Title:

A conceptual model for the analysis of multi-stressors in linked groundwater–surface water systems.

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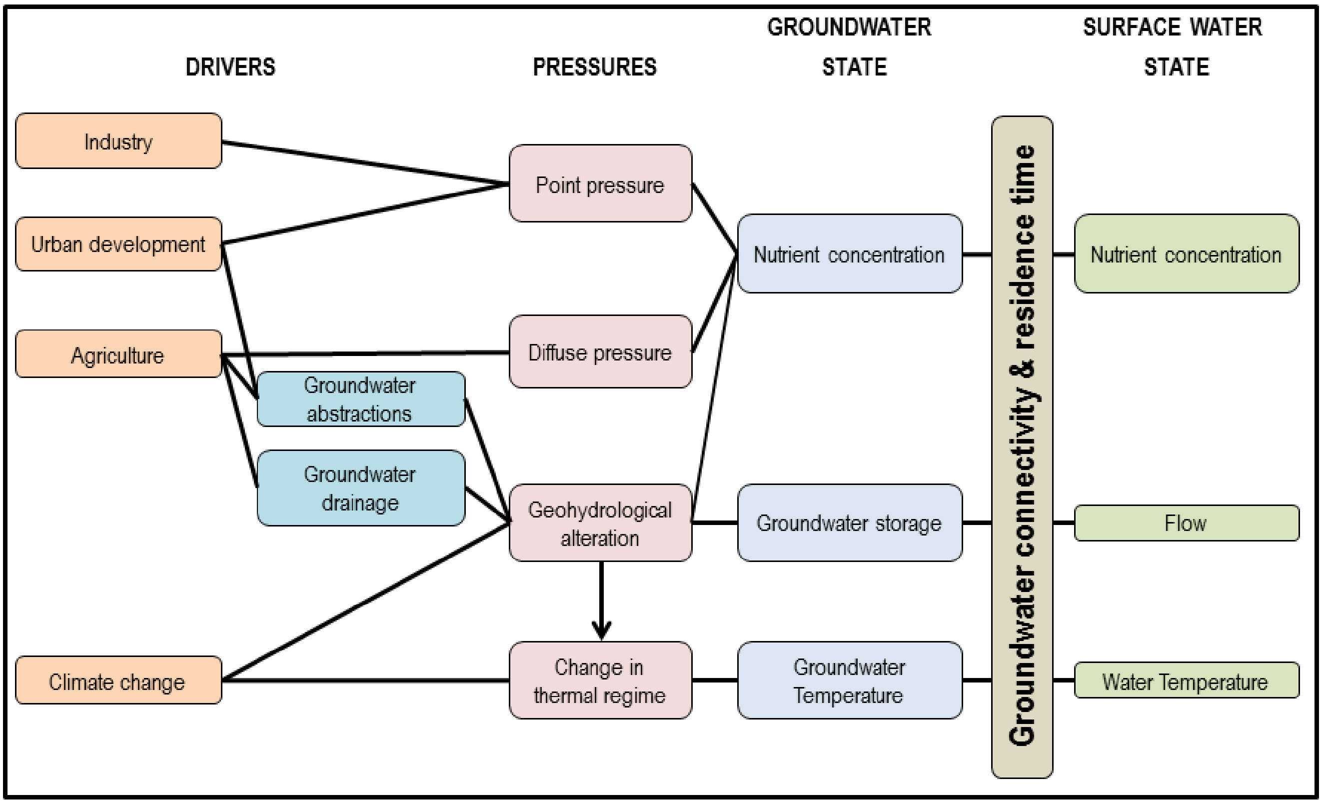
Abstract

Groundwater and surface water are often closely coupled and are both under the influence of multiple stressors. Stressed groundwater systems may lead to a poor ecological status of surface waters but to date no conceptual framework to analyse linked multi-stressed groundwater – surface water systems has been developed. In this paper, a framework is proposed showing the effect of groundwater on surface waters in multiple stressed systems. This framework will be illustrated by applying it to four European catchments, the Odense, Denmark, the Regge and Dinkel, Netherlands, and the Thames, UK, and by assessing its utility in analysing the propagation or buffering of multi-stressors through groundwater to surface waters in these catchments. It is shown that groundwater affects surface water flow, nutrients and temperature, and can both propagate stressors towards surface waters and buffer the effect of stressors in space and time. The effect of groundwater on drivers and states depends on catchment characteristics, stressor combinations, scale and management practises. The proposed framework shows how groundwater in lowland catchments acts as a bridge between stressors and their effects within surface waters. It shows water managers how their management areas might be influenced by groundwater, and helps them to include this important, but often overlooked part of the water cycle in their basin management plans. The analysis of the study catchments also revealed a lack of data on the temperature of both groundwater and surface water, while it is an important parameter considering future climate warming.

Keywords

Multiple stressors, Groundwater – Surface water interaction, Nutrients, Flow, DPSIR framework

Graphical Abstract



Research Highlights

- A framework is proposed to analyse stressors in groundwater-surface water systems
- This is the first application of the DPSIR scheme to groundwater systems
- Groundwater can act as a medium for the propagation of stressors to surface water
- Groundwater can buffer the effect of stressors in time and space
- A need for more ground- and surface water temperature data is identified

1. Introduction

Europe’s groundwaters and surface waters are affected by multiple anthropogenic stressors (Hering et al., 2014) which are having an impact on their status. For example, approximately 20% of Europe’s groundwater bodies have a poor chemical status, while about 50% of the surface water bodies have failing ecological statuses (European Environmental Agency, 2012). Groundwater and surface water are not separate components of the hydrological system, rather they are linked and interact over a wide range of physiographic and catchment settings (Winter et al., 1998; Woessner, 2000; Sophocleous, 2002; Dahl et al., 2007). Consequently, the use and development of or contamination of one or other resource can have an effect on the other component of the system (Sophocleous, 2002). Many aquatic ecosystems in lowland streams are dependent on a supply of groundwater (Brunke and Gonser, 1997; Hatton and Evans, 1998; Power et al., 1999; Wriedt et al., 2007) and together with specific terrestrial ecosystems are referred to as Groundwater Dependent Ecosystems (GDEs) (Hancock et al., 2005; Kløve et al., 2011). Because groundwater-surface water (GW-SW) systems are often so closely coupled, stressed groundwater systems may lead to a poor ecological status of surface waters (Kløve et al., 2011). Much research has already been undertaken on the effect of stressors on surface waters (e.g. Feld and Hering, 2007; Stendera et al., 2012; Nõges et al., 2015; Piggott et al., 2015; Baattrup-Pedersen et al., 2016; Schülting et al., 2016). And although a significant body of research has been developed over the last 50 years or so related to a wide range of aspects of GW-SW interactions – in particular the implications for ecological functioning of the riparian zone (Fleckenstein et al., 2010), Sophocleous (2002) identified the following, still unresolved, research challenge: how to better understand the environmental impacts of multiple processes that affect both groundwater and surface water across multiple spatio-temporal scales. In the same paper, Sophocleous (2002) cited a conceptual model of Brunke and Gonser (1997) who diagramatically illustrated how human induced pressures from contamination, land-use practices and hydro-engineering impacted on one specific GW-SW interaction, colmation - the clogging of stream-bed sediments, and the ecological consequences. Despite this intial problem-specific example, to date no comprehensive conceptual framework has been developed to analyse linked stressed GW-SW systems. The objective of this paper is to address that issue by proposing a framework to help analyse the effect of groundwater on surface waters in multiply stressed systems. This will be illustrated by applying it to four European catchments, the Odense, Denmark, the Regge and Dinkel, Netherlands, and the Thames, UK, and by assessing its utility in analysing the propagation or buffering of multi-stressors through groundwater to surface waters in these catchments.

Here we hypothesise that groundwater affects surface water in a stressed system in two ways: it enables the propagation of stressors spatially and in time through catchments towards surface water, and in addition it acts as a buffer to stressors as they pass through the terrestrial water cycle to surface waters and adds time lags and attenuates stressor signals (in a manner similar, for example, to the attenuation and lagging of naturally occuring droughts in the terrestrial water cycle, Van Loon, 2015). Groundwater functions as a connection between a catchment and connected streams, for example by transmission of time varying heads or by advection and diffusive transport propagating a range of potential stressors towards surface waters. This way, a stressor located somewhere in a GW-SW connected catchment may have an impact on the surface water downstream, even without any direct connection via the surface. However, groundwater may also buffer the effect of stressors as it yields a 'mean' environmental flow and buffers chemistry and temperature in time and space. Streamflow is a mixture of water from different flow routes: overland flow, flow through shallow groundwater including subsurface drains, and deep groundwater flow which have different travel times. The contribution of groundwater to streams and rivers is spatially and temporally heterogeneous, and changes from upstream to downstream (Modica et al., 1997; Gkemitzi et al., 2011). The buffering of stressors by groundwater could mean that groundwater fed surface waters are more resilient to stressors than surface water without a groundwater input.

In order to develop the conceptual framework we focus on three aspects of GW-SW systems, namely the role of groundwater in influencing: streamflow, nutrient chemistry and surface water temperature. Discharge from groundwater is delayed compared to discharge from direct precipitation and overland flow and therefore leads to a more stable streamflow (Smakhtin, 2001). When precipitation infiltrates to groundwater, its temperature quickly equilibrates to around the annual mean when it reaches a depth of generally up to several meters. This is for instance a temperature of 10-11 °C for the Netherlands (Bense and Kooi, 2004) and circa 9 °C for Denmark (Matheswaran et al., 2014). Therefore, as opposed to the seasonally and diurnally fluctuating temperature of surface waters, the direct discharge of groundwater into a stream is characterized by a relatively stable temperature. Seepage of groundwater influences stream temperature through a complex interplay of processes with strong spatial and temporal differences (Conant, 2004; Caissie, 2006). Surface water chemistry is a mixture of the chemistry of all the (groundwater) flow paths it sources from. As such, freshwaters are directly influenced by the quality of groundwater (Rozemeijer and Broers, 2007). Timescales of groundwater flow are important because groundwater with different travel times is characterized by different chemical compositions. The water chemistry is dependent on the flow path and travel time through the subsoil which determine the loading during recharge at source, the chemical interaction with sediments and the time available for chemical reactions.

Following a description of the conceptual framework, the four catchments are briefly described and then each in turn is analysed in the context of the conceptual framework. The framework is then used to compare the drivers, pressures and selected abiotic states between the four catchments. Stressor interactions, propagation and buffering in the groundwater compartment are discussed. Finally, the implications for ecosystem status, management options and needs for future monitoring are considered.

2. Conceptual framework for multi-stressor analysis of GW-SW systems

We propose that the analysis of multi-stressors in linked GW-SW systems and implications for abiotic (and biotic) status of surface waters in lowland catchments can be facilitated by a variant of the Driver-Pressure-State-Impact-Response (DPSIR) model (OECD, 1993; Svarstad et al., 2008). The DPSIR scheme and variants thereof conceptualize and couple natural-social systems and are used for example in European environmental assessments and various large European funded projects (European Environmental Agency, 1999; Kristensen, 2004) as well as extensively in different fields related to: the terrestrial water cycle; marine (Patrício et al., 2016); coastal (Gari et al., 2015); and, onshore systems (Hering et al., 2014; Lange et al., 2017). The models describe a casual cascade of effects from drivers to pressures on the system, which lead to system states, which have an impact which then precipitate a societal response. This response can be linked back to and affect the drivers, pressures, states or impacts. Multiple feedbacks and linkages can be added to the DPSIR scheme, depending on required detail and complexity and it can thus be used to describe for instance connections in a system under multiple-stress (Hering et al., 2014).

For the purposes of the present analysis, and as a first step, we use the framework and focus on the DPS components, where we only consider the abiotic status of groundwater and surface water. The Groundwater DPS framework is presented in Figure 1 and covers key drivers, pressures and states, which relate groundwater to surface water. Here we take the abiotic states of surface water as proxies for the ecological status (as described in Grizzetti, et al., 2015). The groundwater system functions as a bridge between drivers and pressures on the one hand and the surface water state on the other hand, as will be demonstrated using examples later in this paper. The effect of groundwater state on surface water state is governed by the connectivity and residence time of the groundwater. The groundwater DPS framework can be applied to a wide range of scales varying from stream stretch to catchment scale.

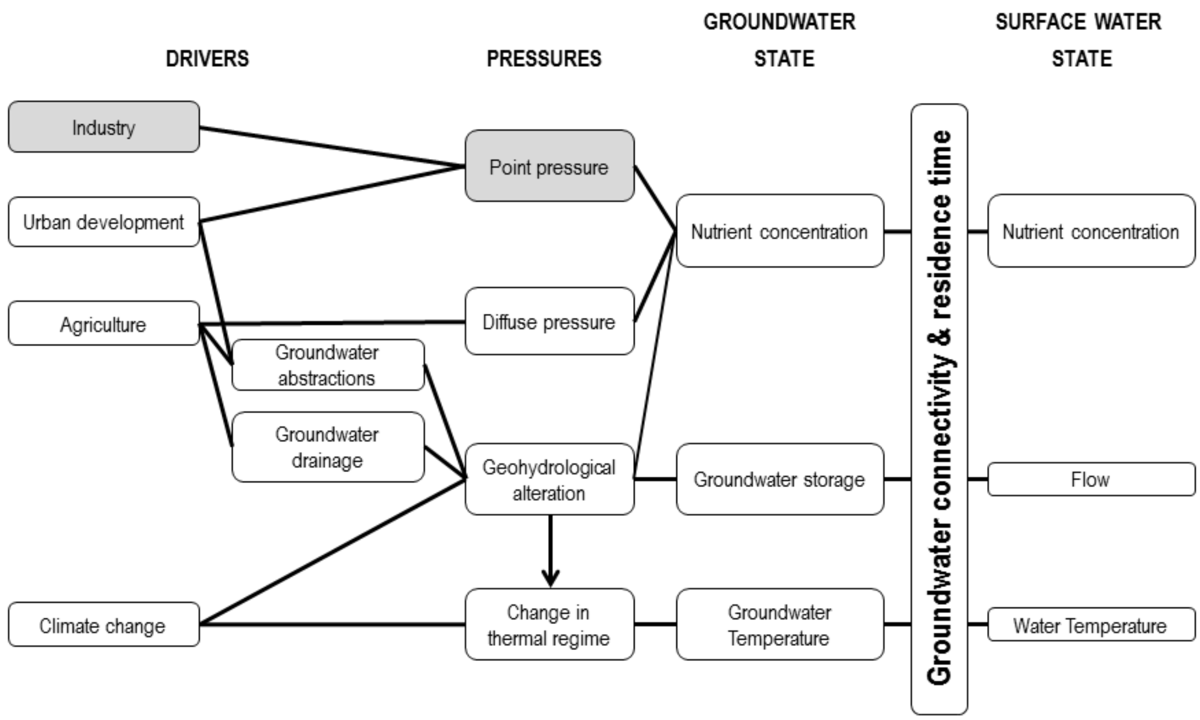


Figure 1. The Groundwater DPS shows how drivers are connected through groundwater with surface waters where they function as a pressure and affect abiotic state. Industry and point pressures will not be part of the analysis in this paper and are therefore marked grey.

Important groundwater drivers of change in connected European GW-SW catchments include urban development, agricultural intensification, climate change and industrialisation of the landscape. For example, both urban development and agricultural intensification of the landscape can result in increased groundwater abstractions, while additionally agricultural intensification can cause modification to groundwater drainage and is a major source of diffuse pollution, including increased loading from nutrients such as nitrogen and phosphorous. It is postulated that these drivers propagate and interact through the groundwater system ultimately affecting surface water quality, quantity and temperature.

The groundwater abstraction, groundwater drainage and climate drivers of change lead to changes in storage and flow of the groundwater system, a pressure designated as geohydrological alteration. The abstraction of groundwater may change groundwater flow, recharge and discharge regimes to the surface water (Zhou 2009), and any water that is removed from a catchment’s water balance is no longer available for surface waters. Groundwater abstractions can reduce stream baseflow

(Henriksen et al., 2008; SKM, 2012; Hendriks et al., 2014) and because of this, groundwater pumping is an important stressor on a stream’s ecosystem. While the abstraction of surface water has a clear immediate effect on stream discharge (Winter et al., 1998; SKM, 2012), the effect of the abstraction of groundwater is delayed in time (Custodio, 2000). In many catchments, part of the groundwater discharge comes from subsurface drainage pipes or small ditches because agricultural and urban areas are often drained intensively. Studies in the USA show examples of catchments where between 41% and 81% of annual stream discharge comes from drainage pipes (Xue et al., 1998; King et al., 2014). Subsurface drainage can be a pressure as it has an effect on groundwater storage and therefore on a catchment’s flow regime. Depending on the local settings, subsurface drainage increases peak flows while reducing baseflow by providing a fast flow path to the surface water (Irwin and Whiteley, 1983; Carluer and De Marsily, 2004) or decreases peak flows and increases baseflow (Irwin and Whiteley, 1983; Schilling and Libra, 2003; Blann et al., 2009; King et al., 2014). The groundwater flow system may also be influenced by climate change because different temperatures and precipitation patterns lead to changes in evaporation, groundwater recharge and groundwater levels (Gkemitzi et al., 2011; Green et al., 2011; Taylor et al., 2013). Increasing evaporation and a decrease in precipitation may reduce groundwater recharge and lower groundwater levels (Singh and Kumar, 2010). If this is the case, the amount of groundwater available to surface waters is lowered and consequently the amount of baseflow provided by groundwater seepage. An additional pressure driven by climate change is a change in thermal regime, i.e. changes in groundwater temperatures. Geohydrological alteration can also be a driver for changes in the thermal regime of the groundwater, as changing flow can lead to a change in the temperature of groundwater. Groundwater temperature changes can ultimately affect surface water temperature by a shift in the temperature of seepage.

Agriculture leads to diffuse pressure due to the application of nutrients while geohydrological alteration can affect nutrient concentrations through changing flow paths and speeds. Anthropogenic nutrient inputs have increased levels of N and P in surface waters by up to a 10-fold

(Vitousek et al., 1997). The most important anthropogenic sources of nutrients are direct into the surface water through waste water treatments plants and diffuse by input from agriculture. Nutrients from agriculture can directly enter the surface water through overland flow but also through subsurface drains and deeper groundwater. Of all groundwater bodies in Europe, about a third has been reported to exceed the guideline values for nitrate, which is acknowledge as a risk in causing nitrate pollution of surface waters (European Commission, 2008). Subsurface drainage has been found to be the most important route for nitrate loss from agricultural fields (Rozemeijer et al., 2010; Blann et al., 2009) as it provides a short-cut towards the surface water and thus provides surface water with groundwater with a short travel time which has had little time for denitrification processes. Phosphorous is considered the most important factor in causing eutrophication because most surface waters are P-limited (Elser et al., 2007). P is easily bound to sediments and is therefore often retarded in the unsaturated zone (Hamilton, 2012). However, phosphorus is transported by the groundwater when it is released within the groundwater or when groundwater levels rise up to and dissolve P-containing sediments (Dupas et al., 2015).

Although point source pressures driven by industry or urban areas are present in many catchments, their effect and behaviour are very case specific and thus for chemical pressures our focus will be restricted to diffuse pollution and specifically nutrients. Future climate change is not part of the case study analyses, but will be included in the discussion.

3. The study catchments

This study focuses on the catchments of the Odense in Denmark, the Regge and Dinkel in the Netherlands, and the Thames in the United Kingdom (Figure 2 and Table 1). All four are permeable lowland agricultural catchments and as such there is interaction between groundwater and surface water over a range of spatial and temporal scales, even though the details of the geological and hydrogeological settings may differ. The catchments range in size from 340 km^2 for the Regge, to 9948 km^2 for the Thames. In all four agriculture is the main land-use (Table 1), while they also include forest and urban areas. The catchments of the Odense, the Regge and the Dinkel include smaller cities such as the city of Odense, Hengelo and Enschede. The catchment of the Thames contains the Greater London area with a population of about 15 million as well as a number of other large urban areas.

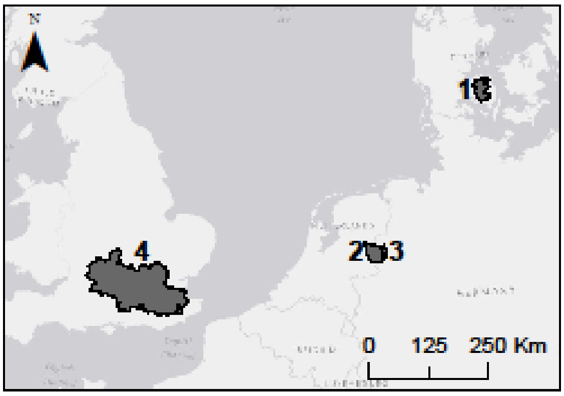


Figure 2. Location of the four study catchments.

Table 1. Selected characteristics of the four study catchment: size, primary land use, average annual air temperature, precipitation and flow, Baseflow Index and geology.

Catchment	Size	Land use	T [°C]	P [mm]	Q [m^{3}s^{-1}]	BFI	Hydrogeology
1. Odense	1061 km^2	Agriculture (68%)	9.0	800	10.4	0.90[*]	Clayey moraines
2. Regge	340 km^2	Agriculture (60%)	9.9	800-850	6.95	0.59	Sand and gravel
3. Dinkel	630 km^2	Agriculture (70%)	9.9	800-850	5.50	0.61	Sand, gravel and clayey moraines
4. Thames	9948 km^2	Agriculture (45%)	10.2	600-900	65.7	0.64	Limestones, low permeability clays and gravels

[*]Outlet of the Odense main river

Table 1 shows that the mean climatology of the catchments is broadly similar, and Figure 3 shows that there is no pronounced seasonality to the precipitation in the four catchments, however there is a strong seasonality to evapotranspiration (ET) and to annual river flows.

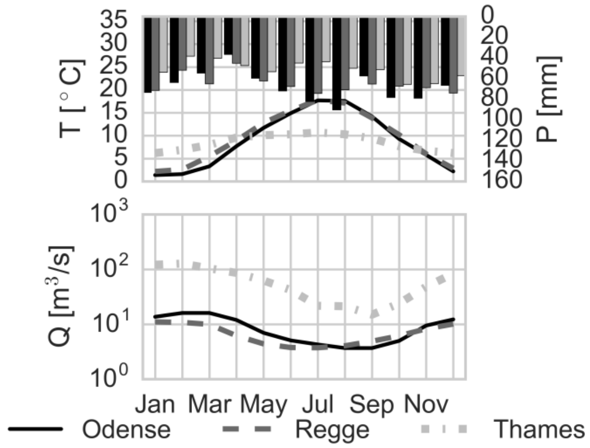


Figure 3. Monthly average air temperature, precipitation and discharge of the Odense, Regge and Thames catchments. Values for the Dinkel are very similar to those of the Regge.

The Odense, Regge and Dinkel catchments are relatively flat, however, there is over 300 m of relief in the Thames Basin, and this, combined with differences in the underlying geology (Table 1) and catchment location with respect to the coast (the Odense and the Thames both discharge to the sea), means that the depth of flow systems and travel times in each of the catchments varies in nature (Figure 3). The Odense is the least geologically complex catchment while the Thames has the most complex geology (Figure 4). The Odense catchment consists of clayey moraines and terminal moraines in the south (Smed, 1982). Sand and gravel deposits form local aquifers (Troldborg et al., 2010) and sandy-loam soils are present throughout the catchment. Groundwater flow systems are typically shallow and relatively rapid; the most extensive aquifer complex only has a thickness of about 10 m. The aquifers in the Odense catchment contain a substantial amount of organic matter, consequently contaminants are typically non-conservatively. The Regge catchment contains mostly sedimentary aquifers up to a depth of about 150 m with multiple clay layers in between (Figure 4). Flow systems may be moderately deep. These aquifers wedge out towards the east, where they only reach a depth of 10 to 20 m below surface. The Dinkel is characterized by sandy deposits located between clayey ice-pushed ridges which have shallow aquifers and flow systems that feed several tributaries such as the Springendalse Beek and Elsbeek. The Thames catchment, being the largest catchment, contains the most variation in geology. The Basin is underlain by two major bedrock aquifers, the Chalk of the Chilterns, Berkshire Downs and North Downs, and the Oolitic Limestones of the Cotswolds in the west of the Basin with a wide range of shallow (fast) to deep (slow) flow systems (Figure 4). These are separated by a series of clay-dominated aquitards (Bloomfield et al., 2009). There is no significant organic matter in the Chalk and Oolitic limestone, consequently contaminants typically act conservatively in these aquifers (Downing et al., 1993a). The bedrock aquifers are overlain by Palaeogene to Holocene gravels and sands along the course of the main drainage channels with relatively rapid, shallow flow systems (Bricker and Bloomfield, 2014).

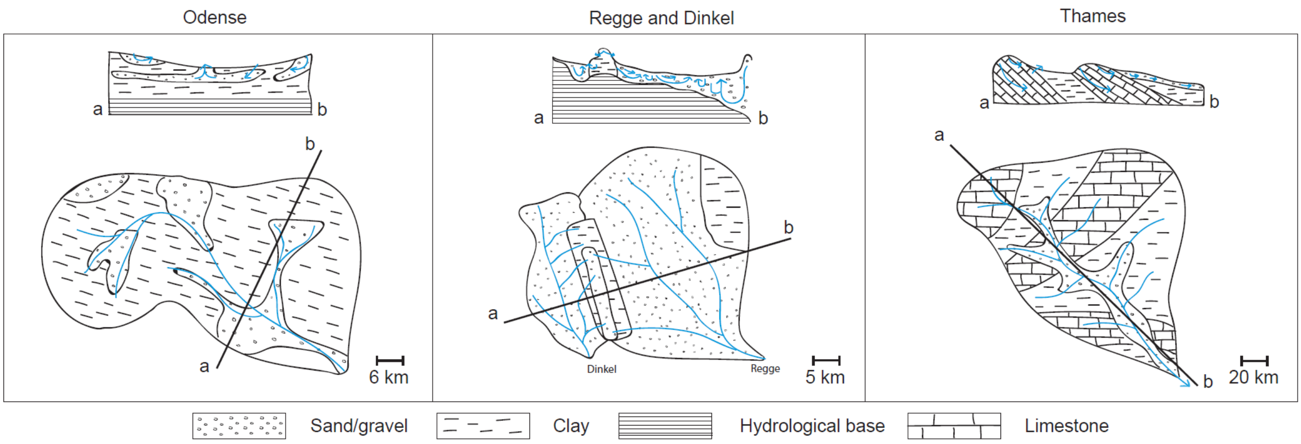


Figure 4. Conceptual drawings of the geology and flow systems of the catchments.

Based on a classification system of groundwater–surface water interaction developed by Dahl et al. (2007), the Odense Fjord catchment falls into the moraine landscape type, with a groundwater system comprising a complexly interbedded sequence of sand aquifers and confining layers of clayey till. The topography controlled water table is dominated by local flow systems; although regional and deep groundwater bodies are also present (Dahl et al., 2007). The Regge and Dinkel catchments also have multiple aquifers and confining layers, with local, regional and deep groundwater systems. On the local scale all riparian flow path types are present but discharge is mostly direct and through drainage systems. The Thames Basin likewise falls into the ‘landscape type’ category, a complexly interbedded sequence of aquifers and confining layers, where groundwater flow systems are principally influenced by factors related to regional geomorphology, hydrogeological setting and aquifer structure and heterogeneity rather than specific riparian zone processes (Bloomfield et al., 2009; 2011; Darling and Bowes, 2016).

Baseflow Index (BFI) (Gustard et al., 1992) is often used as an indicator of the relative contribution of groundwater to stream flow. In all four catchments the long-term baseflow at the base of the catchments is of the order of 0.6 to 0.90 (Table 1) consistent with the mixed groundwater-surface water nature of the catchments. However, all four catchments illustrate spatio-temporal variations in BFI consistent with spatio-temporal variations in groundwater-surface water interactions.

Because of the high level of agricultural land use as well as the degree of urbanization in the catchments, there are pressures on water resources in the study catchments. Agricultural land use was established in the Thames prior to the 20^{th} century, with intensification of farming from the 1940s onwards. The major urban areas were also established before the start of the 20^{th} century in the Thames although peri-urban growth was a continuous process through this period. Many of the major modifications to the drainage structure in the catchment were also in place by the early 20^{th} century. In contrast, in the Odense, Regge and Dinkel catchments most of the land use and stream alterations took place in the 20^{th} century (Larsen et al., 2008; Hendriks et al., 2015a; Lu et al., 2015). All four catchments are now heavily modified with an altered stream network that is artificially regulated. Diffuse agricultural nutrient loss is one of the main threats to aquatic ecosystems in the catchments (Miljø- og Fødevareministeriet, 2016; Molina-Navarro et al., 2018), and agricultural activities are the main pollutant source of Danish groundwater (Blicher-Mathiesen et al., 2014).

As a consequence of the pressures on the water environment, water bodies in the catchments are commonly at poor status. For example, many of the water bodies in the Odense catchment, including the estuary comprised by the Odense Fjord, do not meet European Water Framework Direct (WFD) criteria for good ecological status (Miljø- og Fødevareministeriet, 2016). Of the 600 km of streams only 36% have a good or high ecological status, and for the 17 lakes larger than 5 ha the corresponding number is 12% (Miljø- og Fødevareministeriet, 2016). Likewise, the Regge and Dinkel catchments are classified as heavily modified and most surface waters don’t have good chemical or ecological status. The groundwater body underlying the Regge and Dinkel catchments has an insufficient water quantity status, but a good chemical status, although the chemical status of some local groundwater bodies is unsatisfactory (Ministry of Infrastructure and the Environment, 2015). The Thames Basin contains 489 surface water bodies: 45% of the water bodies are affected by pollution from waste water, and 27% and 17% of water bodies are affected by pollution from rural sources, and from towns, cities and transport respectively. 47 of the water bodies are groundwater bodies, of which 22 have been assessed as having poor quantitative status and 18 have poor chemical status (Environment Agency, 2016a).

4. Analyses of stressor propagation and buffering through catchments

4.1 Odense catchment

An adequate functioning of the aquatic ecosystems in the Odense Fjord catchment including rivers, lakes and transitional waters, depends on keeping a sufficient and persistent flow level and water quality. Here groundwater might play a vital role; however, studies on this topic are scarce. Here we analyse the role of groundwater in the preservation of the aquatic ecosystems in the Odense catchment based on a comprehensive hydrological and nutrient transport modelling carried out with the Soil and Water Assessment Tool (SWAT, Molina-Navarro et al., 2018).

4.1.1 Water storage and flow

Hydrological modelling allows exploring the GW-SW interaction. Figure 5 shows the monthly streamflow subdivided into direct aquifer discharge and contributions via tile drains, surface flow and lateral flow simulated for the period 2001-2010. The average aquifer contribution to total streamflow was around 76%. Such a high aquifer contribution favours a delay in the hydrograph response to precipitation events (Figure 5). The BFI was calculated for each sub-basin to explore the spatial variability of the baseflow contribution. Values varied between 0.63 and 0.92 (Figure 6a), supporting the relevance of the aquifer contribution in ensuring the sustainability of aquatic ecosystems in the Odense Fjord catchment.

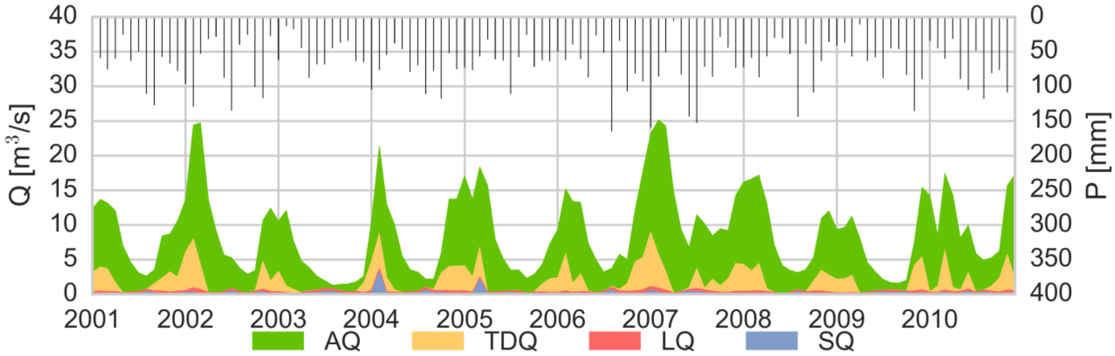


Figure 5. Observed precipitation (P) and simulated flow subdivided (stacked area) into surface flow (SQ), lateral flow (LQ), tile drain flow (TDQ) and direct aquifer discharge (AQ) in the Odense Fjord catchment at a monthly time step (period 2001-2010)

Another parameter that illustrates the GW-SW interaction is Q_{90}, i.e. the flow below the 90th percentile of the flow duration curve divided by the median flow. A Q_{90} value close to 0 means a more extreme flow variation than if the Q_{90} is close to 1. Modelling results yielded Q_{90} values between 0.03 and 0.39 in the Odense Fjord catchment (Figure 6b), with an average of 0.17. This means that, despite the high aquifer contribution that would ensure a more stable flow regime, flow seasonality is highly pronounced in this catchment, partially due to a considerable tile drain flow contribution (discussed below). In the drier months, the streamflow is almost only supported by the direct contribution from the aquifers (Figure 5), becoming a key component of streamflow to provide sufficient water to the aquatic ecosystems in the catchment.

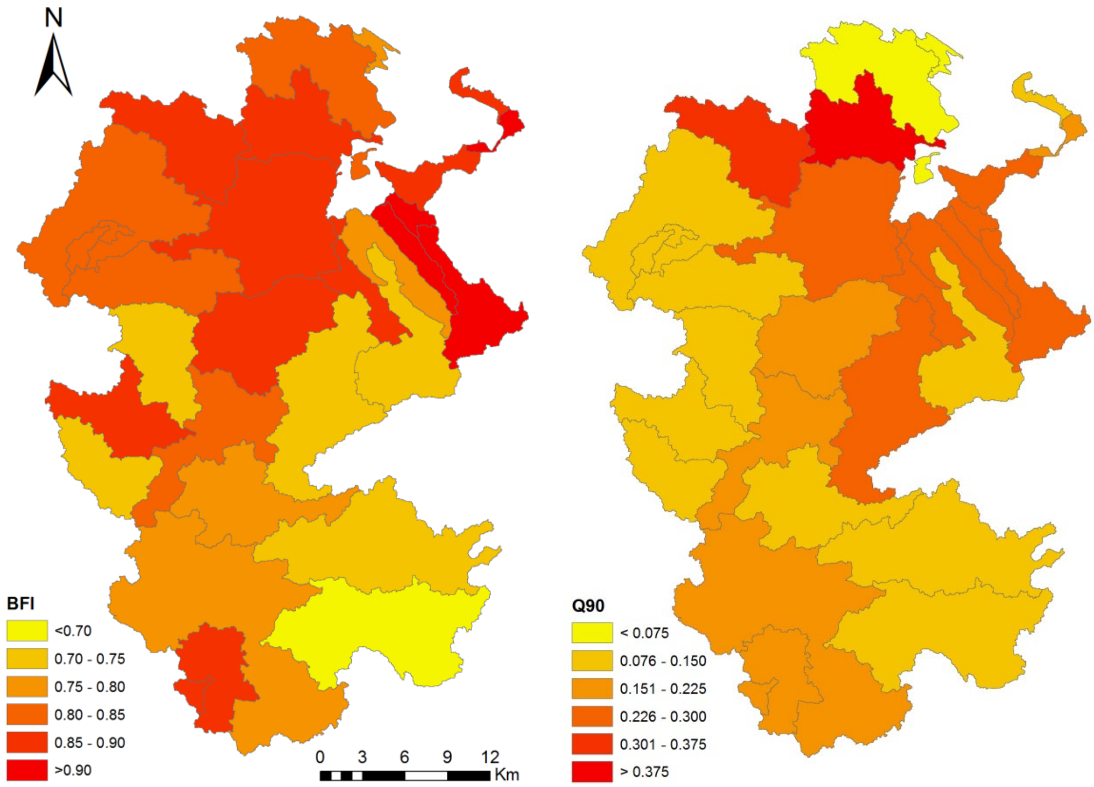


Figure 6. BFI (a) and Q_{90} (b) values in the sub-basins modelled in the Odense fjord catchment.

Hydrological alteration through groundwater abstraction and tile drainage is a major pressure for aquatic ecosystems in the Odense Fjord catchment. The catchment has undergone substantial hydrological and hydro-morphological modifications, including sub-surface tile draining of about half of the agricultural area (Thodsen et al., 2015). As a result, 19% of the total flow in the streams of the catchment comes from tile drains (Molina-Navarro et al., 2018). Simulation of land use changes in the catchment revealed that a decrease of the agricultural area by 21% would lead to a decrease in the drain water contribution to total stream flow from 19% to 14%, conversely increasing the direct aquifer contribution (Molina-Navarro et al., 2018). Tile drains respond quickly to precipitation during autumn and winter when soil in the root zone is close to saturation (Figure 5), altering the natural streamflow in the catchment. Additionally, they reduce aquifer recharge, thus ultimately diminishing water supply to the aquatic ecosystems during the critical summer months.

Groundwater abstraction also plays a major role altering the hydrology of the Odense Fjord catchment. Based on comprehensive hydrological modelling, Henriksen et al. (2008) assessed “sustainable groundwater abstraction” on regional and national scale for Denmark focusing on avoiding significantly negative impacts on both surface water ecology and groundwater quality, in line with the underlying WFD principles. For the Funen Island, of which the Odense catchment is a major part, Henriksen et al. (2008) estimated a sustainable groundwater yield varying from 10 to 29 mm $year^{-1}$, although the lowest value was chosen for the assessment of sustainable abstraction. They also reported an actual abstraction of 12.8 mm $year^{-1}$ (year 2000), which could mean slight over-exploitation. However, an additional evaluation at a sub-area level showed that the Odense Fjord catchment had the highest exploitation rate in the island, with current exploitation at more than two and a half times the sustainable yield, probably due to abstraction for the water supply of Odense city. One result of non-sustainable groundwater abstraction is streamflow depletion with adverse effects on aquatic ecosystems.

4.1.2 Nutrients

Nitrate is by far the largest N fraction being loaded into the Odense Fjord estuary (Molina-Navarro et al., 2018), and thus crucial for the ecological status of the marine ecosystem, where it acts as a limiting nutrient (Conley et al., 2007). Moreover, modelling results suggested that direct aquifer discharge and tile drain flow are responsible of 65% and 30% of the total nitrate yield in the streams, respectively, while direct flow (surface and lateral) transport only 5% (Figure 7). Nitrate leached from the root zone can be reduced during transport via groundwater both by microbial denitrification and by pyrite oxidation denitrification (Blicher-Mathiesen et al., 2014). This attenuation of the nitrate concentration is strongly dependent on hydraulic residence time (Humborg et al., 2015) and the presence of pyrite in the aquifers. For 17 Danish catchments, Andersen et al. (2001) reported groundwater retention of N of 20-80%, with the higher rates in areas with higher residence times. Particularly, for the Odense Fjord catchment, Blicher-Mathiesen et al. (2014) found that nitrate reduction in aquifers and surface waters varied between <40% and up to 70-80% of the root zone N leaching. However, the specific role of groundwater was not analysed.

The SWAT model calculated that 52% of the nitrate that percolates past the base of the soil profile is reduced before reaching the stream as return flow (Ferreira et al., 2016), confirming the relevance of nitrate reduction in groundwater in the Odense catchment, which has been previously pointed out, but scarcely supported by data. Figure 7 illustrates this reduction, and in addition shows how the transport through the aquifer exerts a delay in the nitrate yield back to the streamflow. Results also show how nitrate transport via tile drains is much faster (Figure 7).

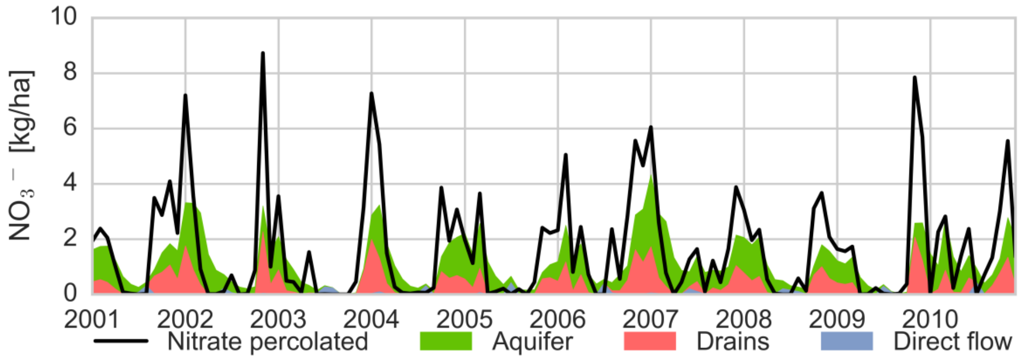


Figure 7. Simulation of nitrate percolated and transported via direct aquifer contribution, tile drain flow and direct (surface and lateral) flow in the Odense Fjord catchment at a monthly time step (period 2001-2010).

On the other hand, phosphorus (P) acts as a limiting nutrient in streams and lakes, and high loads might also threaten the health of the aquatic ecosystems in the Odense Fjord catchment. Monitoring data revealed that dissolved phosphate represents slightly over half of the TP load in the catchment, and the modelling suggested that the main transport pathway for phosphate in the catchment is groundwater (Molina-Navarro et al., 2018). The source of P is partly agricultural by leaching to the upper groundwater and partly naturally occurring P in reduced groundwater (Kronvang et al., 2007). Figure 8 reveals how both the groundwater flow and phosphate follow a nearly-parallel trend.

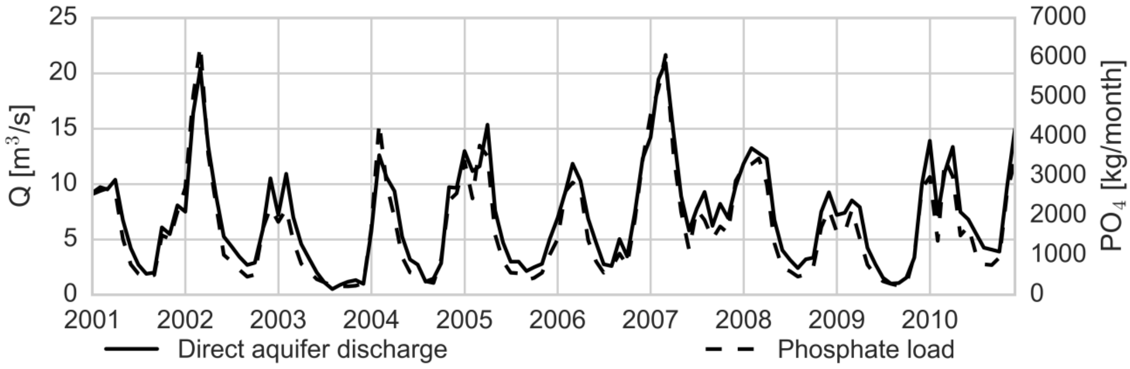


Figure 8. Simulated baseflow and phosphate load in the Odense Fjord catchment at a monthly time step (period 2001-2010).

4.2 Regge & Dinkel catchments

4.2.1 Water storage and flow

The tributaries of the Dinkel show significant variation in their BFI (Table 2) reflecting both differences in underlying geology but also in drainage and land management practices. Using an existing groundwater model (Kuijper et al.,2012; Hendriks et al., 2014), groundwater discharge to the three Dinkel tributaries can be divided into different flow routes: overland flow, subsurface drainage pipes and direct discharge to streams/rivers. Discharge that occurs when the groundwater table rises above the land surface is modelled as overland flow and is highest in the Springendalse Beek due to the fact that many of the springs in the upstream part of this stream are created by local depressions where the groundwater table rises above the surface. The highest amount of discharge from subsurface drains occurs in the Elsbeek and Roelinksbeek (Table 2), which are the catchments with the highest agricultural activity and lowest BFI. Although these catchments also seems to have the highest amount of groundwater outflow though streams, this is mostly through ditches as they have the highest drainage density, while in the Springendalse Beek this water mainly seeps in the main stream course and has longer flowpaths.

Table 2. Calculated Baseflow Indices and model flows for three Dinkel tributaries.

		Modelled groundwater outflow routes		
River / Stream	BFI	Streams incl. ditches	Drainage	Overland flow
Springendalse Beek	0.8	78%	2%	20%
Elsbeek	0.4	85%	7%	9%
Roelinksbeek	0.4	84%	4%	12%

This artificial drainage and pumping of groundwater for irrigation are changes in the hydrology that resulted from agriculture, while pumping of groundwater for the production of drinking water also occurred. Hendriks et al. (2015a) showed that this caused lowering of groundwater levels in the Regge catchment until the 1980s. Figure 9 presents two time-series of groundwater heads of relatively deep wells in the Regge and Dinkel catchments (60 – 80 m below ground surface) and shows the decrease in groundwater heads between 1940 and 1980. Data at measuring location GW-80M shows that deep groundwater levels declined by >15 m between 1950 and 2010. At measuring point GW-60M, groundwater heads have gone down by approximately 3 m between 1965 and 1980 as a result of a drinking water abstraction well. After the 1980s, the abstraction and drainage of groundwater remained at a stable level which led to a new equilibrium of the water balance of the systems. In this equilibrium, groundwater heads at measuring location GW-60m did not return to their 1940-1950 levels but remained at a reduced level. With these lowered groundwater tables, research by Hendriks et al. (2014) showed that the Q95 of streamflow has decreased by and 5-55% as a result of groundwater abstractions, and 16-30% as a result of artificial drainage. Kuijper et al. (2012) found a reduction of the groundwater input to surface waters of 20 to 50% as a result of groundwater abstractions and subsurface drainage. Reduction of groundwater baseflow makes the Regge and Dinkel catchments more sensitive to droughts; during the 2003 drought many streams in the Regge and Dinkel catchment fell dry (Kuijper et al., 2012). One of the few exceptions was the Springendalse Beek, which kept flowing as a result of the high groundwater component of flow.

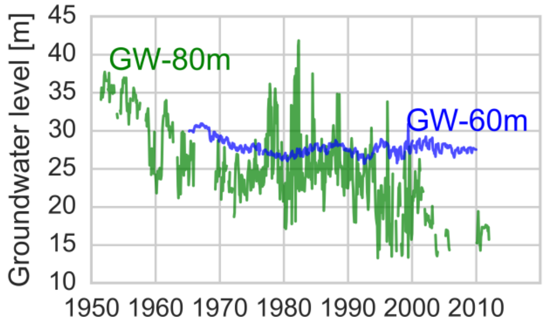


Figure 9. Time-series of groundwater heads in the Regge and Dinkel catchments: measuring points GW-80M (filter 80 m below ground surface*) and GW-60M (filter 60* m below ground surface*). Note these measurements are from different locations.*

4.2.2 Nutrients

Industrial and urban development has led to point and diffuse pollution in the Regge and Dinkel catchments, but the main driver of poor water quality is agriculture, because of the use of manure and pesticides. These nutrients are transported partly through the groundwater to the surface water. The effect of these pressures depends on the connectivity and residence time of the groundwater, as well as the capacity for degradation of nutrients in the subsurface. A good example of the pressure from nutrients can be seen in the Springendalse Beek where downstream nitrate concentrations have gone up until the 1990s as a result of agricultural loading (Figure 10). In the late 1950's, Maas (1959) reported the Springendalse Beek to be a nutrient poor stream and measurements from the 1970s indeed show stream nitrate levels between 0 and 2 mg-N/L (Higler et al., 1981). Between the 1970s and 1990s nitrate concentrations increased significantly (Figure 10) following agricultural intensification in the catchment. Since the aquifer is low in organic matter the capacity for denitrification is limited. Following the Nitrate Directive, nitrogen use by farmers in the Regge and Dinkel catchments has decreased by about 40% between 1994 and 2015 (Centraal Bureau voor de Statistiek [CBS], 2017), and in stream nitrate concentrations have consequently slowly decreased, although they are still higher than natural background levels.

Contrary to this decreasing trend, nitrate levels in the upstream part of the Springendalse Beek have not yet reached their peak (Figure 10). As this stream stretch is directly fed by several springs, it is not surprising that the concentration of nitrate in the groundwater at this location is also elevated

(Figure 10). This groundwater has flowpaths with long travel times and consequently historic nitrate inputs are still in the system. The groundwater functions as a slow connection between the agricultural fields and stream, and historic nutrient input causes long term pollution of the surface water. This is an important consideration for catchment management, since the effect of converting the surrounding agricultural fields to pasture with the consequent reduction in nitrate loading in the late 1990s (Nijboer et al., 2003) has not yet resulted in a response in nutrient concentrations in the stream. Despite the nutrient chemistry of the stream, due to the influence of groundwater on discharge and stream temperature, the upstream stream stretch actually is designated as having relatively high natural value and is a habitat of for instance the brook lamprey (*Lampetra planeri*) (Verdonschot et al., 2002).

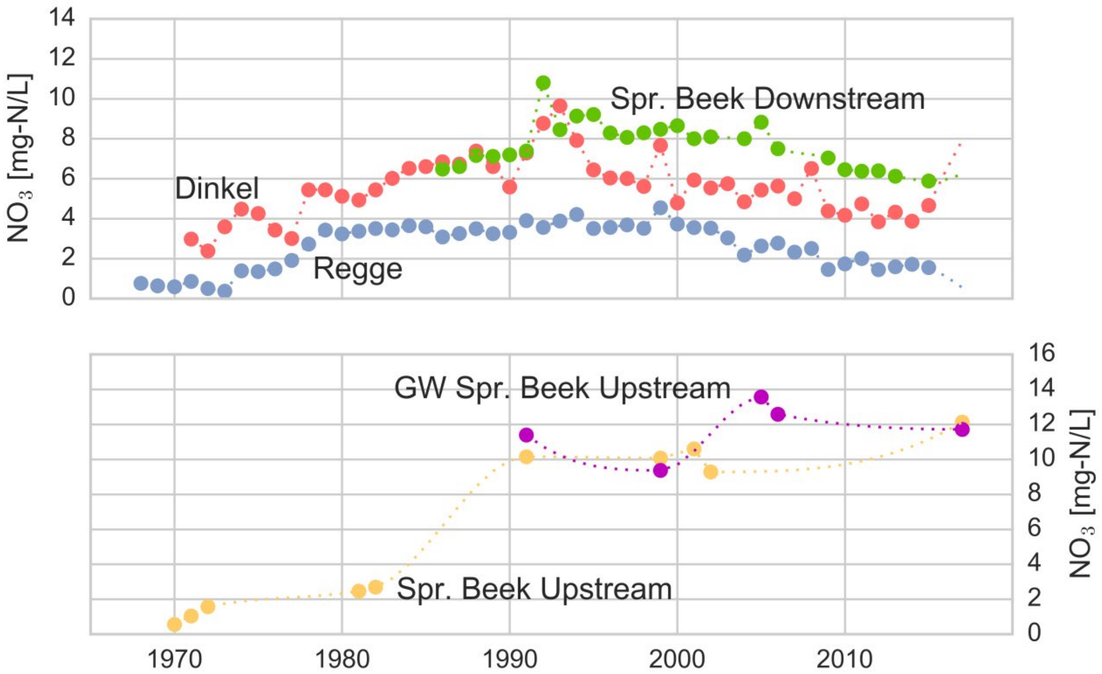


Figure 10. NO3-N concentrations in the Regge, Dinkel and Springendalse Beek downstream (top), and NO3-N concentrations in the Springendalse Beek downstream, upstream and in the groundwater (GW) upstream.

The largest sources of phosphorous in the surface water of the Regge and Dinkel catchments are overland flow from agricultural fields and the outflow of waste water treatment plants. Most of the phosphorous used in agriculture is retained in the unsaturated zone and does not leach to the groundwater. As opposed to the long delivery times of nitrate in the Springendalse Beek (Figure 10), the levels of phosphorous in the stream dropped directly following the abandonment of the agricultural fields upstream, although some phosphorous still seems to leach from the enriched soils (Nijboer et al., 2003).

4.2.3 Water Temperature

Figure 11 shows the daily average temperature of the Springendalse Beek and the Elsbeek during the summer of 2016 and shows that the Springendalse Beek compared to the Elsbeek has a more stable temperature regime with less temperature variation during the summer and lower temperature peaks, which is caused by its higher input of groundwater. Table *3* shows the average monthly temperature for August and November both upstream and downstream of a ~500 m river stretch of the Springendalse Beek and Elsbeek. During summer, the difference in temperature between the up- and downstream parts of the Springendalse Beek is almost 2 °C while for the Elsbeek this difference is only 2/10th of a degree. On the contrary, the difference between up- and downstream is larger in the Springendalse Beek during a cold month. This is the result of the fact that the upstream part of the Springendalse Beek is highly influenced by groundwater and this groundwater has little temperature variation throughout the year. The difference in stream temperature between a warm and cold month is also a characteristic of the temperature regime and influenced by the buffering effect of groundwater. The average downstream temperatures between August and November differ about 5 degrees for the Springendalse Beek and 10 degrees for the Elsbeek, again showing the buffering temperature effect of groundwater in the Springendalse Beek. Scales are important here as well; the influence of groundwater on stream temperature is clearer in smaller streams such as the Springendalse Beek. Groundwater seepage mitigates summer temperature peaks in the Springendalse Beek and creates a thermal habitat where stenothermic species are able to reside (Verdonschot et al., 2002).

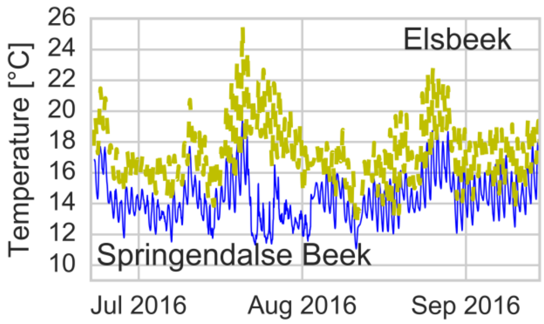


Figure 11. Average daily temperatures of the downstream parts of the Springendalse Beek and Elsbeek during the summer of 2016.

Table 3. Average temperatures [°C] of two Dinkel tributaries in August and November 2016

Stream	Month	Average Temperature [°C]	
		Upstream	Downstream
Springendalse Beek	August	12.45	14.41
	November	9.86	9.69
Elsbeek	August	16.59	16.81
	November	6.90	6.86

4.3 Thames catchment

4.3.1 Water storage and flow

Variations in groundwater heads in the Thames Basin are dominated by seasonal variations in evapotranspiration and inter-annual variability in precipitation, the latter represented by the Standardised Precipitation Index (6 month accumulation) for the Thames Basin (Centre for Ecology and Hydrology, 2017) in Figure 12. This seasonal and inter-annual variability is also seen in the flow of the Thames at the tidal limit of the Basin at Kingston, London, Figure 12. Across the Basin episodes of high precipitation, groundwater level stand and flow have been associated with extensive groundwater flooding (Adams et al., 2010; Hughes et al., 2011; Upton and Jackson, 2011; Macdonald et al., 2012), for example in the winter of 2014 (Ascott et al., 2016a; 2017). Similarly, major episodes of drought, for example in 1975-76, 1988-92 and most recently in 2011-12, have been associated with low groundwater levels, reduced flow, drying up of ephemeral streams in Chalk sub-catchments and lowered groundwater yields (Bloomfield and Marchant, 2013; Folland et al, 2015). Warming and changes in the intensity of storm events in the UK, including the Thames Basin, has occurred over the 20th century (Jenkins et al., 2009) consistent with anthropogenic climate change, but evidence of the effect of these changes on groundwater recharge and heads is illusive (Jackson et al., 2015; Watts et al., 2015).

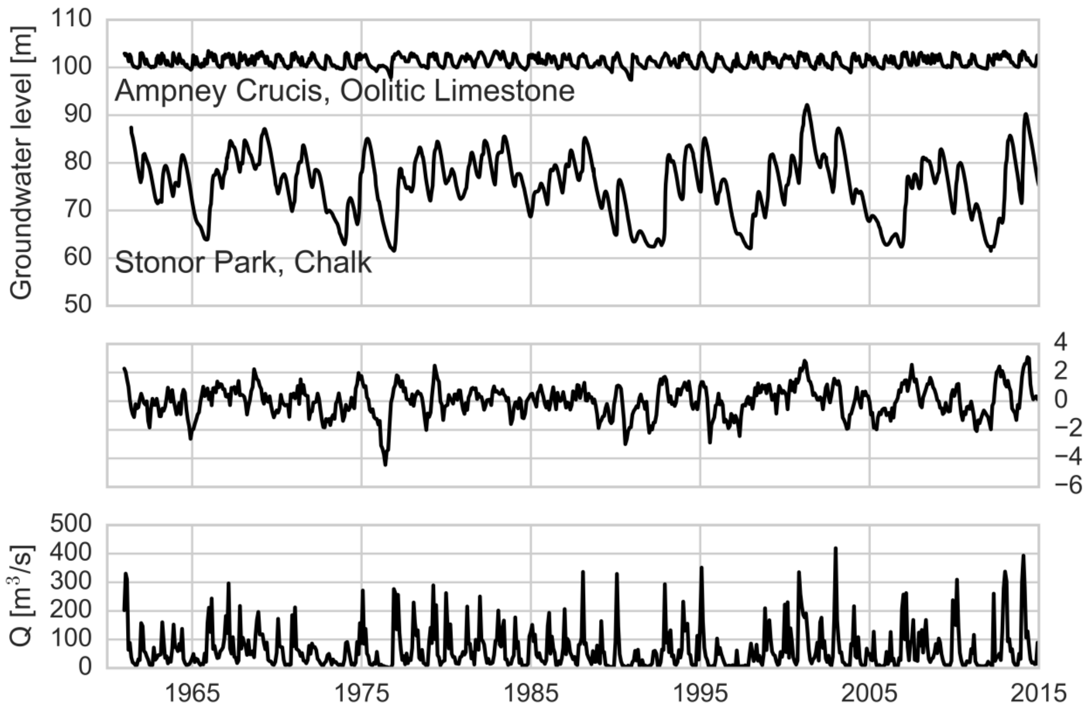


Figure 12. Variation in groundwater levels in the Jurassic Oolitic Limestone and Chalk Aquifer; temporal changes in the average Standardised Precipitation Index (6 month accumulation) across the Thames Basin; and, flow in the Thames at the tidal limit at Kingston.

There is a long history of abstraction from both groundwater and surface water sources in the Thames Basin. As with the rest of England, abstractions from the unconfined aquifer steadily increased from the 1940s reaching a peak in the late 1980s and early 1990s (Downing 1993b), while over a similar period groundwater abstraction from the confined Chalk of London significantly reduced leading to groundwater level rebound (Environment Agency, 2016b). The drought of 1988-1992 resulted in low groundwater levels across the unconfined Chalk of the Thames Basin, and the environmental regular at the time, the National Rivers Authority, identified five groundwater-dominated sub-catchments in the Chalk where over-abstraction had contributed to extreme low flows during this drought (National Rivers Authority, 1993). Low flow alleviation schemes were subsequently put in place (Clayton et al., 2008). Since 2000 across the Basin total abstraction has fallen, Figure 13 (Environment Agency, 2015), mainly due to a reduction in abstraction from surface waters, although during the recent drought of 2011-12 the relative proportion of groundwater abstraction was markedly increased (Figure 13).

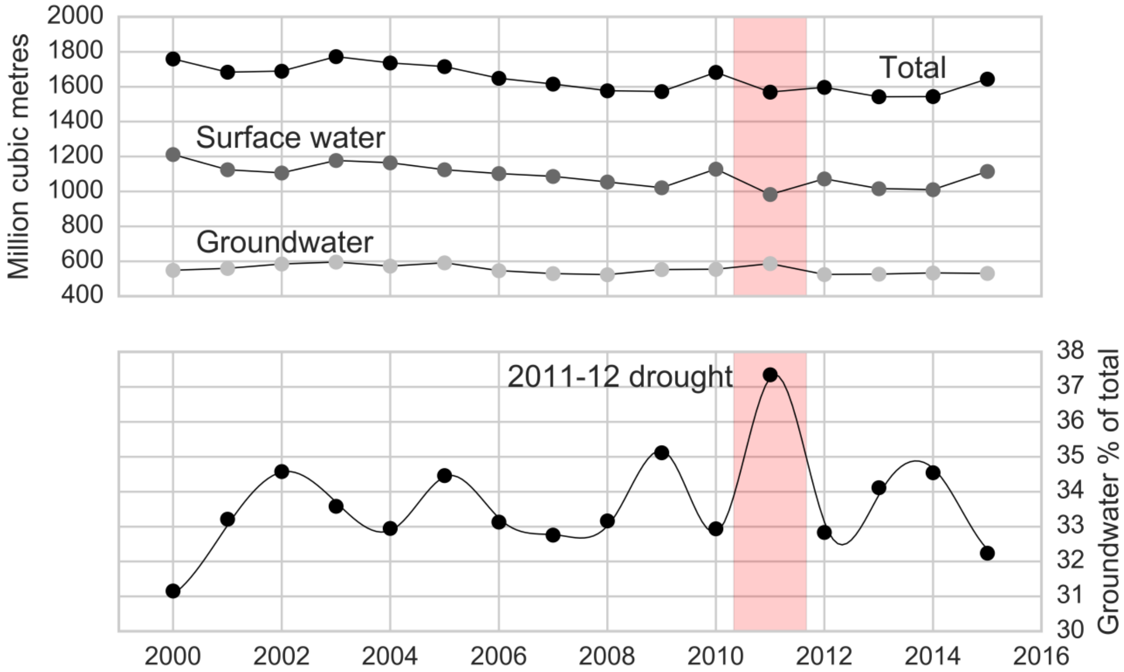


Figure 13. Annual total abstraction and abstraction from surface water and groundwater sources since 2000 and groundwater abstraction as a percentage of total abstraction (the 2011-12 drought is highlighted).

There is some evidence that leakage from the aging water mains network may be modifying the surface water flow regime within the Thames Basin (Bloomfield et al., 2009) and that it may also affect surface water quality (Ascott et al., 2016b; Gooddy et al., 2017).

4.3.2 Nutrients

In an analysis of the nitrogen (N) budget of the Thames Basin from the late 19th century to the present, Worrall et al. (2015a) have shown that before extensive use of inorganic fertilizers the total N budget approximated to steady state, but following widespread use of fertilizers there was net accumulation of N. This resulted in the Basin becoming saturated with respect to nitrate between about 1945 and 1995 (Worrall et al., 2015b) and there was a commensurate increase in nitrate concentrations in the river Thames (Howden et al., 2010; Worrall et al., 2015b). Modelling the fate of nitrate from intensive farming in the Basin has shown that there is still a significant legacy of nitrate in the soil and groundwater compartments within the Basin (Howden et al., 2011), and that restoration of nitrate concentrations in surface waters to levels similar to pre-intensification of farming would require basin-wide changes in land use and management (Howden et al., 2011). Due to the relatively low dissolved organic carbon (DOC) of the Chalk and Oolitic Limestone, once nitrate is in the aquifers the potential for significant denitrification is limited. Consequently, within the major aquifers of the Thames Basin nitrate is relatively mobile and considered broadly conservative in oxic groundwaters (Stuart et al., 2014). Based on a modelled nitrate input function and treating nitrate as a conservative pollutant, Wang et al. (2012; 2013) estimated the arrival of peak nitrate at the water table, and Ascott et al. (2016c) subsequently modelled the quantity of nitrate remaining in storage in the unsaturated zone. Ascott et al. (2016c) identified large areas of the Thames Basin, primarily over the Chalk aquifer of the Chilterns and Berkshire Downs, where substantial quantities of nitrate remain in the unsaturated zone. The peak arrival of that nitrate is estimate to not be due for at least another 50 years (Wang et al., 2012; 2013).

The loading of phosphorus (P) in the aquatic environment is typically considered to be primarily from agriculture (diffuse source) and sewage treatment work effluent (point source). However Ascott et al. (2016b) and Gooddy et al. (2017) have shown that in urbanised areas of the Thames Basin up to 30% of the total flux of P may be from mains leakage (mains supply is dosed with phosphate), with the relative and absolute contribution from this source increasing substantially since 1993 (Gooddy et al. (2017). Like N, P is considered conservative in the Chalk and Gooddy et al. (2017) estimate that for typical shallow groundwater systems P recharged to groundwater 20 years ago may currently be discharging into river networks.

4.3.3 Water temperature

Jenkins et al. (2009) have described a change in average daily mean temperature across the Thames Basin of between 1.4 and 2.1 °C between 1961 and 2006. However, there has been no systematic assessment of the temperature of groundwater in the Thames Basin, primarily due to the absence of suitable monitoring data, and so it is not possible to assess the impact changes in air temperature have had on groundwater temperature. Hannah and Garner (2015) noted that surface water temperatures across the UK, including the Thames Basin, have increased in the latter part of the 20th century, but that this could not simply be attributed to climatic warming since river temperature is a complex response to climate and hydrological drivers, basin properties including groundwater contributions and anthropogenic impacts. Watts et al. (2015) noted that although baseline groundwater temperatures are poorly understood, groundwater contributes much of the summer flow in some rivers, directly influencing water temperature. Using sub-hourly air, river and groundwater temperature data for a site where River Terrace Gravels are in hydraulic connection with the Thames at Wallingford, Habib et al. (2017) showed a lag in temperature between groundwater in the gravel aquifer and the air and river temperature and a seasonal contrast between relatively warm winter groundwater and cooler summer groundwater compared with air and river temperatures (Figure 14).

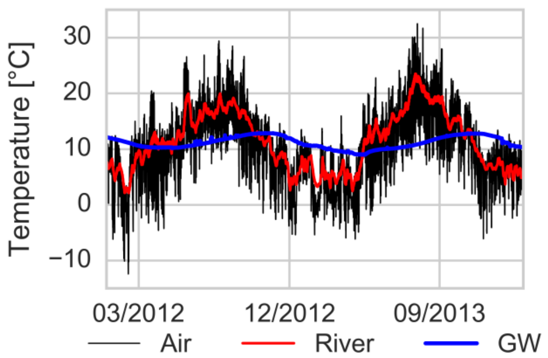


Figure 14. Daily air, Thames (river) and groundwater (GW) temperatures for a two year period from January 2012 for Wallingford, Oxfordshire.

At a range of sites across the Basin the thermal effects of upwelling groundwater in Chalk sub-catchments has been recorded, however these effects are typically highly localised, e.g. House et al. (2015; 2016a; 2016b).

5. Discussion & Conclusions

5.1 Comparison of drivers, pressures and states of the catchments

The proposed DPS framework was used to describe and analyse the effect of stressors on a coupled GW-SW system. It was shown to be a useful basis for this analysis and helped harmonizing the description of the different multi-stressed catchments. In the following paragraphs the drivers, pressures and states of the catchments will be compared.

5.1.1 Water storage and flow

In all four catchments storage and flow from groundwater has been changed by anthropogenic influences such as the installation of subsurface drainage and initiation of groundwater abstractions, which have had differing implications for groundwater levels and flow paths in the catchments. In the Odense catchment, flow is dominated by shallow groundwater systems and groundwater accounts for a major part of the streamflow, which is partly discharged through artificial drainage systems. Groundwater is especially important in sustaining streamflow in the driest seasons, but is threatened by unsustainable groundwater abstraction. In the Regge and Dinkel catchments, flow is likewise dominated by groundwater and heavily influenced by both groundwater and surface water abstractions and artificial drainage systems. These drainage systems have been shown to increase the fast outflow of groundwater and with that increase peak flows. In the Thames catchment, flow is dominated by seasonally varying evapotranspiration and recharge and by interannual variation in driving meteorology. Because of its scale, flow in the Thames catchment is spatially averaged and therefor stresses on the groundwater system can show marked variation between sub-catchments. For example, flow in small groundwater (Oolite and Chalk) dominated sub-catchments, particularly in the upper catchment, can be relatively sensitive to groundwater abstractions especially during drought years.

Changes in flow regimes might not always be apparent from discharge measurements, but this does not mean that groundwater flow paths have not been changed. Changed flow paths could lead to discharge of water of different quality and temperature. For instance, discharge of groundwater from shallow subsurface drains is different from discharge from deep groundwater systems (Rozemeijer et al., 2010), so alterations in groundwater flow paths may also change in-stream habitat conditions.

5.1.2 Nutrients

In addition to causing geohydrological alterations, agricultural intensification can causes diffuse pressure from nutrients and pesticides, of which this paper focussed on nutrients. All four catchments suffer from high levels of historic nutrient loading, which contaminates surface waters both directly through overland flow and though the groundwater. Thick unsaturated zones in the Thames Basin prevent fast leaching of nitrate to the groundwater, and thick aquifers create additional lag time in the effect of this nitrate on surface waters. These lag times caused by the slow flow of groundwater are also documented for the Regge and Dinkel catchments, albeit shorter. Even with reductions in nitrate loading, surface water nitrate concentrations will remain high for the coming decades in the Thames, Regge and Dinkel catchments due to the historical inputs. Long lag times are relatively absent in the Odense catchment due to the shallow groundwater system.

Denitrification can occur in the groundwater system in anoxic conditions in the presence of organic matter or pyrite. These conditions are met in the Odense catchment, but although denitrification has been identified as being active in the Odense, groundwater is still the main transport path for nitrate in the Odense. Denitrification occurs on local scale in the Regge and Dinkel catchments, but is effectively absent in the relatively pure calcium carbonate aquifers of the Thames Basin.

Although in general the main source of phosphorous in the four catchments is effluent from waste water treatment plants, significant amounts of P can also be transported by groundwater. In fact, in the Odense catchment modelling suggested that the main source of mineral P was transport from agricultural fields by groundwater, and in urban areas in the Thames catchment evidence indicates that phosphorous enters the groundwater from leaking water mains network and is subsequently transported to the surface water.

5.1.3 Water temperature

Groundwater influences stream and river temperature by providing an input with a relative constant seasonal temperature. This way, it dampens summer temperature peaks of the surface water, as in the Springendalse Beek in the Dinkel catchment. In addition, at locations with significant groundwater discharge, specific temperature habitats are formed e.g. in the Chalk sub-catchments of the Thames. The effect of groundwater on stream temperature has only been shown for small scale headwaters. In downstream parts, climatic effects seem to be much stronger, highlighting the importance of riparian zones. However, a systematic lack of temperature observations exists both in groundwater and surface waters in the catchments described here.

Table 4. Overview of how groundwater affects flow, nutrients and temperature in the four catchments.

	Characteristic	**Odense**	**Regge&Dinkel**	**Thames**
Flow	GW dependency of flow	++	++	+
	Depth of GW system	-	+	++
	Drainage systems	++	+	-
	GW abstractions	++	+	++
	Spatially averaging due to scale	-	-	++
Nutrients	Nitrate concentrations	++	++	++
	Unsaturated zone lag times	-	-	++
	Saturated zone lag times	+	++	++
	Buffering by denitrification	++	+	-

Temperature	Temperature effect groundwater seepage	?	+	+
	Temperature effect riparian zones	?	+	+

5.2 (Geo)hydrological controls on the DPS cascade

The differences in the effects of the drivers on states between the catchments (Table 4) are the result of differences in a range of factors, such as catchment size, geology, land-use and management practices. The depth of the groundwater system is directly related with the travel times of nutrients, as is shown in the Regge, Dinkel and Thames catchments. Thick unsaturated zones in the Thames catchment increase the time lag in the delivery of nutrients. In addition, relatively small drainage systems, such as in the Odense, Regge and Dinkel catchments, provide a short-cut for the outflow of groundwater, and decrease the travel times and thus lag time in nutrients. The potential for denitrification in aquifers is important in the attenuation of nitrate, and is greatest in the Odense, followed by the Dutch catchments. Groundwater abstractions are a stressor in all four catchments.

5.3 Stressor interaction, propagation and buffering

Figure 15 illustrates how different drivers and pressures can interact within linked GW-SW systems. For example, if streamflow decreases, for a given loading of nutrients the concentration of nutrients will increase. Climate change affects all other stressor by changing temperature, precipitation and evapotranspiration patterns. It is expected that winter peak flows increase and summer baseflow decreases, while temperatures increase (Kuijper et al., 2012, Kløve et al. 2014). A decrease in groundwater discharge will decrease the temperature effect of groundwater, increasing stream temperature in summer and decreasing winter stream temperatures. This will result in a loss of thermal habitats, especially combined with higher air temperatures. On longer timescales the temperature of groundwater will also increase resulting in further shrinkage of summer thermal refugia (Meisner et al., 1988; Isaak et al., 2012). In addition, an increase in groundwater pumping for irrigation during droughts is already occurring in the Thames catchment, and these abstractions may increase with more regular drought periods, resulting in even more lowering of groundwater tables and consequently of baseflow. Conversely, a substantial groundwater contribution to streams could buffer the streamflow response to climate change, sustaining summer flows for elongated periods as opposed to a stream without groundwater input (Tague et al., 2008). Additionally, the presence of groundwater discharge could mitigate part of the effect of an increase in air temperature on streamflow temperature.

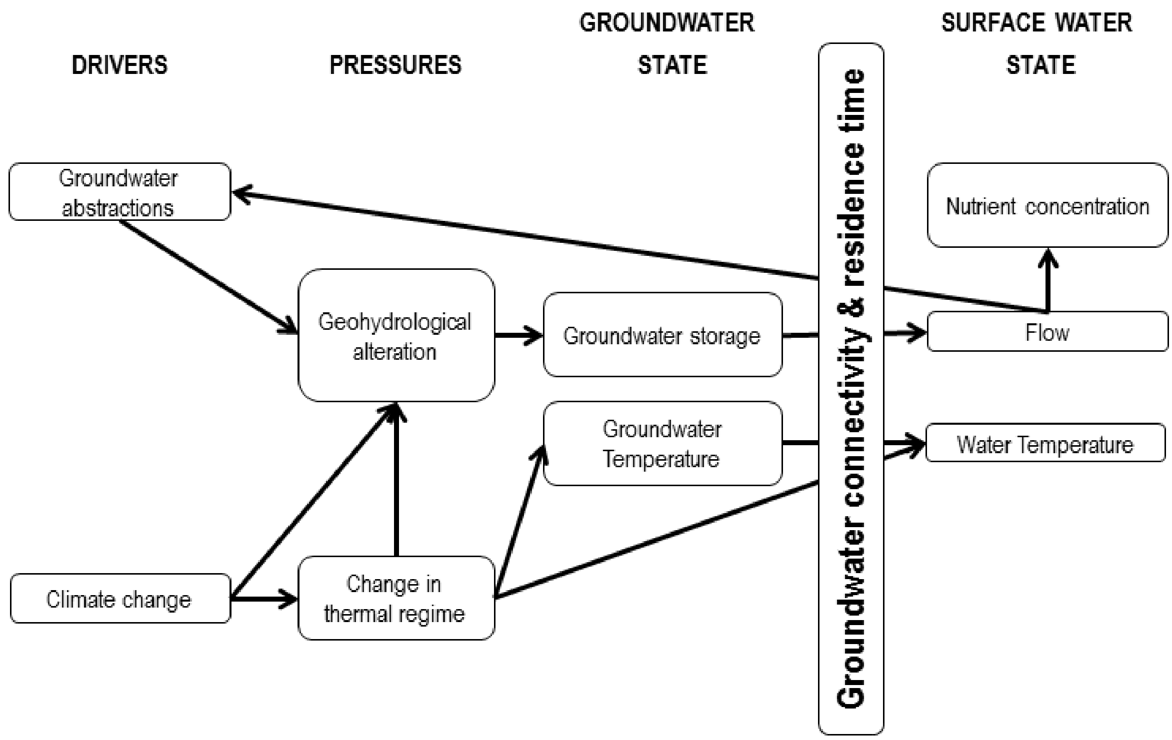


Figure 15. Modified DPS framework showing the interactions and feedbacks following from climate change. Decreased streamflow will lead to increased nutrient concentrations and groundwater abstractions.

This paper started with the hypotheses that groundwater enables the propagation of stressors spatially and in time, and that groundwater acts as a buffer to stressors. Case specific examples have been given, demonstrating both these mechanisms. Groundwater provides baseflow, often good quality water with a stable temperature, and may buffer the temperature increase following climate warming. Groundwater has also been shown to transmit stressors to surface waters, for instance nitrate from agricultural fields to streams. Being a relatively slow system, these stressors are also lagged in time. Groundwater both propagates and buffers stressors, but its effects depend on the local geology, climate, land-use, stressor combinations and scale.

5.4 Implications for ecosystem status

Groundwater influences the status of ecosystems in the catchments by providing water, nutrients and energy to aquatic ecosystems (Bertrand et al., 2012) and that way creates refugia in the surface water to for instance fish (Power et al., 1999). Functioning of aquatic ecosystems is strongly dependent on flow (Arthington et al., 2006; Poff and Zimmerman, 2010), which is frequently influenced by groundwater. Aquatic species have adopted their life strategies to specific flow regimes (Bunn and Arthington, 2002; Lytle and Poff, 2004), but many of these regimes are transformed by human interference in catchments (Feld et al., 2011), including changes in the groundwater system. Changes in flow regime of groundwater dominated rivers lead to more generalist and tolerant species (Blann et al., 2009) because species that evolved in a more variable environment are less vulnerable to environmental changes than habitat specialists (Schlosser, 1990). Groundwater also propagates nutrients in the study catchments, which leads to changing food webs (Blann et al., 2009) and is potentially toxic to aquatic species (Camargo et al., 2005). High nutrients levels can cause eutrophication, leading to low oxygen levels. It was shown that groundwater upwelling can influence water temperature, and is therefore a crucial component in the formation of river habitats by providing thermal refugia for aquatic biota during warm or cold periods of the year (e.g. Power et al., 1999). Additionally, groundwater discharge may buffer the warming effects of climate change.

Linking abiotic to biotic states is challenging. An attempt was made for the Odense catchment using a Danish national dataset comprising 263 variables and 131 observations to evaluate the relationship between different river ecosystem stressors and four ecological status indices for streams. These indices are Danish indices for ecological status assessment for fish fauna, macrophytes and macroinvertebrates, and the widely used Average Score per Taxon for macroinvertebrates (ASPT) (Ferreira et al., 2016). Q_{90} showed a positive correlation (better ecological status) with all the indices, and BFI also showed a positive correlation with the indices except that for macrophytes. These results corroborate the discussion above regarding the relevant role of groundwater in the preservation of the aquatic ecosystems in the Odense Fjord catchment, favouring a higher ecological status. The analysis also included water quality stressors. Total phosphorous (TP) was seen to be relevant in the estimation of ecological status through macroinvertebrate indices and, as expected, exerting a negative influence. Since the model suggested that groundwater was the main source of phosphate, for macroinvertebrates groundwater has mixed effects on the ecological status of the streams in the Odense Fjord catchment by providing a hydrological regime that favours a high status but on the contrary being an important source of dissolved phosphate. Although this empirical modelling shows groundwater-related indicators to be relevant for ecological status, the effect of groundwater is complex and non-linear and at this time there is not enough knowledge and data available to fully understand these linked systems.

5.5 Implications for management

Surface water managers are giving increased attention to the importance of groundwater. For instance, in Australia water managers started integrating groundwater and surface water management in the last decade (Lamontagne et al., 2012). In the US groundwater needs of ecosystems are being taken into account in conservation plans (Brown et al., 2007) and in Europe groundwater is now included in the Water Framework Directive, Groundwater Directive, Habitats Directives and the CIS Working group on Groundwater (European Commission. 2000; 2008).

Important for management is that long groundwater residence times create a time lag in the contribution of historic pollution input which causes pollution even after management interference in a catchment (Nijboer et al., 2003; Hamilton, 2012). Groundwater fed surface waters often contain water with ages of several decades and older (Hamilton, 2012) which means that current pollution inputs will propagate through the groundwater system and form a future pressure on surface water ecosystems. This proposes a problem to water managers who have to deal with restoring such catchments as a good chemical and ecological status cannot be achieved with excessive nutrient levels. However, evidence from the upstream part of the Springendalse Beek suggests that the ecological value of such stream can be high even with high nutrient levels, encouraging water managers to not abandon these sites, even though good ecological status cannot be achieved on the short term. This is also the case in steams in the south of the Netherlands where data seems to indicate that nitrate levels actually show some correlation with a high ecological status (Waterboard Limburg, personal communication), possibly because nitrate in these catchments is an indicator for groundwater influence.

As opposed to nitrate, phosphorous is generally related to surface runoff processes, but examples from the study catchment showed that groundwater should also be taken into account in managing catchments with phosphorous stress. Temperature is an often overlooked parameter, but has been shown to be important for the creating of specific habitats for e.g. stenothermic species (Power et al., 1999).

A groundwater contribution to surface waters provides ecosystem services, by providing habitats for e.g. trout which are fished in the Thames chalk streams, by providing cool water important for some fish spawning, and by providing water and preventing ceasing of flow during droughts.

Freshwater ecosystems are under multiple-stress and groundwater has a crucial effect on many of these ecosystems. Using examples from literature and from four different European catchments, it was shown how groundwater can influence surface waters in stressed systems. Groundwater has essential implications for river basin management and this study thus supports the call to water managers made in the FP7 REFORM project to take groundwater into account in river basin management (Hendriks et al., 2015b). Groundwater should be taken into account in ecological management as a possible component of the total environmental system, which can transport and buffer stressors and their effect. Groundwater may be a crucial component in mitigating the effect of climate change on river ecosystems (Tague et al., 2008; Palmer et al., 2009), which strengthens the need for more integrative management of ground- and surface waters.

A framework has been proposed which shows how groundwater in lowland catchments acts as a bridge between stressors and their effects within surface waters. This framework shows water managers how their management areas might be influenced by groundwater, and helps them to include this important, but often overlooked part of the water cycle in their basin management plans.

5.6 Implications for future monitoring

It was noted that linking abiotic with biotic indices remains a challenge due to a lack of data and system understanding. An obstacle is that data on quantity, quality and ecology of both surface and groundwaters are measured by a myriad of agencies for various legislations. In the Netherlands for instance, surface water quality and ecology are measured by Waterboards and Rijkswaterstaat (Ministry of Infrastructure and the Environment), often with different timing of sampling. Shallow groundwater is monitored by both Waterboards and the RIVM (National Institute for Public Health and the Environment) and deep groundwater by the provinces. To truly understand the connection between groundwater and surface water ecology it is needed to start combined monitoring in a synchronized and coordinated way. The analysis of the study catchments also revealed a lack of data on the temperature of both groundwater and surface water, while it is an important parameter considering future climate warming.

Acknowledgements

This work is part of the MARS project (Managing Aquatic ecosystems and water Resources under multiple Stress) funded under the 7th EU Framework Programme, Theme 6 (Environment including Climate Change), Contract No.: 603378 (http://www.mars-project.eu). Eugenio Molina-Navarro has received additional support from the Ramón Areces Foundation postgraduate studies programme (26th Edition). Dr Bloomfield publishes with permission of the Executive Director, British Geological Survey (NERC).

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