SOILS2SEA DELIVERABLE NO. 6.1

BONUS
SCIENCE FOR A BETTER FUTURE OF THE BALTIC SEA REGION

Thresholds for nitrogen in groundwater and streams - a new concept for improved land use regulation and protection of the Baltic Sea and its coastal waters

SOILS2SEA

Reducing nutrient loadings from agricultural soils to the Baltic Sea via groundwater and streams

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Thresholds for nitrogen in groundwater and streams - a new concept for improved land use regulation and protection of the Baltic Sea and its coastal waters

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SOILS2SEA

Reducing nutrient loadings from agricultural soils to the Baltic Sea via groundwater and streams

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Content

Executive summary

The Water Framework Directive, “WFD” (EC, 2000) and the Groundwater Directive, “GWD” (EC, 2006) stipulate that EU member states have to ensure good status of all European water bodies including groundwater, streams, lakes, transitional and coastal waters. The WFD describes in general terms the need for developing quality standards and estimating pollution loads to ensure good status of surface water bodies e.g. lakes and marine waters, while the GWD in more specific terms requires EU member states to derive and establish groundwater threshold values to ensure compliance with good status objectives of the WFD. The good status objectives include good ecological status of groundwater dependent terrestrial and associated aquatic ecosystems (surface water bodies).

Despite this requirement most EU member states still have not established groundwater threshold values for e.g. total nitrogen or nitrate for protection of groundwater dependent or associated ecosystems neither in the first (Scheidleder, 2012) nor the second river basin management plans (EC, 2015a, b). Member states typically point to the lack of data and knowledge as the main reason for not deriving threshold values based on environmental objectives for ecosystems.

In this report we present some general principles for development of groundwater (incl. drains) and stream threshold values for nitrate (or total nitrogen) and illustrate these with concrete examples from two different catchments in Denmark and Poland selected among the case study sites in BONUS SOILS2SEA. Both catchments are groundwater dominated but differ substantially with respect to hydrogeology, nitrate leaching and regulation requirements. The objective of our study is to analyse and discuss challenges related to implementation of the GWD groundwater threshold policy instrument when used in relation to spatially uniform and spatially differentiated regulation. The aims of the report are i) to provide input to GWD guidances, and ii) to inspire to more related research and studies on derivation of groundwater thresholds.

Based on the analyses of the two cases we recommend that

- The GWD guidances presently focussing on concentrations as thresholds should be extended to include loads (fluxes) of nitrate in streams as possible thresholds.
- The GWD guidances should emphasize that analysis of the groundwater body and its interaction with groundwater dependent terrestrial ecosystems and groundwater associated aquatic ecosystems should be carried out through joint use of monitoring and spatially distributed coupled groundwater/surface water modelling.
- The GWD guidances should emphasize that use of TVs for spatially differentiated regulation cannot be done entirely from scientific principles, but is conditioned by associated political decisions taking both ecological and economical considerations into account.

1 Introduction

Nutrient loadings (mainly N and P) create eutrophication, harmful algal blooms and hypoxia (decreased oxygen levels) in the Baltic Sea and its transitional and coastal waters. The major source is agricultural activities in the Baltic Sea basin, which pollute groundwater, rivers, transitional and coastal waters (HELCOM, 2011). The pollution of the transitional and coastal waters occurs mainly via rivers and to a lesser extent via atmospheric deposition (airborne pollution). A significant part originates from groundwater discharging either directly or via streams to the coastal waters (Hinsby et al., 2008, 2012; EC, 2015a). The Groundwater Directive (GWD) provides a policy instrument (groundwater thresholds) to assess groundwater chemical status based on environmental objectives for legitimate uses of groundwater as well as dependent terrestrial and associated aquatic ecosystems or surface water bodies (EC, 2006).

The objective of the present report is to analyse and discuss challenges related to implementation of the GWD groundwater threshold policy instrument when used in relation to spatially uniform and spatially differentiated regulation. The aims of the report are i) to provide input to GWD guidances, and ii) to inspire to more related research and studies on derivation of groundwater thresholds. The analyses are exemplified through two case studies within the Baltic Sea drainage basin.

2 Legal background

2.1 The Water Framework and Groundwater directives

The Water Framework Directive (WFD) and the Groundwater Directive (GWD) (EC 2000, 2006) are the most important EU directives for inland water bodies stipulating the requirements for compliance with good status objectives and the general principles for assessing the chemical status of water bodies e.g. in catchments of transitional and coastal waters.

The WFD and GWD stipulate that EU member states have to ensure good status of all European water bodies including e.g. groundwater, transitional and coastal waters in 2027 at the latest. The WFD describes in general terms the need for developing quality standards and estimate pollution loads to ensure good status of marine waters. The GWD requires in more specific terms EU member states to derive and establish groundwater threshold values to ensure compliance with good status objectives of groundwater dependent terrestrial ecosystems and groundwater associated aquatic ecosystems (surface water bodies), if existing quality standards do not ensure compliance with these.

2.2 CIS working groups and guidances

The Working Group Groundwater (WGG, previously WG C) of the Common Implementation Strategy (CIS) for the Water Framework Directive, which is currently chaired by the Commission (DG Environment) and the Environment agencies of Austria and the United Kingdom, closely follows the implementation of groundwater related aspects of the WFD and GWD through bi-annual meetings and ad hoc activities. The WGG assesses and advices on the progress of the implementation of the WFD/GWD and develops technical reports and guidances for the member states to ensure proper and harmonized implementation of the WFD/GWD. There is a long list of guidances and reports, which are relevant for the assessment of groundwater chemical and quantitative status according to the WFD. In the context of this study the most important of these guidances is guidance number 18 on groundwater status and trend assessment (EC, 2009). Additionally, the technical report on groundwater associated aquatic ecosystems (EC, 2015a) is highly relevant, when assessing groundwater chemical status based on good status objectives and ecological status of dependent or associated aquatic ecosystems.

2.3 Implementation status

Despite the requirement in the GWD and the associated guidance in the CIS guidance documents, most EU member states still have not established groundwater threshold values for e.g. total nitrogen or nitrate for protection of groundwater dependent or associated ecosystems, neither in the first (Scheidleder, 2012) nor the second river basin management plans (EC, 2015a, b). Member states typically point to the lack of data and knowledge as the main reason for not deriving threshold values based on environmental objectives for ecosystems.

3 Concept for deriving groundwater threshold values

3.1 General concept and related implementation challenges

The general concepts for derivation of groundwater threshold values are described in guidance no. 18 of the Common Implementation Strategy (CIS) of the WFD (EC, 2009) and illustrated by Hinsby et al. (2008). Hinsby et al. (2012) subsequently demonstrated how the concept can be utilised to derive threshold values for concentrations of N and P in groundwater bodies in the catchment to Horsens Fjord in Denmark.

The conceptual model for the Horsens Fjord catchment and the concept (workflow) used by Hinsby et al. (2012) to derive threshold values (TVs) are illustrated in Figure 1.

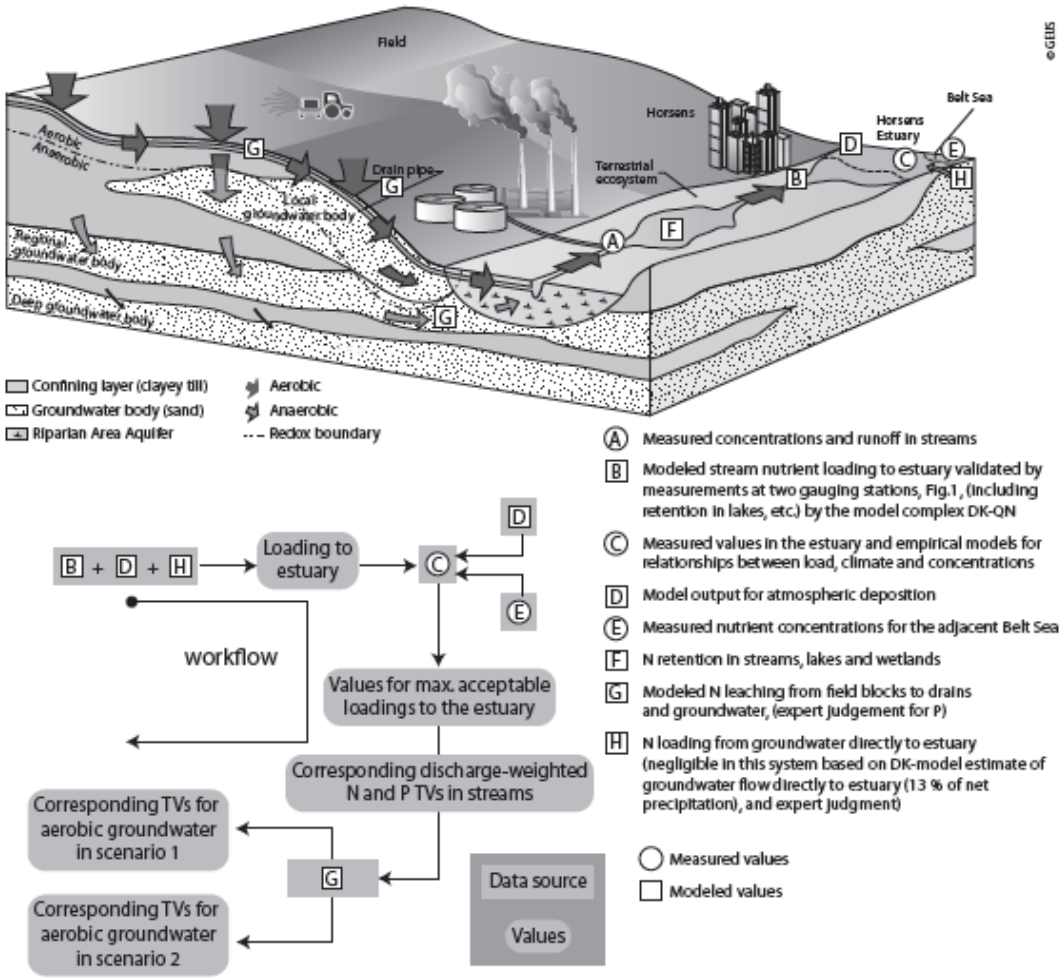

Figure 1. Conceptual model of the catchment of Horsens estuary in Denmark with indication of data and nutrient sources. The work flow in calculation of threshold values (TVs) for streams and groundwater is indicated (Hinsby et al., 2012).

While the idea behind the general concept is clear, and derivation of threshold values appears as an operational policy instrument, the lacking implementation in all member countries indicates that

the countries face challenges that may go beyond the stated “excuses” of lack of data and knowledge. In this respect we will discuss some apparent challenges related to protection of groundwater associated aquatic ecosystems against nutrients (in particular N) loadings from agriculture – seen from a practical implementation point of view.

It is outside the scope of the present report to assess and discuss the challenges related to assessing the aquatic ecosystems and the associated threshold values (TVs) for fluxes and/or concentrations required to achieve good ecological status in the aquatic ecosystems. Instead, we focus on how to derive TVs for the water bodies in the upstream catchment and the challenges related to these “backcalculations”. Here we see three key challenges related to i) the conceptual understanding of the water flow, transport and fate of nutrients; ii) type of threshold value (TV); and iii) the mitigation strategy on how to distribute abatement targets among different water bodies.

Conceptual understanding of flows, transport and fate of nutrients.

A key challenge for establishing relevant threshold values lies in the conceptual understanding of where the water and nutrients coming to the aquatic ecosystem originate. If we look at the example in Figure 1, several aquifers, located at different depths and at different spatial locations within the catchment, contribute with fluxes of water and nutrient. Similarly, a considerable part of the fluxes originates from drain pipes, while some comes as overland flow. To complicate things further, the concentrations differ significantly between and within the various water bodies, both in time and in space. While some of these fluxes may be measured, albeit with large uncertainties, in exceptionally well instrumented research catchments like hydrological observatories (Jensen and Illangesekare, 2011), the only realistic way to quantitively assess these fluxes in real-life catchments is through a combined use of models and standard monitoring data. With the sparse data typically available in most catchments, such model assessments of internal catchment fluxes are quite uncertain.

Measurements of representative concentrations in aquifers are difficult in practise. It is well known that model prediction uncertainties, as well as representativeness of field data, are larger at small support scales than at larger support scales. This implies for instance that the ability of a point observation, such as concentrations in a water sample from a borehole, to represent average value for a small aquifer is much less than measurements of concentrations in a stream or drain pipe draining the aquifer. Hence, to achieve a reliable estimate of the average aquifer concentration samples from many boreholes are required, which may be significantly more costly than measuring in the stream or pipe. Similarly, measurements of fluxes in aquifers are much more difficult than flux measurements in streams or drain pipes.

Hence, in order to design suitable threshold values a significant amount of data and scientific knowledge are required to understand the interactions between the pollutants and the species of the ecosystem.

Type of threshold value

Threshold values for the protection of aquatic ecosystems may be established in two ways:

- as concentration levels to protect certain aquatic species in e.g. specific groundwater associated aquatic ecosystems depending on the ecotoxicity of the specific pollutant towards the species to be protected (Camargo and Alonso, 2006), or
- as target values or maximum total fluxes/loads of the relevant pollutants.

For coastal and marine waters the most commonly used approach in Europe and globally is to establish annual or daily targets or maximum loads for the protected aquatic ecosystem (HELCOM 2011, Timmermann et al., 2010, 2014, Riemann et al., 2016).

In addition to spatial variations, concentrations and fluxes are known to exhibit large temporal variations. Depending on the aquatic ecosystem the relevant threshold may be maximum concentration or fluxes at e.g. daily, monthly or annual support scales.

Mitigation strategy on spatially distribution of targets for lowering of N-loads.
Lowering of N-loads can be achieved in many different ways. A common approach used until now in most countries, is to impose regulations, in terms of requirements for N-leaching from the root zone, uniformly to all areas within the catchment. As the natural N-reduction, taking place along the transport paths between the leaching out of the root zone and the sea, differs very much between different fields, this is not very cost-effective. Another, spatially differentiated, strategy would be to impose uniform targets for N-lowering (% decrease) in N-loads. This last approach is more cost-effective, but will at the same time affect individual farmers differently and hence be politically more challenging. Some of the potential gains and challenges of spatially differentiated regulations have been analysed in BONUS SOILS2SEA (Hansen et al., 2017; Hashemi et al., 2018; Wachniew et al., 2018; Stelljes et al., 2017)

Different choices of mitigation strategy will lead to different threshold values. As the choice of mitigation strategy essentially reflects political priorities, threshold values will always be conditioned by these political priorities.

3.2 Framework

We propose the following methodological framework (Figure 2) for deriving and applying thresholds values (TVs):

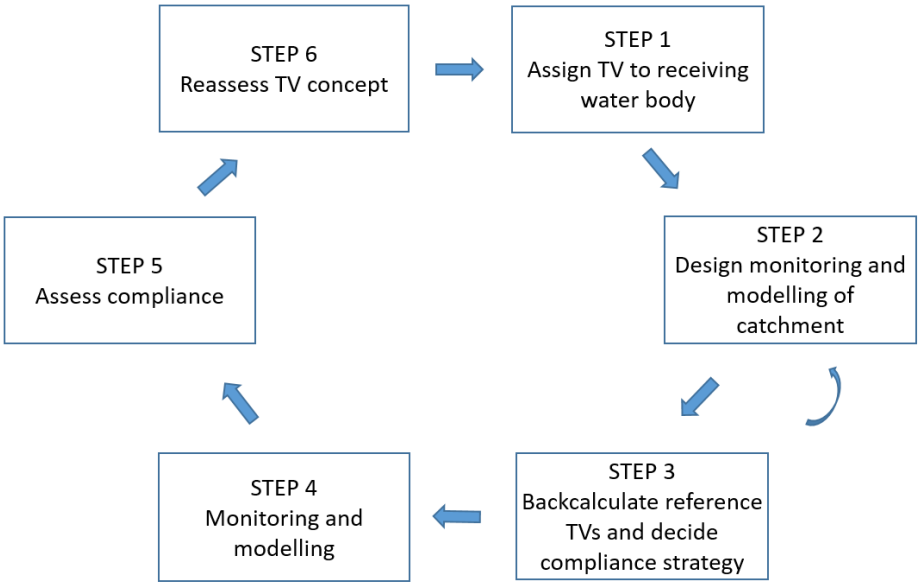

Figure 2. Methodology for deriving and applying threshold values (TV).

STEP 1 – Assign TV to receiving water body

The TV for the receiving water body/receptor ($TV_{ecosystem}$) should be assigned on the basis of ecosystem analyses. The $TV_{ecosystem}$ can be related to concentration values or fluxes and should reflect at which temporal support scale (daily, seasonal, annual, multi-annual) the critical concentration/load applies to. For many coastal ecosystems $TV_{ecosystem}$ is typically expressed as a maximum annual loads, like in the Baltic Sea Action Plans (HELCOM, 2007, 2013).

STEP 2 – Design monitoring and modelling of catchment

The monitoring system should be designed on the basis of the conceptual understanding of the flow and transport processes within the catchment and best possibly match the type of $TV_{ecosystem}$ selected for the receiving water body/receptor. If the $TV_{ecosystem}$ for instance is a load, the monitoring should be based on flux measurements, which are typically made in streams. If the $TV_{ecosystem}$, on the other hand, is a concentration the monitoring may be either concentrations, which, depending on the catchment hydrogeological understanding, may be monitored in streams/drain pipes or in boreholes.

As monitoring in practise is not able to capture all fluxes out of the catchment, e.g. because of ungauged areas or subsurface outflows, it will most often be required to combine the monitoring with modelling. Another benefit of a combined monitoring-modelling approach is that models forced by climate data can quantify the effects of climate variability on the fluxes, and in this way enable a separation of the climate signal and the signal from a mitigation measure (Højberg et al., 2007).

STEP 3 Backcalculate reference TVs and decide compliance strategy

Decisions on how compliance should be maintained or achieved include considerations on at which spatial scale compliance should be monitored. It may be for the entire catchment, or the catchment may be divided into a number of subunits, e.g. subcatchments or specific aquifers, for which individual TVs are assigned. In practise the decisions on division into subunits need to go hand-in-hand with the conceptual understanding of the catchment hydrogeology and will be limited by the possibilities to perform reliable monitoring and modelling at subcatchment scale (STEP 2).

TVs should be specified for variables that can reliably be assessed through combined monitoring and modelling. The two obvious locations on the flow paths between the field and the receiving water body are the leaching from the root zone (TV_{leach}) and the load to the receiving water (TV_{load}). In between the root zone leaching and the load to the receiving water some of the N will be reduced either in groundwater ($N\text{-}reduction_{GW}$) or in surface water ($N\text{-}reduction_{SW}$). Two options for calculating TVs for a reference situation, corresponding to the $TV_{ecosystem}$, are:

- TV_{ref} strategy A – equal leaching. $TV_{leach,ref}$ is kept constant (pr unit area) for all subunits.
- TV_{ref} strategy B – equal load. $TV_{load,ref}$ is kept constant (pr unit area) for all subunits.

Strategy A correspond to a uniform regulation, where all units are treated equally independent of their respective hydrogeological characteristic, while strategy B corresponds to spatially differentiated regulation, where the spatial variation of nature's own capacity to reduce N is exploited. In this respect, it will be required to use maps for N-reduction in groundwater and in surface water (Hansen et al., 2014; Højberg et al., 2017). Such N-reduction maps have similarities to the dilution and attenuation factor described by the CIS guidance on groundwater status and trend assessment (Hinsby et al., 2008; EC, 2009).

The TVs for each subunit within the catchment should be calculated based on the compliance strategy decided in STEP 2 and the knowledge from monitoring and modelling available from STEP 3.

If loads or concentrations need to be decreased to comply with the $TV_{ecosystem}$ assigned for the receiving water body, different mitigation strategies should be considered. In case of decreasing N-loads to a coastal water body, such strategies may include, but not necessarily be confined to, the following options that can be seen as extensions of the above TV_{ref} strategies:

- Mitigation strategy A: Each subunit should make a proportional decrease (in % relative to present situation) in N-leaching from the root zone resulting in $TV_{mit,leach}$.
- Mitigation strategy B: Each subunit should make a proportional decrease (in % relative to present situation) in N-load from the subunit to the coastal water body resulting in $TV_{mit,load}$. In this case the spatial differences in N-reduction between the root zone and all the way to the sea, including the river system, is taken into account.

STEP 4 –Monitoring and modelling

The monitoring is performed during a WFD cycle, and the monitoring results are currently analysed using a combined monitoring-modelling strategy.

STEP 5 – Assess compliance

The compliance is assessed by comparing the assigned TVs to the results from STEP 4. Due to uncertainties in the monitoring and modelling results, including the “noise” induced by climate variability, the conclusion on compliance or not may be ambiguous and should be expressed in probabilistic terms.

STEP 6 – Reassess TV concept

Based on the experiences from applying the concept, including the conclusions from STEP 5, the TV concept emerging from decisions in STEP 1 + STEP 2 should be reviewed and modified as required.

4 Examples

We have applied the TV framework to two of the BONUS SOILS2SEA catchments, Norsminde in Denmark and Kocinka in Poland. Both catchments are groundwater dominated, but otherwise very different. Norsminde has a relatively high pressure from intensive agriculture, very fast travel times from soils to streams and a large requirement from WFD plans to decrease N-loads, and the N-reduction in groundwater is high (larger than 50% averaged over the catchment). Kocinka has a relatively low pressure from present day's agriculture, but substantial nitrate concentrations in the aquifer due to past agricultural activities. The problems due to past activities are still relevant today due to a long residence time for nutrients in the unsaturated and saturated zones (median lag time around 20 years). Kocinka is furthermore characterised by a low N-reduction in the aquifer (15-30%).

While the two examples are based on real catchments applying best possible knowledge on the hydrogeology and nitrate status in the two catchments, it must be emphasised that the "decisions" made during different steps of the TV framework for the two catchments are hypothetical and only serve to illustrate the framework. Hence, they are not in accordance with present government approved WFD plans in Denmark and Poland.

4.1 The Norsminde catchment, Denmark

4.1.1 Description of the catchment

The Norsminde Fjord catchment is located on the east coast of Jutland in Denmark (Figure 3). The 109 km^2 catchment is intensively farmed with about 65% of the catchment area being agricultural land.

The catchment is dominated by a moraine landscape from Weichsel with mainly clayey soils and some sandy soils in the southern part of the catchment. The topography varies from around 100 m above - to sea level. An extramarginal stream valley from Weichsel, running from Southwest to Northeast, divides the catchment into a western more elevated and rather hilly part and an eastern part consisting of a flat low lying plain. The climate is temperate with an average precipitation for the period 1995-2003 of 773 mm/yr. and an average evapotranspiration estimated to 555 mm/yr. Rævs stream and its tributaries contribute to the main part of the discharge from the catchment to the fjord. The average discharge at the most downstream gauging station (area 86 km^2) was 232 mm/yr for the period 1995-2003. The clayey soils in most of the area are typically drained using tile pipe drains. The drains highly influence the subsurface flow paths in the catchment (Jakobsen et al., 2018).

The stratigraphy in the Norsminde area consists of Paleogene and Neogene sediments covered by a sequence of Pleistocene glacial deposits. The Paleogene layers consist of fine-grained marl and clay, which has low permeability. The Neogene layers above comprise a Miocene sequence of marine origin, typically up to 40 m thick. The formation is clay-dominated but with interbedded sand units, which can be more than 10m thick. The Miocene is only found in the western part of the catchment and the glacial deposits are therefore found directly above the Paleogene clay in the eastern part. In some parts of the area, the Paleogene and Miocene deposits are cut by buried valleys, in particular in the southern part of the catchment where the Boulstrup tunnel valley is

found. The glacial sequence consists of both sandy and clayey sediments. The clay deposits include a variety of lithologies from glaciolacustrine clay to clay till, whereas the sandy deposits mainly are of glaciofluvial origin. The clayey sediments dominate the sequence with the sandy sediments occurring as small and distributed units within the clay. The glacial sequence is in some areas heavily tectonically deformed with occurrences of rafts of Paleogene clay (He et al., 2014).

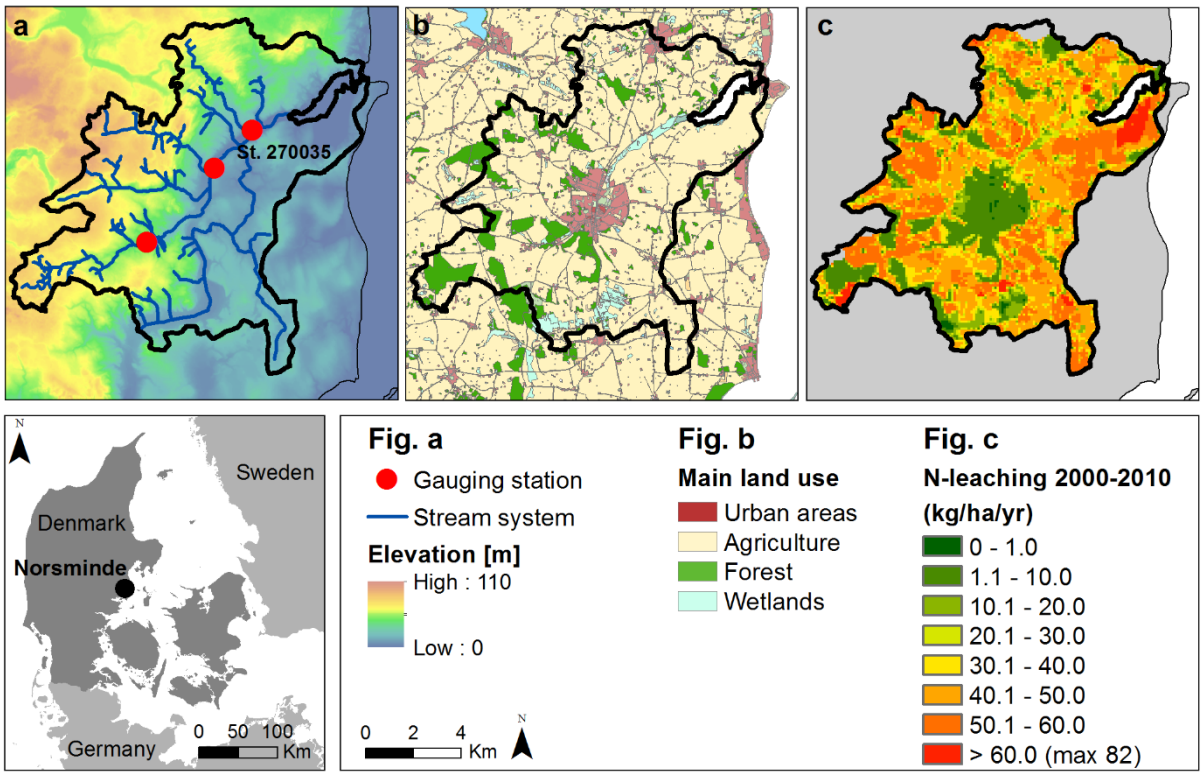

Figure 3. The Norsminde catchment area with A) topography, river network and gauging stations; and B) land use; and C) N leaching. Figure adapted from Hansen et al. (2014).

The area has been subject to comprehensive studies, both with respect to field measurements and modelling. The data used in this example comes from the modelling work undertaken by Hansen et al., 2014, 2017).

For water management purposes Denmark has been delineated into around 3,000 subcatchment of about 15 km^2 (ID15 catchments). There are six ID15 catchments within the Norsminde catchment (Figure 5).

Figure 4 Typical landscape in the Norsminde catchment (Photo: Vibeke Ernstsen)

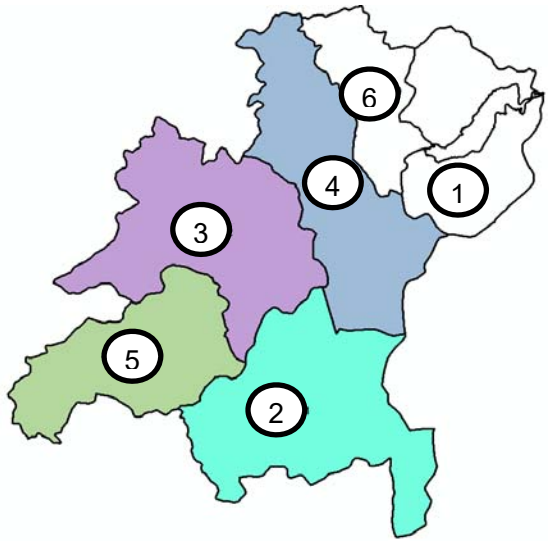

Figure 5 The six ID15 catchments in Norsminde. The numbers 1-6 shown in the figure and used in this report correspond to the official ID15 catchment numbers 43600028, 43600041, 43600042, 43600043, 43600051 and 43600099.

4.1.2 Application of TV framework

STEP 1 - Assign TV to receiving water body

The Norsminde catchment drains to Norsminde Fjord, which is a coastal water body. The average N-load to Norsminde Fjord 2000-2010 has been estimated to 147 t TN/year. According to the latest WFD plan a TV corresponding to a maximum TN load of 62 t TN/year has been calculated. Considering that the nitrate leaching was decreased by around 50% between 1980 and 2010, this additional requirement for a decrease of N-load of 58% is very tough for the agricultural sector. Such decrease will only be possible by abandoning about half of the agricultural area and convert these areas to nature (Ørum et al., 2017). To provide an example that is more representative for Danish conditions, we instead adopt a target for N-decrease of 30% corresponding to a N-load of 103 t TN/yr.

STEP 2 – Design monitoring and modelling of catchments

The main gauging station 270035 (Figure 3) measuring both discharge and nitrate concentrations, i.e. N-flux, is located at the outlet from ID15 catchment 4, and hence measures the load originating from catchments 2+3+4+5. The other two gauging stations shown in Figure 3, located at the outlet of ID catchments 3 and 5, only measure discharge and not nitrate. Thus, most of the six subcatchments are ungauged with repect to nitrate flux and many of them also for discharge. The estimates of nitrate fluxes at ID15 catchment level therefore have to be made by combining the existing monitoring data with a spatially distributed hydrological model and empirical models for leaching of nitrate from the root zone.

The estimates of N-leaching from the root zone are made at field level using the NLES model (Kristensen et al., 2008) using input data on crop rotation, fertilizer and manure, N-fixation, percolation, soil type and soil content of clay and organic matter (Højberg et al., 2015; Hansen et al., 2017). The transport and fate of nitrate was subsequently simulated using a coupled groundwater-surface water hydrological model based on MIKE SHE (Hansen et al., 2017). This model was used to derive maps for N-reduction in groundwater. The estimates of N-reduction in surface water between the catchments and Norsminde Fjord was performed using a simple empirical approach (Hansen et al., 2017) that resulted in numbers of the same order of magniture as estimated in a national nitrate modelling study (Højberg et al., 2015). The models made use of all available data in the catchment either as input data or as calibration/validation data to evaluate the model performances.

***STEP 3* – Backcalculate reference TVs and decide compliance strategy**

The threshold values for the six ID15 catchments in terms of maximum annual fluxes of total nitrogen are calculated for the two reference strategies $TV_{ref,leach}$ and $TV_{ref,load}$ assuming constant distribution of leaching and loading, repectively, throughout the Norsminde catchment. The values are shown in Table 1.

Table 1. Total areas, agricultural areas, N-leaching and N-loads in the six ID15 catchments for the baseline period and threshold values (TVs) calculated for two reference conditions assuming equal distribution of leaching ($TV_{ref,leach}$) and load ($TV_{ref,load}$), respectively, throughout the catchment.

Catchment	**1**	**2**	**3**	**4**	**5**	**6**	**Norsminde**
Total area (ha)	1438	2451	2314	2130	1627	901	10861
Agricultural area (ha)	1068	1637	1240	1612	1008	523	7088
Agricultural area (%)	74%	67%	54%	76%	62%	58%	65%
N-reduction$_{GW}$ (%)	56%	58%	56%	40%	71%	43%	54%
N-reduction$_{SW}$ (%)	10%	27%	20%	11%	28%	9%	17%
Baseline (2000-2010)							
N-leaching (tN/year)	61	89	71	85	53	27	387
N-reduction in GW (tN/year)	34	52	40	34	38	12	209
N-reduction in SW (tN/year)	3	10	6	6	4	1	30
N-load to Norsminde Fjord (tN/year)	24	27	25	46	11	14	147
N-leaching (kg N/year/ha)	43	37	31	40	33	30	36
N-load (kg N/year/ha)	17	11	11	21	7	15	14
Threshold Values							
$TV_{ecosystem}$							103
Strategy A - equal leaching - $TV_{ref,leach}$							
N-leaching (tN/year)	36	61	58	53	41	22	271
N-leaching - compared to baseline	-41%	-32%	-18%	-38%	-24%	-16%	-30%
N-load (tN/year)	14	19	20	28	8	12	103
N-load - compared to baseline	-41%	-32%	-18%	-38%	-24%	-16%	-30%
Strategy B - equal load - $TV_{ref,load}$							
N-leaching (tN/year)	34	76	62	38	74	16	*301*
N-leaching - compared to baseline	-44%	-15%	-12%	-56%	39%	-39%	-22%
N-load (tN/year)	14	23	22	20	15	9	103
N-load - compared to baseline	-44%	-15%	-12%	-56%	39%	-39%	-30%

Since the baseline N-load (147 tN/year) is larger than the target for the receiving water body, $TV_{ecosystem}$, (103 tN/year), a mitigation strategy has to be adopted. This decision is political, because it in various ways will have adverse impacts on the farmers in the catchment.

A strategy following the $TV_{ref,leach}$ logic, allowing equal N-leaching per ha throughout the Norsminde catchment, is close to the commonly used uniform regulation strategy, where all farmers are treated the same way with respect to requirements to agricultural practice and N-leaching. It differs, however, somewhat beause some catchments, e.g. #4, has a smaller fraction of agricultural area and hence of baseline N-leaching than others, e.g. #1 (54% against 74%) implying that farmers in catchments with relatively large urban and forest areas benefit compared to farmers in catchments with relatively large fraction of agricultural land.

A strategy following the $TV_{ref,load}$ logic, allowing equal N-load per ha throughout the Norsminde catchment, exploits the spatial differences in the natural N-reduction in groundwater and surface water. This is a spatial differentiation strategy aiming at a more cost-effective mitigation. The effectiveness is reflected by the fact that it allows 11% larger N-leaching (from 271 to 301 tN/year) to achieve the same N-load. The distribution of N-load and N-leaching between catchments is, however, very different. Catchments with large natural N-reduction (e.g. #2, #3 and #5) are

"rewarded" by smaller abatement targets than the other catchments making conditions for farmers in these catchment more favourable, unless some associated compensational measures are adopted. For catchment #5 the very large N-reduction percentages even lead to a possibility for increasing the N-load by 39% as compared to baseline. As this is not likely to be part of a mitigation strategy, the increase in load in #5 (4 tN/year) could for instance be distributed among the other five catchments.

At the end of the day, the mitigation strategy will be a political decision, where cost-effectiveness has to be balanced against legal and political possibilities. Based on the adopted mitigation strategy threshold values, TV_{mit}, need to be calculated for each of the six catchments.

STEP 4 – Monitoring and modelling

Monitoring will be performed during a WFD period. The monitoring results will be analysed concurrently using models to simulate the effects of climate variability on discharge, N-leaching and N-load.

***STEP 5* – Assess compliance**

Statistical tests will be performed to assess to which extent compliance, in terms of N-loads less than TV_{mit}, have been achieved. The use of dynamic models to simulate effects of climate variability (STEP 4) can help reduce the "noise" from climate variability and hence enhance the power of the statistical test (Højberg et al., 2007).

***STEP 6 – Reassess* TV concept**

Based on the experiences from applying the concept, the TV concept emerging from decisions in STEP 1 + STEP 2 will be reviewed and recommendations will be prepared for modifications with respect to the overall concept as well as to the monitoring and modelling system.

4.2 The Kocinka catchment, Poland

4.2.1 Description of the catchment

The 258 km^2 Kocinka catchment is located in the south of Poland (Figure 6) within the Oder river catchment. The 40.2 km long Kocinka river discharges into the Liswarta river. Dominant soils are mainly sandy and clayey soils. The topography is slightly undulating with elevations varying between 185 to 317 m a.s.l (Figure 6). The climate is temperate with an average annual precipitation of 600-700 mm/yr and average air temperatures between 7.5 to 8°C. The average discharge and the baseflow discharge at the gauging station (Figure 6D) were 218 mm/yr and 158 mm/yr, respectively for the period 1974 - 1983. The catchment is mostly agricultural with pine forests dominating in the lower reach (Figures 7 and 8).

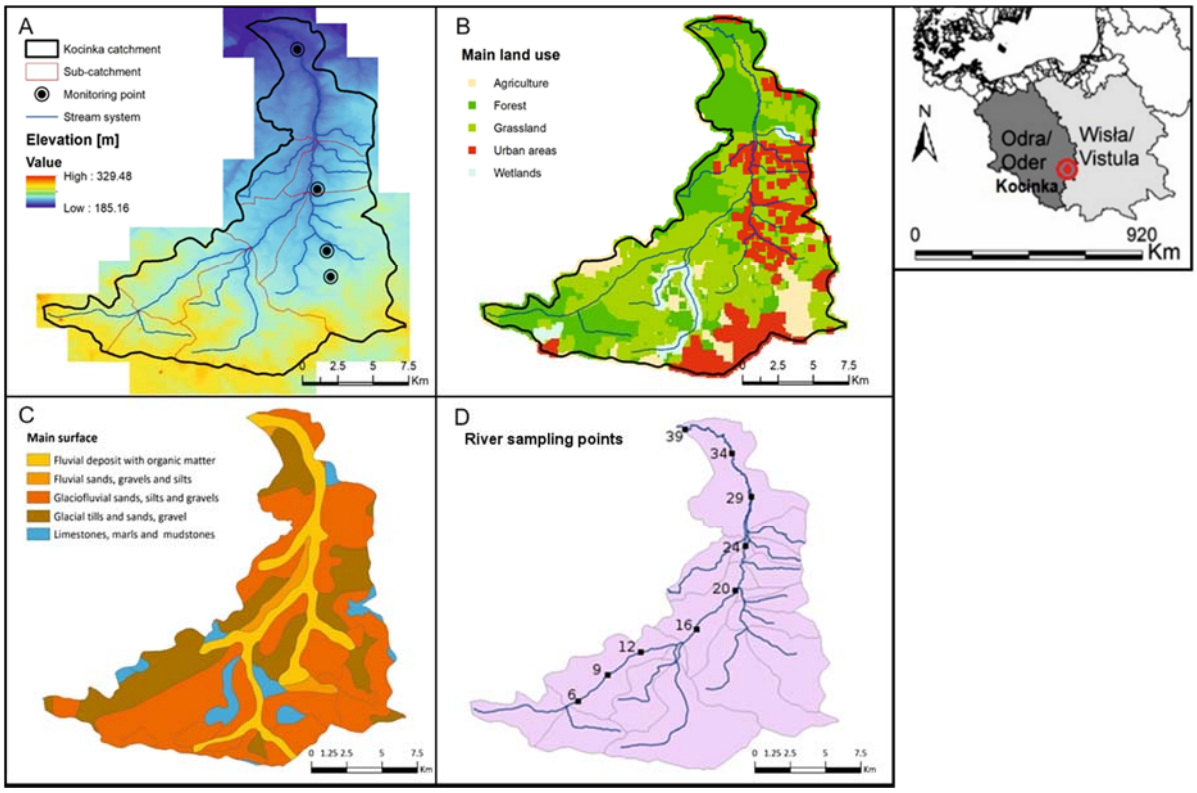

Figure 6. Kocinka catchment. A: Elevation, stream system and monitoring stations. B: land use. C: surface geology. D: location of the main sampling points with numbers denoting kilometres from the source of the river. Fish ponds are located at kilometres 11 and 33 and an automatic river gauging station at kilometre 34.

Figure 7. Regulated stretch of the Kocinka and the riparian forest in the lower part of the river.

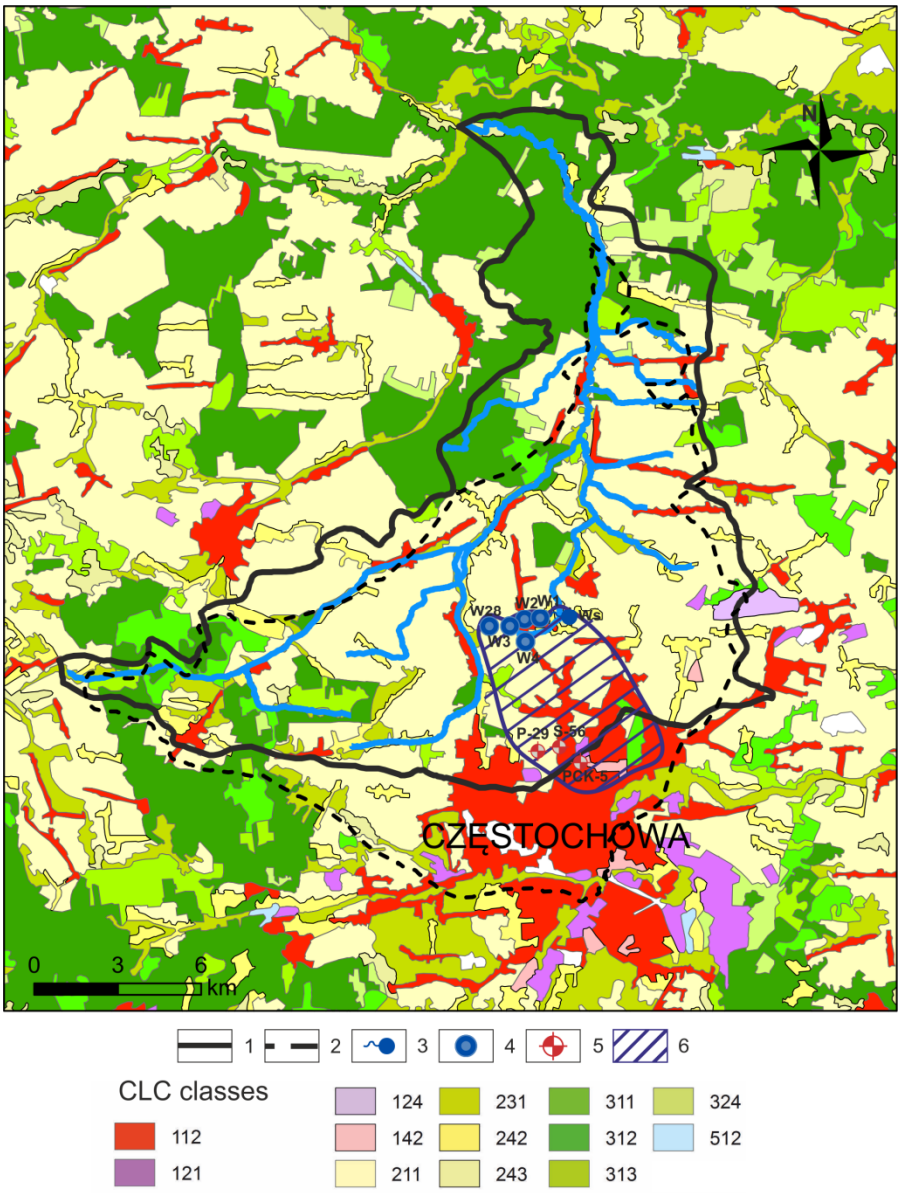

Figure 8. Land use - CORINE Land Cover (CLC v.2006) classes. Green colour denotes forests, yellow colour agricultural land and red color urban areas.

Explanations: 1 – Kocinka surface catchment; 2 – Kocinka groundwater catchment; 3-6 – Wierzchowisko waterworks: 3 – spring; 4 – exploitation well, 5 – monitoring well; 6 – recharge area

The catchment is covered by 1 - 33 m thick Quaternary deposits (Paczyński and Sadurski, 2007) of glaciofluvial and aeolian origin underlain by Upper Jurassic limestones. A geological crossection along a dominant flow line is shown in Figure 9. Here it is seen that the Quaternary deposits comprise relatively thin aquifers and aquitards, while the underlying Jurassic limestone aquifer is more than 200 m thick. The Jurassic strata contain one of the largest groundwater bodies in Poland – the Major Groundwater Basin 326 (MGWB-326).

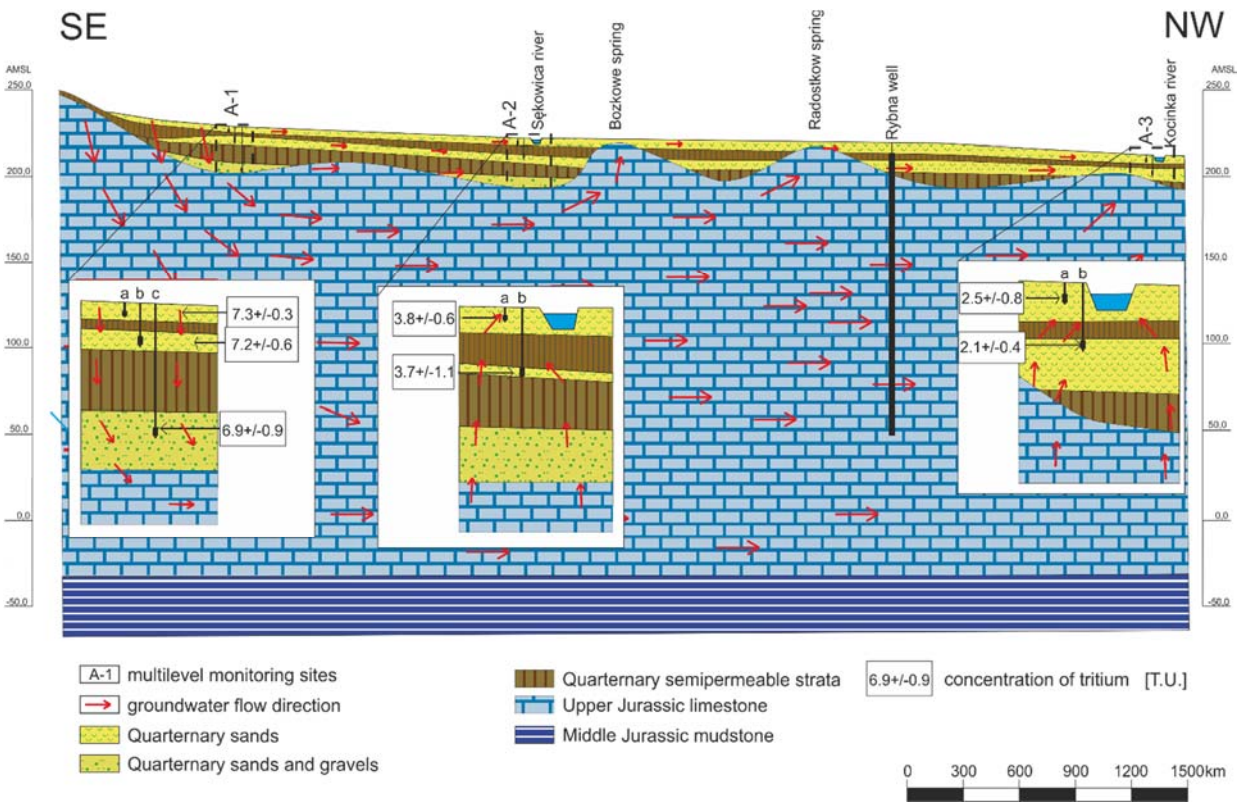

Figure 9. Hillslope cross-section showing positions of well nests (A-1, A-2, A-3 shown as the three southern "monitoring points" in Figure 6A). The figure shows the conceptual understanding of the groundwater flow paths in the upper Quaternary deposits and the deeper Jurassic limestone aquifer.

The aquifers in the Kocinka catchment are polluted by nitrate with the 50 mg NO_3/l limit for drinking water being exceeded in many parts of the aquifer. Figure 10 shows concentrations for wells in the recharge area of the Wierzhowisko waterworks (Figure 8) near Częstochowa. Here the nitrate concentrations have been above 50 mg NO_3/l for two decades, why a water treatment plant employing microbial denitrification was installed in 2005 (Wachniew et al., 2018). The main source of the nitrate in groundwater is assessed to be excessive use of fertilizer and manure in the decades prior to 1990 (Wachniew et al., 2018), while the N-leaching from today's agriculture is significantly less with around 25 kg N/ha/year, which is around 2/3 of the N-leaching in Norsminde (Olesen et al., 2018). It should however, be noted that due to a low amount of net precipitation the groundwater recharge is so low that the 25 kg N/ha/year leaching results in average NO_3 concentrations significantly above 50 mg/l in the water recharging the upper Quaternary aquifer. The average concentration of nitrate in groundwater discharging to the Kocinka river is 25 mg/l, which is not likely to change significantly during the coming decades due to the large amounts of nitrate already in the aquifer and the high concentrations in leaching from the present day's agricultural practice (Wachniew et al., 2018).

Due to the generally deep unsaturated zone and the long travel times for water in the aquifers, there is a very considerable lag time from a nitrate particle leaves the root zone until it reaches the river system (Wachniew et al., 2018; Figure 11). The nitrate removal potential in groundwater is in the range 15 – 30 %, where the higher values are associated with occurrences of clay layers in Quaternary strata. Nitrate reduction in the deeper Jurassic strata is not important due to prevailing aerobic conditions.

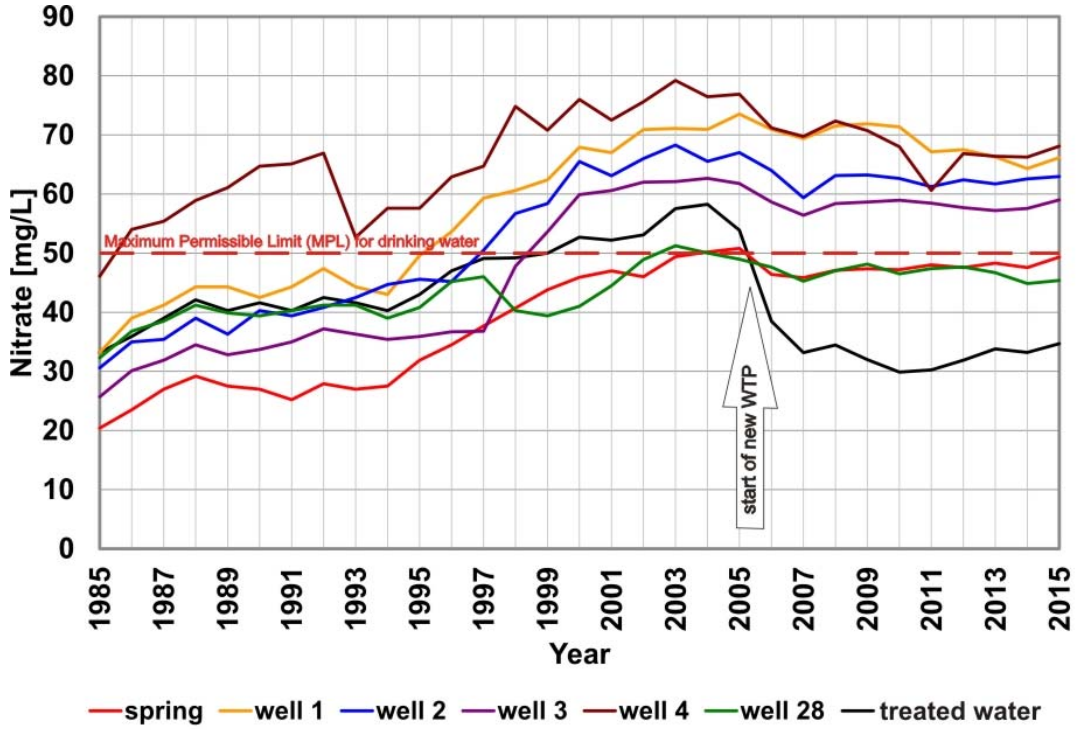

Figure 10. Trends in the yearly averaged nitrate concentrations at Wierzchowisko waterwork consisting of five wells (W1; W2; W3; W4; W28 at Fig.8) and one spring (Ws at Fig.8). A significant drop of nitrate concentrations in treated water occurred after the opening of the new Water Treatment Plant (WTP) that employs microbial denitrification (Wachniew et al., 2018).

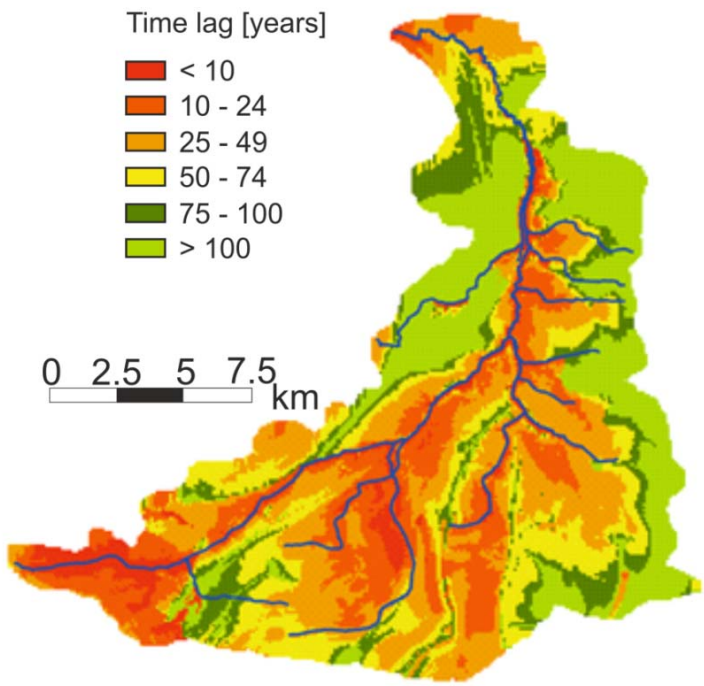

Figure 11. Cumulative water residence times in the Kocinka catchment reflect delays (in years) between application of measures in agriculture and response arriving to the river.

Groundwater controls the water quality in the Kocinka because the groundwater contribution makes up more than 80% of the total discharge. Figure 12 shows the concentration of nitrate in the Kocinka river measured during five campaigns in 2014-2016. The groundwater induced enrichment of nitrate is visible in the upper part of the river, down to 20 km from the source. Self-purification is significant only in the through-flow pond (11 km). Small drop in nitrate concentrations (around 10%) is observed in the lower part of the river, while nitrate removal in other parts of the river, if it occurs at all, is masked by groundwater inflows. Overall, the Kocinka has limited self-purification potential. The average concentration at the outlet is about 20 mg NO_3/l.

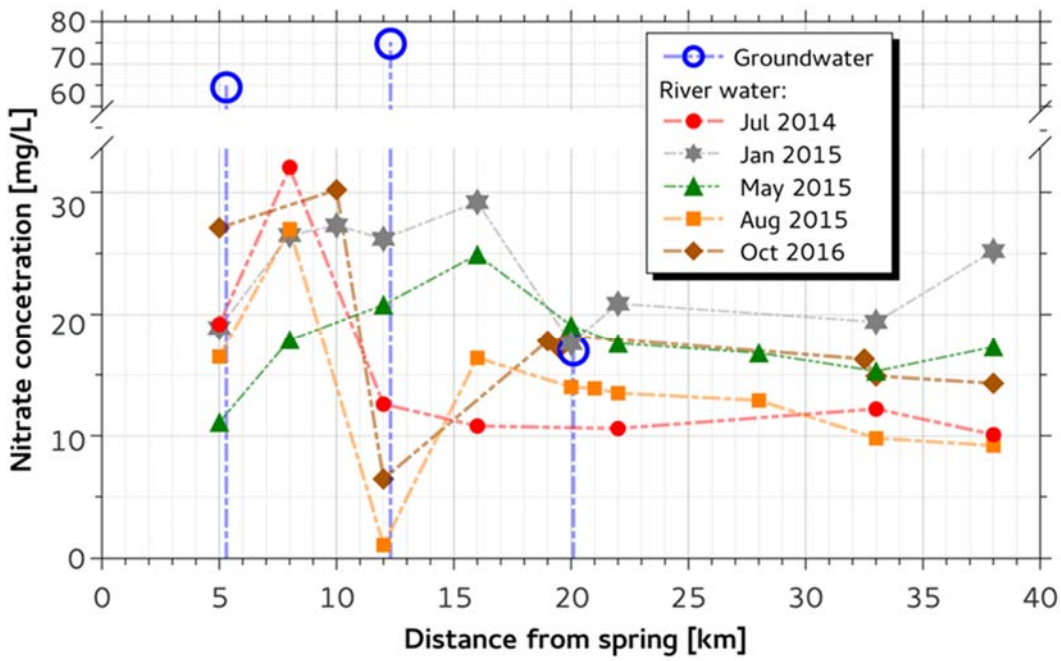

Figure 12. Longitudinal changes of nitrate concentrations in the Kocinka river and springs discharging close to the river channel (groundwater).

4.2.2 Application of TV framework

STEP 1 - Assign TV to receiving water body

The three critical receiving water bodies that should be considered when assigning a $TV_{ecosystem}$ are: i) the Baltic Proper Sub-basin of the Baltic Sea, where the Oder river discharges its water and nutrients; ii) the river Kocinka; and iii) the Kocinka aquifer.

- *Baltic Sea Proper*. The Baltic Sea Action Plan (HELCOM, 2007, 2013) prescribes a maximum allowable TN input to the Baltic Proper of 325,000 tonnes TN/year as compared to a reference input of 423,921 tonnes TN/year, corresponding to a decrease of 23%.
- *Kocinka river*. Nitrate concentration of 10 mg/l was established in Poland as a threshold value for surface flowing waters (rivers) above which eutrophication occurs (Decree of the Ministry of Environment). However, given that the Kocinka river ecosystem is valuable as trout fishery, the specific TV could be set at another level.
- *Kocinka aquifer*. Here the drinking water limit of 50 mg NO_3/l applies.

Among these three criteria, the requirements to the Kocinka river ecology, with a threshold of 10 mg NO_3/l is the most critical. Given that the current average levels of nitrate in groundwater and river are 25 and 20 mg/l, respectively, reduction of nitrate concentrations in the river down to 10 mg/l would thus require lowering of groundwater levels to 12.5 mg/l. If compliance is achieved for this water body, compliances will automatically be achieved also for the two other water bodies. Consequently, we shall in this example assign the TV to the Kocinka aquifer as a maximum concentration value equal to 12.5 mg NO_3/l.

We also note that due to the history, the long residence times in groundwater and hence the lag time between introduced measures and the compliance with established targets (Meals et al., 2010), it seems completely unrealistic to achieve compliance until 2027 as required by the WFD.

STEP 2 – Design monitoring and modelling of catchments

Groundwater concentrations are monitored in around 30 wells with about half of the wells screened in the Quaternary aquifer and the other half in the Jurassic aquifer. This is not in itself sufficient to obtain a full overview of the concentrations in the aquifer. Therefore, the comprehensive model of groundwater flow and nitrate transport and fate based on the MODFLOW and MT3DMS codes developed by Wachniew et al. (2018) will be used as supplement to the monitoring data to provide a more complete spatial and temporal picture of the nitrate concentrations in the aquifer.

In order to achieve relatively fast improvements in river water quality, measures need to be implemented primarily in areas characterised by short travel times (Figure 11). Therefore, the aquifer is divided into two units based on Figure 11, one with travel times less than 10 years and the other with travel times larger than 10 years. The monitoring wells in each of the two units are grouped to analyse for compliance in each of the units.

***STEP 3* – Backcalculate reference TVs and decide compliance strategy**

In this example the TV is simply a maximum concentration of 12.5 mg NO_3/l in each of the two units.

STEP 4 – Monitoring and modelling

Monitoring will be performed during a WFD period. The monitoring results will be analysed concurrently using models to simulate the effects of climate variability on discharge, N-leaching and N-load. This is relevant for the Quaternary aquifer unit, where the concentration is dominated by present days leaching, because the leaching will vary considerably from year to year depending on the actual climate of the year.

***STEP 5* – Assess compliance**

The compliance for each of the two units will be assessed applying the usual WFD procedures for evaluating compliance, when there are samples from several locations within the same water body. The use of dynamic models to simulate effects of climate variability (STEP 4) on nitrate

concentrations in the quaternary aquifer can help reduce the “noise” from climate variability and hence provide a more clear assessment. The projected climate changes underpins the need for improved tools for combined and iterative uses of groundwater monitoring and modelling for groundwater status assessment and compliance testing according to EU policy and guidance.

***STEP 6 – Reassess* TV concept**

Based on the experiences from applying the concept, the TV concept emerging from decisions in STEP 1 + STEP 2 will be reviewed and recommendations will be prepared for modifications with respect to the overall concept as well as to the monitoring and modelling system.

5 Discussion, conclusions and recommendations

5.1 The present study

The Groundwater Directive (GWD) stipulates that EU member states must establish groundwater threshold values to ensure compliance with good status objectives in groundwater dependent terrestrial ecosystems and groundwater associated aquatic ecosystems (surface water bodies), if existing quality standards do not ensure compliance with these. The concept of groundwater threshold values is clear in its general idea and potentially a powerful policy instrument for groundwater dominated catchments within the Baltic Sea drainage basin, such as most catchments in Denmark, Germany and Poland. Nevertheless, the lack of implementation in most EU member countries until now indicates that the concept may be difficult to handle in practice.

We have analysed how groundwater threshold values for nitrate could be derived in two catchments in Denmark and Poland that have been used as study areas in BONUS SOILS2SEA. Both catchments are groundwater dominated and face problems with high nitrate loads and concentrations, but otherwise they are quite different. The Danish catchment, Norsminde, has a relatively high pressure from intensive agriculture and a requirement from River Basin Management Plans to significantly decrease N-loads due to a groundwater associated aquatic ecosystem (Norsminde Fjord). The depth of the redox interface (transition from oxic to anoxic conditions) in Normsinde is shallow (on average 3-4 m), which means that nitrate is only present in the upper oxidized aquifer system, where the mean residence time is 1-2 years, and that N-reduction in groundwater (below the redox interface) is large. The Polish catchment, Kocinka also has a considerable pressure from agriculture today, and in addition still suffers from excessive use of fertilizer and manure before 1990. The Kocinka aquifers are deep with typical residence times around 20 years and small N-reduction in groundwater. To facilitate the analyses of the threshold concept in these two examples we have created a framework comprising six steps, where we have adopted a somewhat broad interpretation of the GWD threshold concept.

The following sections summarise the key findings and the associated recommendations from our study. Our two examples are real-life cases with large differences regarding hydrogeological and ecological conditions as well as socio-economic challenges, and some reservations should be made with respect to the general validity of our conclusions derived from analysis of only two cases. However, the approach and overall principles should be applicable in a wide range of settings across Europe as well as globally.

Recommendation #1: *The findings and recommendations from our study (described below) should be tested against conditions in other case areas before they are adopted as part of the GWD policy guidances.*

5.2 TVs based on concentrations or fluxes

One of the challenges in the GWD and the original guidances is that the thresholds are conceptualised as derived from concentration values in boreholes, as e.g. illustrated by Hinsby et al. (2008). In many cases the critical issue for a groundwater dependent terrestrial ecosystem and

even more often for a groundwater associated aquatic ecosystem is the flux of nitrate rather than the concentration of nitrate in the inflowing groundwater. This implies that concentrations need to be combined with fluxes of water to form a relevant criterion. Concentrations sampled in boreholes face another challenge in practice, namely that the spatial variability is typically very large and a water sample represents only a small water volume, while concentrations in e.g. streams represent a much larger spatial volume and hence are much less uncertain in terms of representing a water body.

Hence, concentration values from boreholes may not be practical to use in many cases, because they would require too many boreholes for sampling and still be very uncertain. As most of the water flowing in small streams in groundwater dominated regimes originate from groundwater, measurements in streams will often be a good measure of the conditions in the groundwater body itself. Furthermore, measurements at a single gauging station in a stream draining a small catchment will be much cheaper than measurements in many boreholes, and because stream measurements are integrated responses for the entire catchment, rather than a point value like a borehole sample, the stream measurement will provide a more reliable estimate of the conditions of the groundwater body. Due to large temporal variations in concentrations throughout a year, and to some extent even between years, frequent and preferably continuous measurements of both flows and concentrations are required. These data should then be compiled to a load, which is equivalent to flow weighted concentrations times average flow during the period of interest, e.g. day, month or year. This approach is also applied in the American Clean Water Act, and many Chinese studies, where Total Maximum Daily Loads is the threshold or target to comply with for protection of aquatic ecosystems (Boesch, 2002; Wang et al., 2014).

Recommendation #2: *The GWD guidances presently focussing on concentrations as thresholds should be extended to include annual, daily and monthly loads (fluxes) and corresponding flow-weighted average concentrations of nitrate in drain pipes, ditches and streams as possible thresholds.*

5.3 Combined monitoring and modelling

Measurements are, for good reasons, often considered to be more reliable than model predictions. It would therefore be ideal to base a compliance testing entirely on measurements. This is however, not possible in practise. First of all, measurements are relatively costly. Secondly, the large spatial variability of hydrogeological properties in all catchments require many monitoring sites to ensure a reliable estimate of the condition of a water body. And thirdly, some variables, e.g. N-leaching out of the root zone or subsurface outflow from a catchment, simply cannot be measured in practice but need to be inferred in another manner. Therefore, the only practical way forward is to combine monitoring with hydrological modelling (Højberg et al., 2007). In relation to nitrate in groundwater dominated catchments, spatially distributed hydrological models integrating groundwater and surface water are required. In our two examples we have used such models, where we using different modelling codes and combining modelling of N-leaching from the root zone at field level with modelling of flows and transport in the unsaturated and saturated zone as well as the stream-aquifer interaction to improve the system understanding.

From the combined use of monitoring and modelling it is possible to derive total maximum daily or montly loads and corresponding flow-weighted average concentration values for drains and streams, which may be used in compliance testing of groundwater chemical status by the farmers, relevant consultants and/or authorities. When the concept is used at small scales, such as drain catchments or fields, it must however be recognised that model predictions, based on commonly available data density and quality, most often are so uncertain that they cannot in a meaningful manner be used for water management decisions (Refsgaard et al., 2016).

Recommendation #3: *The GWD guidances should emphasize that analysis of the groundwater body and its interaction with groundwater dependent terrestrial ecosystems and groundwater associated aquatic ecosystems should be carried out through joint use of monitoring and spatially distributed coupled groundwater/surface water modelling, when threshold values for total loads of nutrients and other contaminants have to be assessed and established.*

5.4 TVs and spatially differentiated regulation

Our examples illustrate that specification of threshold values (TVs) is far from straight forward. It will always be required to make some decisions that are not entirely science based, but include elements of political priorities. This is in particular the case, if TVs in a catchment/groundwater body have to reflect that N-loads need to be decreased to achieve compliance in a receiving water body.

The examples provided by Hinsby et al. (2008, 2012) showed that it is possible in a scientifically based manner to assign TVs as average values for an entire catchment upstream a receiving water body. The complications occur when a TV for a receiving water body, like Norsminde Fjord in our Danish example, has to be backcalculated to TVs for six upstream subcatchments. How do we do that? We have calculated TVs using two different strategies, namely that each of the catchments should be allowed the same N-leaching or the same N-load (per ha catchment area). Good political arguments may be made to support both strategies, but they lead to quite different TVs. The latter strategy of allocating equal N-loads to the catchments is a spatially differentiated regulation strategy that has been analysed in the BONUS SOILS2SEA project. This strategy exploits the differences in the natural N-reduction in groundwater and surface water and is therefore attractive from a purely cost-effective point of view. However, this strategy leads to a larger degree of differences in treatment of farmers between the six subcatchments and will therefore have political complications in implementation.

The Polish case study shows a different picture, where TVs are assigned to average concentrations in two groundwater units. In our example, we have assigned the same TV of 12.5 mg NO_3/l, and we have motivated the differentiation into two units as the need to have one unit, with relatively small transport time (< 10 years), which will be able to show relatively fast improvements. This differentiation may be associated with a uniform regulation with the same requirements to mitigation measures in both unites or with a spatially differentiated regulation strategy, where more comprehensive mitigation measures are applied to one of the units, e.g. the unit with fastest response. It is recognised that the choice of mitigation strategy also in this case will be a political decision.

Thus, it must be realised that when moving from uniform regulation with one groundwater body upstream a receiving water body towards a division into several sub-aquifers or sub-catchments, no unambiguous scientific method for backcalculating TVs exists. Consequently, the use of TVs in a spatially differentiated regulation context cannot be done without making political choices.

Recommendation #4: *The GWD guidances should emphasize that use of TVs for spatially differentiated regulation cannot be done entirely from scientific principles, but is conditioned on associated political decisions.*

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