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Socio-hydrogeological assessment of EU Nitrates Directive application in the Lombardy plain (Italy)

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List of acronyms

AET	Actual evapotranspiration
ARPA	Regional Agency for Environmental Protection
CH	Constant head
DA	Deeper aquifer
EU	European
GHB	General head boundary
IA	Intermediate aquifer
I_{rr}	Irrigation from surface water
K	Hydraulic conductivity
K_{crop}	Crop coefficient
LHU	Lower Hydrogeological Unit
n	Porosity
N	Nitrogen
ND	Nitrates Directive
N_{Den}	Denitrification
N_{Dep}	Atmospheric N deposition
N_{Fert}	Synthetic N fertilizers applied to agricultural lands
N_{Fix}	Agricultural N_2 fixation associated with N-fixing crops
N_{Harv}	N exported by crop harvesting
N_{Man}	N in livestock manure
nNVZs	non-Nitrate Vulnerable Zones
NO_3^-	Nitrate
NSE	Nash-Sutcliffe coefficient of efficiency
$N_{surplus}$	Nitrogen surplus on agricultural lands
N_{Vol}	NH_3 volatilization
NVZs	Nitrate Vulnerable Zones
P	monthly mean precipitation
PET	Potential evapotranspiration
pNVZs	partially Nitrate Vulnerable Zones
Q_{ag}	Agricultural water withdrawal
R	Rainfall
R^2	Coefficient of determination
RMSE	Root–mean-square error
S	Storage coefficients
SA	Shallow aquifer
SSB	Soil System Budget
S_y	Specific yield
UHU	Upper Hydrogeological Unit
w	Land use coefficient

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SUMMARY

After almost 30 years since the European Nitrates Directive (ND) was enacted, nitrates contamination of groundwater resources is still relevant and poses serious threats to human health and aquatic ecosystems. Unrealized remediation expectations may generate growing discontent and frustration among the actors involved, undermining the legitimacy of ND itself. Understanding the causes of this delay is therefore a crucial task from an environmental, management and political perspective.

The delay between the ND application and the observation of measurable results in groundwater is rooted in both socio-economic and hydrogeological aspects. Nevertheless, multidisciplinary assessments considering both struggle to be implemented.

The overall aim of this thesis is to assess the effectiveness of ND in the Lombardy plain (northern Italy) and the reasons behind its successes and failures, by adopting a socio-hydrogeological approach, i.e. by integrating an hydrogeological assessment with the investigation of socio-relational dynamics affecting the ND implementation.

To this aim, the effectiveness of ND application in the Lombardy plain is investigated firstly by conducting a contamination trend analysis aimed to uncover possible critical situations, and secondly by building up a hydrogeological flow and transport model for a representative sector of the plain. Environmental criticalities of ND are subsequently evaluated in the light of the analysis of the governance process supporting its application. Governance components (i.e. the stakeholders involved, their relationships and emerging social dynamics) strengthening or hindering the ND implementation, are identified by the means of a quali-quantitative social network analysis approach.

Previous analysis of nitrate contamination, conducted consistently with the ND requirements, pointed out an overall stability on nitrate concentrations across time, suggesting a rather positive success as regards the prevention of further contamination. Differently, the assessment carried out in this thesis points out that relevant groundwater resources and already impaired waters present critical increasing trends. More attention should be paid to significant increasing of concentrations affecting supposedly resistant and resilient aquifers, as they can highlight the weakening of the system's ability to self-recover. By providing a comprehensive understanding of both the entire system evolution and the components of the hydrogeological system actually governing the overall nitrate mass balance, the hydrogeological model permits to detect a slow and constant increase of the nitrate amount stored in the aquifers. Moreover, the resulting estimation in terms of absolute amounts of the nitrogen stored and moving in the aquifers permits to account for the groundwater role in nitrogen mass balance computations and finally connect the dynamics of contamination occurring in ground and surface water at the catchment scale. As a result, beyond the poor effectiveness, wider considerations on the current assessment process of the ND, its tools and procedures, emerge.

According to the analysis of governance, the improvement of ND implementation is not only a management issue. At present, the possibilities of ND success are strongly limited by socio-relational factors, among which the overall lack of perceived transparency in governance process, the weak knowledge-transfer process supporting the ND application and the control-based strategy on which the Directive relies on. Without overcoming these obstacles, any change in legislation or management actions may be useless.
The socio-hydrogeological approach is confirmed a powerful tool to achieve a broader view on the reasons behind the poor effectiveness of enforced regulations on groundwater protection and, consequently, to increase the chances to accomplish the ND goals. His effectiveness is here proven also when the environmental characteristics and dynamics of the system are quite well known, as in the case of nitrate contamination in the Lombardy plain.

RESUM

Després de gairebé 30 anys de la promulgació de la Directiva Europea de nitrats (DN), a contaminació per nitrats de les aigües subterrànies encara representa una greu amenaça tan per la salut humana com pels ecosistemes aquàtics. El no assoliment de les expectatives de remediació pot agreujar el descontentament i la frustració dels actors implicats, perjudicant la legitimitat de la DN. Per tant, comprendre les causes d'aquest retard és una tasca clau d'una perspectiva ambiental, de gestió i política. Un cop iniciada l'aplicació de la DN, l'endarreriment en l'observació de resultats mesurables en aigües subterrànies té el seu origen en aspectes socioeconòmics i hidrogeològics. No obstant, avaluacions multidisciplinàries tenint en compte ambdós aspectes han rebut poca atenció.

L'objectiu general d'aquesta tesi és avaluar l'efectivitat de la DN a la plana de la Lombardia (nord d'Itàlia) i els motius dels seus punts forts i febles, mitjançant un enfocament socio-hidrogeològic. És a dir, integrant l'avaluació hidrogeològica amb la investigació de les dinàmiques socio-relacionals que influencien la implementació de la DN.

Per tal d'assolir aquest objectiu, l'eficàcia de l'aplicació de la DN a la plana lombarda ha estat avaluada en primer lloc analitzant les tendències de contaminació, permetent detectar possibles situacions crítiques, i, posteriorment mitjançant un model de flux i transport hidrogeològic en una àrea representativa de la plana. Posteriorment, s'han avaluat les criticitats ambientals de la DN considerant el procés de governança que dóna suport a la seva aplicació. Els components de la governança (és a dir, els grups implicats, les seves relacions i les dinàmiques socials que se'n originen) que reforcen o dificulten la implementació de la DN s'identifiquen mitjançant un anàlisi qualitatiu i quantitatiu de les estructures socials.

Estudis realitzats segons els requeriments de la DN anteriors a aquesta tesi, han assenyalat una estabilitat general de les concentracions de nitrats al llarg del temps, fet que suggereix un èxit més aviat positiu de la prevenció de l'augment de la contaminació. Contràriament, l'avaluació realitzada en aquesta tesi assenyala que els recursos d'aigua subterrània més rellevants i les aigües deteriorades amb anterioritat presenten tendències creixents de contaminació. Aquests resultats evidencien que una major atenció s'ha d'efectuar en l'augment significatiu de concentracions en aqüífers considerats resistents i resilients a la contaminació, ja que poden mostrar el debilitament de la capacitat d'auto-recuperació del sistema.

El model hidrogeològic, proporciona una comprensió completa tan de l'evolució del sistema com dels components que governen el balanç de masses de nitrogen, permetent, d'aquesta manera, detectar un augment lent i constant de la quantitat de contaminant emmagatzemat en els aqüífers. D'altra banda, l'estimació de les quantitats absolutes de nitrogen emmagatzemades i desplaçades en els aqüífers permet tenir en compte el paper de les aigües subterrànies en el càlcul del balanç de masses del nitrogen. I, finalment, connectar les dinàmiques de contaminació que es produeixen en aigües subterrànies amb les

d’aigües superficials a escala de conca. Com a resultat, més enllà de la baixa efectivitat de la DN, sorgeixen consideracions més àmplies sobre l‘actual procés d'avaluació, les seves eines i procediments.
Segons l’anàlisi de governança, la millora en la implementació de DN no només és un problema de gestió. Actualment, les possibilitats d’èxit de DN estan fortament limitades per factors determinats per les relacions socials, entre els quals: la falta generalitzada de transparència percebuda en el procés de governança, el feble procés de transferència de coneixement que dóna suport a l’aplicació de la DN i, l’estratègia basada en el control en què es basa la Directiva. Sense superar aquests obstacles, qualsevol canvi en la legislació o accions de gestió pot ser estèril.
Finalment, l’enfocament socio-hidrogeològic es confirma com una eina potent per tal d’obtenir una visió àmplia sobre les raons de la baixa efectivitat de les regulacions aplicades sobre protecció de les aigües subterrànies i, conseqüentment, una eina per tal d’augmentar les possibilitats d’assolir els objectius de la DN. Tal i com aquesta tesi mostra, la seva efectivitat també es veu demostrada quan les característiques i les dinàmiques ambientals del sistema són prou conegudes, com en el cas de la contaminació de nitrats a la plana llombarda.

1 INTRODUCTION

1.1 The EU Nitrates Directive

Nitrate contamination is a long-standing environmental challenge in Europe. Since the mid-1970s the maximum concentration limit of 50 mg/L of nitrates was set by European authorities, with the main purpose of protecting water intended for human consumption (EU Commission 1975; EU Commission 1980; Frederiksen 1995; Oenema et al. 2011). Subsequently, in 1991, due to both the increase of nitrate concentrations in several European countries and the rising awareness of the remarkable contribution of nitrates from the agricultural sector, the Nitrates Directive (ND; EU Commission 1991) was promulgated. The ND established two main goals that EU countries must achieve: (i) reducing water pollution caused or induced by nitrates from agricultural sources, and (ii) preventing further such pollution. Accordingly, Member States are required to designate Nitrate Vulnerable Zones (NVZs), namely areas likely to contribute to surface or ground waters contamination to a level at or above 50 mg/L. Within the NVZs, specific mandatory protection measures had to be adopted, by the means of so-called "action programmes", and the maximum limit of 170 kg nitrogen applied per hectare per year from organic manure was established. Furthermore, within the non-vulnerable territory (nNVZs) the Member States had to propose a set of measures to be implemented on a voluntary basis (Monteny 2001), mainly regarding the periods and weather conditions for fertilizers' application. The ND is also one of the Statutory Management Requirements that European farmers are required to respect in order to receive the subsidies provided for the cross-compliance system of the Common Agriculture Policy (EU Commission 2013a; EU Commission 2013b). Individual benefits are reduced proportionally to the noncompliance detected.

At present, the ND is still the main reference in Europe for the protection of water threatened by over-exploitation of agricultural land and the consequent nitrate contamination. Nevertheless, while nitrate concentrations have steadily declined in freshwaters, the efforts required to the agricultural sector have so far delivered poor results in groundwater bodies: about half of the European monitoring stations shows no significant change in nitrate concentrations and 26% of them presents increasing trends (EU Commission 2018). Average concentrations in aquifers almost returned to the values of 1992, revealing that efforts are still required to restore groundwater quality across Europe (EEA 2015). Nitrates are still the main pollutants of European groundwater resources (EEA 2018) and agriculture, responsible for more than 50% of total nitrogen reaching the aquifers, their primary source (EU 2010).

1.2 Nitrates Directive effectiveness: the keys to success

Factors influencing the achievement of appreciable decrements in groundwater nitrate concentrations can be grouped in two main categories.

On one side, the inherent characteristics of both unsaturated and saturated zones play a key role in the long-term impact of nitrates in the aquifers (Vero et al. 2018). Grain size, redox conditions, groundwater residence time, permeability, clay or organic content, thickness of unsaturated zone, microbial communities, among others, can certainly influence the retention or removal of nitrates (e.g. Acutis et al. 2000; Tesoriero and Puckett 2011; Sacchi et al. 2013; Mas-Pla and Menció 2019). On the other side, social, economic and political contexts can strongly control the time required for nitrates reduction in aquifers (Davidson et al. 2015; Davidson et al. 2016; Van Grinsven et al. 2016). The societal demand for groundwater quality, the economic wealth and the environmental awareness can be main driving forces of nitrate concentrations decrease (Hansen et al. 2017), as well as the wide set of relational, cultural, demographic and ethical issues influencing decision-making by farmers (Blackstock et al. 2010; McGuire et al. 2013; Compagnone and Hellec 2015). Legitimacy, participation, proactive and sustained engagement, among others, have been also identified as key factors to guarantee the effective and rapid achievement of groundwater quality objectives (Barthel et al. 2017; Inman et al. 2018).

In the recent literature, the complexity of the multiple social, economic and political factors, acting synergistically at different scales, is enclosed in the "groundwater governance" concept. Groundwater governance is defined as "the overarching framework of groundwater use laws, regulations, and customs, as well as the processes of engaging the public sector, the private sector, and civil society that shapes how groundwater resources are managed and how aquifers are used" (Megdal et al. 2015). Differently from management, which indicates specific day-by-day practical actions implemented by a limited number of stakeholders to ensure groundwater protection, governance includes a wider range of actors (individuals, organizations, agencies, administrations etc.) who act, consciously or not, based on multiple objectives, through more or less direct, formal or informal relationships. The definition of governance suggests its broader nature of both framework and process in which all the dynamics influencing the resource occur (Villholth and Conti 2017).

The role played by governance in the correct implementation of good practices has recently been strongly emphasized (Theesfeld 2010; OECD 2015; Garrick et al. 2017). Foster and Garduno (2013) observe how the failure of groundwater management is often the result of an inadequate governance configuration, which produces inefficiencies in the implementation system, rather than the lack of knowledge related to the vulnerability of aquifers and the hydrogeological dynamics. Although the ND strategy is originally focused on the identification and application of management actions, the same European Commission, in its latest report on the ND implementation, reaffirms the need to strengthen the governance (EU Commission 2018).

1.3 The socio-hydrogeological approach

In light of the varied nature of factors influencing both quality and quantity of groundwater resources, integrated approaches are recognized as the most effective to ensure their long-term sustainability (McDonnell 2008; Akhmouch and Correia 2016). Nevertheless, the implementation of multidisciplinary investigations in hydrogeological science is still far from being common practice (Foster and Ait-Kadi 2012; Re and Misstear 2017) and the increase in the use of social science keywords is not reflected in the way hydrogeological research is carried out (Barthel and Seidl 2017).

To foster the inclusion of the social dimension in hydrogeological investigations, the socio-hydrogeological approach has been recently formalized (Re 2015). Accordingly, socio-economic assessments should be systematically integrated in hydrogeological research, with the ultimate goal of understanding the mutual relations between people and groundwater (i.e., the impact of human activities on groundwater and the impact of scarce and/or polluted groundwater on human wellbeing).

The main aspects of the socio-hydrogeological approach are:

- the assessment of the impact of human activities on groundwater resources;
- the identification of the stakeholders involved in the groundwater issue at stake, their relationships, especially conflicts;
- the evaluation of (socio-economic) impact of groundwater resources (and its changes in terms of both quality and quantity) on human life and well-being;
- the promotion of a better use of the outcomes of a hydrogeological investigation;
- the attempt to bridge the gap between science and society;
- the demystification of science and scientists.

The major implication of this approach is that hydrogeological outcomes are also based on real needs and local knowledge, by strengthening the likelihood of subsequent dissemination activities and, finally, of policies' implementation. Moreover, as stated in the last point, this implies the possibility to enhance the general trust in scientists and, finally, in management action resulting from hydrogeological investigations. The two previous studies that explicitly adopt this new approach were carried out in the rural context of the Grombalia Basin, in Tunisia (Re et al. 2017; Tringali et al. 2017). With the overall aim of identifying relevant actors for groundwater quality protection and creating a momentum for dialogue and capacity building with local stakeholders, hydrogeochemical investigations were coupled with a stakeholder analysis, a social network analysis and structured interviews, Thanks to the socio-hydrogeological approach, new possible pathways for improving the implementation of sustainable practices were identified, as well as the stakeholders practically supporting the dissemination of management actions, based on their role within the social system. But, above all, relevant information to better interpret hydrogeological data were obtained, guaranteeing the correct assessment of contamination sources. Currently, socio-hydrogeological investigations in industrialized contexts are lacking.

1.4 The INTEGRON project

This thesis is developed in the wider framework of the INTEGRON project (Cariplo Foundation, Grant number: 2015-0263) which aims at evaluating the role of groundwater as a temporary or permanent sink or as a source term in nutrient mass balances at the catchment scale in two key sub-basins of Po River (northern Italy), the Adda and the Ticino basins (Figure 1).

Worldwide calculated nutrients' budgets have shown that a high amount of nutrients can be retained or lost within the watershed (Mulholland et al. 2008; Schlesinger 2009). Loads of nitrogen exported at the closing sections, for example, are often equal to 25-40% of the estimated nitrogen excess, and the fate of the missing amount is often unclear (Soana et al. 2011). As regards phosphorus, its dynamics in the watersheds are possibly even less known, despite recent studies pointed out critical phosphate saturation values of soils, expected to contribute to water resources pollution (Jarvie et al. 2013; Hendricks et al. 2014; Schoumans and Chardon 2015). Therefore, groundwater role as a temporary or permanent sink for nutrients has been reconsidered (Böhlke 2002). Soil and aquifers characteristics (e.g. thickness of unsaturated zone, redox conditions, groundwater residence time, permeability, clay or organic content, microbial communities etc.), besides the nutrients input, can play a role in preserving or removing them from groundwater (Acutis et al. 2000; Tesoriero and Puckett 2011).

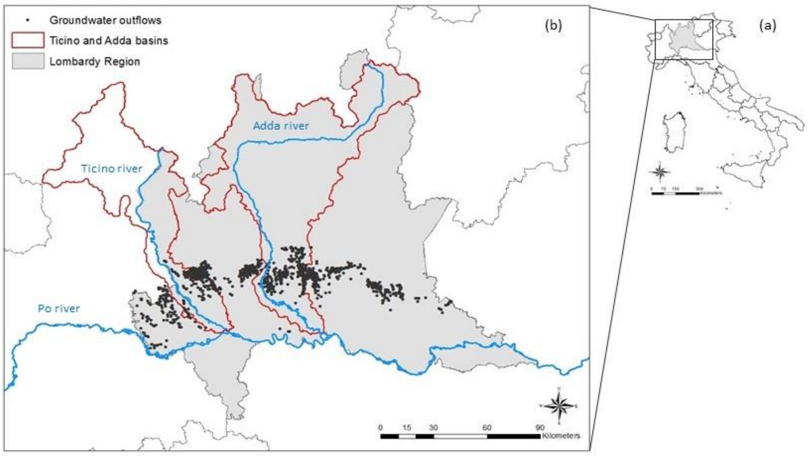

Figure 1.1 (a) Location of the Lombardy Region in Italy; (b) basins (in red) and rivers (in blue) studied in the INTEGRON project. Each black dot represents one of the groundwater outflows (fontanili) forming the so-called springs belt (see par. 3.2 for more detail).

In the two studied basins, the uneven distribution of nitrates in groundwater, not fully matched to any of the urban, industrial and agricultural sources, suggests that local features and processes (e.g. depth of water table, land use, denitrification processes etc.) play a key role in preserving or removing nitrogen from water (Sacchi et al. 2013). Therefore, an innovative and integrated approach was proposed and currently tested, considering both surface and groundwater, combining hydrogeology, biogeochemistry and socio-hydrogeology, and targeting both inorganic nitrogen and phosphates species. In particular, the project synergistically investigated five aspects (Figure 1.2):

- the nutrients mass balance at the catchment scale
- the groundwater contamination and residence time in the aquifers
- the hydrological responses and processes occurring at the outflow
- the environmental factors controlling the N and P dynamics in groundwater
- the stakeholders involved, their relations and possible existing conflicts

Details on the approach and on each task are reported in Musacchio et al. (2019b; Supplementary Materials).

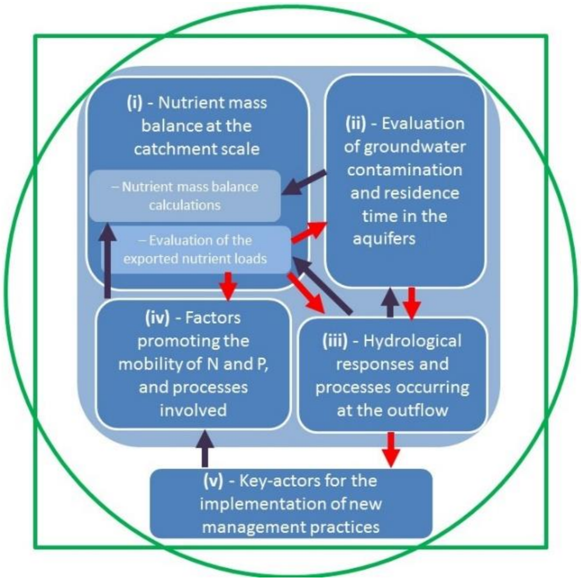

Figure 1.2 Relationships between the different components of the integrated approach. Red arrows: information input; blue arrows: information output and feedback. In green, the logo of the project.

2 THESIS RATIONALE AND RESEARCH GOALS

After almost thirty years since the ND was promulgated, both the reduction of nitrate contamination in groundwater and the prevention of nitrates from agricultural sources, are still far from their full achievement. Unrealized expectations may generate growing discontent and frustration among the actors involved (Meals et al. 2010). Understanding the causes of this delay and determining its duration is therefore a crucial task from an environmental, management and political perspective. Although the general awareness that the delay between the ND application and the observation of measurable results in groundwater is rooted in both hydrogeological and governance aspects, multidisciplinary assessments struggle to be implemented.

The overall aim of this thesis is to assess the effectiveness of ND in the Lombardy plain (northern Italy) and the reasons behind its successes and failures, by adopting a socio-hydrogeological approach. The study area is one of the main European basins in terms of groundwater storage (BGR/UNESCO 2008) but it is also subject to one of the highest nitrate inputs in Europe (Leip et al. 2011; Eurostat 2012). By virtue of the wide range of social aspects potentially influencing the ND, the social assessment addresses directly the overarching governance process.

After a general overview of the study area (Chapter 3), the thesis is organized in two chapters, followed by a general discussion. In Chapter 4 the effectiveness of ND across time is investigated by conducting a contamination trend analysis aimed to uncover possible critical situations. Then, the stakeholders involved in groundwater governance and ND implementation, and the socio-relational dynamics between them, are identified through a social network analysis. In Chapter 5 the major dynamics (both anthropic and natural) controlling nitrate contamination in a portion of the Adda basin, as well as the magnitude of their contribution, are examined by using a three-dimensional flow and transport model. In order to connect hydrogeological dynamics of contamination to social patterns unveiled by the governance investigation, the model is coupled with a nitrogen budget estimation, permitting to estimate the real contribution of agricultural activities to the amount of contaminant reaching the aquifers. Since until now hydrogeological models were mainly used to understand mechanisms influencing nitrate concentrations or to explore alternative management actions, special attention is paid to the possible contribute of hydrogeological modelling to the governance process.

3 STUDY AREA

3.1 Geographical setting

A vast alluvial plain, commonly known as “Po plain”, characterizes the north of Italy. The Po plain, named after the Po River watershed, the largest in Italy (71,057 km^2), is bordered by the Alpine and the Apennine mountain belts at its northern and southern limits, respectively, and by the Adriatic Sea at its eastern boundary (Figure 3.1a). In this area, agricultural activities have historically played a key role. Since Roman times, the original broad-leaved forest has been drastically reduced for agricultural purposes, substantially modifying the landscape (Bracco and Marchiori 2001; Marchetti 2002). Currently, intensive agricultural and livestock activities are associated with a high population pressure (20 million of inhabitants) and several industrial districts. Nowadays, the Po plain supports more than 50% of the Italian Gross National Product and it is one of the areas experiencing the highest nitrogen input in Europe (Figure 3.1b; Leip et al. 2011; Martinelli et al. 2018; Viaroli et al. 2018).
The Lombardy plain, focus area of this thesis, covers 13,400 km^2, representing 30% of the Po plain and 49% of the Lombardy Region. Urban and industrial areas cover about 22% of the land and include the city of Milan. The remaining area is devoted to maize and wheat (70%), and rice (11%) cultivation (Figure 3.2; Brenna et al. 2004). The climate is temperate continental (average annual temperature ~13°C, average precipitation ~800 mm), with cold winters and hot summers, spring and autumn being characterized by the highest precipitation amounts (Fratianni and Acquaotta 2017).

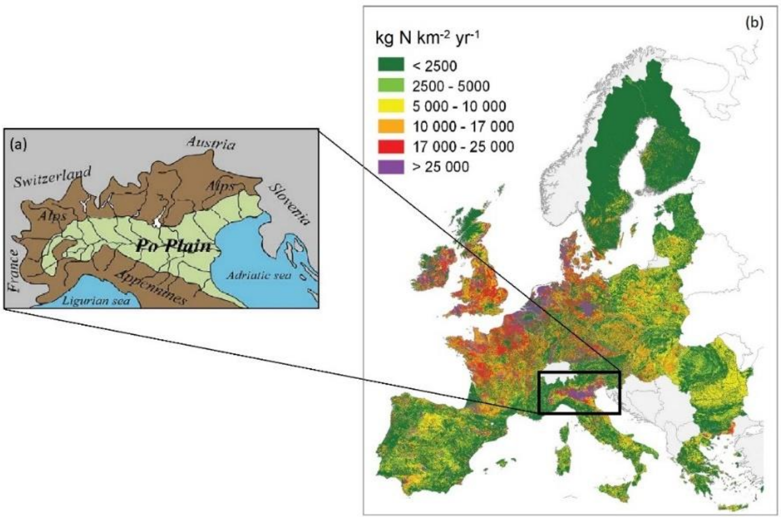

Figure 3.1 (a) Location of the Po plain (from Martinelli et al. 2018); (b) Nitrogen input to agricultural soils in the Po plain and in the rest of Europe (EU27) for the year 2002 (modified from Leip et al. 2011).

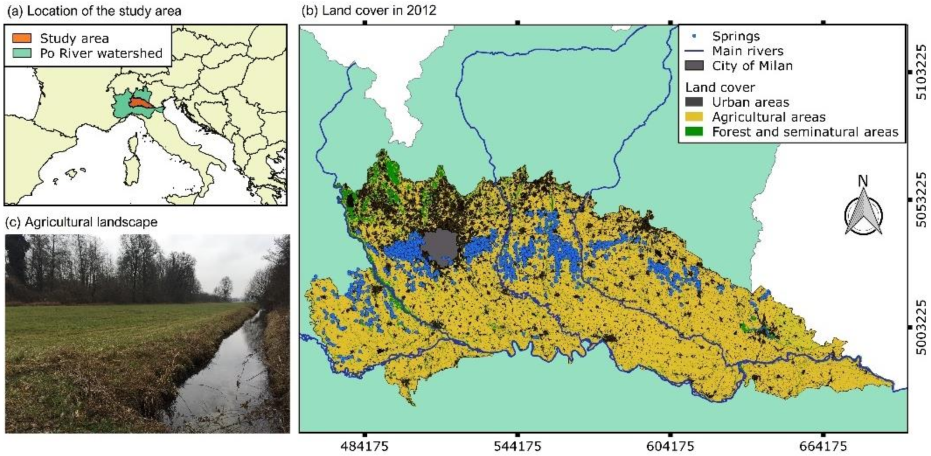

Figure 3.2 (a) Location of the study area Lombardy plain, (b) Land cover in 2012 (coordinates refer to WGS 1984 e UTM Zone 32 N projection), (c) Agricultural landscape of the Lombardy plain (photo by Arianna Musacchio)

3.2 Hydrogeological setting

The Po plain originated during Quaternary by deposition of continental sediments. This large sedimentary basin ranges in thickness from a few dozen to several hundred meters. As illustrated in the block diagrams (Figure 3.3, modified from Martinelli et al. 2018), sediments are generally coarser at the foothills of the Alpine and Apennine mountain belts and become progressively finer towards the centre of the plain and towards the Adriatic Sea coast.

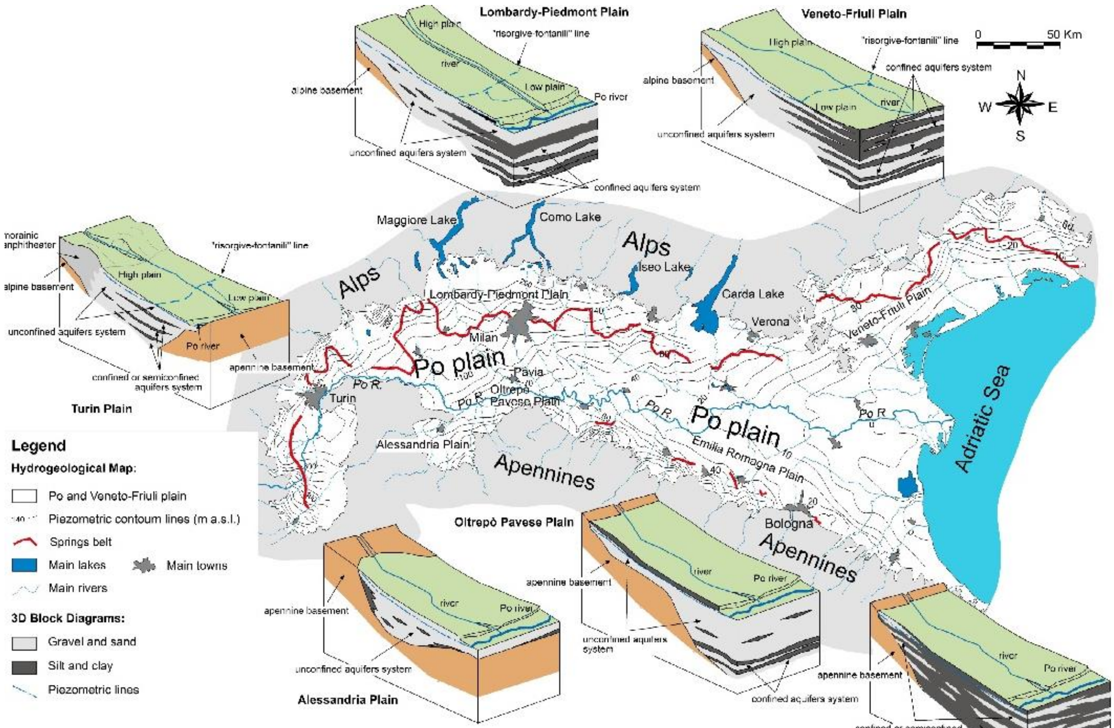

Figure 3.3 Simplified geological and hydrogeological settings of the study area (modified from Martinelli et al. 2018).

At the transition between coarser and finer sediments, the Po plain develops the so-called “springs belt” (“*fascia dei fontanili*”), i.e. an east-west directed area crossing the plain where hundreds of springs (“*fontanili*”) occur. Some *fontanili* are natural groundwater outflows while others have been created since the Middle Ages as irrigation water supply sources, by excavating dips and/or installing tubular artefacts (Balestrini et al. 2016). In both cases, the “*fontanili*” return groundwater, partially mixed with recently infiltrated irrigation water, back to the surface water compartment (Balderacchi et al. 2016). Commonly the springs belt is considered the boundary between the two main sectors of the Po plain, the higher and the lower plain, characterized by different geological, hydrogeological and hydrogeochemical properties. The grain size decrease with distance from the mountain ranges (i.e. from N to S and from W to E) also influences the aquifers' permeability, which is higher near the Alpine and Apennine foothills, and lower downgradient (Sacchi et al. 2013).

In the Lombardy plain, the alluvial sequence, made of gravels and sands with interbedded clay layers and reaching a depth exceeding 500 m, creates a multilayer aquifer system of great importance for the lateral extension and for hydraulic conductivity (about 10^{-3} m s^{-1}; Martinelli et al. 2018). Based on stratigraphic and piezometric data, Éupolis Lombardia (2015) distinguished three aquifer groups (Figure 3.4a,c):

- The shallow aquifers, made of coarse gravel and sand, generally phreatic and with cumulative thickness of the water bearing layers from 20-30 m, in the lower plain, to approximately 150 m, in the higher sector. The groundwater resources of these aquifers are commonly considered the most vulnerable from both a qualitative and quantitative point of view. These aquifers are recharged by direct infiltration of precipitation and irrigation water, and in turn they recharge the underlying aquifers.
- The intermediate aquifers, made of coarse to fine sand, which cumulative thickness increases from north to south, from 50-100 m to 600 m. In the northernmost sectors of the higher plain this group is absent and the shallow aquifers directly overly the deeper aquifer. In the higher plain the intermediate aquifers are hydraulically connected with the shallow ones, while they become semi-confined in the centre of the plain. Groundwater resources of this group are considered strategic for the Lombardy Region domestic water supply as they showed an overall quantitative stability during the last thirty years and are generally less vulnerable to contamination.
- The deeper aquifer made of sand with interbedded fine layers, generally confined and deep. This group is found only in the northern sector of the plain and it also represent a key resource for regional water management, from both a qualitative and a quantitative point of view. The stratigraphic lower boundary of this group is currently unknown, due to the great depth it can reach.

Groundwater flow in the shallow and intermediate aquifers is directed towards the Po River and is strongly controlled by the draining action of the Po rivers itself and its tributaries. The water table depth varies from ~30 m b.g.l. in the northwest to 2-3 m in the southeast (Lombardy Region 2006).

The recharge areas of all the aquifers are located in the Alpine and Apennine foothills but the shallow group is also significantly recharged by precipitation and irrigation water. The latter is diverted from the tributaries of the Po river, approximately from May to August, and is then distributed in an extensive network of channels. As these channels are mostly unsealed and irrigation is traditionally carried out with low efficiency, large volumes of water seasonally infiltrate into the shallow aquifers, producing an increase of the water table in the norther plain which, in turn, feeds rivers and *fontanili* (Balderacchi et al. 2016; Balestrini et al. 2016; Rotiroti et al. 2019). Instead, a general equilibrium of piezometric levels, with some slight decrease, characterizes the lower plain (Figure 3.4b; Éupolis Lombardia 2015).

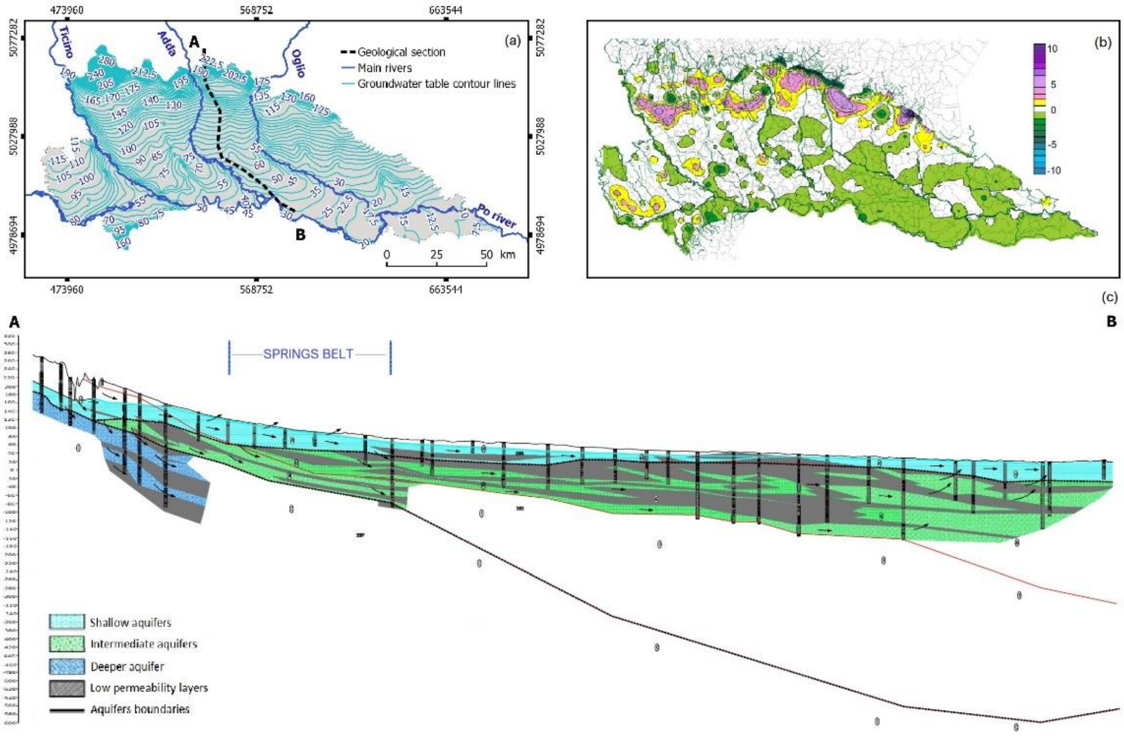

Figure 3.4 (a) Main rivers and piezometric levels (September 2014) of the shallow aquifer (Éupolis Lombardia 2015). The black line represents the location of the hydrogeological section shown below. Coordinates refer to WGS 1984 e UTM Zone 32 N projection; (b) differences in meters between the piezometric levels of the shallow aquifer measured in September 2014 and those measured in April-May 2014; (c) hydrogeological section representing the three main groups of aquifers of the Lombardy plain (modified from Éupolis Lombardia 2015).

3.3 Nitrate contamination and the Directive implementation

In the Lombardy plain the more critical areas, in terms of contamination values and vulnerability, are located in the northern sector of the plain, where unfavourable geological, hydrogeological or hydrogeochemical features (i.e. the high permeability of unsaturated zone and the aquifers, the great depth of water table, the absence of continuous aquitards effectively separating the aquifers, the absence of denitrification processes) are combined with high anthropic pressure (Sacchi et al. 2013; Stevenazzi et al. 2015; Stevenazzi et al. 2017). Several wells of this sector exceed the maximum permitted concentration of 50.0 mg L^{-1} (Figure 3.5).

On the contrary, in the lower plain groundwater resources are generally less vulnerable to nitrate contamination. Low nitrate concentrations are mainly due to the low permeability of soils and the shallow water table, which jointly support the occurrence of denitrification phenomena (Masetti et al. 2008; Sacchi et al. 2013). At the transition between the higher and the lower plain, the “springs belt” represent a major discharge of the norther contaminated aquifers, and this significant groundwater outflow can strongly affects the quality of surface water, as demonstrated in the Oglio river basin (Bartoli et al. 2012; Delconte et al. 2014).

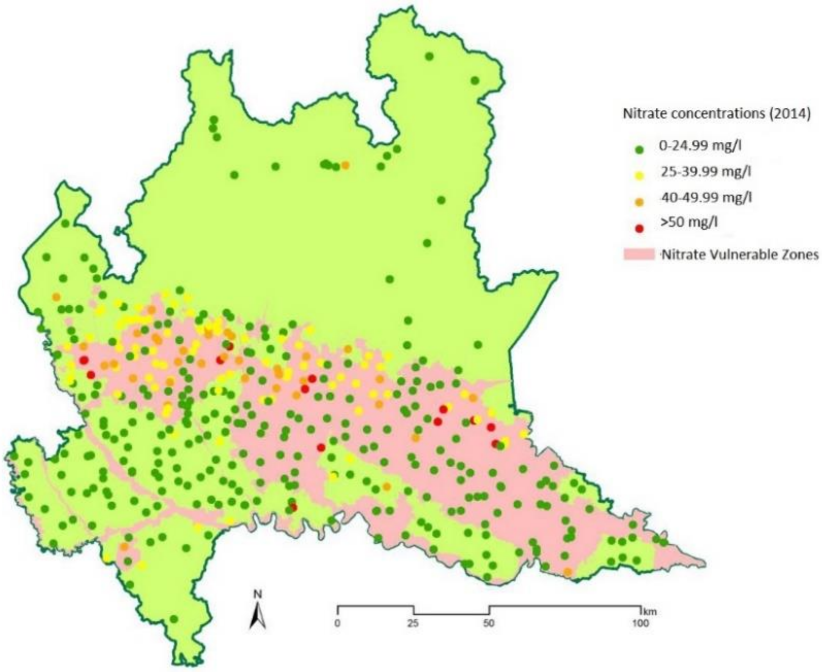

Figure 3.5 Mean nitrate concentrations in the Lombardy Region (2014). Modified from ERSAF et al. (2015).

The analysis of contamination trends, carried out by regional authorities in accordance with ND criteria (i.e. comparing mean concentration values of the monitored four-year periods), points out that 70% of monitoring wells show stable or decreasing trends, with maximum and minimum values of ±1 mg L^{-1}. The percentages of wells belonging to the considered concentration classes are mostly stable over time, especially for those wells which concentrations are lower than 40.0 mg L^{-1}. The number of wells with higher mean concentrations shows small fluctuations (ERSAF et al. 2015).

According to Viaroli et al. (2018), in the Lombardy plain the nitrogen input has historically exceeded the nitrogen export due to crops, volatilization or denitrification in soils, increasing the risk of surface and ground waters contamination. Over the last fifty years, the major variation in nitrogen input has been driven by changes in land use and farming practices, resulting from the substitution of traditional agricultural practices with large scale farming. In particular, the increase in livestock density coupled with the enforcement of less restrictive standards in manure and sewage usage in the Lombardy Region, compared to the neighbouring Emilia Romagna Region, strongly affects the SE sector of the plain. Moreover, the direct relationship between increasing nitrate concentrations in groundwater and urban development in the last decades has been recently demonstrated (Stevenazzi et al. 2015).

As regards the ND implementation, NVZs currently cover about 52% of the plain. Vulnerability is defined based on a municipal scale, although some municipalities only in part subject to NVZs regulations are classified as partially Nitrate Vulnerable Zones (pNVZ). However, as in other European areas (Frederiksen 1995; Goodchild 1998), also in Italy the ND application has been slow and troubled. Despite the Directive has been transposed at national scale in 1994 (Law n. 146, 22 February 1994), its full implementation only occurred in 2006, after two decisions of the European Court of Justice and one infringement proceeding. Before, both the designation of NVZs and the definition of mandatory management actions were missing or insufficient. Similarly, in the Lombardy region, although a previous implementation designated 15% of agricultural lands as vulnerable already in 1993, the full implementation was achieved only in 2006 (Favilli 2008; ERSAF et al. 2015; Figure 3.6).

Since 2011, a derogation was granted by the European Commission to the Lombardy region, together with other territories in the Po plain (EU Commission 2011). The derogation allows, under certain conditions, to exceed the limit of 170 kg ha^{-1} y^{-1} of nitrogen from manure spreading up to 250. Higher management standards are required to farmers who benefit from derogation (Sacchi et al. 2013; EU Commission 2018). Currently, only a small number of farmers applied for derogation.

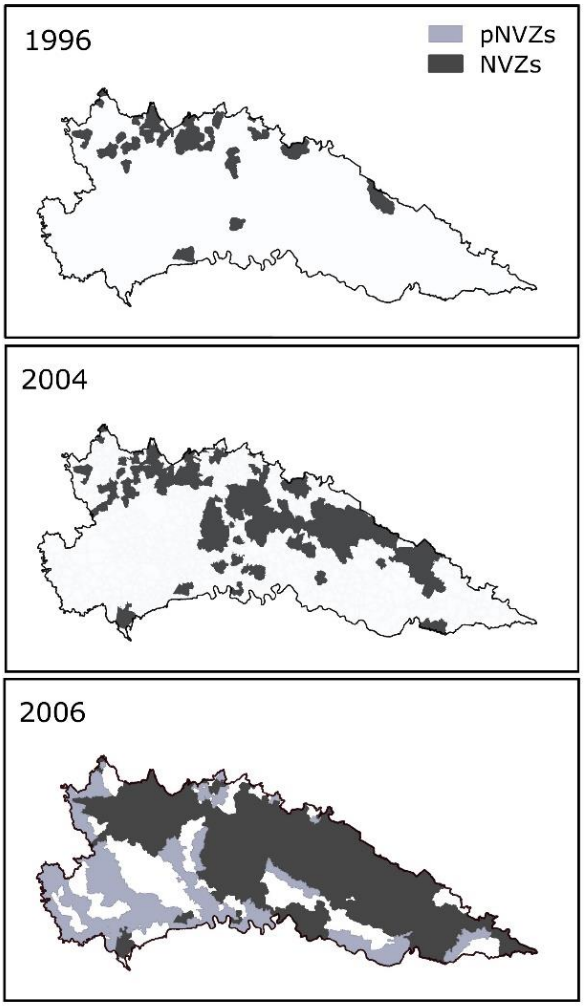

Figure 3.6 Increase in coverage of Nitrates Vulnerable Zones (NVZs) and partial Nitrates Vulnerable Zones (pNVZs).

4 ENVIRONMENTAL EFFECTIVENESS OF THE NITRATES DIRECTIVE ACROSS TIME AND INFLUENCE OF REGIONAL GOVERNANCE ON ITS PERFORMANCE[1]

4.1 Background

The achievement of appreciable declines in groundwater nitrate concentrations is influenced by both intrinsic factors (i.e. the biogeochemical, pedological, and hydrogeological characteristics of the system) and multiscale groundwater governance. Recently it was observed that failure in groundwater management is often the result of an inadequate governance configuration, rather than the lack of knowledge related to aquifer vulnerability or hydrogeological dynamics (Foster and Garduno 2013; Garrick et al. 2017).

The main objectives of this chapter are to assess (i) whether groundwater contamination has decreased since the ND has been applied, and (ii) whether regional groundwater governance supports the application of the ND correctly, with the aim of looking for possible links between nitrate trends and governance actions.

To address these aims, a contamination trend analysis is conducted, followed by the identification of the stakeholders involved in groundwater governance and ND implementation and the socio-relational dynamics between them, through a social network analysis. Finally, management and governance implications are discussed.

4.2 Theoretical framework

The concept of governance refers to "the range of political, social, economic and administrative systems that are in place to regulate development and management of water resources and provisions of water services at different levels of society" (Rogers and Hall 2003). Unlike management, in its broadest sense governance includes the complexity of the regulatory processes that result from the interaction between the different actors who help to define the legal framework and then implement the environmental policy and its tools (Pahl-Wostl 2009). Therefore, governance is an ongoing process in which multiple actors on different scales, with multiple purposes and priorities, interact more or less directly through formal and informal relationships.

[1] This chapter contains the information published in: **Musacchio, A.**, Re, V., Mas-Pla, J., & Sacchi, E. (2019). EU Nitrates Directive, from theory to practice: Environmental effectiveness and influence of regional governance on its performance. Ambio, 1-13. The original publication is available at https://link.springer.com/article/10.1007/s13280-019-01197-8

To understand the governance structure and the main relational patterns influencing ND application, a social relational approach is adopted using social network analysis (Bodin and Prell 2011). According to this approach, investigating how the actors contribute and influence natural resource governance requires a more systemic perspective, namely exploring how the actors are framed in the wider social context, instead of considering them separately. The sociological theories underpinning the social relational approach (Sawyer 2002) assume that the cultural, economic and political properties of a system are not the mere sum of the attributes of its components (i.e. the actors and their actions); rather, they are new emerging properties, determined by both the relational structure and the way the actors are tied to and positioned in the social system. In other words, patterns of relationships and configurations of governance can constrain or promote different attitudes and behaviours of both the actors involved and the system as a whole. From a methodological perspective, the social relational approach is implemented by means of the social network analysis, permitting to understand the social system in its complexity by visualising it as a graph. The nodes in the graph represent the actors, and the links the relationships between them. Qualitative and quantitative analysis of this graph provides insights into the influence of governance structure on observed behaviours and patterns.

Although social network analysis was shown to be an effective method for these aims, its application is quite a novelty in hydrogeological assessments. To our knowledge it was only used in two studies that explicitly focused on groundwater governance, both of which were carried out in rural developing regions (Kuzdas et al. 2015; Tringali et al. 2017).

4.3 Methods

4.3.1 Nitrate contamination values and trends

Current nitrate levels and contamination trends are assessed using data collected by the Regional Agency for Environmental Protection (ARPA) through the regional monitoring network for groundwater quality. The wells belonging to this network are selected by ARPA based on the standards required by the Italian Law (n. 30, 16 March 2009, Annex 4) in application of the European Groundwater Directive (2006/118/CE). Accordingly, the location and density of monitored wells are based on (i) the current hydrogeological conceptual model, (ii) the existing pressures, (iii) the extent and geometry of aquifers and (iv) unambiguous belonging to the aquifers. Based on risk and prevention criteria, the presence in the network of wells intercepting the shallow aquifers is potentially favoured (ARPA Lombardy 2018c). Field sampling and laboratory procedures are defined by ARPA consistently with the standards required by the Italian Institute for the Environmental Protection and Research (ISPRA 2018).

To evaluate current contamination the mean nitrate concentrations detected in 258 wells (168 tapping the shallow aquifers, 58 the intermediate and 31 the deeper aquifers; ARPA Lombardy 2018) in 2016 are calculated and a Wilcoxon rank-sum test is performed to determine significant differences between aquifers (Helsel and Hirsch 2002).

The nitrate trend analysis uses ARPA monitoring data from 2006 (i.e. the year the ND was fully implemented on a regional scale) to 2016 (ARPA Lombardy 2018b). The non-parametric Mann Kendall test is applied to statistically detect significant monotone trends (Mann 1945; Kendall 1955), using a 95% significance level, and to categorise the wells as decreasing, increasing or non-detected trends. The latter corresponds to a non-statistically significant trend. The Mann Kendall-test has the advantage of not assuming any distribution for the data and it has similar power and efficiency to parametric methods (Serrano et al. 1999; Batlle Aguilar et al. 2007). Nevertheless, measurements need to be taken at regular intervals. This is not the case for the wells belonging to the regional monitoring data in Lombardy, as these wells may have been sampled a maximum of four times per year. Therefore, available data are selected, based on Hirsch et al. (1991) suggestions, and a list of suitable wells is obtained. Firstly, for each well only data collected between April and June and between October and December are selected, due to the higher number of samples collected during these months. The lack of a seasonal effect between the two groups is previously demonstrated by means of a Wilcoxon rank-sum test (Kent and Landon 2013). The wells are then selected by dividing the study period into three sub-periods (2006-2009, 2010-2013, 2014-2016) and the percentage of collected data with respect to the number of data potentially available for each sub-period is calculated, considering an average data collection of 2 samples per year. Only wells with a higher coverage than 25% in each sub-period are included in the analysis.

For detected trends, the Sen slope estimator is calculated to assess their magnitude, namely nitrate increase or decrease (in mg/L per year) (Sen 1968). The Wilcoxon rank sum test is applied to explore significant differences in trend magnitudes (i) between aquifers and (ii) between four concentration classes (<10; 10-25; 25-50; >50 mg/L; Eurostat 2012). Finally, the distribution of trends in NVZs, pNVZs and nNVZs is explored.

All the analyses are performed using the statistical software Rstudio version 1.0.153 (RStudio and Team 2015) and the packages *imputeTS* (Moritz 2017) and *trend* (Polhert 2017).

4.3.2 Governance framework

To identify governance structure and dynamics, a social network analysis is carried out using the participative network mapping tool called "Net-map" (Schiffer and Hauck 2010), which obtains both network data and qualitative descriptions of relationships and roles in the network. Qualitative data, also called "network narratives", provide information on intersubjective meanings attributed to network components and shared or predominant perceptions (Fuhse and Mutzel 2011; Hauck et al. 2015).

Between October 2016 and June 2017, in-depth focus groups with five groups of key informants are carried out: (i) authorities (members of the Regional Directorates for Agriculture and for Environment, and of ARPA), (ii) farmers, (iii) breeders, (iv) organisations (representatives of a farmers' trade union and a Water Consortium), (v) scientists actively involved in research projects (Figure 4.1; details on selection of key informants in Supplementary Materials). Each group is interviewed separately in order to avoid possible bias due to power differences or intimidation. During the meetings, the interviewees are asked to draw the social network involved (directly or indirectly) in nitrate contamination. The guiding questions for creating the social network are: "Who can influence groundwater pollution reduction in the Lombardy plain?" and "Who can influence the implementation of new groundwater protection actions based on the outcomes of the hydrogeological and biogeochemical investigation?". Basically, the interviewees are first asked to list all the actors involved in groundwater nitrate contamination and in ND application (e.g. contractors, consultants, consortia, breeders, citizens) for each category of actors (i.e. authorities, individual actors, research institutions, industries, organisations), and then to depict all the links between the actors for each category of link (i.e. authorisation and control, technical information, advice and best practices, money, conflicts; Table S2.1). The categories of actors and links are previously defined by researchers, with the main aim of facilitating and inspiring the discussion. This allows the interviewees to orient themselves in a wide and varied governance context. At the same time, no further description of links is provided, compared to Table S2.1, to also gain a deep understanding of the meanings that each group associates to the governance dynamics. Finally, the interviewees are asked to quantify the potential influence of each actor, according to their perception. This is evaluated on an influence-value scale, from 1 to 5 (i.e. 1 is the lowest influence, 5 the highest; decimal values are permitted). Description of the role of the actors, relationships and reasons for the influence values are encouraged, both in order to validate the information and to obtain qualitative data. When disagreement on effective involvement of an actor, link or on influence values occurs, the interviewees are asked to discuss the matter further until a consensus is reached within the group. Each focus-group lasts between 2-3 hours.

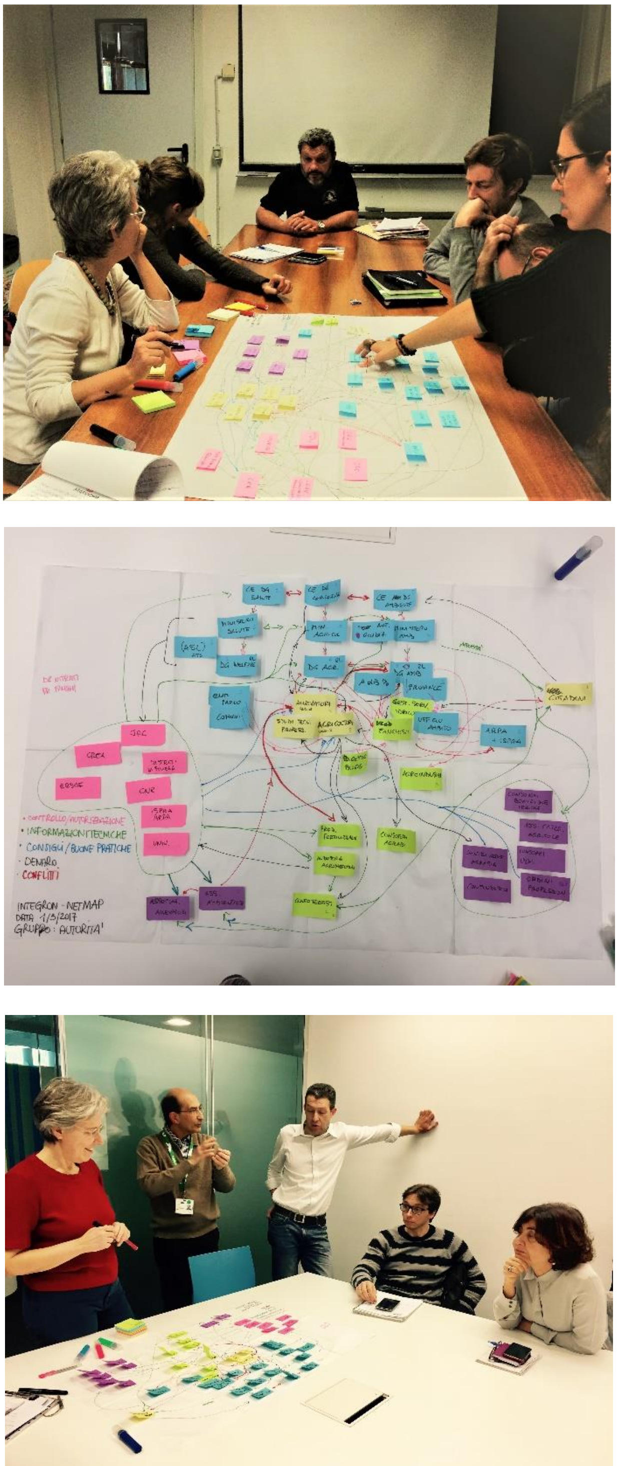

Figure 4.1 In depth focus groups with key informants and an example of the social network they drawn. Photos by Arianna Musacchio.

The five networks are merged the average influence levels of each actor are calculated (more details in Supplementary Materials), by also considering when actors are not listed by all focus-groups (i.e. considering an influence value equal to zero for each network in which that actor is not mentioned).

By virtue of both the quantitative and qualitative nature of collected data, structural and content analyses are combined (Hauck et al. 2016). The structural role of each actor is evaluated by measuring degree centrality (i.e. the total number of ingoing and outgoing links of an actor; Wasserman and Faust 1994) and betweenness centrality (i.e. how many times an actor is found on the shortest path connecting other actors who are otherwise disconnected; Freeman 1978). Actors with a degree centrality, betweenness centrality or influence higher than the 75th percentile are considered as the most central, connecting or influent actors. Average influence, centrality measures and description of actors in network narratives are compared to detect discrepancies between structural and perceived roles. As regards content analysis, a description of each kind of link is obtained by using recordings and transcriptions from focus groups to define the way that relationships support or hamper ND application according to stakeholders' opinions. The social network is displayed and analysed using the software Visone (Brandes and Wagner 2004).

The focus groups reveal a distinct difference in the perception of meanings and network components between groups of key informants; therefore, the analysis is enhanced by measuring (i) how many times each actor is mentioned by the different groups of key informants, (ii) the percentage of actors listed by each focus group and included in the merged network, and (iii) the difference between the minimum and maximum value of influence assigned to each actor by different focus groups. Actors mentioned less than 3 times are considered scarcely perceived. Differences in influence of three or more are considered as significant in terms of divergence in perception between groups of key informants. Consistently, the descriptions of relationships are integrated with data on differences in perception of structure and meanings associated to each kind of link.

4.4 Results

4.4.1 Current values of nitrates and concentration trends

Results highlight significant differences between the nitrate levels in the shallow aquifers compared to the intermediate and deeper ones ($W_{shallow\text{-}intermediate}$=3426, p=0.002; $W_{shallow\text{-}deeper}$=3426, p=0.007; $mean_{shallow}$=26.5 mg/L, $mean_{intermediate}$=18.5 mg/L, $mean_{deeper}$=17.7 mg/L; Figure 4.2a). 43.4% of the wells are currently above the "concern" threshold of 25 mg/L (Eurostat 2012), and 4.7% are above the statutory limit of 50 mg/L.

Out of the 162 wells suitable for trend analysis (118 in shallow, 31 in intermediate and 13 in deeper aquifers), 46.3% are classified as non-detected, 28.4% as increasing and 25.3% as decreasing trends (Figure 4.2b; Table 4.1). Overall, in the three aquifers, approximately half of the wells are classified as non-detected trends (Table 4.1). Regarding the trend magnitudes, the annual increase and decrease ranges between +0.05 and +1.7, and between -0.08 and -3.4 mg/L, respectively. On average, these variations are higher in wells tapping shallow aquifers (Table 4.1), although these differences are not statistically significant. Trend magnitudes show both higher average and higher maximum annual concentration changes in the class with concentrations >50mg/L (mean:0.41; max:1.74; $mean_{shallow}$:0.68; $max_{shallow}$:1.74 mg/L per year Figure 4.3), although differences between classes are not significantly different. Decreasing average values are found in the class 25-50 mg/L in both analysed groups (mean=-0.15; $mean_{shallow}$=-0.24 mg/L). A higher proportion of increasing trends was detected in the municipalities currently classified as nNVZs and pNVZs, compared to those currently classified as NVZs. These wells generally have low concentrations and are mostly located in the southern sector (Figure 4.2). The trend magnitude of increasing wells located in nNVZ is on average higher than that of wells located in NVZ (Table 4.2; Figure 4.2b, c).

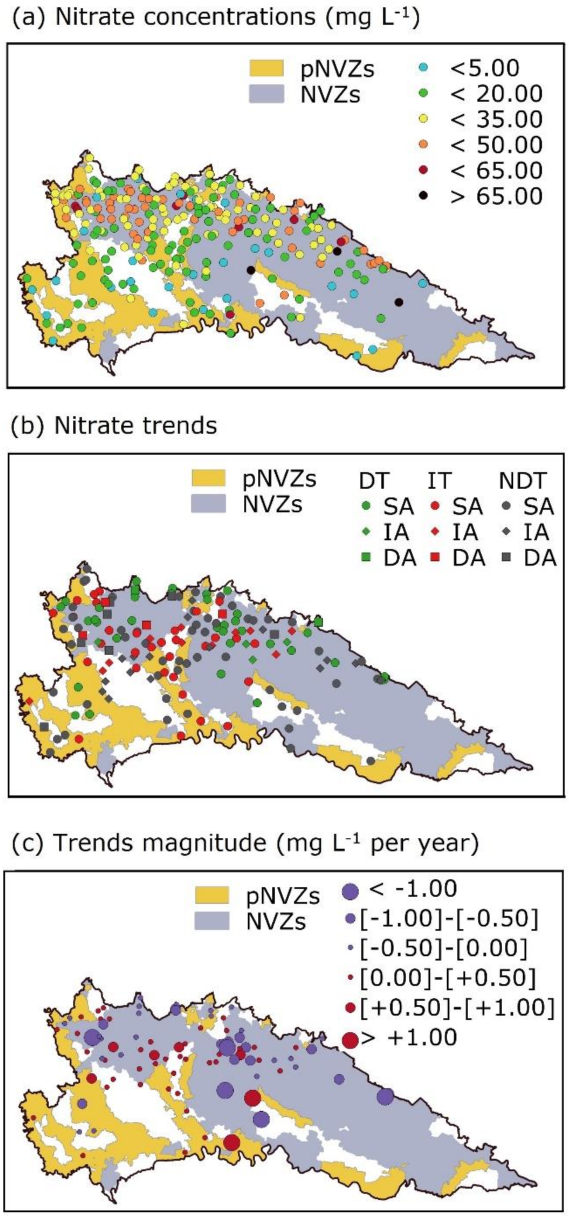

Figure 4.2 (a) Mean nitrate concentrations in 2016 (ARPA monitoring data); (b) trends of nitrate concentrations in the three groups of aquifers, from 2006 to 2016 (SA: shallow aquifers, IA: intermediate aquifers, DA: deeper aquifers); (c) magnitude of nitrate trends. Nitrates Vulnerable Zones (NVZs) and partially Nitrates Vulnerable Zones (pNVZs) are reported.

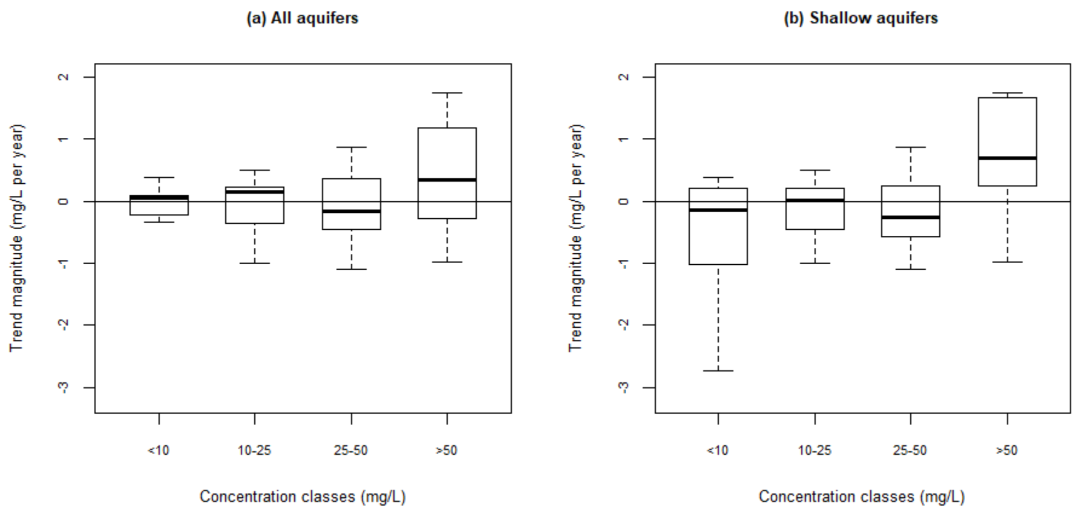

Figure 4.3 (a) Trend magnitudes for nitrate concentration classes; (b) trend magnitudes for nitrate concentration classes in shallow aquifers.

Table 4.1 The number (%) of wells having non-detected, increasing and decreasing trends, in each group of aquifers, and the minimum, mean and maximum trend magnitudes.

Aquifers	wells				mg/L per year in increasing trends			mg/L per year in decreasing trends		
	n°	% non-detected	% increasing	% decreasing	min	mean	max	min	mean	max
Shallow	118	44.1	26.3	29.7	0.1	0.4	1.7	-0.1	-0.7	-3.4
Intermediate	31	51.6	35.5	12.9	0.1	0.3	0.8	-0.1	-0.3	-0.8
Deeper	13	53.8	30.8	15.4	0.1	0.2	0.4	-0.8	-0.2	-0.04

Table 4.2 The number of wells (%) having non-detected, increasing or decreasing trends within nitrates vulnerable zones (NVZ), partially vulnerable zones (pNVZ) and non-vulnerable zones (nNVZ), and their average magnitude of trends.

Vulnerability	wells				Trend magnitude (mg/L per year)	
	n°	% non-detected	% decreasing	% increasing	decreasing	increasing
NVZ	92	40.2	35.9	23.9	-0.60	0.38
pNVZ	29	55.2	13.8	31.0	-0.32	0.25
nNVZ	41	53.7	9.8	36.6	-1.05	0.43

4.4.2 Governance framework

The governance network supporting the application of ND includes 33 actors, distributed across the four main governance levels (Figure 4.4; Table 4.3). Among them, 10 were not mentioned more than twice (Table 4.4). The median percentage out of 33 actors listed by each focus group was 54.5% (Table S2.2, S2.3, S2.4). 27 actors show relevant differences in influence values (Table 4.4). Only farmers, breeders and national research institutions were highly mentioned and, at the same time, similarly perceived by interviewees in terms of influence. 11 out of the 33 actors were shown to be more relevant than the others (Table 4.4); except for the agricultural retailers, all of them were mentioned at least four times.

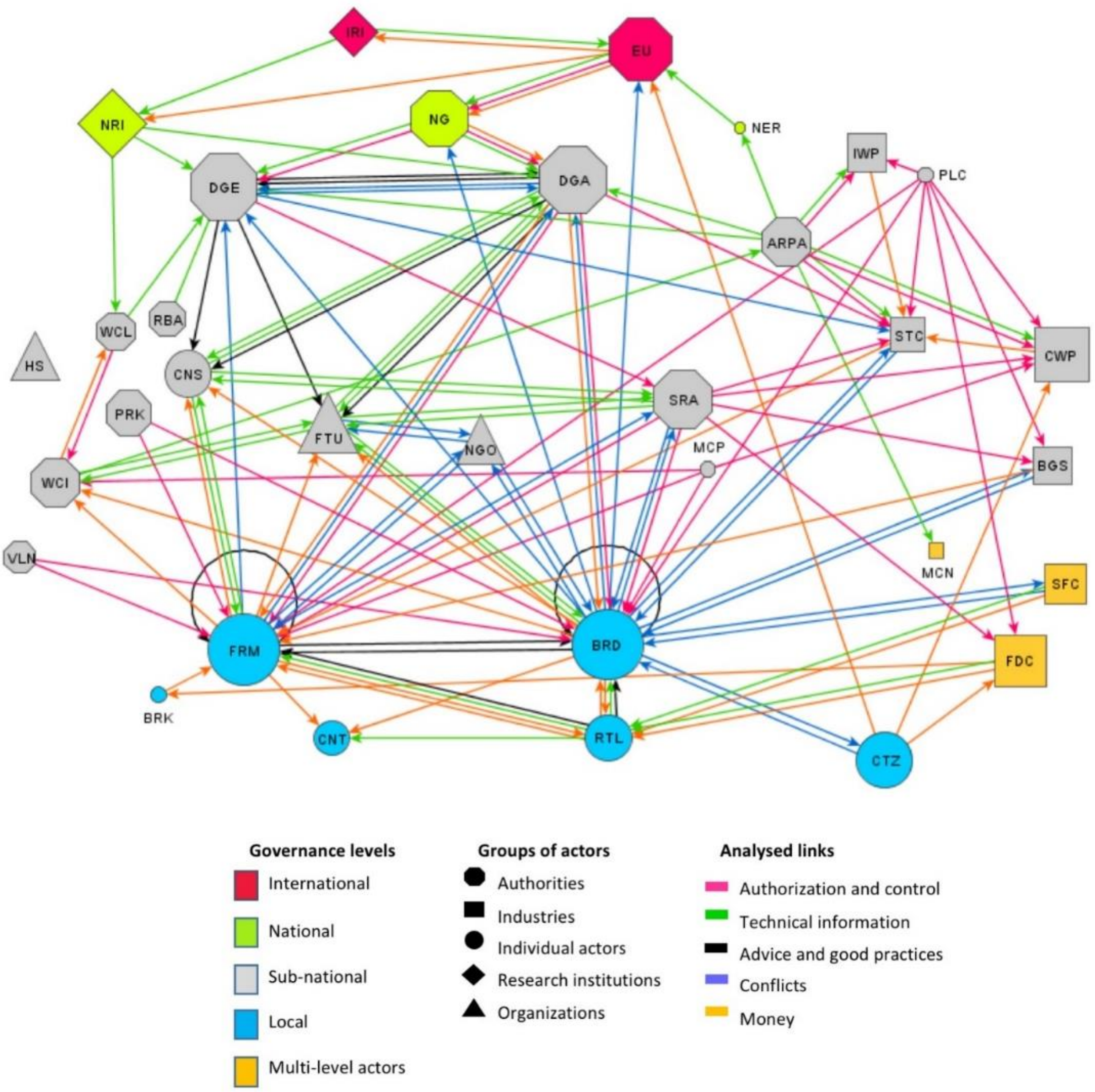

Figure 4.4 Final network representing the governance framework. The size of the nodes corresponds to the perceived influence of each actor. List of acronyms used in the map in alphabetical order: ARPA: Regional environmental protection agency, BGS: Biogas and compost plants, BRD: Breeders, BRK: Brokers, CNS: Agricultural consultants, CNT: Agricultural contractors, CTZ: Non-farm residents, CWP: Civil wastewater treatment plants, DGA: Regional DG Agriculture, DGE: Regional DG Environment, EU: European Commission, FDC: Food companies, FRM: Farmers, FTU: Farmers' trade unions, HS: High school, IRI: International research institutes, IWP: Industrial wastewater treatment plants, MCN: Agricultural machinery manufacturers, MNC: Municipalities, NER: National Institute for Environmental Protection and Research, NG: National government, NGO: Environmental NGO, NRI: National research institutions, PLC: Environmental Police, PRK: Parks, RBA: Po River Basin Authority, RTL: Agricultural retailers, SFC: Seed, fertilizers, animal feed companies, SRA: Sub-regional administrations, STC: Sludge Treatment Companies, VLN: Environmental volunteers, WCI: Water consortia (irrigation), WCL: Water consortia (lakes).

Table 4.3 Network composition in terms of groups of actors and kind of links.

Groups of actors	%	Links	%
Authorities	42.4	Technical information	27.5
Individual actors	21.2	Authorisation and control	23.2
Industries	21.2	Money flows	21.7
Organizations	9.1	Conflicts	18.8
Research institutions	6.1	Advice and best practices	8.7

Table 4.4 Results concerning the average influence of the actors, the betweenness and degree centrality, the number of times the actors were mentioned and the difference between the maximum and minimum value of influence assigned to them by different groups of key informants. Asterisks identify the most relevant actors, i.e. having highest values of influence, centrality or betweenness, based on 75th percentile. For these actors, asterisks are reported only on those measures which value result higher than the 75th percentile.

Actor	Acronym	Influence	Degree centrality (%)	Betweenness centrality (%)	Times mentioned	Max-min influence
Farmers*	FRM	4.2*	10.5*	10.5*	5	2
Breeders*	BRD	4.2*	13.4*	25.0*	5	2
Regional DG Agriculture*	DGA	3.8*	8.0*	6.9*	5	4
Regional DG Environment*	DGE	3.6*	5.8*	6.0*	5	4
Sub-regional administrations*	SRA	2.8*	5.1*	6.4*	5	3
European Commission*	EU	3.2*	3.3	6.3*	4	4
Farmers' trade unions*	FTU	3*	5.1*	3.6	5	4
National research institutions*	NRI	4.0*	1.8	3	5	2
Agricultural retailers*	RTL	1.8	4.7*	1.9	3	5
Regional environmental protection agency*	ARPA	2	4	7.1*	4	4
Water consortia (irrigation) *	WCI	2	2.9	10.6*	5	3
Agricultural consultants	CNS	1.8	3.6	0.1	3	5
Agricultural contractors	CNT	1	1.1	0	2	4
Agricultural machinery manufacturers	MCN	0.2	0.4	0	1	1
Biogas and compost plants	BGS	1.2	1.8	0	3	4
Brokers	BRK	0.2	0.7	0.1	1	1
Non-farm residents	CTZ	2.6	1.8	0.9	4	5
Civil wastewater treatment plants	CWP	2.4	2.5	0.1	3	5
Environmental NGO	NGO	2	2.2	0	3	4
Environmental Police	PLC	0.2	2.5	0	1	1
Environmental volunteers	VLN	0.8	0.7	0	1	4
Food companies	FDC	2.2	2.2	2.3	4	4
High schools	HS	2	0	0	2	5
Industrial wastewater treatment plants	IWP	1.2	1.4	0	2	5
International research institutes	IRI	2	1.1	0	3	5
Municipalities	MNC	0.2	1.4	0	3	4
National government	NG	2.6	3.3	2.4	3	4.5
National Institute for Environmental Protection and Research	NER	0.1	0.7	0.4	2	4
Parks	PRK	1.7	0.7	0	3	4
Po River Basin Authority	RBA	1	0.4	0	2	4
Seed, fertilisers, animal feed companies	SFC	1.4	1.4	0	3	3
Sludge Treatment Companies	STC	1	4	4.6	1	5
Water consortia (lakes)	WCL	1.2	1.4	1.5	3	4

Seven of these relevant actors were both highly influential and central or connecting, and their roles and responsibilities were also described by different groups of key informants in the same way:

- farmers and breeders are the most influential actors according to key informants' perception (i.e. given the potential impact of agricultural and breeding activities on water quality) and centrality measures.
- Agricultural trade unions (hereafter referred to as "trade unions") are the link between authorities and farmers in terms of information and administrative flows.
- European, regional and sub-regional authorities have a key role in defining and applying the ND, supplying Common Agriculture Policy subsides, coordinating and implementing controls. The European Commission is not central; its high perceived influence is actually confirmed by its betweenness value, consistent with its connecting role described by key informants. The lack of a high centrality degree for this actor could be expected considering the regional scale of ND application targeted.

The high influence that key informants uniformly assign to national research institutions is not confirmed by centrality measures (Table 4.4). The central role of retailers and the connecting capacity of ARPA and Irrigation Water Consortia do not correspond to high perceived influence. Centrality measures reveal that trade unions do not have a highly connecting capacity, though their influence is mainly associated to this function in interviewees' opinion.

As regards relationships, 138 links were identified (Table 4.3). In the light of the similar role played by farmers and breeders, from now on we will refer to both as "farmers", unless otherwise indicated. We will also refer to information and advice flows jointly, as "knowledge flow". The network narratives highlight the following:

- During all focus groups, difficulties in fully mastering both authorisation and control ties (more details in Supplementary Materials; Figure S4.1) and the tasks of several authorities (i.e. municipalities, provinces, ARPA, River Basin Authority and Water Consortia) were detected. All interviewees reported that control-based strategies represent the only real tool to guarantee the adoption of sustainable practices by farmers but, at the same time, it is extremely difficult to perform systematic and widespread controls due to the number of farms and the associated costs. Farmers observed that recurrent controls may be a life-long learning opportunity if associated to structured capacity building and to a mutual trust relationship between farmers and authorities.
- European direct payments associated to the ND, research funds, costs for wastewater treatment and economic relations between the agricultural and industrial sectors are the main money links (Figure S4.2). Only authorities and farmers showed a detailed knowledge of the money flows that farmers are involved in, despite their relevance in defining farmers' decision-making. To comply with the limits defined by the ND, breeders usually give superfluous manure to farmers. This manure relocation neither corresponds to money flows nor is tracked by authorities.
- Two knowledge flows are present: one based on a traditional knowledge transfer approach (i.e. information is generated in science and transferred to farmers through authorities and trade unions; Figure S4.3; Blackstock et al. 2010), the other connects the seed, food and fertiliser companies to farmers through retailers (Figure S4.4). Authorities inform farmers about rules for ND application and periods in which fertilisation is forbidden or permitted, via official websites, newsletters and smartphone applications. Trade unions provide farmers with administrative support regarding ND and subsides. Occasionally they produce dissemination materials on ND. Authorities, researchers and organisations disregard the advice generated by the industrial sector. In general, a conceptual diversity emerges between groups of key informants with regards to the advice flow. Authorities,

researchers and organisations see no or little difference between “information” and “advice”, as both aim to improve water quality. Farmers distinguished the two flows: they associate technical information with the compliance with the rules required to obtain subsides and the bureaucratic procedures (i.e. documents quantifying nitrate inputs, provided by farmers to authorities through trade unions); on the other hand, reliable advice that can change farm strategies is provided by neighbouring farmers and retailers. The retailers establish business relationships locally, based on direct meetings with farmers. In farmers’ opinion, frequency, informality and consistency with their production goals makes the relationships that occur on a local scale (i.e. with retailers and other farmers) stronger and relevant compared to those with the actors of other governance scales. Additionally, agricultural magazines as a trustworthy information source and agricultural high schools may have a key role in enhancing environmental awareness in the agricultural sector, according to farmers.

- Conflicts (Figure S4.5) between farmers and authorities on a regional, national and European scale are mainly due to slurry or manure spreading and to ND restrictions; conflicts between farmers, non-farm residents and sub-regional authorities are mainly caused by smells produced by effluent spreading.

4.5 Discussion

There has been overall stability in groundwater contamination in the Lombardy plain since the full implementation of the ND. Most wells in the three aquifers present no significant trends in nitrate concentrations. Current nitrate concentrations are not particularly alarming, often being lower than the threshold levels. Nevertheless, contamination is ongoing, as testified by increasing trends detected in about a third of the monitoring wells (Table 4.1). Moreover, current application of the ND is not sufficient either to reduce nitrate contamination in previously impaired areas, or to preserve the higher groundwater quality of resistant and resilient hydrogeological units. Indeed, the in-depth trend analysis revealed three critical issues:

(i) Increasing trends, mostly in wells that already exceed threshold values (50 mg/L; Figure 4.3).

(ii) Deteriorating groundwater quality in the supposedly protected intermediate and deeper aquifers, highlighted by both the high proportion of wells with increasing values and the magnitude of their trends (compared with those of the shallow aquifers). Intermediate and deeper aquifers are generally considered less vulnerable to contamination (Éupolis Lombardia 2015), and are extremely relevant as a strategic resource in the light of possible droughts driven by climate change and/or overexploitation of shallow aquifers.

(iii) Increasing trends within the nNVZs located in the southern plain, which is generally more resilient. In this area, the intrinsic characteristics of the aquifers and the agricultural practices promote denitrification, i.e. the system’s ability to self-recover (Sacchi et al. 2013; Martinelli et al. 2018), which justifies the presence of extended nNVZs. Most wells with increasing trends are not located in areas characterised by recent urban sprawling or population growth (Stevenazzi et al. 2015), and their low concentrations (Figure 4.2) do not suggest local phenomena of point-source pollution. Therefore, we can hypothesise that nitrogen input from agricultural activities is slowly contributing to exceeding the self-recovery capacity of the area. This excess may (paradoxically) be promoted by manure relocation adopted by breeders to comply with the ND.

As regards groundwater governance, the implementation of management practices is mainly based on the activity of a small number of relevant actors (i.e. farmers, trade unions, European, regional and sub-regional authorities), whose influence and roles are uniformly described and confirmed by the centrality measures (Table 4.4). Besides these few actors, the analysis reveals the divergence between the governance perceived and its structure, and the lack of a common vision on governance components and dynamics. The role of some actors (i.e. ARPA, retailers, Irrigation Water Consortia) with high centrality or betweenness values is not perceived, as indicated by their low perceived influence; likewise, the influence of the national research institutions is not supported by an appropriate position in the network. The lack of clear, shared knowledge of the whole governance framework is also testified by the low percentage of the entire network perceived, the differences between influence values assigned by the groups of key informants (Table 4.4), and the difficulties in fully mastering even supposedly formal relationships (i.e. control links) and those strongly relevant in farmers' decision making (i.e. advice or money flows involving farmers). Overall clarity on governance structure and dynamics is therefore missing, although it is considered one of the key principles of good and adaptive water governance as it can affect all the processes requiring high coordination and networking (i.e. the information and advice flow or the authorisation and control system; OECD 2015).

From a relational perspective, groundwater governance mainly depends on knowledge dissemination and control-based strategies. Control-based approaches are often criticised due to both the negative effects on the social-environmental resilience and the lack of long-term benefits (Mazmanian and Kraft 2009; Cox 2016). Moreover, in our study control-based strategies are strongly limited by practical and economic issues because of the large number of farms, as reported in network narratives. The poor capacity to conduct systematic and widespread controls affects the effectiveness and credibility of this strategy in the stakeholders' perception. Besides, it should be noted that the reduction of subsidies from the Common Agriculture Policy may impair the whole control system since controls are based on the information provided by farmers to obtain subsidies. As regards knowledge dissemination, the lack of advice flow in all networks, except in those drawn by farmers, indicates the failure to consider knowledge that originated in the industrial sector although it is strongly relevant in farmers' decision-making. The tools chosen by trade unions and authorities to communicate the ND substantially differ from those commonly used by farmers. This underpins a poor consideration of the informality, frequency and clear consistency with production goals, which characterise reliable sources according to farmers (i.e. other farmers and retailers). Besides, agricultural high schools are completely marginal (Figure 4.4), and the opportunity to promote long-term change in farmers' attitudes is not exploited (McGuire et al. 2013).

4.6 Management and governance recommendations

Firstly, detected trends highlight that reducing nitrate inputs is required to halt increasing concentrations as fast as possible in wells with exceeding threshold values, in intermediate and deeper aquifers (Mas-Pla and Menció 2018). Although the estimate of groundwater residence time will complement the evaluation of ND environmental performance, it is clear that an input reduction is urgently needed to recover the water quality. Moreover, the effect of the increasing water withdrawal from intermediate and deeper aquifers should be monitored as it can induce nitrate migration to these deeper aquifer units. To this end, the number of monitoring wells tapping the intermediate and deeper aquifers should increase. Water abstraction policies should thus be defined in accordance with the objectives and the nitrogen inputs established by the ND. Finally, even where overall stability has already been detected, in-depth trend estimations (i.e. considering distribution in both space and concentration classes) should be carried out, including and constantly monitoring wells with concentrations lower than 25 mg/L. Although the ND currently allows these wells to have less intensive monitoring programs (i.e. every eight years; EU

Commission 1991), our study confirms that they can provide useful information on possible unforeseen side effects of current regulations (e.g. manure relocation).

As regards manure, its relocation produced by the occurrence of different protection levels in neighbouring municipalities (i.e. NVZs and nNVZs) should be also monitored. In doing so, controls could be strengthened by cross-checking nitrogen input, and the contribution of agriculture to increasing trends, compared to other sources, could be better understood by enhancing the accuracy of nitrogen input estimates. Overall, the whole-territory approach adopted by other EU nations (Smith et al. 2007) would avoid this relocation of nitrate contamination compared to the discrete zones designation.

The desirable strengthening of the control-based strategies on which governance is strongly based, cannot disregard an increase in adaptive capacity. In fact, adaptive governance is required to deal with both the uncertainty and the speed of environmental changes, and potential modifications to regulations. As increasing adaptive capacity needs to pursue a real shift in farmers' values (de Snoo et al. 2013), the way farmers select the sources of knowledge and the presence of multiple knowledge flow should be considered (Munoz-Erickson and Cutts 2016; Inman et al. 2018). In this respect, actors who interact with farmers locally are required, because of the effectiveness of frequent and informal relationships. This change in the current knowledge-transfer strategy would be also consistent with the necessity of new social learning spaces highlighted by Nguyen et al. (2014). The comparison between structural and perceived influence in the network allows us to identify some suitable actors, whose role can be improved in this respect (e.g. Water Consortia, ARPA). To reach different age groups, agricultural high schools should be included in the knowledge flow, for example through dissemination activities required by funded research and conservation projects.

Finally, effective and adaptive socio-environmental systems also need to go beyond the unclear and ill-defined idea of governance dynamics by the actors involved themselves. Management changes should hence be coupled with a greater awareness of the governance structure (i.e. clearness of roles and responsibilities), as it could enhance the legitimacy of leadership and improve relationships (Bodin and Prell 2011; Akhmouch and Correia 2016). Therefore, dissemination activities should not only include 'which' agricultural practices are required, but also 'who' is involved in governance processes and 'how'. A simpler governance arrangement would strongly support this process.

It is worth mentioning, however, that constrains and opportunities emerged in this study may not fully represent the case of other European regions. This is due, on the one hand, to the site-specific nature of both hydrogeological and socio-relational dynamics. On the other hand, based on national variations in implementation and/or stakeholders involved, other parts or functioning of the ND may be described differently. Indeed, the network approach used in our study focused on identifying the most relevant features of the governance system, based on the perception of the actors involved. Therefore, the identification of all relevant elements of the impact of legislation should also include other case studies and other approaches.

Although other studies on ND went beyond an environmental or agronomic perspective, they only dealt with specific aspects of governance or management, namely the integration between local and scientific knowledge (Nguyen et al. 2014), the factors influencing farmers' collaborative arrangements for manure exchange (Asai et al. 2014), and the use by farmers of tools for balanced fertilization (Ravier et al. 2016). Differently, our study frames specific or local aspects influencing the ND implementation in the multi-relational context in which they occur. Thanks to this approach, the analysis of specific governance or management issues is improved by detecting otherwise hidden criticalities, precisely produced by the coexistence of several socio-relational dynamics in the governance system. In the Lombardy plain, for example, the necessity of both rethinking knowledge transfer processes and thoroughly considering the way farmers select information and collaborations, also highlighted by previous studies, cannot ignore the need to consider (i) the solid relationship between farmers and industries, (ii) the relative importance of knowledge transfer and control activities, and (iii) the definition of the adaptive capacity that we want to preserve or obtain, to achieve a fully resilient socio-environmental system.

In this light, a governance-oriented debate on ND is currently missing, although it could enhance the current knowledge on the Directive performance, at present only partially understood, thus hampering its environmental success. Therefore, we believe that Member States should be required to provide to the EU Commission an assessment of the governance dynamics supporting the Directive implementation together with environmental monitoring data.

5 USING HYDROGEOLOGICAL MODELS TO ASSESS THE EFFECTIVENESS OF NITRATES DIRECTIVE AND FULFIL ITS GOVERNANCE GAPS[2]

5.1 Background

The invisibility of groundwater and the inscrutability of the aquifers in their entirety are real challenges for effective governance and management (Kroepsch 2018). Despite currently available technologies provide high quality data on sub-surface characteristics and properties, these data are limited to restricted portions of the aquifers (Tidwell and van den Brink 2008). Three dimensional hydrogeological models have been increasingly used over the last decades as an approximation of reality to simulate groundwater flows and contaminant transport. By overcoming the limitations of a patchy knowledge of aquifers, they allow us to represent and understand the sub-surface water dynamics (Baalousha et al. 2008). As regards groundwater contamination by nitrates, a closer look to the recently developed models reveals that they were mainly used with two aims. The first is to understand processes and mechanisms influencing nitrate transport and concentrations (e.g. Jang et al. 2017; Paradis et al. 2018). The second is to explore the implication of alternative management actions to be adopted, by using predictive scenarios in order to anticipate the responses of the system to changes (e.g. Krause et al. 2008; Zhang and Hiscock 2011; Bailey et al. 2015).

On the contrary, to our knowledge, recent literature does not explore their role to assess the effectiveness of the current policies. Similarly, although the urgency to shift from management to governance has been strongly underlined in hydrogeological literature (Mukherji and Shah 2005; Foster and Garduno 2013; Garrick et al. 2017) and the strengthen of governance is required to enhance ND effectiveness (EU Commission 2018), within the governance-oriented debate hydrogeological models are generally seen and used as tools to provide management recommendations (Bear et al. 1992; Baalousha et al. 2008; EU Commission 2018).

Therefore, considering the capacity of models to overcome a fragmentary vision of sub-surface dynamics, this chapter aims (i) to evaluate ND performance by using hydrogeological flow and transport models, and (ii) to consider if and how hydrogeological models contribute to improve the governance process supporting ND application.

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[2] This chapter contains the information of: **Musacchio, A.**, Mas-Pla, J., Soana, E., Re, V., & Sacchi, E. Hydrogeological modelling provides essential knowledge to fulfil governance gaps in applying EU Nitrates Directive. Insights from the Lombardy Plain (N Italy) *In prep.*

To this end, a groundwater flow and transport model of a large portion of the Adda basin is developed. The model is built up as a mean to contribute to groundwater governance, providing a broad comprehension of the major dynamics (both anthropic and natural) controlling nitrate contamination as well as the magnitude of their contribution, rather than the specific responses of the system to possible management decisions. In order to connect hydrogeological dynamics of contamination to agricultural practices, the model is coupled with a nitrogen budget estimation on agricultural soils. Finally, besides the well-functioning of the model, data are introduced so as to allow the model to deal with potential variations of the aquifer stresses that derive from the regional application of the ND (i.e. municipal scale).

5.2 Conceptual framework

To speculate on the role of an hydrogeological model in sustaining the governance involved in ND application, its contribution to fulfill governance gaps is evaluated. To consider governance as a mean to ensure the protection and sustainable use of groundwater in the long term, implies to assume that governance performance can reach different degrees of effectiveness in supporting policies at stake (Akhmouch et al. 2018). Several integrated frameworks are currently available to analyse how "good" are the governance processes (Grafton et al. 2019). Among all, the 'Multi-level Governance Framework' developed by the Organization for Economic Co-operation and Development was adopted as reference point (OECD; Akhmouch 2012). This analytical framework, already employed in 30 OECD and non-OECD countries, aims to diagnose governance challenges, by grouping them in seven major gaps: policy, administrative, information, accountability, objective, funding and capacity gaps. Details on each one of them is reported in Table 5.1. The vision on governance enclosed in this set of aspects is hence used as a conceptual reference to understand the role of hydrogeological models in supporting and improving the governance process of ND application.

Table 5.1 Key implementation gaps in water governance (modified from Akhmouch and Correia 2016).

Gap	Description
Administrative gap	Geographical "mismatch" between hydrological and administrative boundaries. This can be at the origin of resource and supply gaps
Information gap	Asymmetries of information (quantity, quality, type) between different stakeholders involved in water policy, either voluntary or not
Policy gap	Sectoral fragmentation of water-related tasks across ministries and agencies
Capacity gap	Insufficient scientific, technical, infrastructural capacity of local actors to design and implement water policies as well as relevant strategies
Funding gap	Unstable or insufficient revenues undermining effective implementation of water responsibilities at sub-national level, cross-sectoral policies, and investments requested
Objective gap	Different rationales creating obstacles for adopting convergent targets, especially in case of motivational gap (referring to the problems reducing the political will to engage substantially in organising the water sector)
Accountability gap	Difficulty ensuring the transparency of practices across the different constituencies, mainly due to insufficient users' commitment' lack of concern, awareness and participation

5.3 Study area

The study area concerned by the hydrogeological model is located East of Milan and includes a large part of the Adda basin (Figure 5.1 b and c). The Adda river is the fourth in length among the Italian rivers, and the second contributor to the Po river in terms of discharge.

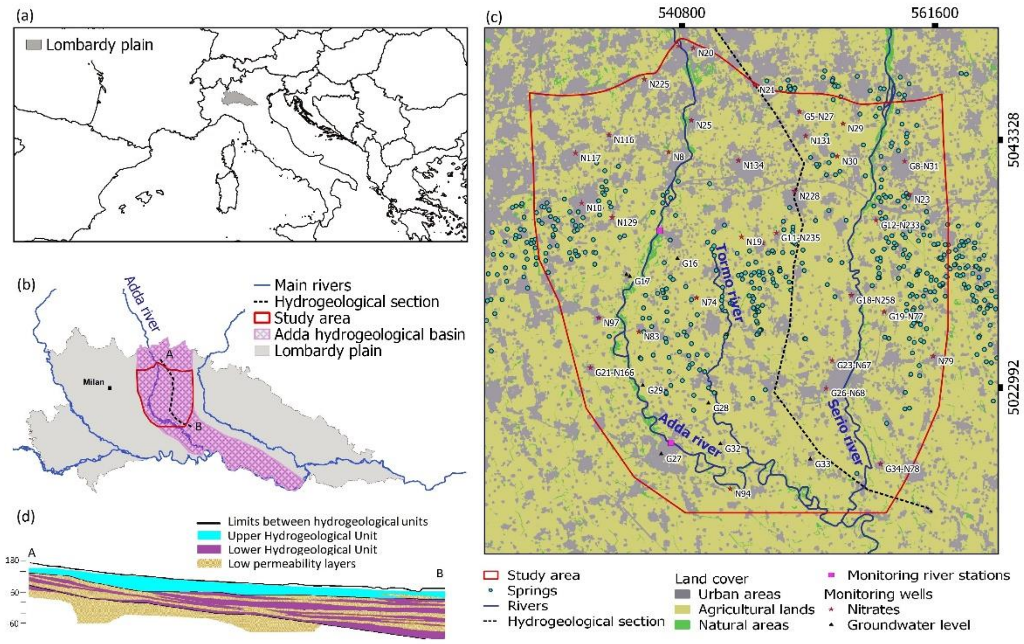

Figure 5.1 (a) Location of Lombardy plain; (b) model domain within the Lombardy plain and the Adda hydrogeological basin; (c) main hydrological characteristics, land uses and monitoring wells for both groundwater levels and nitrate concentrations; (d) hydrogeological section simplified from Éupolis Lombardia 2015.

The river outflows from the Como lake, and is regulated by a dam. This allows the water storage during winter and release during summer, when it is distributed to the fields through a dense network of channels. Irrigation water represents a significant portion of the recharge of the shallow aquifers: for example, in the adjacent basin of the Oglio river, with very similar hydrogeological features and land use, Rotiroti et al. (2019) estimated it provides between 55 and 88% of the aquifer recharge, and positively affects groundwater quality in the higher plain by diluting groundwater nitrate contents. Natural discharge of the shallow aquifers occurs through the numerous outflows in the springs' belt area (Figure 5.1c), through the draining rivers, especially the Adda river (Alberti et al. 2016; De Caro et al. 2017), through recharge towards the intermediate and deeper aquifers, and through recharge of the lower plain aquifer. Moreover, numerous wells for drinking and agricultural purposes are located in the area, constituting an additional artificial discharge of both the shallow and intermediate aquifers.
The area boundaries for the hydrogeological model were set as follows:

- the northern boundary corresponds to the piezometric contour line of 145 m derived from the hydrogeological map realized by Éupolis Lombardia (2015) for the May 2014 campaign (high piezometric level; Figure 3.4);
- the western and the eastern boundaries correspond to the groundwater divides identified in the hydrogeological study of the Lombardy plain by Éupolis Lombardia (2015);
- the southern boundary is set to include the southern portion of the middle plain, where natural denitrification processes are known to occur (Sacchi et al. 2013).

The model considers the shallow and intermediate aquifers, hereinafter respectively referred as Upper Hydrogeological Unit (UHU) and Lower Hydrogeological Unit (LHU). A schematic N-S hydrogeological section with the considered aquifers is reported in Figure 5.1d. Although not fully covering the Adda basin, the study area encompasses the transition between the high and the middle plain, and is therefore representative of the surface-groundwater exchanges occurring in the springs' belt area throughout the Po plain. Land use (Fig. 1c) is mostly agriculture (74.8%), where the dominant crop is maize, and urban/industrial (19.7%).
According to Viaroli et al. (2018), the discordant environmental policies at regional scale caused a resource relocation in the watershed, leading nutrient sources to concentrate in few hot spots, one of which corresponds to the study area under investigation.

5.4 Methods

5.4.1 Nitrogen surplus estimation

Nitrogen surplus estimation is conducted according to the soil nutrient budget approach proposed by (Oenema et al. 2003) and based on comparing nitrogen inputs and outputs affecting the agricultural lands within the study area. Estimations are performed at municipal scale and then aggregated. Municipal scale is the resolution at which both census data are usually available and the ND restrictions are applied. Budget estimations of those municipalities partially included in the area are weighted based on the agricultural surface amount. The budget is calculated on an annual basis as follows:

$$N_{surplus} = N_{Man} + N_{Fert} + N_{Fiz} + N_{Dep} + N_{Harv} + N_{Vol} + N_{Den}$$

where:
N_{Man}: N in livestock manure generated in the study area
N_{Fert}: synthetic N fertilizers applied to agricultural lands
N_{Fix}: agricultural N_2 fixation associated with N-fixing crops
N_{Dep}: atmospheric N deposition
N_{Harv}: N exported by crop harvesting
N_{Vol}: NH_3 volatilization
N_{Den}: denitrification

Both input and output terms, as well as the overall budget, are expressed in unit of mass in time (t N y^{-1}). The budgets per unit of area are calculated by dividing annual fluxes in each municipality by the productive agricultural surface available (t N ha^{-1} y^{-1}).
As regards input terms, N_{Man} is estimated by means of data of livestock density (divided in 8 major categories and more than 35 sub-categories according to their type, age, and purpose; ISTAT 2010), average live weights and N excretion rates of each animal category (DM 07/04/2006, decree of the Italian Ministry of Agricultural and Forest Policies about agronomic utilization disciplinary, Table I). Due to lack of data on manure relocation between farmers, we assume that livestock manure is only used for spreading in agricultural lands of the municipality in which the farms are located. The annual N input by synthetic fertilizers (N_{Fert}) is estimated by using provincial data on fertilizers sales (ISTAT 2014) by considering the N content of each type of fertilizer and the extent of agricultural lands potentially fertilized in each municipality (i.e. permanent crops plus arable land with the exception of N-fixing crops-alfalfa, soybean and legumes, only in case of N fertilizations). Data on agricultural lands used for crops production are obtained from the regional database (Agricultural Information System of Lombardy Region, SIARL 2014,

www.siarl.regione.lombardia.it) which is yearly updated. To estimate the N_{Fix} associated with alfalfa, soyabean, permanent grasslands and pastures the production per unit of surface (i.e. the yield of each specific N-fixing crop; ISTAT 2014) is multiplied by the N content in the harvested portions (Salvagiotti et al. 2008; Borreani et al. 2009; Sulieman and Schulze 2010). N input from atmospheric deposition (N_{Dep}) is calculated by multiplying average values of atmospheric oxidized N deposition by the total agricultural surface. The first term is setted in 8.5 kg N ha-1 yr-1, accordingly to both the data extrapolated from the EMEP-Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (http://www.emep.int) and the Italian Code of Good Agricultural Practices (Decree of the Italian Ministry of Agricultural and Forest Policies 04/19/1999).
As regards output terms, the estimate of N_{Harv} considers the area of each crop type (SIARL 2014), their N content and annual yield (ISTAT 2014). NH_3 volatilization produced by fertilizers application (N_{Vol}) is calculated by using ammonia volatilization factors (expressed as the average percentage lost of the supplied N) of 32.5%, 16%, 6% and 2% associated to animal manure, urea plus ammonium sulphate, nitrates and other N fertilizers, respectively. These values were coherent to direct emission estimations performed in the Italian territory, as reviewed by Minoli et al. (2015). About 60% of the volatilised ammonia was assumed to be re-deposited locally and only the remaining 40% was considered as a true N output from the agricultural system. N output from denitrification in soils (N_{Den}) is calculated as 10% of the supplied N as both synthetic fertilizations and animal manure, spanning the main soil types of the Po River, according to a review of a vast body of literature (Castaldelli et al. 2013). Additional information on nitrogen budget estimation is provided in the Supplementary Materials.

5.4.2 Numerical flow and transport model

The numerical flow and solute transport model uses the MODFLOW (McDonald and Harbaugh 1988) and MT3DMS (Zheng and Wang 1999) codes, respectively, under the processing software Groundwater Vistas v6.96 (Rumbaugh and Rumbaugh 2011). Data treatment to create input files, and result postprocessing was conducted using the QGIS software (QGIS Development Team 2018).
Numerical models are based on a finite difference grid with horizontal cell size of 100x100 m, and variable depth according to the thickness of the six layers that represent the hydrogeological units. In particular, the UHU is discretized in 3 layers. Their cells have minimum and maximum thicknesses of 3.41 and 75.34 m. Similarly, the LHU is also divided in 3 layers with a range of cell thicknesses between 11.57 and 154.26 m. The elevation of the upper layer is taken from the digital terrain model with 5-m grid width, supplied by Lombardy Region (2015), and the boundary between the two units is obtained from Éupolis Lombardia (2015). References to the behaviour of the UHU will relate to model layer 2, as layer 1 becomes dry in some areas because of topographic elevations.
The total number of active cells in the model is of 642,237. Time discretization is based on monthly stress periods that describe the recharge and groundwater withdrawal regimes. A 10 years simulation from 2008 to 2017 is simulated in this analysis with a total simulation time of 120 months, as stress periods.
Hydrogeological zones have been determined according to the geological information resulting from the published geological map (Lombardy Region 1990), and from former original research (Éupolis Lombardia 2015). Hydraulic conductivity (K), storage coefficients (S), specific yield (S_y), and porosity (n) values and their spatial distribution were initially taken from published geological data (Canepa 2011; Alberti et al. 2016; De Caro et al. 2017), and later on calibrated based on the available potentiometric data provided by the Regional Agency for Environmental Protection (ARPA). The property zones for layers 2 and 5, and the corresponding values for K, S_y and n are illustrated in Figure 5.2. The zonation of the UHU, represented by layer 2, reflects the shape of the distinct geological units. It has a larger detail than zones in the LHU,

represented by layer 5, where the progression to finer sediment units towards the Po River plain has been prioritized. K is considered anisotropic with a vertical anisotropy ratio (K_h/K_v) of 10:1 (Alberti et al. 2016). As regards the modelling boundary conditions, the northern limit of the model is represented by a general head boundary condition (GHB) in all layers. In the southern limit, a constant head boundary condition was imposed in the first three layer (UHU) in those cells located between the Adda and Serio rivers' watercourses. The eastern and western edges of this boundary are set as no-flow boundaries, giving preference to the groundwater flow along the most recent alluvial floodplain. LHU cells in the southern limit are defined as constant head cells. Existing potentiometric maps (Éupolis Lombardia 2015) determine the head values assigned to each cell in these boundaries. This potentiometric information also sets the eastern and western hydrogeological limits of the modelled region, defined as no-flow boundaries according to the regional flow lines that finally drain to the Po River alluvial aquifer.

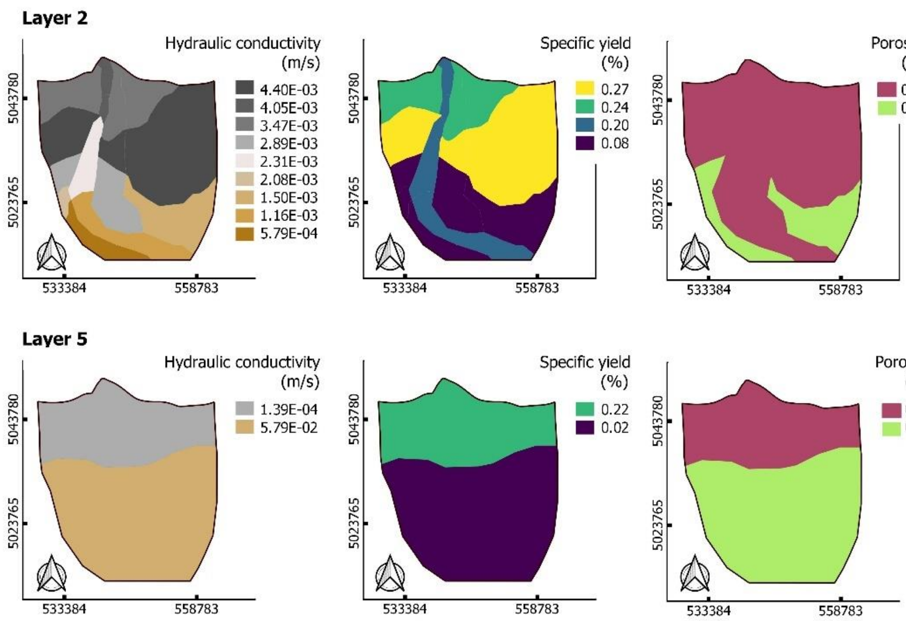

Figure 5.2 Zones of layers 2 and 5, values of hydraulic conductivity (K), specific yield (Sy) and porosity (n).

The net recharge term in the upper layer is estimated as a combination of all parameters that determine the water budget at the model surface; namely, rainfall (R), actual evapotranspiration (AET), irrigation from surface water (I_{rr}), and agricultural water withdrawal (Q_{ag}); that is, Net recharge = (R + I_{rr}) – (AET + Q_{ag}). The two terms related to irrigation (I_{rr} and Q_{ag}) are available at a municipality scale in the last national General Census of Agriculture (ISTAT 2010). Indeed, recharge zones that include natural as well as land use-related parameters, are defined according to the municipality limits, as the existing information (i.e., data bases of irrigation from surface and ground water, groundwater withdrawal, nitrogen input) are estimated at this administrative level. This is a useful setup for models devoted to advice governance actions, as the ND is administratively applied at a municipal level and the local scale plays a key role in disseminating good agricultural practices (Musacchio et al. 2019a). Given the areal extension of the 125 municipalities in this part of the Lombardy Plain (8.57±0.64 km^2), this "zone unit" has an acceptable size to represent the spatial variability of the water and solute balance components.

Monthly rainfall and temperature data were obtained from ARPA Lombardy (2018) from 6 observatories located within or nearby the study area. Actual evapotranspiration (AET) using the Zhang et al. (2001) approach was based on the values of the monthly mean potential evapotranspiration (PET), monthly mean precipitation (P) and a coefficient for each land use (w). PET was estimated by the well-known Thornthwaite equation at each weather observatory using the monthly mean temperature data. Land use coefficient values (w) were adopted from those listed in Zhang et al. (2001) and modified to the K_{crop} coefficient in agricultural lands (Gharsallah et al. 2013; Perego et al. 2014).

Mean annual precipitation is of 972.8±91.2 mm. Mean annual AET represents 40.0±2.7% of the mean annual rainfall. Mean agricultural groundwater extraction from the UHU and irrigation rates using surface water during the crop-growing season amounts to 13.5±3.6 and 182.3±10.9 mm y^{-1}, respectively. Monthly variability of these terms in all municipalities for the period 2008-2017 is shown in Figure S3.1 (in Supplementary Materials).

Groundwater for urban supply is withdrawn from wells located in the LHU. The annual exploitation regime of each municipality is extracted from the databases of the National Institute of Statistics for year 2015 (ISTAT 2015), and it is considered constant over the simulation period. Mean and median values for this component considering all municipalities are 2,115.1±230.8 and 1178.1 m^3 d^{-1}, respectively.

Main streams (Adda, Serio and Tormo rivers) are defined using the River routine of the MODFLOW code. The stage of each river cell was set equal to the mean ground elevation of the river given by the digital terrain model and the depth of the rivers estimated by using topographic data. Cell river conductance has been calibrated through the simulation process, ranging from 75,000 to 4,500,000 m^2 d^{-1}, with a mean value of 1,132,199±26,090 m^2 d^{-1}, and a median of 1,200,000 m^2 d^{-1}. 85% of the river conductance values are below 2,000,000 m^2 d^{-1}.

Fontanili are relevant hydrological components in the Po Plain that represent a groundwater outflow in the transition zone between the northern part of the plain, characterized by coarser materials, and the southern plain, where sediments are finer. Some of them are natural groundwater outflows, mostly due to the decrease in permeability from north to south, while others have been created since the Middle Ages as irrigation water supply sources by excavating dips and/or installing tubular artefacts (Balestrini et al. 2016). In both cases, they presently drain water from the hydrogeological system. Discharge from *fontanili* is partially used for local irrigation, and the non-used resource feeds the drainage network through channels or natural streams. The Drain routine is assigned to these singular cells located in the model upper layer. Drain conductance values range from 2,700 and 7,350,000 m^2 d^{-1}, with a mean value of 215,079±28,163 m^2 d^{-1}, and a median of 62,550 m^2 d^{-1}. 85% of the conductance values in the drain cells are below 300,000 m^2 d^{-1}.

Regarding the transport model, a general longitudinal dispersivity value is set at 15 m after its relationship with the scale of the field site (Gelhar et al. 1992); in this case, the scale is taken as the cell width (100 m). Transversal dispersivity is considered as 1/3 of the longitudinal value. Initial nitrate concentration is estimated at a municipal scale by interpolating the 2006-2007 field data (ARPA Lombardy 2018b). These values have been slightly adjusted during the calibration especially in the northern boundary. Initial nitrate mean concentrations finally used for each hydrogeological unit are as follows, UHU: 25.72±9.25 mg L^{-1}; LHU: 17.21±8.87 mg L^{-1}, while the original data statistics were UHU: 26.67±7.69 mg L^{-1}; LHU: 18.31±8.70 mg L^{-1}.

Nitrogen monthly inputs through infiltration, as nitrate, are introduced as a concentration linked to the monthly net recharge. The model assumes that the annual mass of nitrate reaching the water table estimated from municipality records can be equally distributed on a monthly basis. Therefore, the monthly input concentration varies proportionally to the monthly net recharge rate, so that the same nitrogen mass is introduced every month. Consequently, recharge concentration is larger in dry than in wet months. Moreover, the model disregards potential soil nitrogen stocks from previous fertilization periods. This assumption is consistent with previous studies which show that nitrate concentration in groundwater is strictly related to weather variability and water use for agricultural purposes, permitting to exclude the

possibility of significant nitrogen retention in agricultural soils (Balestrini et al. 2016; INTEGRON Project 2019; Rotiroti et al. 2019).
The nitrogen surplus value estimated at municipality level using data of 2010 and 2014 is uniformly applied along the entire simulation (2008-2017), which hampers the accurateness of the 10-year long simulation. Additionally, nitrate concentration at the GHB is assigned equal to the initial concentration at that cell for the entire simulation. Some initial nitrate concentrations along the GHB have been calibrated to fit with the observed data in monitoring wells. Transport simulation used nitrate units, yet results are translated to mass of nitrogen in the discussion.
Denitrification occurring in the southern part of the study area has been estimated in percentages between 25 and 80% using multi-isotopic data (Sacchi et al. 2013). Considering the results obtained by Carroll et al. (2009), nitrate half-life has been set to 0.96-1.92 y^{-1} in UHU zones where denitrification had been reported. The flow model has been calibrated using potentiometric data from 19 wells, all of them monitored at least 33 months out of the 120 simulated by the model. In turn, the transport model uses nitrate data from 33 wells for calibration, each of them sampled from 1 to 3 times per year (ARPA Lombardy 2018b). The model performance is evaluated using the coefficient of determination (R^2), the root–mean-square error (RMSE), and the Nash-Sutcliffe coefficient of efficiency (NSE; Moriasi et al. 2007). Mean river and drain (*fontanili*) discharge were compared with field data collected during the INTEGRON Project, ARPA monitoring data, and results of previous studies conducted in the Lombardy plain (Alberti et al. 2016; De Caro et al. 2017).

5.5 Results

5.5.1 Nitrogen surplus estimation

The annual nitrogen input to agricultural land is 32,079 t, corresponding to a mean pressure of 366.7 kg N ha^{-1} y^{-1}. At municipal scale, the input ranges from 102.8 to 939.4 kg N ha^{-1} y^{-1}, with a median value of 338.9±11.2 kg N ha^{-1} y^{-1} (Figure 5.3). In most municipalities classified as vulnerable according to the ND application, the annual N input produced by fertilizers and manure widely exceeds the ND restrictions, and the mean input in these municipalities is higher than that of the non-vulnerable ones (Table 5.2). Considering the entire area, livestock manure represents the main input (56.9%; Figure 5.4), followed by synthetic fertilizers (35.0%). Biological fixation and atmospheric deposition are minor terms, respectively 5.8% and 2.3%. At municipal scale, livestock manure constitutes 49.3±4.4% of the budget, with maximum of 81.1%, and synthetic fertilizers are 41.1±3.7%, with maximum of 87.7%.
The total annual nitrogen export from agricultural lands corresponds to 71.0% of the input (22771 t; 260.3 kg N ha^{-1} y^{-1}). At the municipal level, exports range from 80.21 to 421.5 kg N ha^{-1} y^{-1} (median: 254.1 kg N ha^{-1} y^{-1}). The main export term is represented by crop harvesting, both considering the whole area (73.2%) and the municipalities (75.1±6.7%). The contribution to the export of NH_3 volatilization and denitrification in soils is roughly the same, respectively 13.8% and 12.9% for the entire area, and 12.5±1.1% and 12.4±1.1% at the municipal scale.
The overall nitrogen budget is positive, with nitrogen input exceeding export, and generating an annual surplus of 9,308 t, equal to 29.9% of the input and corresponding to 106.4 kg N ha^{-1} y^{-1}. At municipal scale, the surplus ranges from -40.2 kg N ha^{-1} y^{-1} to 517.9 kg N ha^{-1} y^{-1}, with a median value of 90.9 kg N ha^{-1} y^{-1}, and municipalities classified as vulnerable present on average higher surpluses (Table 5.3).

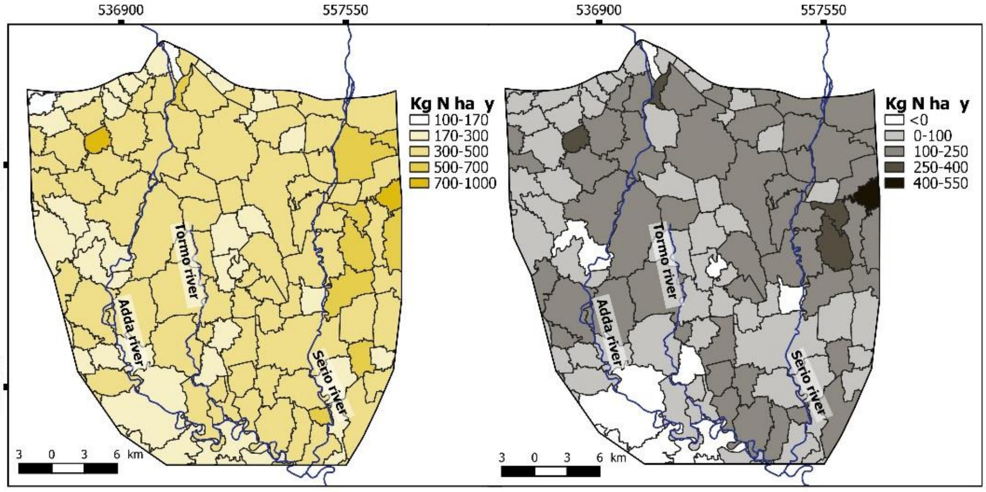

Figure 5.3 Nitrate input (a) and surplus (b) distribution at municipal level.

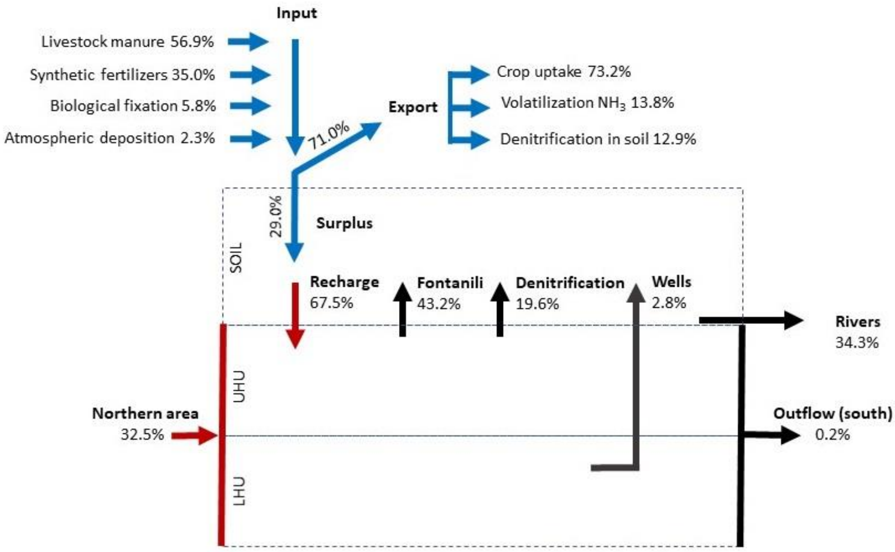

Figure 5.4 Diagram of the N mass balance in the studied area of the Lombardy Plain. Blue arrows (and the associated percentages) refer to the budget of nitrogen on the agricultural soils, according to the Soil System Budget. Red arrows refer to the input to the aquifers, and black arrows to the output from the aquifers according to the numerical transport model. Percentages associated to blue arrows refers to the soil nutrient budget estimation; percentages associated to red and black arrows are referred to the mean annual inflow and outflow mass budget estimated by the numerical model.

Table 5.2 Input of nitrogen (kg N ha-1 y-1) from manure and fertilizers in the non-vulnerable (nNVZs), partially vulnerable (pNVZ), and vulnerable (NVZ) municipalities. The percentage of municipalities exceeding the ND limit for N input from livestock manure (170 kg N ha-1 year-1 in NVZs and pNVZs, and 340 kg N ha-1 year-1 in nNVZs) is reported.

	Limit ND (kg N ha^{-1} y^{-1})	Manure and fertilizer input (kg N ha^{-1} y^{-1})			Municipalities exceeding ND limits (%)
		Min	Mean	Max	
nNVZ	340	140.3	285.9±28.8	758.2	19.0
pNVZ	170	177.4	301.1±29.0	409.6	100
NVZ	170	86.6	331.6±13.0	910.5	94.8

Table 5.3 Surplus of nitrogen (kg N ha-1 y-1) from manure and fertilizer in the non-vulnerable (nNVZs), partially vulnerable (pNVZ), and vulnerable (NVZ) municipalities. The percentage of municipalities having a surplus is reported.

	Manure and fertilizer surplus (kg N ha^{-1} y^{-1})			Municipalities with a surplus (%)
	Min	Mean	Max	
nNVZ	-40.2	85.3±18.7	385.7	90.5
pNVZ	-31.3	73.6±21.7	151.4	87.5
NVZ	-30.3	105.5±8.7	517.9	92.7

5.5.2 Numerical flow model

The hydraulic head distribution simulated by the flow model (Figures 5.5 and 5.6) reproduces the observed potentiometric data with accuracy (Figure 5.7). Observed versus simulated head values fall on a 1:1 slope line (R^2: 0.99; RMSE: 1.99; NSE: 0.99; Fig. 5.7a). The mean head difference between observed and simulated values is of -0.94±0.04 m, and with 87.1%% of the data points within a difference range of ±3.0 m which is considered acceptable for a regional scale model.

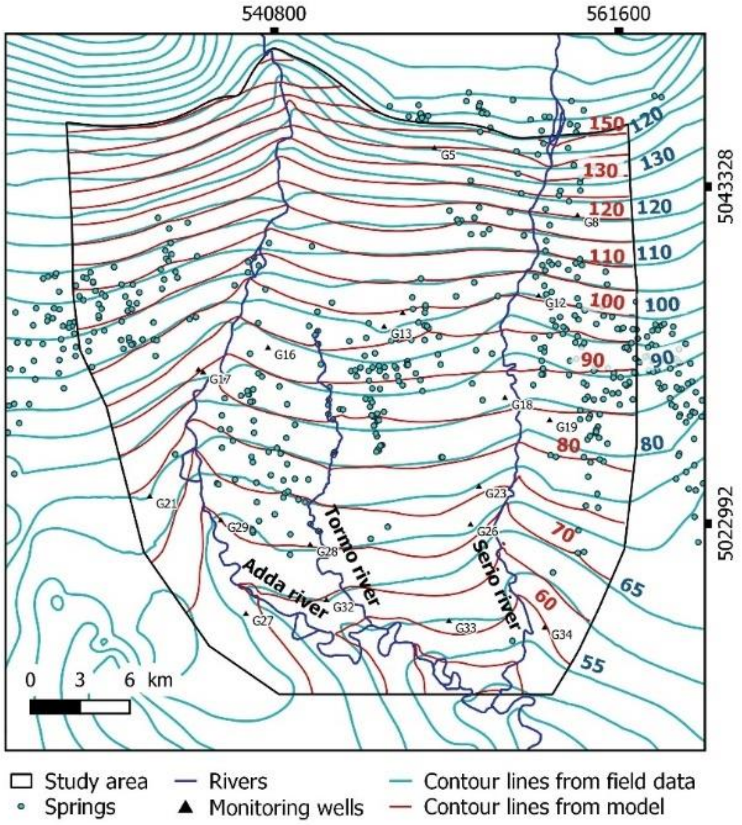

Figure 5.5 Contour maps of groundwater levels (m) estimated by field data (red lines and values) and from model data (light blue lines and values).

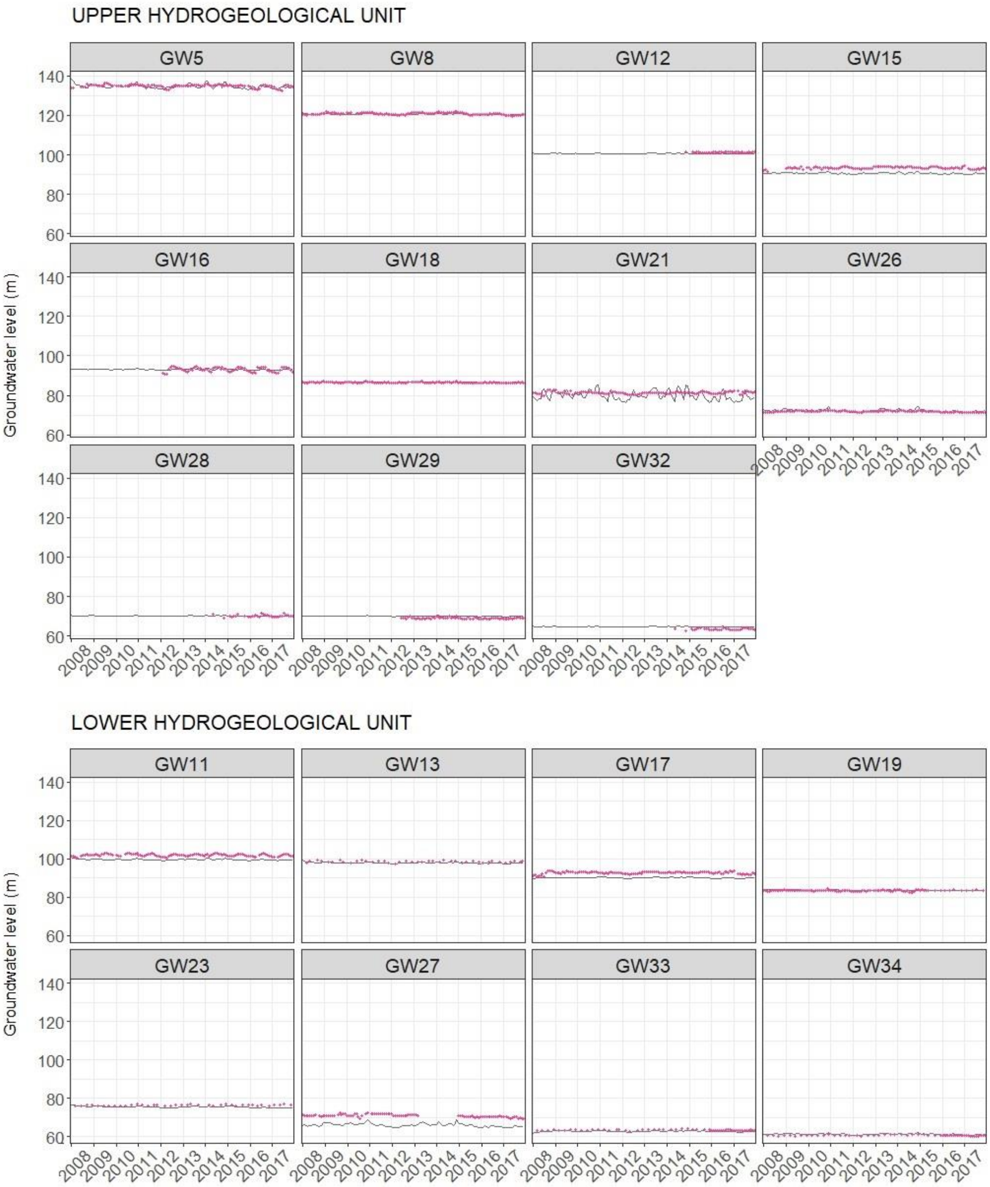

Figure 5.6 Observed (pink dots) versus simulated (dark grey line) potentiometric head values for representative control wells at the UHU and LHU.

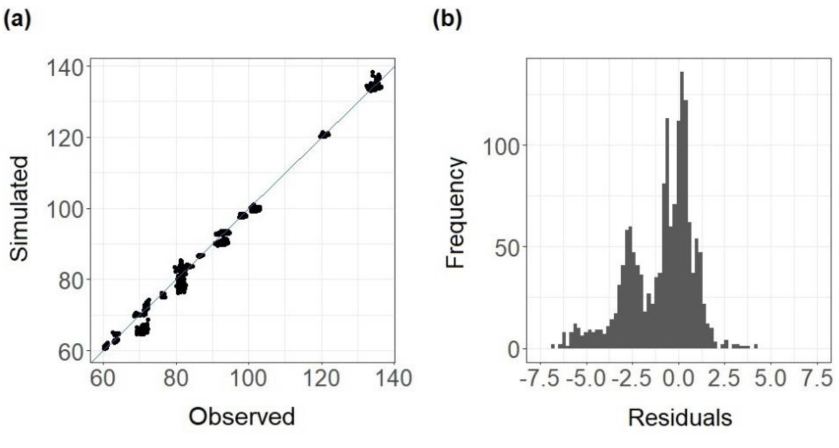

Figure 5.7 (a) Observed versus simulated potentiometric head values for representative control wells at both UHU and LHU; (b) Frequency plot of the residuals, as the difference between simulated minus observed values.

Rivers, where they recharge the aquifer, represent the third recharge term in magnitude (1.73±0.03 $hm^3\ d^{-1}$, 27.6%). Nevertheless, simulated rivers mainly act as gaining streams with a mean net flow term (outflow minus inflow) of 1.6±0.1 $hm^3\ d^{-1}$. In turn, *fontanili*, as drains, have a mean output rate of 2.28±0.08 $hm^3\ d^{-1}$. Both terms correspond to 53.1% and 36.4%, of the total outflow from the model domain, respectively. Aquifer exploitation from the LHU for urban supply, given by the well term, has a mean rate of 0.2 $hm^3\ d^{-1}$ (3.3% of the total outflow). Finally, storage represents 7.4% of the total inflow (0.46±0.07 $hm^3\ d^{-1}$), and 6.6% of the total outflow (0.42±0.05 $hm^3\ d^{-1}$).

Estimated river and *fontanili* outflows satisfactorily fit with available measured data (Table 5.4). Moreover, the mean flow rate of Adda river measured by ARPA and Adda Consortium from 2008 to 2015 in two monitoring stations (Figure 5.1) is 13.6 $hm^3\ d^{-1}$.

The vertical flow between UHU and LHU is mainly advantageous to the UHU, as there is a differential flow regime of 0.24 $hm^3\ d^{-1}$ per year from the LHU to the UHU.

Table 5.4 Comparison between the flow of rivers and fontanili ($hm^3\ d^{-1}$) estimated by the model and those estimated by other authors. The rivers outflow estimated by the model corresponds to the outflows minus the inflows of Adda, Serio and Tormo rivers; the outflow estimated by other authors refer to Adda rivers and to a wider section of its basin. Outflow of drains were estimated by the INTEGRON partner CNR-IRSA, using field data of 2016 and 2017.

	Model	Other data	Data source
Rivers	1.70	1.08	Alberti et al. 2016
		1.37-3.29	De Caro et al. 2017
Fontanili	2.35	3.55	INTEGRON Project 2019
		3.01	De Caro et al. 2017

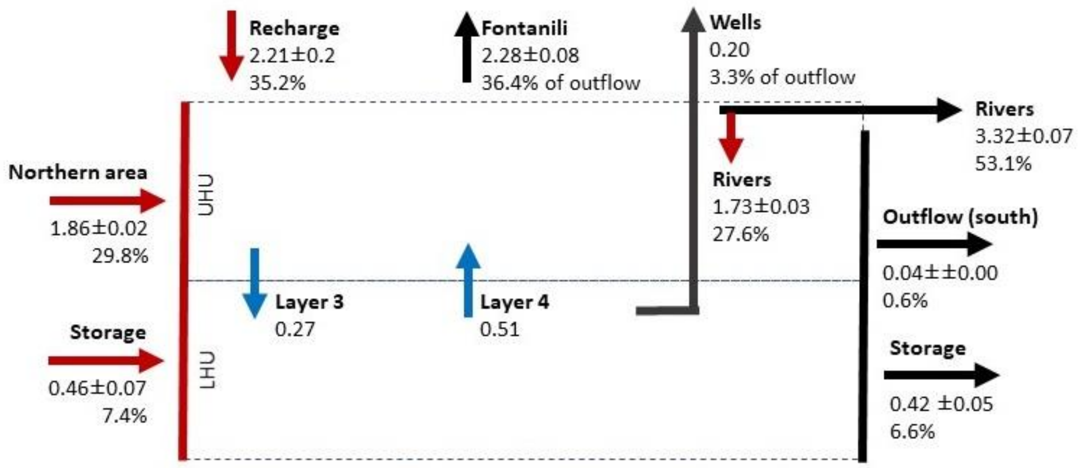

Figure 5.8 Diagram of the water mass balance. Input (red arrows) and output values (black arrows) and errors are calculated based on the mean annual values ($hm^3\ d^{-1}$). Percentages refer to the total inflow or outflow, according to the mass balance component. Blue arrows indicate the flow between hydrogeological units. Additional information are provided in the Supplementary Materials.

5.5.3 Numerical solute transport model

Nitrate distribution in the UHU, as reported by the numerical transport model, highlights the main hot-spots of nitrate pollution (Figure 5.9). Nitrate concentrations in selected monitoring wells of the UHU (Figure 5.10) display a large variability of values, usually around 20-30 mg NO_3 L^{-1}, yet even higher concentrations above the 50 mg NO_3 L^{-1} threshold also appear. Conversely, nitrate concentration in LHU is seldom found at values larger than 20 mg NO_3 L^{-1}, and some wells show a very low content (below 5 mg NO_3 L^{-1}). Interestingly, field data measured by ARPA Lombardy (2018b) show almost no variation on the nitrate concentration on the period 2008-2017 in many monitoring wells in both hydrogeological units. This uniformity has been fairly reproduced by the model, as well as nitrate increasing or decreasing tendencies wherever they were observed. Furthermore, nitrate mass exported by rivers and *fontanili* in the model highly suits available field data (Table 5.5).

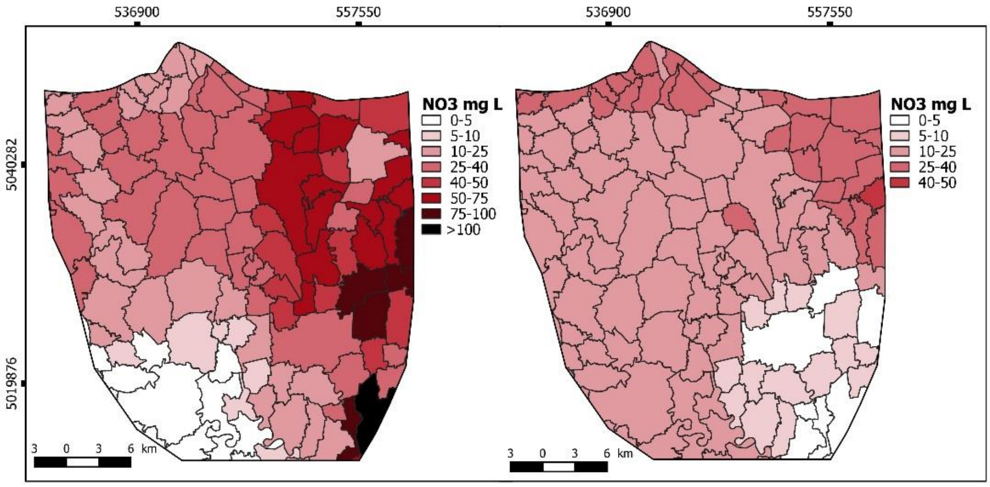

Figure 5.9 Nitrate distribution after the 10-year simulation in the UHU (model layer 2) and the LHU (layer 5).

Table 5.5 Comparison between N output mass from rivers and springs estimated by the transport model and by the INTEGRON Project using ARPA field data (2013-2014). River data estimated by the INTEGRON Project refer to the Adda River, while model data refer to Adda, Serio and Tormo rivers. Nitrogen units are expressed in t N y-1 x 104.

	Model	Other data	Data source
Rivers	2.01	1.52	INTEGRON Project 2019
Fontanili	2.52	2.19	INTEGRON Project 2019

Indeed, the relationship between observed and simulated nitrate concentrations also fall on a 1:1 slope line (R^2: 0.81; RMSE: 7.11; NSE: 0.79), with a mean difference of 1.3±0.3 mg NO_3^- L^{-1}, and with 70.0% of the data points within a difference range of ±5.00 mg NO_3 L^{-1} (Figure 5.11). All these satisfactory results determine our trust in the model and allow processing the nitrate mass balance results to describe the actual qualitative status of the hydrogeological system, under present land use and exploitation regimes.

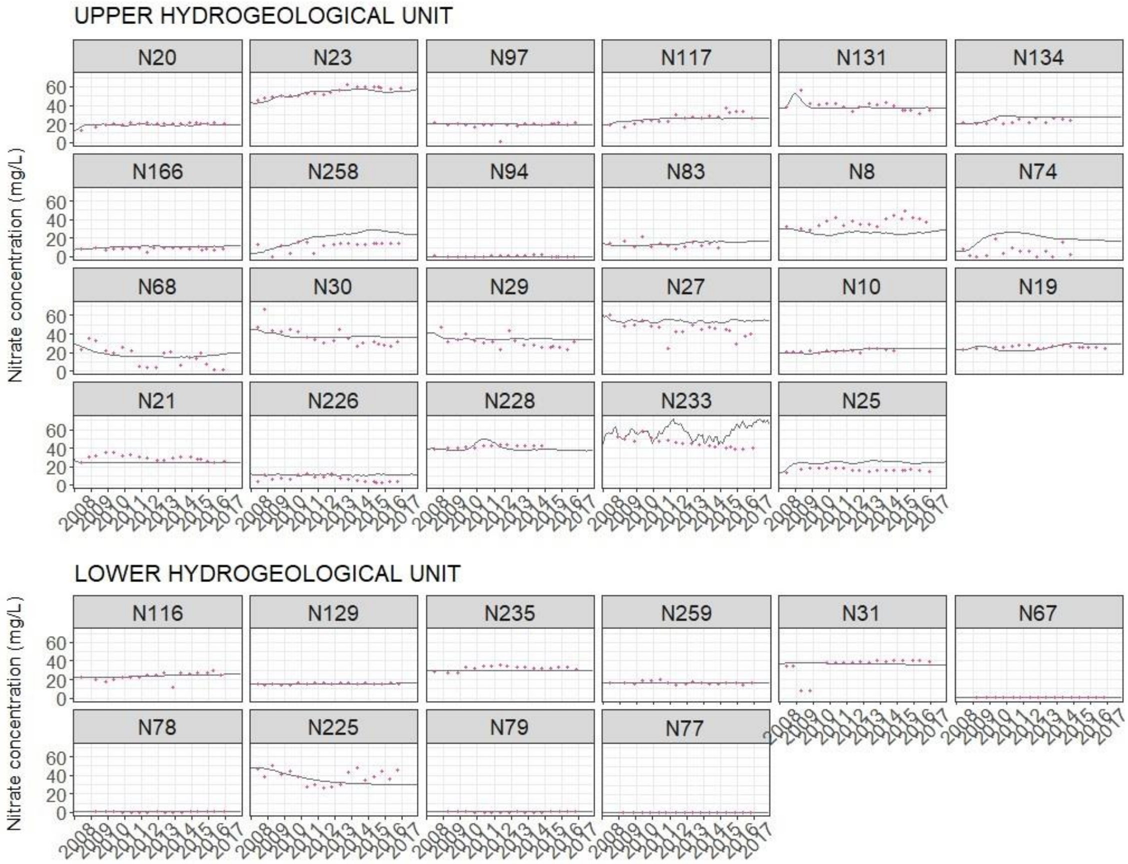

Figure 5.10 Nitrate observed (pink) and simulated (dark grey) evolution in the most representative wells from the ARPA database.

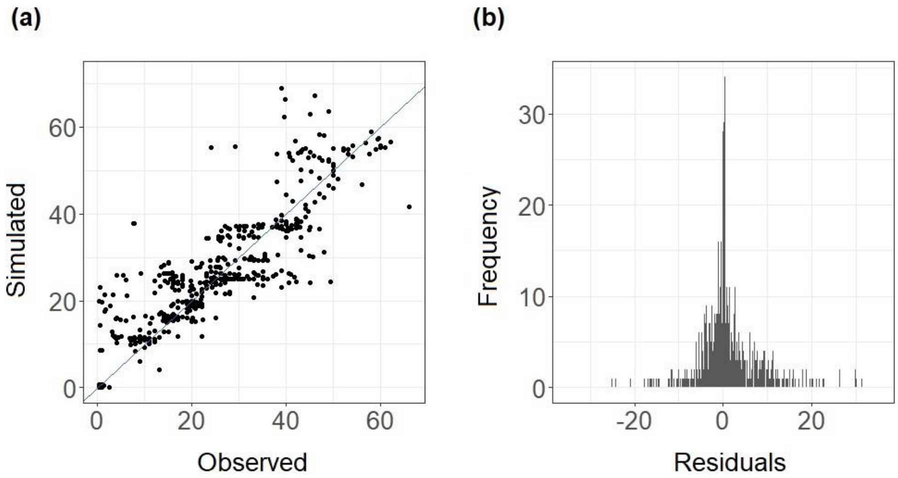

Figure 5.11 Simulated versus observed nitrate concentrations using the available field measurements in the ARPA database and the 1:1 line are shown in plot (a), and the residuals distribution, as the differences of simulated minus observed values (b).

According to the transport model (Figure 5.4, Table 5.6), the nitrogen surplus transferred from the agricultural soil to the aquifers by the net recharge term represents 67.4% of the mean annual nitrogen input.

The remaining part of the nitrogen mass (32.6%) enters the system with the groundwater flow through the northernmost GHB. The annual mean input of nitrogen to the aquifers is 13,953±38 t, which corresponds on average to the 8.7±0.1% of the mean nitrogen present in groundwater (160,147±1074 t; Table 5.7).
Nitrogen is mostly exported out from the aquifers or lost by denitrification; yet a smaller amount is stored in groundwater (Table 5.7). Exported N mass (13,309±541 t y^{-1}), on average, equals a 95.5±4.1% of the total annual amount introduced by the net recharge term. This N mass represents 8.3±0.4% of the mean total nitrogen stored in the aquifers throughout the simulation period. Consequently, a 4.5±4.1% of the annual input remains stored in the system. This increase of stored N mass represents a mean positive variation of 0.4±0.4% of the total mass existing in groundwater.
N export mainly takes place through *fontanili* (43.2±1.4%) and rivers (34.3±0.4%). The estimated N mass exported by the *fontanili* produces a median concentration throughout the 10-years period of 28.7 mg NO_3^- L^{-1}. Considering stream discharge records for 2014 and 2015 from the Adda River gauging station (data provided by the Adda Consortium), the nitrate amount exported to rivers generates a median concentration of 4.48 mg NO_3^- L^{-1}. Both values satisfactorily match with the field data reported by Musacchio et al. (2019b). Denitrification occurring in the southern sector of the UHU reaches a 19.6±1.7% of the exported N mass. Due to this geochemical process, the N losses is only 0.2±0.0% of the total exported N mass. Finally, N exported by groundwater withdrawal from the LHU for urban supply amounts a 2.8±0.0% of annual aquifers N losses. Percentages of the N mass balance reported in Table 5.6 are quite uniform over time.
The continuous variation of stored nitrogen mass in the whole modelled domain shows a significant decline during the first 3 years, and a sudden increase at annual rates larger than 1% since 2011 (Figure 5.12; Table 5.7). For caution, we accept that the early steep decline that lasts 1.4 years could be due to the early adjustment of the numerical model to the flow and transport initial conditions. Nevertheless, the continuous increase during the following months is attributed to the overall response of the system to the set boundary terms, sources and sinks that rule the model. Figure 5.13 shows the monthly output N mass from *fontanili* and river cells, being the two largest hydrogeological terms that export nitrate from the aquifers. This plot points out the larger, and more variable, contribution of *fontanili* with respect to the export capacity of the drainage network.

Table 5.6 Percentage of each hydrogeological term in the total n mass input and output flows according to the numerical transport model.

	INPUT TERMS		**OUTPUT TERMS**				
	Northern GHB	Net recharge	Drains	Rivers	Denitrification	Wells	Southern CH bound.
2008	32.3	67.7	33.95	32.48	31.27	2.13	0.17
2009	32.5	67.5	37.69	33.09	26.53	2.51	0.18
2010	31.9	68.1	42.60	33.59	21.06	2.57	0.18
2011	32.8	67.2	42.69	33.63	20.46	3.03	0.19
2012	32.8	67.2	43.61	34.18	18.95	3.08	0.18
2013	32.5	67.5	47.51	34.18	15.42	2.71	0.18
2014	31.1	68.9	48.73	34.35	14.14	2.60	0.18
2015	33.6	66.4	45.20	35.78	15.82	3.01	0.19
2016	32.3	67.7	45.07	34.74	16.90	3.10	0.18
2017	33.6	66.4	44.45	36.72	15.46	3.19	0.19
MEAN (±ERROR)	32.5 (±0.2)	67.5 (±0.2)	43.2 (±1.4)	34.3 (±0.4)	19.6 (±1.7)	2.8 (±0.1)	0.2 (±0.0)

Table 5.7 Nitrogen mass flow into and out of the aquifers per year, and cumulative total mass in aquifers as given by the numerical transport model. Percentages refer to the input, output and stored mass fraction related to the total N mass within the aquifers at a given year.

	Total N mass inflow ($t\ N\ y^{-1}$)	Total N mass outflow ($t\ N\ y^{-1}$)	Total N mass in aquifers (t N)	Input (%)	Export (%)	Stored (%)
Initial value	--	--	*$1.598x10^5$*	--	--	--
2008	$1.387x10^4$	$1.705x10^4$	$1.568x10^5$	8.84	10.87	-2.03
2009	$1.393x10^4$	$1.444x10^4$	$1.563x10^5$	8.92	9.24	-0.32
2010	$1.382x10^4$	$1.417x10^4$	$1.560x10^5$	8.86	9.08	-0.22
2011	$1.404x10^4$	$1.204x10^4$	$1.578x10^5$	8.90	7.63	1.27
2012	$1.405x10^4$	$1.184x10^4$	$1.602x10^5$	8.77	7.39	1.38
2013	$1.386x10^4$	$1.358x10^4$	$1.605x10^5$	8.64	8.46	0.18
2014	$1.375x10^4$	$1.417x10^4$	$1.603x10^5$	8.58	8.84	-0.26
2015	$1.410x10^4$	$1.193x10^4$	$1.623x10^5$	8.69	7.35	1.33
2016	$1.404x10^4$	$1.202x10^4$	$1.644x10^5$	8.54	7.31	1.23
2017	$1.406x10^4$	$1.184x10^4$	$1.668x10^5$	8.43	7.10	1.33
Mean: (±error)	1.395x104 (±38)	1.331x104 (±541)	1.601x105 (±1.074x103)	8.72 (±0.05)	8.33 (±0.38)	0.39 (±0.36)

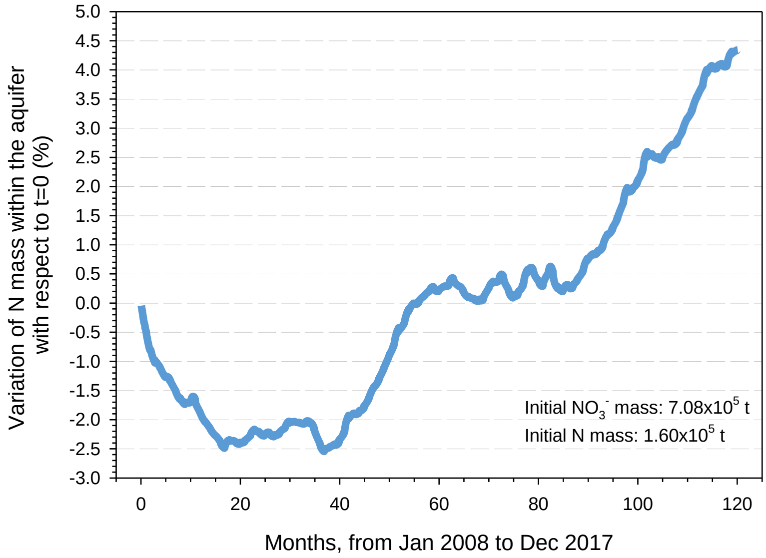

Figure 5.12 Monthly changes, in percentage, on N mass stored in the aquifers with respect to the mass at the beginning of the simulation (t=0).

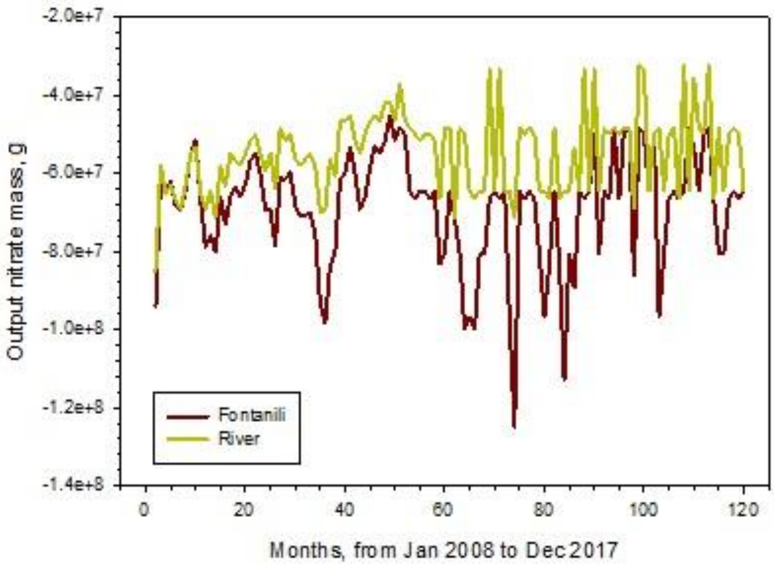

Figure 5.13 Monthly nitrate mass output through the fontanili and streams (drain and river boundary conditions, respectively). Negative mass expresses outputs from the system in grams.

5.6 Discussion

Hydrogeological models have been increasingly used over the last decades due to their benefits for groundwater resources protection. As regards groundwater contamination by nitrates, until now the use of hydrogeological models has been mostly restricted to the analysis of mechanisms and processes influencing nitrate contamination (e.g. Jang et al. 2017; Paradis et al. 2018) and to the identification of suitable management practices (e.g. Krause et al. 2008; Zhang and Hiscock 2011; Bailey et al. 2015).

Here, the broad comprehension and quantification of the major dynamics controlling nitrate contamination, on one side, represents a different approach to evaluate the ND performance, permitting to go beyond the general stability detected using a trend analysis (Musacchio et al. 2019a) and to identify the main components of the system that, among others, hinder the ND goals. On the other side, this allows to improve several aspects of governance supporting the ND implementation: the definition of site-specific objectives as a basis for a well-grounded strategy to address nitrate contamination, the provision of clear and unambiguous hydrogeological knowledge coherent to governance needs, the possibility to finally evaluate ND performance in its entirety by strengthening its legitimacy.

In the following sections, we discuss the main results obtain by the model and, finally, how they allow to identify and fill the governance gaps.

5.6.1 Nitrogen surplus on agricultural soils

Since the ND was fully implemented in the Lombardy Plain, agricultural practices have not been properly modified with the aim of diminishing their impact on groundwater quality. As highlighted by the nutrient budget, nitrogen input dramatically exceeds the requirements by crops in about all municipalities, whether located in vulnerable or not vulnerable zones (Table 5.2). Indeed, almost one third of the annual nitrogen imported to the agricultural soils cannot be assimilated by crops, volatized as NH_3, or denitrified (Figure 5.5). Thus, balanced fertilization is still far from being a common practice.

The main input is due to livestock manure, which is applied in amounts exceeding the ND restrictions (i.e. 170 kg N ha^{-1} y^{-1}) in most of the vulnerable zones (Table 5.2). However, it is worth noting that synthetic fertilizers correspond to nearly one third of the overall input (35%; Figure 5.5); that is, a percentage comparable to the exceeding N amount in the area (29%). This means that ND application has not produced a significant reduction in livestock density permitting to achieve the ND goals. Given the constant reduction of agricultural lands in the Lombardy plain (i.e., lands available for manure spreading; (ISPRA. 2018), this failure can become increasingly critical for groundwater quality issues.

5.6.2 Nitrogen balance within the hydrogeological system

The N surplus from agricultural activities represents almost 70% of the annual N load to groundwater, whereas inflow from the northern sector of the Adda basin contributes to the remaining part (Figure 5.5). Once in the aquifers, N is stored, denitrified or exported to surface waters. Given the present state of recharge rate and N input, the annual amount exported or eliminated is actually high, as only a 0.4% of the annual input remains stored in the system (Table 5.7). This small increment indicates a high capacity of the system to preserve its groundwater quality, yet it does not allow for attaining the ND goals. Moreover, such increment of the groundwater stored N mass acts in detriment to surface water quality, as a large part of the N export takes place through gaining streams and *fontanili* outflow. In mass balance terms, N export to surface water averages 8.3% of the mass currently stored in the aquifers, and 95.5% of the annual N input. On the other side, denitrification guarantees the final removal of 19.6% of annual N input, reducing N storage and preventing higher pollution levels in the southern sector of the basin.

N mass balance results point out that the annual amount retained in the aquifers (i.e., 0.4% of the annual N input), although small in comparison with the amount exported or denitrified (Table 5.6, Table 5.7), causes a constant increase of the total accumulated nitrogen (Figure 5.14). Therefore, the overarching estimation obtained by using the model confirms that the overall stability observed in nitrate concentrations across time (Musacchio et al. 2019a) cannot be only attributed to inherent characteristics of the aquifers (e.g. groundwater residence time). Every year an exceeding input contributes to contamination persistence in groundwater and determines the delay between the ND implementation and the observation of measurable improvements. This also suggests that stable trends of nitrate concentrations should be discussed separately from decreasing trends in the European reports, as their apparently small variations can conceal critical situations.

5.6.3 Implications of N export to surface water

As already stated, rivers and *fontanili* have a key role in regulating groundwater quality. The resulting median nitrate concentrations of both rivers (4.5 mg L^{-1}) and *fontanili* (28.7 mg L^{-1}), calculated using the model estimations of N export, are consistent with field data which range from 1.8 to 4.4 mg L^{-1} for the Adda River and correspond to 25.6 mg L^{-1} in the *fontanili* (Musacchio et al. 2019b). Both act as N sources into the Po River and finally contribute to the eutrophication of the northern Adriatic Sea (Penna et al. 2004; Viaroli et al. 2018). Thus, the reduction of the surface water nitrate content stands as a priority, although preventive remedial actions should focus on groundwater since most of contaminant comes from the subsurface. Promoting and enhancing in-situ denitrification processes by the means of wetlands and vegetative buffer strips are recommended actions to avoid the subsidiarity of downstream N pollution in the Po River basin (Balestrini et al. 2016; Castaldelli et al. 2018).

Finally, it should be noted that the export of nitrates through rivers and *fontanili* heavily relies on the maintenance of a regional hydraulic gradient, which in turn depends on the contribution of both precipitations and surface irrigation to the net recharge (Balderacchi et al. 2016). In light of the local effects of climate change, an increase of groundwater vulnerability to nitrate contamination has to be expected over time. In fact, although a quite good resilience should characterize the groundwater resources as a whole, the reduction of meteoric and irrigation recharge has been predicted, as well as the increase in both number and length of low groundwater table episodes (De Caro et al. 2017; Oberto et al. 2018). Given the vulnerability of *fontanili* outflows to variations in both groundwater depth and water quantity employed for irrigation, a decrease in the nitrogen load exported can be expected, with consequent increase in nitrogen amount stored in the aquifers.

The use of "inefficient" irrigation practices has been already recommended in the area, as it dilutes groundwater nitrate by increasing the net aquifers recharge and helps to maintain the necessary hydraulic gradient and water table elevation that feed gaining streams and *fontanili* (Balderacchi et al. 2016; Rotiroti et al. 2019). While the underlying reasoning is realistic in its attempt to control nitrate pollution, its viability on the long term should be carefully evaluated. The current use of inefficient irrigation practices, which takes place under the present socio-economic conditions, only confirms its feasibility on the short term. Nowadays, current conflicts on water use (and their associated costs), although reported in the Lombardy plain (Sacchi et al. 2013; Musacchio et al. 2019b), are probably not so relevant to change irrigation practices. However, under the foreseen reduction of water resources due to climate change, new conflicts and needs could emerge or fester, and the opportunity cost of water use estimated on the wider basin scale (i.e. Po river basin) may become disadvantageous. Therefore, inefficient irrigation should not be the main strategy, in the light of this uncertainty and the non-linear evolution of water crisis (Parisi et al. 2018).

5.6.4 Governance implications of numerical groundwater modelling

In the light of the aspects above discussed, the developed hydrogeological model appears to mainly contribute to fulfil three governance gaps in the study area: the objective gap, the information gap and the accountability gap.

As regards the objective gap, by going beyond a fragmentary vision the model highlights (and quantifies) three main priorities to address groundwater nitrate contamination in the area, namely: (i) to reduce the nitrogen input in order to avoid the surplus, (ii) for as long as possible, to sustain nitrates export through rivers and *fontanili* by artificially maintaining high values of net recharge, (iii) to support denitrification processes once nitrogen has been exported to surface water. According to the model, nitrogen input and surface waters, every year account for respectively 67.5% of the nitrogen currently entering in the aquifers and 77.5% of the nitrogen currently exported by the aquifers, respectively. Finally, in light of the expected changes in water availability (De Caro et al. 2017; Oberto et al. 2018) and the consequent changes in social and environmental conditions, it is urgent to define a clear timeline for each of the priorities indicated above. Their identification and their potential contribution to the achievement of ND goals draw attention on the current lack of regional objectives that can consider site-specific objectives of groundwater contamination. Indeed, the existing objectives are those defined at European scale, i.e. (i) reducing water pollution caused or induced by nitrates from agricultural sources, (ii) preventing further such pollution. Nevertheless, these rather wide European objectives do not take into account site-specific characteristics of groundwater resources. Consequently, site-specific management actions (and their relative contribution) are currently not framed in a well-grounded strategy to address nitrate contamination. The overall lack of clarity on governance dynamics detected in the area (Musacchio et al. 2019a), affecting its effectiveness, can be partially caused by this governance gap. Moreover, this objectives gap may not be an isolated incident, as it is consistent with ND requirements. In fact, the Directive framework does not explicitly require the definition of ad hoc objectives which consider the specific characteristics of groundwater resources influencing nitrate contamination.

Information gap is often described in literature as the presence of asymmetric information (in type, quality or quantity) between different stakeholders or the inability to share the existing data (Mirzaei et al. 2017; Akhmouch et al. 2018). However, the invisibility of groundwater and the inscrutability of the aquifers in their entirety sharpen the challenge, as data are often limited to restricted portions of the aquifers (Tidwell and van den Brink 2008). By overcoming the limitations of a patchy knowledge of aquifers, three dimensional models allow us to represent and understand the sub-surface water dynamics (Baalousha et al. 2008; Tidwell and van den Brink 2008; Kroepsch 2018). The main limitation of our model is its poor capacity to describe in detail processes at local scale. The specificity of each well is not always completely reproduced. Nevertheless, consistently with the aim of the study, it provides a view from afar of the studied system. By both identifying the main components which determine nitrogen distribution and quantifying how the nitrogen surplus is stored and exported, the model represents an overarching diagnosis which describes the dimensions of the problem and the extent of agricultural impact on ground and surface waters. This permits to provide hydrogeological knowledge in a clear and handy format. However, to obtain a hydrogeological knowledge coherent to governance needs is only a first step to obtain a successful interchange of information. As underlined by the previous governance analysis, the way farmers select the sources of knowledge has to be considered (Musacchio et al. 2019a).

Finally, regarding the accountability gap, we believe that three-dimensional hydrogeological models widely increase the possibilities to provide trustable evidence in ND performance (Mirzaei et al. 2017), compared to the currently used trend analysis. Using monitoring data to assess variation in nitrate concentrations across time is in fact the institutionally recognized approach. However, due to the simultaneous presence of increasing, decreasing and stable trends, it is difficult to draw conclusions on the overall ND performance. By estimating the whole amount of nitrogen currently stored in the aquifers and its annual increase, the model unveils that, overall, nitrogen is slowly but constantly increasing in the area. Although trend analysis was useful to detect critical situations (Musacchio et al. 2019a), as regards the overall ND assessment it leaves the door open to uncertainty. Accountability is a key aspect of governance effectiveness as it supports legitimacy and trustfulness, fundamentals of many informal relationships.

6 GENERAL DISCUSSION AND CONCLUSIONS

The assessment of the ND, based on the socio-hydrogeological approach, is first and foremost a key opportunity to improve the application of the Directive, and to increase the chances of its success. The proper evaluation of ND environmental performance, together with the investigation of the reasons behind its successes and failures, permits reconsidering the enforced management regulations and enhancing procedures and tools. In addition, the assessment stage is a valuable chance to enrich the analysis with new information.

In this thesis, the environmental performance of the ND in the Lombardy plain is evaluated by both estimating the variation of nitrate concentration across time, and building up a hydrogeological numerical model. According to the socio-hydrogeological approach (Re 2015), data on contamination trends, nitrate input from agricultural activities, retention and removal of nitrate mass in and from the aquifers, are interpreted also in the light of the socio-relational dynamics at stake influencing the ND implementation.

In addition to taking stock of the main results of the thesis and naming their management implications, the purpose of this general discussion is to share wider reflections related to groundwater governance, the application of the ND and the prospects for its success. Possibilities for future multidisciplinary investigations are also presented.

In Figure 6.1 the main methods, assumptions and results of this thesis are summarised.

6.1 Effectiveness of Nitrates Directive and tools to measure it

As provided for by the European Commission, the ND effectiveness in the Lombardy plain is evalutated by assessing the variations of nitrate concentrations across time using monitoring data (EU Commission 1991). According to this evaluation, 70% of wells shows stable or decreasing concentrations (ERSAF et al. 2015), suggesting a rather positive success as regards the prevention of further contamination (i.e. the second objective stated by the Directive).

However, the assessment carried out in this thesis points out that:

- current nitrate contamination of groundwater mainly affects the already impaired situations and particularly relevant groundwater resources (as those located in intermediate and deeper aquifers; Chapter 3);
- behind an overall stability of contamination trends, a slow and constant increase of the nitrate amount stored in the aquifers can be concealed (Chapter 4).

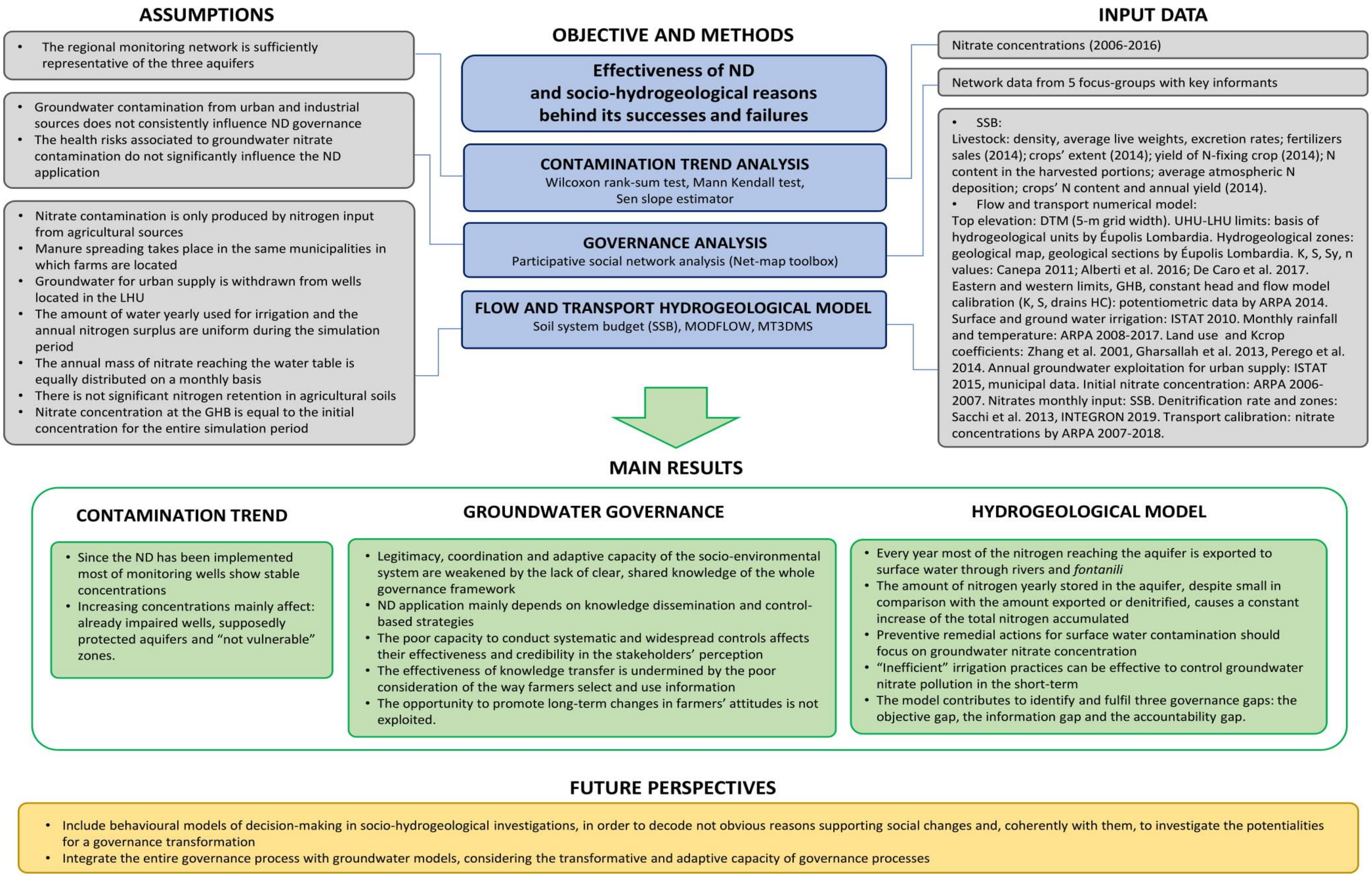

Figure 6.1 Assumptions, input data and results for each method applied. Major perspectives for future investigations are reported.

Therefore, the practice of evaluating the ND by comparing current nitrate values of monitoring wells with those of the previous four-year period, generally used in the study area and in European reports (ERSAF et al. 2015; EU Commission 2018), is insufficient. Explicitly looking for poorly evident critical situations while carrying out the statistical analysis (e.g. the worsening of relevant groundwater resources, of already impaired situations and of theoretically less vulnerable aquifers; Chapter. 4) is paramount, as they are highly informative of the real ND effectiveness, besides allowing to detect contamination issues at local scale. Nevertheless, in the Lombardy plain the contamination risk is currently more considered in the monitoring network design, jeopardizing the representativeness of the network, than during data analysis. Moreover, as ostensible small variations in terms of mg L^{-1} y^{-1} can hide a significant increase in the long-run, ND assessment should be supported by a robust analysis looking for statistically significant changes. Mann Kendall test and Sen slope estimator are, for example, rather simple statistical tools, pretty common in scientific literature (Serrano et al. 1999; Zhang et al. 2006; Batlle Aguilar et al. 2007; Hansen et al. 2011; Iital et al. 2014; Koh et al. 2017), that may be integrated in the official evaluation of ND performance carried out every four years (EU Commission 2018). Monitoring data, in turn, should be collected consistently with these purposes and with the data analysis to be performed, by guaranteeing the comparability, on one side, of data from a same set of wells over time, and on the other side, of different portions of a region subject to different hydrogeological dynamics. On the contrary, in the Lombardy plain, each well of the monitoring network may have been sampled from zero to four times per year, revealing a large effort that does not grant quality data for statistical testing. Furthermore, the sampling campaigns are conducted independently of the seasonal variations of groundwater table, differently affecting the higher and lower plain (Figure 3.4).
As regards the tools currently used to assess ND performance, it is necessary to evaluate whether they are in all respects exhaustive for this purpose. Although a trend analysis permits to detect critical situations and the general evolution of groundwater quality, it does not permit to estimate if the amount stored in the whole aquifer is finally increasing or not, which is the knowledge we ultimately need to protect groundwater resources. Information provided by trend analysis should be integrated with a comprehensive understanding of both the entire system evolution and the components of the hydrogeological system actually governing the overall nitrate mass balance, while taking into account local singularities, such as e.g. the *fontanili*. This information is crucial, especially when most of the monitored wells show stable trends; otherwise, the overall stability may prove to be a “false positive”. In this respect, hydrogeological modelling represents a fairly compelling tool in support of monitoring data, permitting (i) to detect small increases of nitrogen stored in the aquifers, (ii) to understand how the system is responding to the input generated by the agricultural sector, and (iii) to identify the natural features that could support a contamination reduction. Moreover, the estimation in terms of absolute amounts of the nitrogen stored and moving in the aquifers, as provided by the model, permits to account for the groundwater role in nitrogen mass balance computations and finally connect the dynamics of contamination occurring in ground and surface water at the catchment scale (Dunn et al. 2012; Sacchi et al. 2013).

6.2 Some keys to success of Nitrates Directive

The improvement of ND implementation is not only a management issue, exclusively concerning the regulations on the quantity and timing of nitrogen application to soils. The adoption of a governance perspective on ND application makes explicit that agricultural practices are only the last step of a wider process (Barthel et al. 2017; Villholth and Conti 2017).

Farmers are not isolated stakeholders, and their decisions regarding nitrogen input is influenced by several components and socio-relational dynamics of the governance network, which sometimes are not directly related with the farmers themselves or under farmers' control. The integrated analysis of governance and hydrogeological dynamics brings to light several key aspects that, although indirectly, can strengthen ND implementation and facilitate a change in agricultural practices, to fulfil the objectives which are currently not achieved.

This firstly implies to define (and make clear to the actors involved) a set of regional objectives, by considering the socio-hydrogeological characteristics of the system and the needs of the stakeholders involved that both influence the nitrate persistence. The contextualization of management choices within a grounded strategy based on site-specific goals has several advantages (Davies and White 2012; Dutra et al. 2015; Klenk and Wyatt 2015; Akhmouch et al. 2018):

- it forms the essential basis to build up common purposes among the involved stakeholders;
- it strengthens the legitimacy of both the changes required in agricultural practices and the actors supporting the Directive implementation;
- it provides a starting point for the transparent assessment of ND performance, which in turn contributes to the legitimacy of its governance process.

However, the definition of regional objectives is necessary, but not sufficient. In fact, it must be coupled with a high clarity on the roles and dynamics supporting the application of the Directive (Akhmouch and Correia 2016), by overcoming the idea that dissemination of information on ND should only be focused on management practices and groundwater protection relevance. The knowledge of the governance system in which the stakeholders are called to participate is a key element for strengthening their degree of trust and, ultimately, the likelihood of their positive contribution to ND goals. Moreover, it has to be matched with a turnaround in communication strategies, which aims, targets and tools should be the result of a specifically designed analysis (Inman et al. 2018). In the last 30 years we have indeed verified that the identification of good management practices is not enough.

As regards the research supporting ND application, it is required to take up a deeper social challenge (de Snoo et al. 2013). How do retailers gain trust of farmers? Which factors influence decision-making by farmers about nitrogen use and manure management? Which factors make them more prone to pro-environmental behaviours? Is decision-making by farmers mainly influenced by customs or by social norms rooted in their reference network? If it is mainly influenced by social norms, which are the expectations behind them? These can be some of the questions that should be answered in future studies in order to progress in groundwater protection, and more in general in the protection of commons governed by the agricultural sector, going beyond the "reduction of input" recipe. This implies to avoid simplistic analyses of human choices, by engaging into interdisciplinary investigations that include behavioural models of decision-making, in order to decode not obvious reasons supporting social changes and, coherently with them, investigate the potentialities for a governance transformation (Bicchieri 2016; Pahl-Wostl 2017). Otherwise, any change in legislation or management action may be ineffective.

Overall, the need of multidisciplinary approaches, such as socio-hydrogeology, to achieve sustainability goals is confirmed once again (Newing 2010; Re et al. 2017), also when the environmental characteristics and dynamics of the system are quite well known, as in the case of nitrate contamination in the Lombardy plain. In light of this, it would be relevant to investigate the major constrains behind the general reticence showed by hydrogeologists to open up to the integration of social methods (Barthel and Seidl 2017).

6.3 Future perspectives: the integration of governance processes into hydrogeological models

The persistence of nitrates in groundwater underlines the need to urgently find effective and feasible solutions to reduce their concentrations, as well as the associated health and environmental risks (Seitzinger and Phillips 2017). Numerical simulations are typically used to explore the effectiveness of alternatives management strategies, with the ultimate aim of identifying changes in human activities that enhance the possibilities of preventing or minimizing groundwater pollution. The responses of the aquifers to different management options (or a combination of them) are simulated and made explicit in terms of nitrate concentrations, by including major processes of contaminant attenuation or removal (Almasri 2007). The ability of studied options to deal with nitrate contamination is often evaluated by comparing aquifers' responses with a baseline, i.e. the commonly called "do nothing" or "business as usual" scenario (Almasri and Kaluarachchi 2007; Bailey et al. 2015; Zhang et al. 2019). The studies aimed at developing management alternatives to hinder nitrate contamination in groundwater, can be classified in two board categories: (i) studies that directly deal with a set of management options, (ii) studies that try to integrate the effect of other drivers, beyond the management per se, potentially influencing the feasibility or the level of implementation of the management options.

In the first group, the most commonly simulated management scenarios include changes in land use, reductions in nitrogen loading on agricultural soils and implementation of different irrigation strategies (Krause et al. 2008; Bailey et al. 2015; El Khattabi et al. 2018; Sidiropoulos et al. 2019; Zhang et al. 2019). In these studies management actions are chosen so as to be realistic or potentially relevant (e.g. the application by farmers of a balanced fertilization), and the full implementation of management actions by the actors involved is generally assumed; alternatively, several theoretical degrees of implementation are explored (e.g. basic, moderate, aggressive implementation levels as in Bailey et al. 2015).

Studies of the second group broaden the perspective on nitrate issue by focusing on the economic drivers that condition farmers' or managers' decision-making. Almasri (2007), for example, proposes a multi-criteria decision framework in order to prioritize the possible management options to simulate, so as to consider both economic and environmental goals. Other authors develop the so-called hydro-economic models, in which results from agronomic or agricultural economic models are integrated to hydrogeological models, mainly to consider the impact of studied management actions on the agricultural income (Graveline and Rinaudo 2007; Peña-Haro et al. 2010).

Although the economic drivers are certainly crucial in determining both the adoption of management strategies and the level of their implementation, many more factors determine the success of (ground) water management, and the governance concept exactly aims at capturing the complexity of their coexistence (Pahl-Wostl 2015).

So, to do justice to the complexity of the real-world processes, hydrogeological numerical models designed to test, discuss and finally identify effective management strategies should also integrate two facets of governance. On the one side, the effects of governance process, in its entirety, on nitrogen input and ultimately on groundwater quality. On the other side, the transformative and adaptive capacity of governance (Pahl-Wostl 2017) to both groundwater quality (as socio-relational dynamics and legislative frameworks can change in response of quality issues) and other external pressures, likely to become part of governance in a near future (e.g. climate changes, effects of Brexit on EU direct payments etc.).

Given the characteristics of the governance networks used in this research, they may be an effective tool to fulfil these requirements. In fact, the identification and visualization of the most influential actors and relationships, without neglecting the entire governance process, can possibly be the basis to quantify the effects of governance dynamics on nitrogen input and, ultimately, on groundwater quality. Also, the graphical rapresentation of a wide range of relational drivers affecting groundwater governance may deal with the cascade effect of future scenarios, namely the transformative capacity of governance systems. More generally, it is critical to investigate whether and how, from the methodological point of view, governance networks can be used to develop integrated models, by guaranteeing the systemic approach required for the investigation of governance dynamics as well as the analytical rigour (Araral and Wang 2013).

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SUPPLEMENTARY MATERIALS

- **S1** Supplementary information for the Introduction
- **S2** Supplementary information for Chapter 4
- **S3** Supplementary information for Chapter 5

S1 Supplementary information for the Introduction

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Squaring the cycle: the integration of groundwater processes in nutrient budgets for a basin-oriented remediation strategy

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ABSTRACT

Long term projection of N and P concentrations in surface and ground waters cannot be entirely achieved unless groundwater features and processes affecting nutrients at the watershed scale are considered. This work presents the general approach of the INTEGRON project whose aim is to "square the cycle" in nutrient budgets, evaluating the role of groundwater as a temporary/permanent sink or as a release term at the catchment scale, in the Adda and the Ticino basins (northern Italy). An integrated approach is currently tested, which considers surface and groundwater, N and P species combining hydrogeology, biogeochemistry and socio-hydrogeology. The availability of nutrient excess, the nutrient increase in rivers during the irrigation period, and nitrate trends in different groundwater bodies confirm that groundwater in the area acts as both sink and source term of nutrients. Given the complexity of the studied system, the proposed multidisciplinary approach can permit to effectively implement science-based management strategies for water protection that consider both the social and environmental domains.

KEY WORDS: Po plain, nutrient mass balance, surfacegroundwater interaction, socio-hydrogeology, groundwater dating.

INTRODUCTION

Diffuse contamination of surface- and ground-waters resulting from the green revolution represents a major threat to the long-term sustainability of water resources worldwide. Nutrients' budgets, calculated in a wide array of river basins, have shown that a high amount of nutrients can be retained or lost within the watershed, where site-specific characteristics contribute to define their fate (Mulholland et al., 2008; Schlesinger, 2009). In the Lombardy plain (North Italy), the uneven distribution of nitrates in groundwater, not fully matched to any of the urban, industrial and agricultural sources, suggests that local features and processes (e.g. depth of water table, land use, denitrification processes etc.) play a key role in preserving or removing nitrogen from water (Fig.1; Laini et al., 2011; Sacchi et al., 2013). Based on the outcomes of previous investigations in the Oglio river basin (Bartoli et al., 2012; Delconte et al., 2014), it was hypothesized that the high soil permeability in the higher plain promotes the leaching of nitrogen excess to the subsoil and its subtraction from the shallow environment. Nevertheless, the nitrogen residence in the aquifers is only temporary, since groundwater naturally outflows in correspondence of the transition between the higher and lower plain, in the so-called "fascia dei fontanili" (springs belt). Here, numerous semi-natural outflows ("fontanili") are present, which return the deep groundwater, partially mixed with recently infiltrated irrigation water, and the dissolved nutrients back to the surface water compartment. By contrast, in the lower plain ground-water is deprived of nitrates due to high rates of bacterial denitrification, hence when contributing to river recharge it dilutes its nitrate content. In this framework, the INTEGRON project aims at evaluating the role of groundwater as a temporary or permanent sink or as a source term in nutrient mass balances at the catchment scale in two key sub-basins of Po River, the Adda and the Ticino (Fig. 2).

MATERIALS AND METHODS: THE INTEGRON APPROACH

An innovative and integrated approach is proposed and currently tested, considering both surface and groundwater, combining hydrogeology, biogeochemistry and socio-hydrogeology, and targeting both inorganic nitrogen (N) and phosphates (P) species (Fig. 3) as detailed below.

CALCULATION OF THE NUTRIENTS MASS BALANCE AT THE CATCHMENT SCALE

The nutrient surplus in the agricultural lands and the loads exported by rivers at the closing section are calculated, in order to quantify the nutrient amounts retained within the basin. The nutrient budgets are calculated by applying the Soil System Budget approach (Oenema et al., 2003), converting statistical data (agricultural surfaces, livestock density, etc.) into N and P loads by agronomic coefficients (i.e. livestock N and P excretion factors, crop yields and N and P contents). The N and P surplus are determined as a difference between the total inputs (livestock manure, synthetic fertilizers, atmospheric deposition, biological fixation- the latter term only for N budget) and the total outputs (crop uptake, ammonia volatilization and denitrification in soils, the latter two terms only for N budget) across agricultural surface.

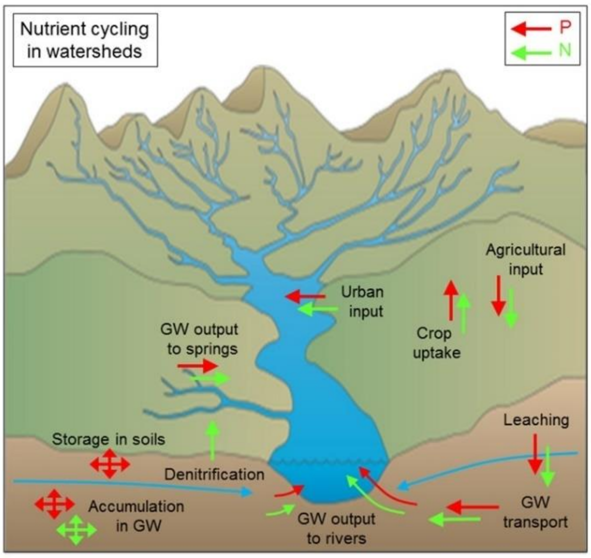

Figure S1.1 Schematic diagram illustrating the nutrient fluxes (red = P; green = N) at the catchment scale that are investigated and quantified in the project.

ASSESSMENT OF GROUNDWATER CONTAMINATION AND RESIDENCE TIME IN THE AQUIFERS FROM THE HIGHER PLAIN

To infer the residence time of nutrients, groundwater dating is performed. In the study area the alluvial sequence creates a multilayer aquifer system, mainly composed by gravels and sands in the higher plain, becoming progressively finer towards the lower plain. Groundwater depth also decreases from north to south, ranging from more than 70 m depth to less than 2 m. Aquifers are separated by aquicludes which are discontinuous at regional scale. The shallow unconfined aquifers, the most vulnerable, reach a cumulative thickness of more than 100 m in the higher plain and 20-30 m in the lower one. Based on the available piezometric maps and well logs, transects of aligned wells tapping the unconfined aquifer and extending from the foothills to the springs belt area are identified. Groundwater is sampled from 5 wells per transect and analyzed using CFCs and SF_6, allowing to establish the groundwater residence time and apparent flow velocity. The combination of the chronological information and the nitrate concentration trends in time permits to infer the nutrient dynamics in the aquifers and to estimate the timing for groundwater recovery in different watershed management scenarios.

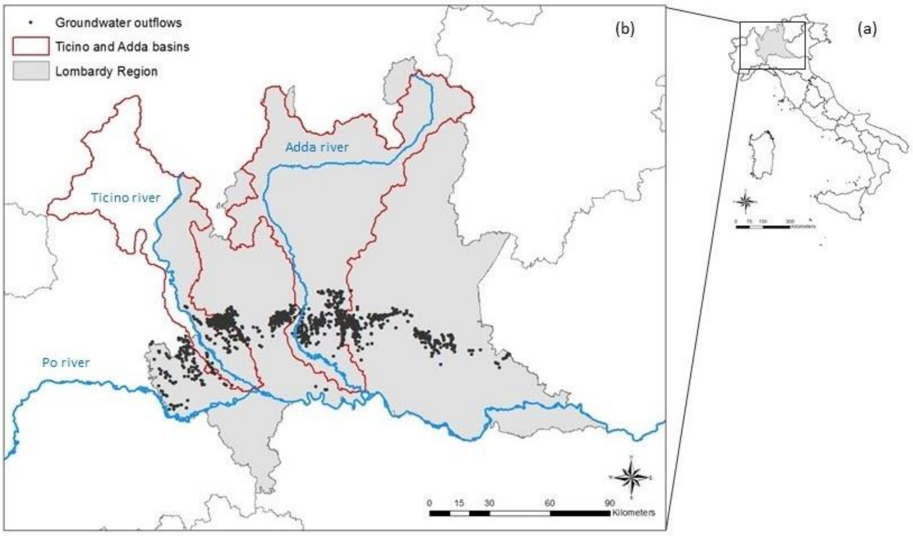

Figure S1.2 (a) Location of the Lombardy Region; (b) Studied basins (in red) and rivers (in blue). Each black dot represents one of the groundwateroutflows (fontanili) composing the so-called springs belt.

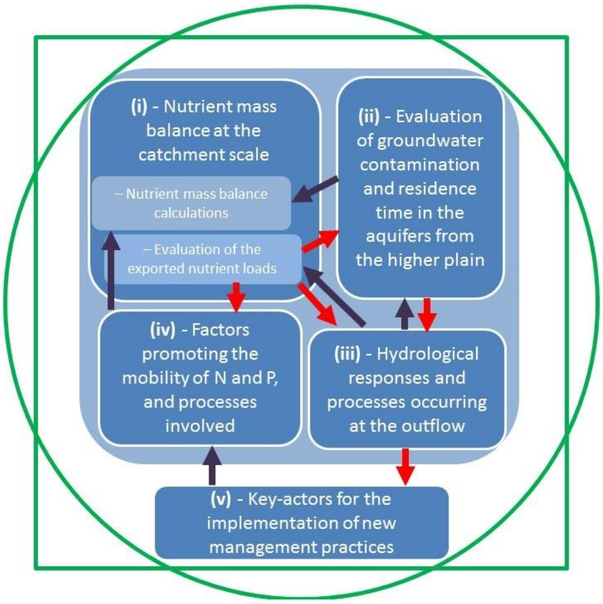

Figure S1.3 Relationships between the different components of the integrated approach. Red arrows: information input; blue arrows: information output and feedback. In green, the logo of the project.

EVALUATION OF THE HYDROLOGICAL RESPONSES AND PROCESSES OCCURRING AT THE OUTFLOW

The amount of nutrients exchanged between surfaceand ground-waters is estimated and the quantity and quality of recently infiltrated irrigation water evaluated. Finally, the processes occurring at the interface which can affect nutrient loads (e.g. nitrification, denitrification) are identified. In this task, both discrete outflows such as springs and diffuse input to the Adda and Ticino rivers are investigated. During an extensive seasonal monitoring of nutrient contents and discharge, groundwater outflows are classified according to their hydrological response type. This classification permits the selection of a few representative springs to be monitored more frequently during an irrigation season. River water sampling and discharge measurements are performed via open-channel approach (Izagirre et al., 2008) during irrigation and nonirrigation periods in a segment of each river within the river-groundwater interaction zone.

INVESTIGATION OF THE ENVIRONMENTAL FACTORS CONTROLLING THE N AND P DYNAMICS IN GROUNDWATER

The investigation of the factors promoting the retention or removal of nutrients (e.g. denitrification, P adsorption) is performed. The seasonal evolution of nitrate and P concentrations in groundwater, coupled to nitrate isotope determinations is studied in rice paddies from the Ticino basin and in the lower plain of the Adda basin. Results permit to identify the conditions and the pathways for denitrification in groundwater, in order to exploit its full potential while reducing the negative impacts on the global environment. In addition, the environmental conditions and processes affecting P mobilization, transfer to groundwater and recycle to surface waters, with implications on their trophic status, are investigated as a function of redox potential and soil texture.

IDENTIFICATION OF THE STAKEHOLDERS INVOLVED, THEIR RELATIONS AND POSSIBLE EXISTING CONFLICTS

To identify key actors for the implementation of new management practices a socio-hydrogeological analysis is carried out (Re, 2015; Tringali et al., 2017). This is performed using the Net-Map toolbox (Schiffer & Waale 2008), an interview-based tool method, facilitating the identification of all the actors involved in a given issue (including marginal ones) while also highlighting their power relations, their influence and their main goals. Different groups of key informants are involved in drawing a so-called Influence Network Map (INM), depicting the social network affected directly or indirectly by groundwater contamination. Each of these INMs permits to visualize the social network perceived by the key informants, the stakeholders involved and their perceived influence, while also highlighting the relationships among them (money, conflicts, authorization and control, advice and technical information flows). Degree of centrality and network density are computed to identify relevant actors in the studied issue and to evaluate the differences in perception among the groups of key informants (Hauck et al., 2015).

PRELIMINARY RESULTS N AND P MASS BALANCES AT THE CATCHMENT SCALE

Agriculture land covers a similar area in the Ticino and Adda basins (19% and 23%), but the two watersheds differ in terms of livestock density (0.7 and 2.3 livestock units per ha of agricultural land) and main crops (rice for Ticino and maize and fodder crops for Adda). The two watersheds are similar in terms of population density (257 and 264 inhabitants per km2). N and P availability in agroecosystems is in excess compared to crop demand, resulting in a different average areal surplus between the two basins: 71 kg N ha-1 yr-1 and 24 kg P ha-1 yr-1 for Adda and 40 kg N ha-1 yr-1 and 6 kg P ha-1 yr-1 for Ticino. The main sources of nutrients are manure spreading due to high livestock density for Adda and synthetic fertilizers for Ticino.

ESTIMATION OF NITRATES TRENDS

Comparing the mean NO3 - values detected in the monitoring network during the period 2006-2008 with those detected during the period 2014-2016 (data of the Regional Agency for the Environmental Protection) differing trends emerge between the shallow and the deeper aquifers. An overall stability of nitrates contamination in the shallow aquifers emerges. In fact, the percentage of wells in which a decrease higher than -1.00 mg/L was recorded (42.6%; mean: -17.4 mg/L) is similar to the percentage of wells in which an increase greater than +1.00 mg/L (44.3%; mean: +11.6 mg/L) was observed. By contrast in the deeper aquifers 26.8% of wells show decreasing concentrations (mean: -18.7 mg/L) and 51.2% increasing ones (+7.0 mg/L).Finally, monitoring data confirm a general reduction of groundwater contamination at the transition between the lower (mean 2006-2008: 20.7 mg/L; mean 2014-2016: 19.8 mg/L) and the higher plain (mean 2006-2008: 30.4 mg/L; mean 2014-2016: 29.1 mg/L).

GROUNDWATER OUTFLOWS: SPRINGS AND INPUT TO ADDA AND TICINO RIVERS

During the non-irrigation period, water NO3 -concentration in both rivers is constant (< 5.5 mg/L) along the reaches in correspondence of the transition between the higher and lower plain, where rivers cross the springs belt area. During the irrigation period, water NO3 – concentration in both rivers increases along the same reaches from 1.8 to 4.4 mg/L for Adda River and from 1.3 to 4 mg/L for Ticino River. Groundwater fed springs (fontanili) located in the Adda basin show higher median NO3 - concentrations (25.6

mg/L) compared to the Ticino basin (14.3 mg/L). Median values of NO3 - concentration, measured in the Ticino and Adda river water, result lower than those measured in spring waters (Ticino river, 5.4 mg/L; Adda river, 4.6 mg/L), indicating a difference between river and groundwater chemistry.

DENITRIFICATION AND P MOBILITY IN GROUNDWATER FROM RICE PADDIES

Monitoring conducted during one irrigation season in very shallow groundwater (1-2 m below ground) below rice paddies detected elevated P (up to more than 2.2 mg/L) and NO3 - (up to more than 250 mg/L) contents, but also limited denitrification. The direct relationship between the two nutrients and measured Eh suggests their input from the surface related to the use of fertilizers. The investigation is presently addressing deeper portions of the aquifer in order to assess the presence and extent of denitrification, and its relationship with the total P contents.

MARGINALITY OF INFLUENT ACTORS AND DIFFERENCES IN PERCEPTION

Three groups of key informants were involved: authorities (A), researchers (R) and farmers (F), and three preliminary INMs obtained (Fig. 4). Differences in perceptions between groups were mainly associated to (i) the number of actors identified to compose the network (A: 44, R: 36, F: 20), (ii) their influence level and (iii) the density and kind of links. The lack of correspondence between the actors with a higher influence level and the ones with a higher degree centrality value, recurring in all INMs, points out the commonly perceived marginality of influent actors. Deep differences in number (A: 302; R: 295; F: 55) and kind of the perceived links between actors emerged: in each INM different kinds of links prevail (authorization and control, and money flows in the farmers' network; technical information and advice in the authorities' network; conflict flow in the researchers' network); the efforts carried out by the authorities in communicating technical information and good practices is poorly perceived by the farmers.

CONCLUSIONS

The protection and management of water resources requires to identify and to quantify which features and processes could affect the nutrients removal and retention within the watersheds. The proposed integrated approach suggests a multiple role played by groundwater in defining the fate of nutrients in the studied basins, depending on extremely variable and site-specific processes. A nutrient excess compared to the crop uptake requirements is available every year in the agricultural lands. The connection between surface and ground waters, strengthened by the hundreds of springs placed in the transition between higher and lower plain, defines an increase in NO3 - concentration in rivers, promoted by the contribution of the irrigation water. Temporal and spatial trends of groundwater contamination confirm that remarkable differences in nutrient values are present between the higher and the lower plain. In conclusion, the integrated evaluation of all contributing factors and processes provides a suitable strategy to deal with the complex task of "squaring the cycle" and to move towards a more effective management of water resources, especially in areas where regulations seem not to be effective. Using a socio-hydrogeological approach permits to point out the inherent limits of the social context to the implementation of good practices in groundwater management, as well as the marginality of the influent actors or the deep differences in perception between the involved groups, and to highlight feasible strategies to overcome these criticalities.

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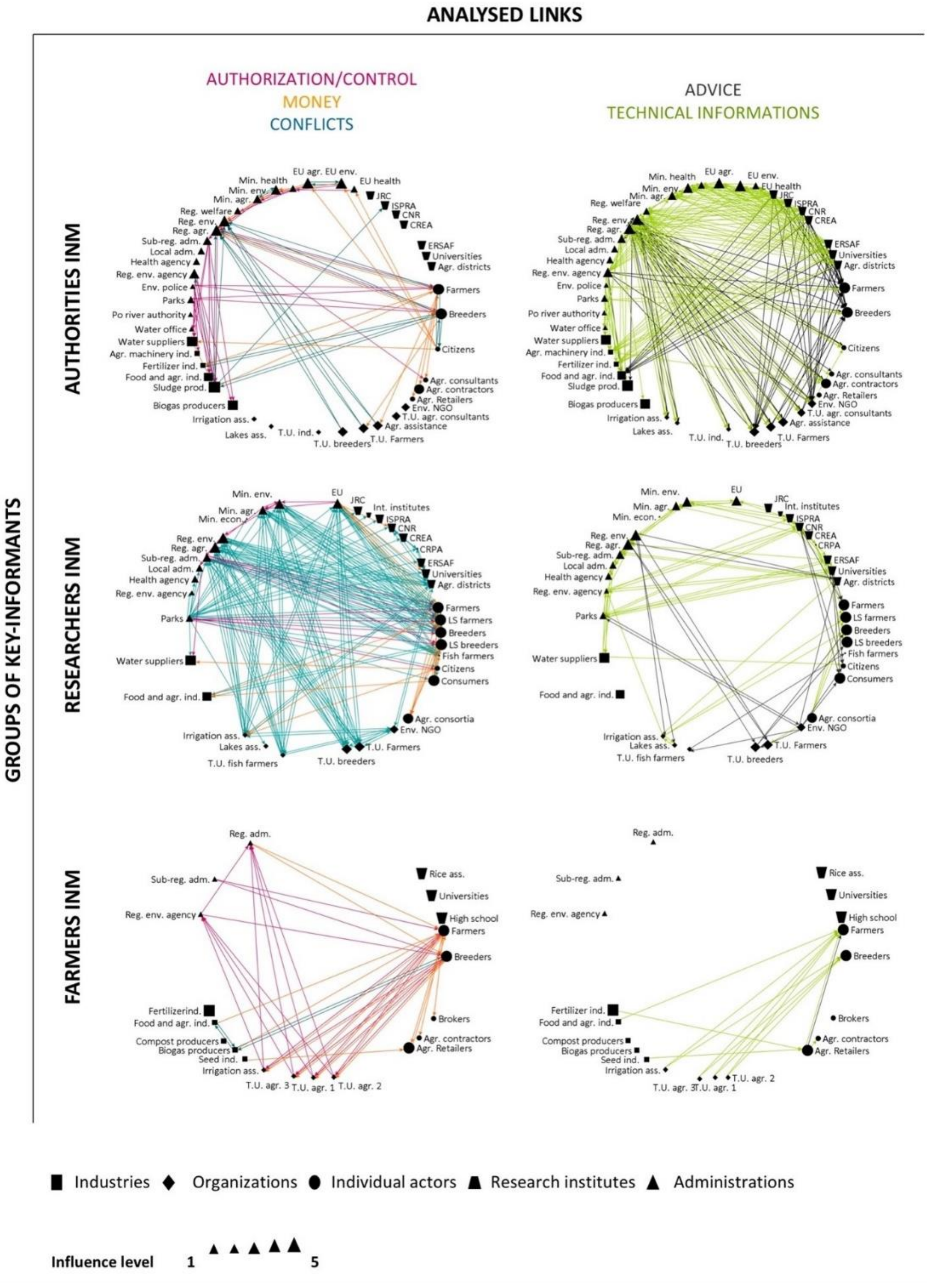

Figure S1.4 Influence Network Maps (INMs) of the three groups of key informants. In the right column advice and technical information links perceived by each group are represented. In the left one the relationship of authorization and control, money and the existing conflicts are reported. Abbreviations: Agr. (Agricultural), Ass. (Association), Adm. (Administration), Econ. (Economy), Env. (Environmental), Ind. (Industries), Int (international), LS (large-scale), Reg. (Regional), Sub-reg. (Sub-regional), T.U. (Trade Unions). Research insitutes: CNR, CREA, CRPA, ISPRA, ERSAF.

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S2 Supplementary information for Chapter 4

Social network analysis

The Net-map method is based on a participatory approach, which contributes to the in-depth understanding of both the dynamics at stake within a studied network and the differences in perspectives of each stakeholder (Bodin and Crona 2009; Newing 2010; Fuhse and Mutzel 2011; Hauck et al. 2015; Sayles and Baggio 2017).

Firstly, a list of potential interviewees was drafted. The selection was based on (i) the level of involvement in the decisional and management processes concerning the studied issue, (ii) their level of expertise in their fields, (iii) the practical possibility of contacting and meeting them. Then selected interviewees were invited by email and phone. This list included: 7 authorities (members of the Regional General Directorates for Agriculture and for Environment, and of ARPA), 2 farmers, 2 breeders, 6 organisations (representative of farmers' trade unions, environmental organizations, water consortia, natural parks), 7 scientists actively involved in research projects. The number of farmers and breeders was strongly limited by their difficulties in attending the meeting. All the authorities, farmers, breeders and researchers, and 2 organizations we invited, attended the meetings.

Before merging the 5 networks into a single network, we assessed the relevance of (i) the stakeholders not listed by all groups of key informants, (ii) those whose function had not been clearly explained by the interviewees and (iii) the actors whose role was differently described by the groups of key informants. To this end, we used the information available in official documents and on web sites (i.e. documents provided by the regional, provincial or municipal authorities). Neither the actors who were not involved in the studied issue nor those who were only marginally involved (e.g. fish farmers or health authorities) were included in the final network. When a stakeholder was not listed by all groups and its role in official documents and web sites could not be verified, it was only included in the final network if it was mentioned by someone with whom the stakeholder shares a relational link (i.e. the brokers who were mentioned by the farmers). The members belonging to the same institution or organisation and those whose difference in role and functions were not relevant to the studied regional scale, were merged to create the final network (e.g. all the Italian ministries, the European Commissions or large/small-scale farmers/breeders). The same approach was also applied for the links. If (i) a link was not listed by all groups of key informants, (ii) the description provided was not clear or (iii) the link was described differently by the groups of key informants, the link was verified by means of official documents and web sites. If it was an informal link, it was only included in the final network when it was mentioned by key informants directly involved in the same relationship (i.e. a link listed and adequately described by the farmers, in which the same farmers were involved). When there was discrepancy on informal relationships between groups of key informants (e.g. in the case of the advice links), a conservative approach was applied and only ties with a satisfactory convergence of views were included in the final network. Moreover, a few days after the interview, the network produced by each group was shown to the key informants involved so that they could make changes and in order to share additional remarks. All deletions and changes made to the list of actors initially compiled by the key informants are reported in Table S2.5.

Tables

Table S2.1 Description of links provided to interviewees.

Link	Description
Authorisation and control	Ties representing control activities on groundwater use and nitrogen input, the authorisation of activities which can cause groundwater contamination and the definition of the limits to these activities.
Technical information	Ties representing dissemination of practical information related to groundwater and fertiliser use.
Advice and best practices	Ties representing dissemination of practical or theoretical knowledge aimed to improve groundwater and fertiliser use.
Money	Ties representing the exchange of money directly or indirectly influencing groundwater use and nitrogen inputs.
Conflicts	Ties representing practical or theoretical disagreements or incompatibility that can influence groundwater or fertiliser use.

Table S2.2 Actors reported by the five focus groups. The number of actors mentioned and also included in the final network is reported in brackets.

Focus group 1 (20)	**Focus group 4 (19)**
Farmers	Farmers
Breeders	Breeders
ARPA	ARPA
Farmers' trade unions	Farmers' trade unions
Breeders' trade unions	Environmental NGO
Environmental NGO	River basin authority
Health agency	Citizens
Citizens	EU Commission
National Research Institute	Municipalities
EU Commission	Water Consortia (Irrigation)
Municipalities	Water Consortia (Lakes)
Agricultural consortia	Parks
Water Consortia (Irrigation)	Provinces
Water Consortia (Lakes)	Lombardy region
Consumers	Agricultural consultants
CREA (Research Institute)	Universities
DG Agriculture	Environmental volunteers
DG Enviroment	Civil wastewater tr. plants
Agricultural districts	Industrial wastewater tr. plants
International research institutes	CNR
ERSAF (research institute)	Foundations
Water suppliers	Hydroelectric power plants
Large breeders	
Food industries	
ISPRA	
Fish farmers' trade unions	
Fish farmers'	
JRC	
Large farmers'	
Ministry of agriculture	
Ministry of environment	
Ministry of economy	
Parks	
Provinces	
CRPA (research institute)	
Universities	

Table S2.3 Actors reported by each group of key informants. The number of actors mentioned and also included in the final network is reported in brackets.

Focus group 3 (29)	
Farmers	Biogas producers
Breeders	Fertilizers' producers
ARPA	Provinces
Farmers' trade unions	Agricultural retailers
Breeders' trade unions	Agricultural consultants'
Environmental NGO	Water companies (civil)
Health agency	Universities
River basin authority	
Environmental police	
Agricultural trade unions' offices	
Citizens	
CNR (National research institute)	
Municipalities	
Industries' trade union	
Water Consortia (Irrigation)	
Water Consortia (Lakes)	
Contractors	
CREA (Research Institute)	
EU Commision - agriculture	
DG Agriculture	
EU Commision - environment	
DG Enviroment	
EU Commision - Health	
DG Health	
Agricultural districts	
ERSAF (Regional research institute)	
Sludge Treatment Companies	
Civil wastewater treatment plants	
Agricultural machinery manufacturers	
Food companies	
ISPRA	
JRC	
Ministry of Health	
Ministry of agriculture	
Ministry of environment	
Agricultural consultants' trade unions	
Parks	

Table S2.4 Actors reported by each group of key informants. The number of actors mentioned and also included in the final network is reported in brackets.

Focus group 2 (17)	**Focus group 5 (15)**
Farmers	Farmers
Breeders	Breeders
Farmers' trade unions	ARPA
Breeders' trade unions	CIA (farmers' trade union)
Health agency	Irrigation water consortia
Citizens	Contractors
EU Commission	Food companies
Water Consortia (Irrigation)	Agricultural high schools
Non-food companies	Brokers
Food companies	Seeds companies
Agricultural High Schools	Biogas producers
Research Institute	Provinces
Biogas producers	Lombardy region
Fertilizers producers	Agricultural retailers
Provinces	Universities
Lombardy region	Compost producers
National government	Fertiliser producers
Agricultural consultants	National institute for rice cultivation
Universities	COLDIRETTI (farmers' trade union)
	CONFAGR.farmers' trade union)

Table S2.5 List of the actors not included in the final network or not included as mentioned by the key informants.

Actor	Notes
Large-scale breeders	included in breeders
Consumers	included in citizens
Water suppliers	included in CWP
EU agriculture	included in EU
EU environment	included in EU
EU health	included in EU
CIA	included in FTU
Agricultural assistance centre	included in FTU
COLDIRETTI	included in FTU
CONFAGRICOLTURA	included in FTU
JRC	included in IRI
Ministery of health	included in NG
Ministry of agriculture	included in NG
Ministry of environment	included in NG
Ministry of economic development	included in NG
ERSAF	included in NRI
National Research Council	included in NRI
Rice Association	included in NRI
CREA	included in NRI
CRPA	included in NRI
ISPRA	included in NRI
Universities	included in NRI
Fertiliser companies	included in SFC
Seed companies	included in SFC
Water management office	included in SRA
Large-scale farmers	included in FRM
Health Agency	not relevant
Regional DG Welfare	not relevant
Agricultural consultants' associations	not relevant
Breeders' trade unions	not relevant
Fish farmers' unions	not relevant
Fish farmers	not relevant
CONFINDUSTRIA	not relevant
Agricultural districts	not relevant
Compost producers	not relevant
Hydroelectric power plants	not relevant
Regional administration	reported as DGA, DGE

Figures

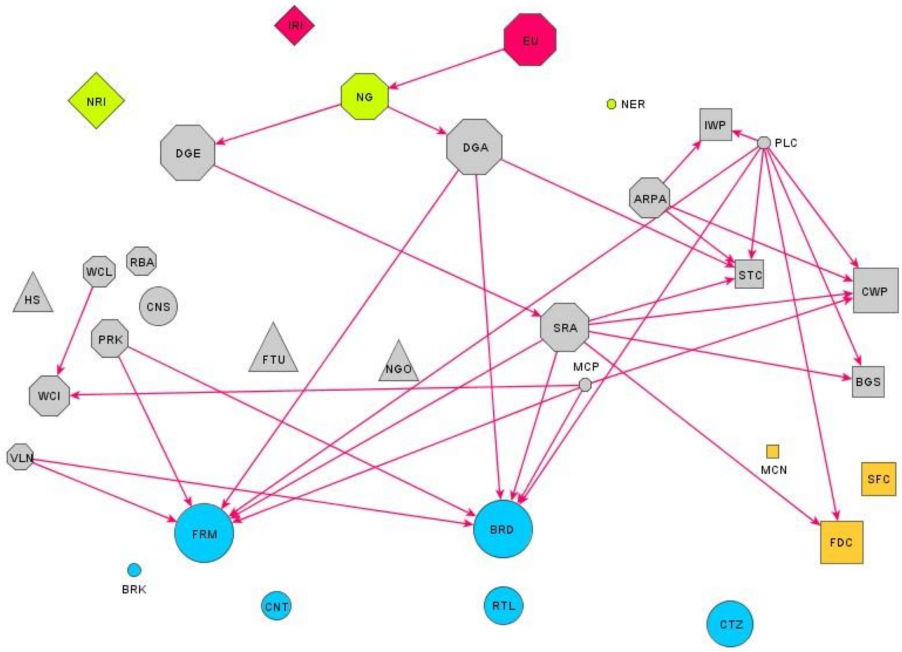

Figure S2.1 Authorization and control links in the governance framework related to groundwater contamination in the Lombardy plain. The colours of the nodes correspond to the levels of governance; pink: international, green: national, grey: sub-national, light blue: local. Orange nodes represent multilevel actors. The size of the nodes corresponds to the perceived influence of each actor.

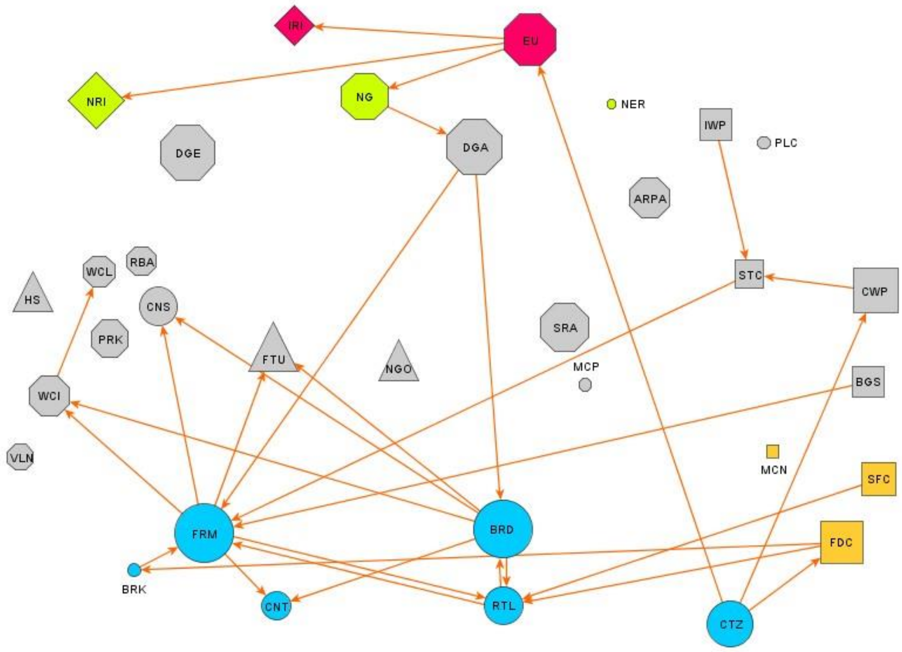

Figure S2.2 Money links in the governance framework related to groundwater contamination in the Lombardy plain. The colours of the nodes correspond to the levels of governance; pink: international, green: national, grey: sub-national, light blue: local. Orange nodes represent multilevel actors. The size of the nodes corresponds to the perceived influence of each actor.

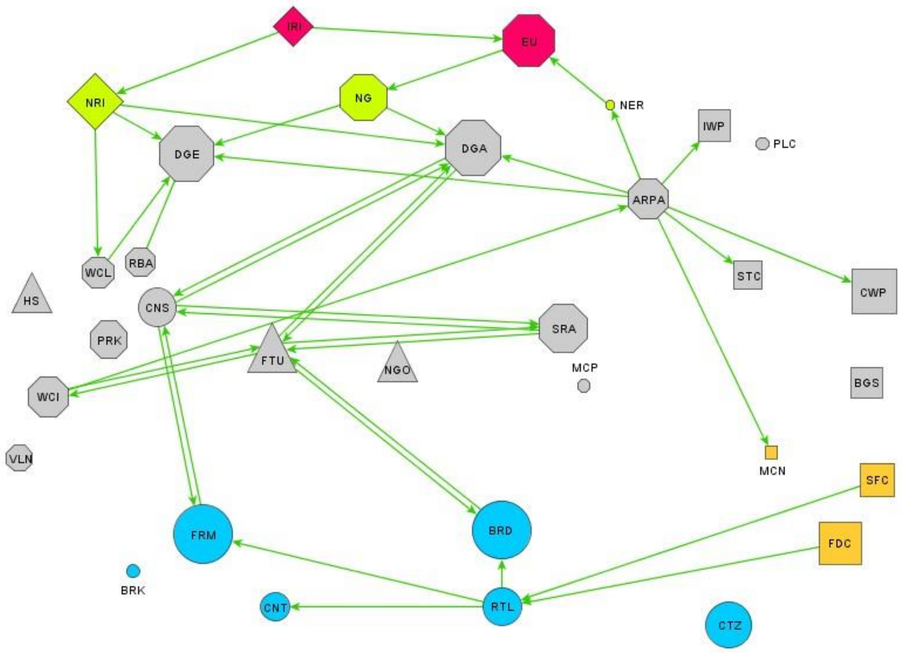

Figure S2.3 Technical information links in the governance framework related to groundwater contamination in the Lombardy plain. The colours of the nodes correspond to the levels of governance; pink: international, green: national, grey: sub-national, light blue: local. Orange nodes represent multilevel actors. The size of the nodes corresponds to the perceived influence of each actor.

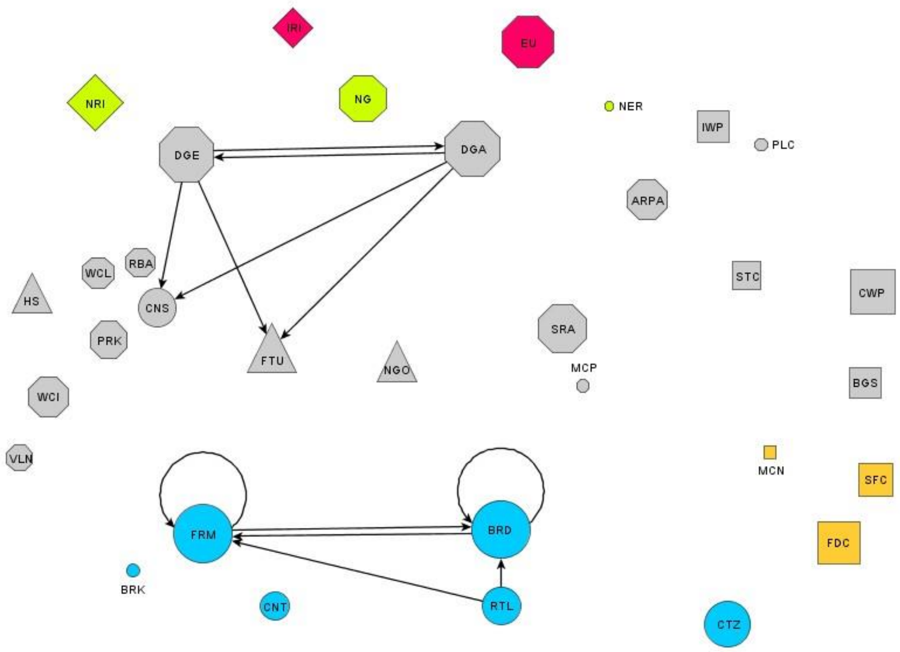

Figure S2.4. Advice links in the governance framework related to groundwater contamination in the Lombardy plain. The colours of the nodes correspond to the levels of governance; pink: international, green: national, grey: sub-national, light blue: local. Orange nodes represent multilevel actors. The size of the nodes corresponds to the perceived influence of each actor

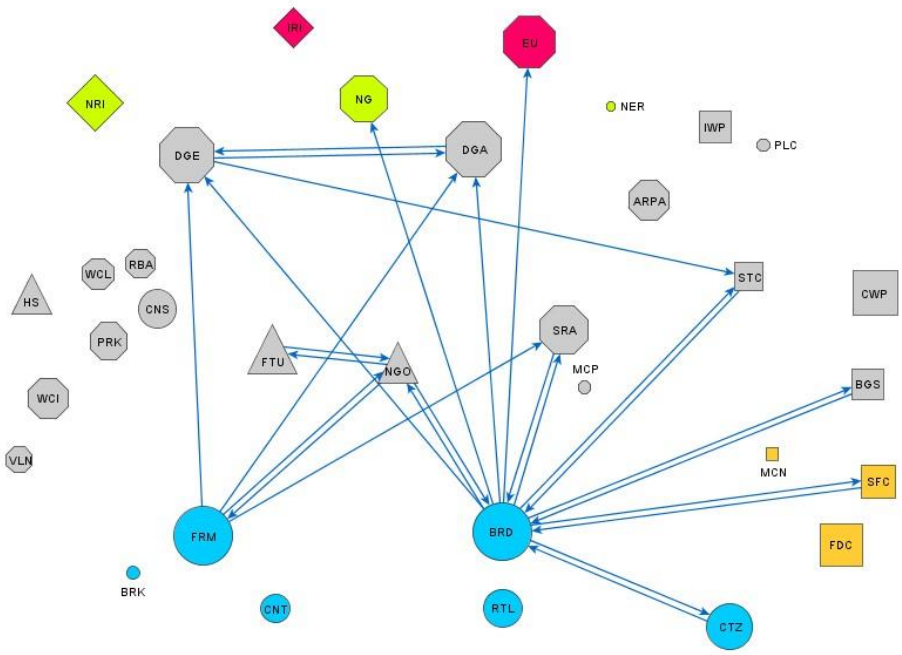

Figure S2.5 Conflict links in the governance framework related to groundwater contamination in the Lombardy plain. The colours of the nodes correspond to the levels of governance; pink: international, green: national, grey: sub-national, light blue: local. Orange nodes represent multilevel actors. The size of the nodes corresponds to the perceived influence of each actor.

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S3 Supplementary information for Chapter 5

Nitrogen budget calculation

N budget terms were first calculated for each of the 125 municipalities totally or partially included within the basin boundaries according to the following equations. N budget terms at the municipality scale were then corrected for the portion of each municipality area included within the basin and then aggregated at the watershed scale by GIS analysis (QGIS software 2.18) to obtain watershed scale values. The calculation were based on agriculture and farming data reported by the National Statistics Institution (6th Agricultural Census 2010; http://dati-censimentoagricoltura.istat.it).

N input from livestock manure was estimated by means of data of livestock density (divided in 8 major categories and more than 35 sub-categories according to their type, age, and purpose), average live weights and N excretion rates of each animal category (DM 07/04/2006, decree of the Italian Ministry of Agricultural and Forest Policies about agronomic utilization disciplinary, Table S3.1). We assumed that manure was used only for spreading on croplands within the municipality where each farm was located.

N input from manure was calculated as follows:

$$\boldsymbol{N_{Man}} = \sum_{\boldsymbol{livestock}} (\boldsymbol{Exc} * \boldsymbol{LW} * \boldsymbol{N})$$

where:

Exc = N excretion rate of each livestock category (kg t^{-1} live weight yr^{-1})

LW = live weight of each livestock category (t)

N = livestock density for each category

Until now, official agricultural statistics data on yearly synthetic fertilizers distribution are available only for N and P, and at the provincial level. These data were converted into nutrient amounts by means of average N and P contents and then weighted by the proportion of agricultural areas fertilized within each municipality (i.e. permanent crops plus arable land with the exception of N-fixing crops-alfalfa, soybean and legumes, only in case of N fertilizations). Data included simple mineral N (calcium cyanamide, nitrates, ammonium sulphate, urea, and others N fertilizers), simple P fertilizers (simple superphosphate, triple superphosphate, and other P fertilizers), compound mineral fertilizers (NP, NK, KP and NPK compounds), organic and organo-mineral fertilizers and soil amendments.

N input from synthetic fertilizers was calculated as follows:

$$\boldsymbol{N_{Fert}} = \frac{\sum_{\boldsymbol{Fert}}(\boldsymbol{F_{typ}} * \%\boldsymbol{N})}{\boldsymbol{UAA_{prov}}} * \boldsymbol{UAA_{tot}}$$

where:

F_{typ} = amount of each type of synthetic fertilizer (t)

%N = N content of each type of synthetic fertilizer (%)

UAA_{prov} = total agricultural surface in the province (ha)

UAA_{tot} = total agricultural surface in the municipality (ha)

Rates of biological N fixation associated with alfalfa, soyabean, permanent grasslands and pastures were estimated by multiplying the production per unit of surface (i.e. the yield of each specific N-fixing crop) by the N content in the harvested portions. This calculation provided the rates of N ascribed to the harvested

aboveground tissues, that was then corrected for a multiplicative factor expressing the ratio of total biomass produced with respect to harvested biomass. As reported by Anglade et al. (2015), we used the values of 1.7 for forage (alfalfa, permanent grasslands and pastures) and 1.3 for grain legumes (soya bean) corresponding to, respectively, 40% and 25% of total plant N as belowground N. According to the Rural Development Program of the Region, no N fertilization is allowed for N-fixing crops. The obtained N-fixation rates resulted in the range of those reviewed by Herridge et al. (2008), and were extended to the total surface of each N-fixing crop in the basin. Rice is not an N-fixing crop but is cultivated under conditions where N fixation by cyanobacteria occurs, thus a literature range of rates was used (20-90 kg N ha^{-1} yr^{-1}; Herridge et al., 2008).
N input from symbiotic biological fixation was calculated as follows:

$$\boldsymbol{N_{Fix}} = \sum_{\boldsymbol{N-fixing\ crops}} (\boldsymbol{Y} * \boldsymbol{Upt} * \boldsymbol{BGN} * \boldsymbol{UAA})$$

where:
Y = yield of the harvest portion of each N-fixing crop (t ha^{-1})
Upt = N content of the harvest portion of N-fixing crop (kg t^{-1})
BGN = factor expressing the ratio of total biomass produced with respect to harvested biomass
UAA = surface occupied by each N-fixing crop (ha)

For non-symbiotic N-fixation in woody crops and other arable land, we adopted literature rates of 1-10 and 1-8 kg N ha^{-1} yr^{-1}, respectively (McKee and Eyre, 2000; Herridge et al., 2008; Butterbach-Bahl et al., 2011). Other N-fixing crops (e.g. pulses) were not considered since the occupied surfaces were negligible in the investigated basin.
Average values of atmospheric oxidized N deposition were set in 8.5 kg N ha^{-1} yr^{-1}. N input from atmospheric deposition was calculated as follows:

$$\boldsymbol{N_{Dep}} = \boldsymbol{A_{Dep}} * \boldsymbol{UAA_{tot}}$$

where:
A_{Dep} = areal values of atmospheric deposition of N (kg ha^{-1} yr^{-1})
UAA_{tot} = total agricultural surface (ha)

N output from crop harvest was calculated by means of the surface, the N content and the annual yield in the harvested portions of each crop (Table S3.2).
N output from crop harvest was calculated as follows:

$$\boldsymbol{N_{Harv}} = \sum_{\boldsymbol{crops}} (\boldsymbol{Y} * \boldsymbol{Upt} * \boldsymbol{UAA})$$

where:
Y = yield of the harvest portion of each crop (t ha^{-1})
Upt = N content of the harvest portion of each crop (kg t^{-1})
UAA = surface occupied by each crop (ha)

Fertilizer application to agricultural soils can result in significant losses via ammonia volatilization and denitrification (Hofstra et al., 2005; Pan et al., 2016). Direct measurements of N losses to the atmosphere via ammonia volatilization and denitrification are very limited for the Italian agriculture, in terms of national territory coverage, representativeness of the employed method, and type of fertiliser and application strategies tested (Arcara et al. 1999; Minoli et al. 2015). According to the budget methodology previously applied to other sub-basins of the Po River system (Soana et al. 2011; Castaldelli et al. 2013), ammonia volatilization factors (expressed as the average percentage lost of the supplied N) of 32.5%, 16%, 6% and 2% were associated to animal manure, urea plus ammonium sulphate, nitrates and other N fertilizers, respectively. These values were coherent to direct emission estimations performed in the Italian territory, as reviewed by Minoli et al. 2015. About 60% of the volatilised ammonia was assumed to be re-deposited locally and only the remaining 40% was considered as a true N output from the agricultural system.

N output from NH_3 volatilization was calculated as follows:

$$\boldsymbol{N_{Vol}} = (\boldsymbol{N_{Man}}\,, \boldsymbol{N_{Fert}}) * \%\ \boldsymbol{loss}$$

where:

N_{Man} = N in livestock manure applied to agricultural soils

N_{Fert} = synthetic N fertilizer applied to agricultural soils

% loss = percentage lost of the supplied N

N output from denitrification in soils was calculated as 10% of the supplied N as both synthetic fertilizations and animal manure, spanning the main soil types of the Po River lowlands, according to a review of a vast body of literature (Castaldelli et al., 2013). Since coupled nitrification–denitrification has been suggested to be a relevant N sink in rice paddy soils, emission factor for the applied N was set here to 30% (Nicolaisen et al., 2004; Liu et al., 2008).

N output from denitrification is soils was calculated as follows:

$$\boldsymbol{N_{Den}} = (\boldsymbol{N_{Man}}\,, \boldsymbol{N_{Fert}}) * \%\ \boldsymbol{loss}$$

where:

N_{Man} = N in livestock manure applied to agricultural soils

N_{Fert} = synthetic N fertilizer applied to agricultural soils

% loss = percentage lost of the supplied N

Table S3.1 Livestock categories, live weights, and N excretion rates.

Category	**Sub-category**	**Live weight (kg)**	**N excretion rate (kg N t^{-1} live weight yr^{-1})**
Cattle	cattle under one year old: males	220	67
	cattle under one year old: females	220	67
	cattle one year old and over but less than two: males	350	84
	cattle one year old and over but less than two: females	300	120
	cattle two years old and over: males	800	84
	cattle two years old and over: heifers for breeding	600	120
	cattle two years old and over: heifers for slaughtering	600	84
	cattle two years old and over: dairy cows	600	138
	cattle two years old and over: other cows (for meat and/or work)	600	120
Buffaloes	buffalo calves	200	67
	famale buffaloes	600	138
	other buffaloes	300	84
Equidae	Horses	350	69
	other (donkeys, mule and hinny)	200	69
Goats	Goats	50	99
	other goats	25	99
Sheep	breeding females: dairy sheep	60	99
	breeding females: other sheep	50	99
	other sheep	35	99
Pigs	piglets having a live weight of under 20 kg	15	110
	pigs with a live weight between 20 and less of 50 kg	40	110
	pigs for fattening: 50 to less than 80 kg	65	110
	pigs for fattening: 80 to less than 110 kg	95	110
	pigs for fattening: 110 kg and above	120	110
	male for reproduction	250	110
	mated sows	180	101
	other sows	180	101
Poultry	Broilers	1	250
	laying hens	1.9	230
	Turkeys	6.5	167
	guinea fowl	0.8	240
	Goose	6.5	211
	poultry - other	1	211
	Ostriches	90	165
Rabbits	rabbit: breeding female	3.5	143
	other rabbits	1.7	143

Table 3.2 Crop parameters used in the calculation of N budget (main crops, yields, N content of the harvested portions).

Type	**Crop**	**Yield (t ha^{-1})**	**N content (kg N t^{-1})**
Cereals	common wheat and spelt	6.0	21.0
	durum wheat	5.5	22.8
	rye	4.0	19.3
	barley	5.9	18.1
	oats	4.0	19.1
	grain maize	11.5	14.9
	sorghum	7.0	15.9
	other cereals	5.0	18.0
Industrial Crops	potato	34.0	4.2
	sugar beet	74.0	2.2
	rape and turnip rape	3.0	3.4
	sunflower	3.5	28.0
	soya beans	4	58.2
Fresh Vegetables	tomato for processing	70.0	2.6
	other fresh vegetables	32.5	5.0
Temporary grassland	alfalfa	11.2	27.0
	other multi-annual temporary grass	10	21.5
	annual grass: green maize consumed directly	60	4.0
	annual grass: green maize for silage	40	4.0
	other monophytous annual grass, cereal	30	4.0
	other annual grass	8.0	26.0
Permanent grassland	meadows	6.0	19.7
	pastures	3.0	8.0
Permanent woody crops	vineyard	10.5	2.0
	oil olives	19.2	10.0
	apple-tree	27.1	0.6
	peach-tree	19.3	1.3
	apricot-tree	12.6	1.3
	other fruits from temperate climate zones	15.2	1.0
	pear-tree	18.1	0.6
	nectarine-tree	21.1	1.4
	kiwi-tree	17.4	1.5
	nursery	10.0	2.0
	other permanent woody crops	10.0	6.0

Numerical flow model

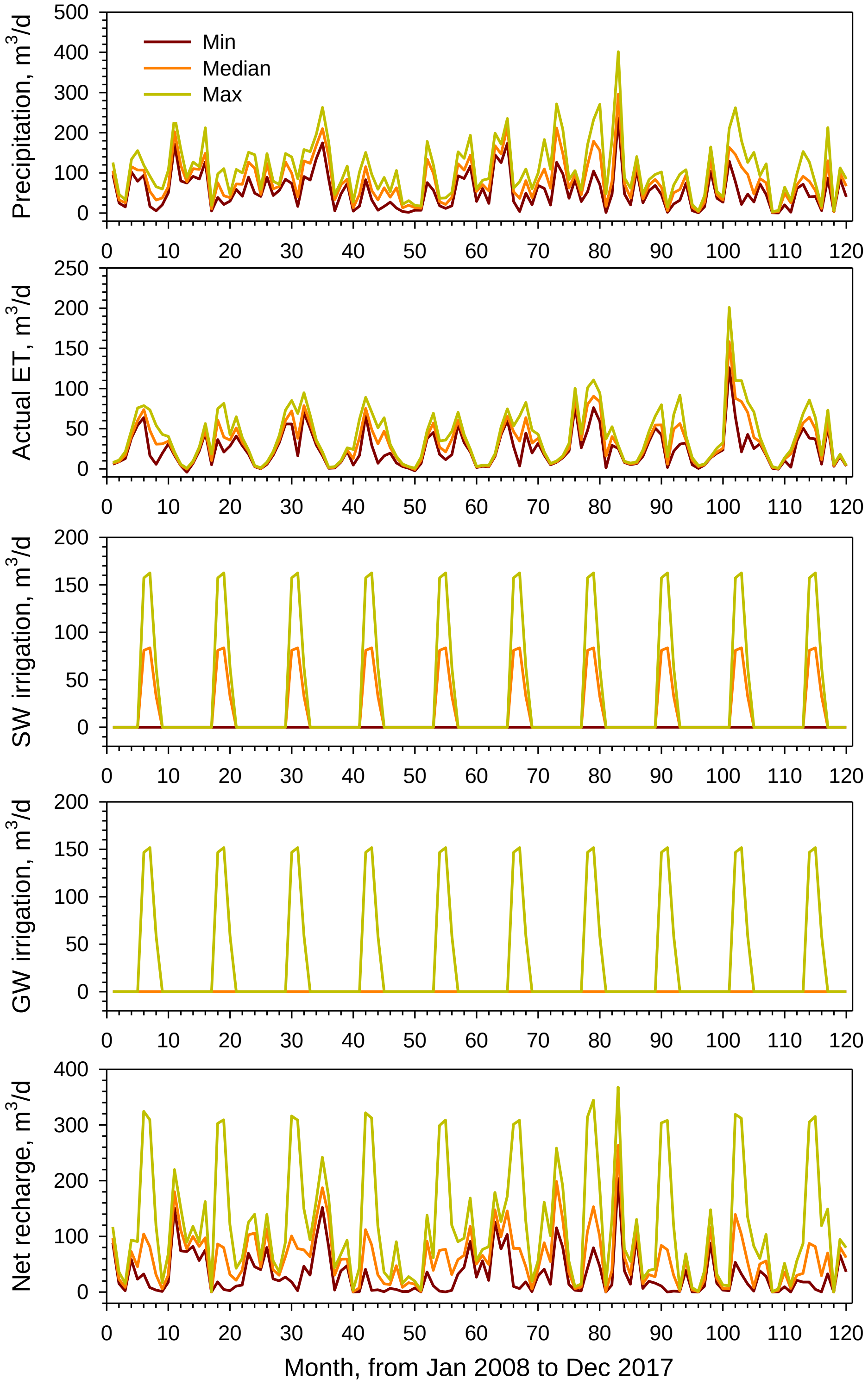

Figure S3.1 Monthly variability of the hydrological components embedded in the net recharge term: Precipitation, Actual evapotranspiration (ET), Surface Water Irrigation, and Groundwater irrigation, given as the monthly minimum, median and maximum values of all municipalities. As data correspond to municipalities, the volume is estimated for all their area. SW: Surface Water; GW: Groundwater.

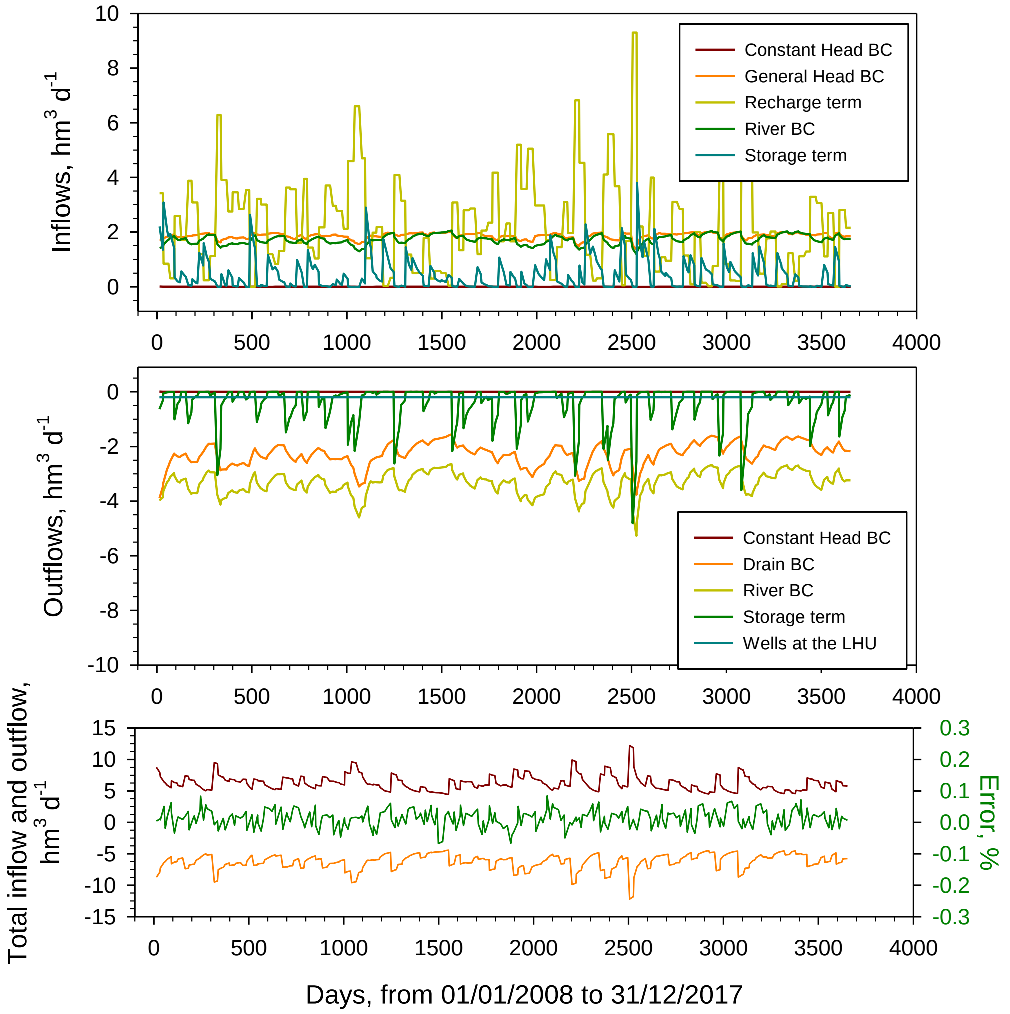

Figure S3.2 Monthly results of the inflow and outflow terms from the flow model mass-balance.

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