 2018b. FAQs zu Nitrat im Grund- und Trinkwasser Was ist der Unterschied zwischen Trinkwasser. [accessed 2019 Apr 9]. https://www.umweltbundesamt.de/faqs-zu-nitrat-im-grund-trinkwasser#textpart-11.
UNEP. 2008. UNEP ECOSYSTEM MANAGEMENT PROGRAMME - an ecosystem approach.
VALERIE. 2017. Crop rotation, soil cover management and integrated pest management. [accessed 2019 Apr 10]. http://www.valerie.eu/index.php/themes/crop-rotation-soil-cover-management-and-integrated-pest-management.
Villamagna AM, Angermeier PL, Bennett EM. 2013. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecol Complexity. 15:114–121. doi:10.1016/j.ecocom.2013.07.004
Vitousek PM, Hättenschwiler S, Olander L, Allison S. 2002. Nitrogen and nature. Ambio. 31:97–101. doi:10.1579/0044-7447-31.2.97
Vitousek PM, Mooney HA, Lubchenco J, Melillo JM, Series N, Jul N. 1997. Human domination of earth's ecosystems peter. Science. 277:494–499. doi:10.1126/science.277.5325.494
Vitousek PM, Naylor R, Crews T, David MB, Drinkwater LE, Holland E, Johnes PJ, Katzenberger J, Martinelli LA, Matson PA, et al. 2009. Nutrient imbalances in agricultural development. Science. 324:1519–1520. doi:10.1126/science.1170261
von Ruschkowski A. 2018. Farm record system 2008–2018.
von Zezschwitz E. 2001. Waldböden des Lipper Berglandes. In: Landschaftsverband Westfalen-Lippe. Geologie und Paläontologie in Westfalen. Münster: LINDEN Print & Media GmbH.
Weißhuhn P, Müller F, Wiggering H. 2018. Ecosystem vulnerability review : proposal of an interdisciplinary ecosystem assessment approach. Environ Manage. doi:10.1007/s00267-018-1023-8.
Welte E, Timmermann F. 1985. Düngung und Umwelt. Materialien zur Umweltforschung herausgegeben vom Rat von Sachverständigen für Umweltfragen. Stuttgart (Mainz): Verlag W. Kohlhammer GmbH.
Wendland F, Keller L, Kuhr P, Kunkel R. 2015. Regional differenzierte Quantifizierung der Nährstoffeinträge in das Grundwasser und in die Oberflächengewässer Mecklenburg-Vorpommerns unter Anwendung der Modellkombination GROWA: endbericht.
Wilcke W, Boy J, Hamer U, Potthast K, Rollenbeck R, Valarezo C. 2013. Current regulating and supporting services : nutrient cycles. Part of the Ecological Studies book series (ECOLSTUD, volume 221).
Willemen L, Jones S, Carmona NE, DeClerck F. 2017. Ecosystem service maps in agriculture. In: Burkhard B, Maes J, editors. Mapping ecosystem services. Sofia: Pensoft Publishers; p. 374.
Wisner B, Blaikie P, Cannon T, Davis I. 2004. At risk : natural hazards, people's vulnerability and disasters. 2nd ed. London: Routledge.
Zurlini G, Amadio V, Rossi O. 1999. A landscape approach to biodiversity and biological health planning: the map of italian nature. Ecosyst Health. 5:294–311. doi:10.1046/j.1526-0992.1999.09948.x

Appendices

Appendix A. Crop type specific nutrient profiles on the field scale. Diverse data sources are specified in Table 2

	Wheat		Barley		Triticale		Silage maize	
	Former system	Current system	Former system	Current system	Former system	Current system	Former system	Current system
Average harvest (dt/ha)	90		80		80		500	
Residuals remaining on the field (%)	30		30		30		[-]	
Average N-content in plant biomass (kg N/dt FM)								
In grain	1.81		1.65		1.65		0.38*[a]	
In straw	0.5		0.5		0.5		[-]	
Ratio primary to secondary product (1:X)	0.8		0.7		0.9		[-]	
Nitrogen input (kg N/ha)								
Agricultural calculation	164	173.4	154	141	162	157	84	107
Ecological calculation	210.2	211.6	194.2	214.2	208.2	195.2	168.2	202
Nitrogen output (kg N/ha)								
In harvest of grain	162.9		132		132		[-]	
In harvest of straw	25.2		19.6		25.2		[-]	
In total harvest	188.1		151.6		157.2		190	
Nitrogen budget (kg N/ha)								
Agricultural calculation	−24.1	14.7	2.4	−10.6	4.8	−0.2	−106	−83
Ecological calculation	22.1	23.5	42.6	62.6	51	38	−21.8	12

	Rapeseed		Field bean		Grass/clover	
	Former system	Current system	Former system	Current system	Former system	Current system
Average harvest (dt/ha)	40		45		80 (DM)	
Residuals remaining on the field (%)	30		30		[-]	
Average N-content in plant biomass (kg N/dt FM)						
In grain	3.35		4.1		0.5*[a]	
In straw	0.7		1.5			
Ratio primary to secondary product (1:X)	1.7		1		0.2*[b]	
Nitrogen input (kg N/ha)						
Agricultural calculation	200.56	172.8	197.5	197.5	[-]	234

(Continued)

(Continued).

	Rapeseed		Field bean		Grass/clover	
	Former system	Current system	Former system	Current system	Former system	Current system
Ecological calculation	270.9	254.1	212.7	212.7	[-]	248.3
Nitrogen output (kg N/ha)						
In harvest of grain	134		184.5		[-]	
In harvest of straw	33.3		31.5		[-]	
In total harvest	167.3		216		232	
Nitrogen budget (kg N/ha)						
Agricultural calculation	33.24	5.48	−18.5	−18.5	[-]	2
Ecological calculation	103.58	86.78	−3.3	−3.3	[-]	16.3

[a]whole plant.
[b]dry mass per fresh mass.

Appendix B. Classification scheme of nutrient regulation ES demand, nitrate leaching and denitrification potential as input data for the assessment of the relative vulnerability and calculated vulnerability values/index and respective relative vulnerability classes

Nutrient regulation ES demand		Nitrate leaching potential		Denitrification potential		Relative vulnerability	
In kg N/(ha*a)	Reclassified	In %/a	Reclassified	In kg NO_3-N/(ha*a)	Reclassified	Calculated index	Reclassified
< 20	1	< 80	1	< 10	1	< 3.4	Very low
20 to < 30	2	80 to < 100	2	10 to < 30	2	3.4 to < 4.8	Low
30 to < 40	3	100 to < 120	3	30 to < 50	3	4.8 to < 6.2	Moderate
40 to < 50	4	120 to < 140	4	50 to < 150	4	6.2 to < 7.6	High
≥ 50	5	≥ 140	5	≥ 150	5	≥ 7.6	Very high