Relating wellfield drawdown and water quality to aquifer sustainability – A method for assessing safe groundwater abstraction

Nørskov Gejl, Ryle; Løgstrup Bjerg, Poul; Henriksen, H. J.; Bitsch, K.; Trolborg, L.; Schullehner, J.; Rasmussen, J.; Rygaard, Martin

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1 Relating wellfield drawdown and water quality to aquifer
2 sustainability – a method for assessing safe groundwater
3 abstraction

4

5 Authors: Gejl, R.N.^{1,2}, Bjerg, P.L.¹, Henriksen, H.J.³, Bitsch, K.², Trolborg, L.³, Schullehner, J.³,
6 Rasmussen, J.² and Rygaard, M.¹

7 Institutions: DTU Environment¹, HOFOR A/S² and GEUS³

8 E-mail address: ryln@env.dtu.dk

9

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- 12 - Sustainable groundwater abstraction
- 13 - Groundwater impact assessment
- 14 - Water utility management
- 15 - EU Water Framework Directive

16

17 **Abstract**

18 Planning for sustainable groundwater abstraction requires realistic and reliable assessments of
19 groundwater drawdown in aquifers as well as impacted groundwater-dependent terrestrial and
20 aquatic ecosystems. Present-day impact assessments allocate water for the environment and
21 ecosystems in the form of environmental groundwater requirements (*EGWRs*). However, securing
22 sustainable groundwater abstraction also requires stable groundwater quality for consumers and
23 receptors (e.g. streams and groundwater-dependent ecosystems). We suggest dividing *EGWR* into
24 two parts, namely *EGWR_{flow}*, related to impacts on surface water, and *EGWR_{wq}*, related to impacts on
25 groundwater quality. We then propose a method for modelling maximum safe groundwater
26 abstraction rates based on a relationship between water quality and piezometric heads. The method
27 provides estimations of sustainable abstraction rates that secure stable water quality and maintain
28 water security. Using hydrological modelling, we estimated spatially differentiated *EGWR_{wq}*, for
29 Zealand, Denmark, based on three different conditioned drawdowns of 3, 6 and 9 m. We found that

30 depending on the boundaries and the different conditioned drawdown $EGWR_{wq}$ poses a 2.5 – 83%
31 share of the groundwater recharge. We found that a redistribution of water abstraction would make
32 27 Mm³ (million m³) available compared with actual abstraction, dependent on scale, while keeping
33 within a conditioned drawdown of 3 m. Our results demonstrate how conditioned drawdown can be
34 implemented with the objectives of evaluating actual abstraction and securing stable water quality,
35 hence comprising a step towards more reliable and relevant groundwater impact assessments.

36 1. Introduction

37 1.1 Background/Motivation

38 Freshwater is vital for ecosystems and is the base of human hygiene, food production and well-being
39 (WWAP, 2016). In many places around the world, groundwater is essential for securing freshwater
40 for ecosystems and humans (Zektser and Everett, 2004). To preserve stable groundwater resources
41 for future generations, there is a growing interest for answers to essential questions like: How does
42 our water abstraction affect the environment and the groundwater resource? How much water does
43 nature need? And of what quality? These questions served as motivation for this study. In addition
44 there is a pressing need to develop methods for water managers as The Water Framework Directive
45 provides a structure for protecting streams, lakes, coastal waters and groundwater, in order to
46 prevent further deterioration (European Union, 2006), but without describing a methodology or
47 approach.

48 Methods for quantifying water stress, as well as what objectives they should serve, are discussed
49 widely in the field internationally (e.g. Hoekstra, 2016; Nunez et al., 2016; Pfister et al., 2017).

50 Evaluating water abstraction is often based on components of the water balance, e.g. Available-

51 **Water-Remaining**, AWaRe (Boulay et al., 2017), **Withdrawal-To-Availability**, WTA (Alcamo and
52 Henrichs, 2002), **Water Stress Index**, WSI_{index} (Pfister et al., 2009), **Water Stress Indicator**, WSI

53 (Smakhtin et al., 2004) and groundwater footprint (Gleeson et al., 2012). Most assessments include

54 an estimation of environmental water requirements (EWR), interpreted as the amount of water

needed to sustain flow in streams and maintain wetlands etc. (International River Foundation, 2007). Water stress assessments focus generally on large-scale, e.g. global and regional, aquifers (Gleeson et al., 2012) with an average area of 50 000 km² and therefore give little consideration to local water abstraction challenges in smaller groundwater catchments (Hybel et al., 2015). The indicators mentioned above generally focus on surface water, which can result in misrepresentations of the groundwater impact (Gejl et al., 2018). Throughout Europe, 50% of drinking water supplies relies on groundwater (Völker and Borchardt, 2019), and it is a significant resource for industrial and agricultural water demands (Jones and Anthony, 2011). This leads to a need for trustworthy indicators revealing groundwater stress that can secure relevant evaluations of resource availability. To overcome the focus on surface water and the large-scale perspective, we developed AGWaRe, the **A**vailable **G**round**W**ater **R**emaining, an indicator designed to assess groundwater impact and which can be applied on any scale (Gejl et al., 2018). However, AGWaRe does not consider potential impacts on groundwater quality that may be influenced by abstraction.

1.2 Environmental groundwater requirements

Internationally, there have been different approaches for ensuring and including the concept of Environmental Groundwater Requirements (*EGWR*). In Australia, for instance, focus has fallen on groundwater-dependent ecosystems and their water demands (Doeg et al., 2012), concentrating particularly on impacts on the shallow groundwater level. Another example, is in a study of the exploitation of global aquifers, *EGWR* were included to ensure environmental flows by setting *EGWR* high enough to sustain Q_{90} in streams, i.e. the monthly streamflow that was exceeded 90% of the time compared to a reference period (Gleeson and Wada, 2013). Such methods aim to limit the effects on river runoff and aquatic ecosystems. Danish water management plans relate abstraction to a percentage of the groundwater recharge (Henriksen et al., 2008), in order to screen for aquifer sustainability, based on a correlation between groundwater recharge (GWR) and aquifer health. A recent initiative involved evaluating how groundwater abstraction affects ecological flow indicators (Graeber et al., 2015), which was a positive step towards obtaining actual impacts on an endpoint

level; however, these indicators only consider in-stream ecological flow requirements and incorporate neither groundwater-surface water interactions nor groundwater quality.

Gejl et al. (2019) looked further into how groundwater abstraction influences groundwater quality. However, the question of how much water should be reserved to prevent changes in groundwater quality remains. To our knowledge, no one has defined *EGWR* in terms of sustaining groundwater-dependent ecosystems, flow and water quality in aquifers. Inspired by environmental flow requirements for river systems (International River Foundation, 2007), we define *EGWR* as “*the water from groundwater resources needed to sustain flows, preserve groundwater-dependent ecosystems and maintain good groundwater quality*”. Through such a definition, *EGWR* includes the adverse effects of groundwater abstraction on groundwater level and quality, including risks related to increased vulnerability due to a lowered piezometric surface (Henriksen and Refsgaard, 2013).

Due to the duality in the intentions of *EGWR*, we suggest separating environmental water requirements into:

EGWR_{flow}: Groundwater reserved to sustain base flow in streams and groundwater-dependent ecosystems (surface water)

EGWR_{wq}: Groundwater reserved to sustain aquifer health related to water quality affected by water abstraction in the aquifer (groundwater)

Several publications address the flows needed to sustain groundwater-dependent ecosystems (*EGWR_{flow}*) (e.g. Graeber et al., 2015; Hoekstra et al., 2012; Smakhtin et al., 2004), and therefore we focus on *EGWR_{wq}*, which is critical for managing water supplies based on groundwater resources.

1.3 Impacts from abstraction

For a groundwater system, the water balance is a function of groundwater abstraction, the hydrogeological setting, precipitation and evapotranspiration, whilst for an aquifer, the water balance is comprised of three main components: 1) groundwater recharge (GWR), 2) in- and

outflows over the aquifer boundaries (including abstraction) and 3) aquifer storage. Groundwater recharge diminishes lower down in the groundwater system, due to flows out of the groundwater compartments. Under natural conditions, groundwater systems are in a dynamic equilibrium where long-term average recharge equals long-term average discharge. Intense pumping will induce new recharge dynamics and impacts on the piezometric head and capture, causing drawdown, the depletion of springs and streamflow, loss of wetland, etc., and a new equilibrium may be established many years henceforth (Bredehoeft, 2002; Gleeson et al., 2012; Henriksen and Refsgaard, 2013); consequently, it is difficult to define consistent system boundaries for assessing the effects of groundwater abstraction. Drawdown can affect groundwater quality, e.g. pyrite oxidation, saltwater intrusion and the mobilisation of residual saltwater (Henriksen et al., 2008; Larsen and Postma, 1997; Postma et al., 2016).

In a study based on long-term data (1900-2015) of the piezometric head, abstraction and water quality for well fields abstracting water from confined aquifers around Copenhagen, we found that sulphate was the most sensitive parameter to drawdown and can be used to indicate overexploited aquifers (Gejl et al., 2019). For these well fields, we used sulphate concentrations as a proxy for evaluating sustainable abstraction in relation to the water quality of natural constituents. We found that from the mid-1980s and onwards, as water abstraction decreased and the piezometric head increased, sulphate concentrations decreased and stabilised. We suggested that water abstraction in this period (1985 - 2015) was sustainable from both a water quantity and a quality perspective. However, practical implementation in terms of managing water abstraction and any corresponding drawdown at various scales is still unresolved.

1.4 Aim

In the following, we demonstrate a method to assess safe abstraction from aquifers, based on the relationship between long-term drawdowns and potential groundwater quality impacts. The method aims to address the following questions:

130 - How can we implement a groundwater drawdown constraint that sustains quality on
131 multiple geographical scales over time?
132 - How much groundwater is available for abstraction without compromising aquifer water
133 quality?
134 - What suitable indicators in groundwater impact assessments could be used to account for
135 potential impacts on groundwater quality?
136 - What drawdown represents local groundwater stress?
137 The method will assist groundwater-based water utilities in their decisions, as well as the
138 management of water resources in well fields and on a regional scale. With the AGWaRe-compatible
139 method, the assessment can also feed into life cycle impact assessments considering groundwater
140 use.

141 2. Method

142 2.1 Study area

143 The study was carried out for Zealand, Denmark, an area with decades-long pressure on
144 groundwater resources caused by large water demands of Copenhagen, Denmark's largest city.
145 Zealand has an area of 7000 km² and a population of 2.3 million, of which 1.6 million live in the
146 Greater Copenhagen area. This has forced water suppliers to abstract groundwater far from the city,
147 in competition with local water supplies in the northern and eastern parts of Zealand. The majority
148 of the groundwater on Zealand is currently abstracted from limestone aquifers (82%), and the
149 elevations of the tops of these limestone aquifers vary (Figure 1).

Figure 1: The location of Copenhagen on the island of Zealand, and the elevations of the tops of the limestone aquifers, from which the majority of water is abstracted. The blue dotted lines represent the locations of HOFOR's well fields. The map was adapted from Højberg et al. (2015).

2.2 A generic concept of defining $EGWR_{wq}$ from the maximum allowed drawdown

Drawdown is defined as the difference in the modelled potentiometric head (termed the 'piezometric head' from now on) between pristine conditions (no abstraction) and actual abstraction:

$$Drawdown = head_{pristine} - head_{abstraction} \quad (1)$$

A conditioned drawdown is defined as a maximum allowed drawdown in the main aquifer from where the wells withdraw groundwater. From a conditioned drawdown, we can estimate the corresponding abstraction, which represents the maximum abstraction that is expected not to change water quality. We then define $EGWR_{wq}$ as the difference between the groundwater recharge under sustainable groundwater abstraction, GWR_{sus} , and sustainable groundwater abstraction, WU_{sus} , both defined from a conditioned drawdown:

$$EGWR_{wq} = GWR_{sus} - WU_{sus} \quad (2)$$

Formulating $EWGR_{wq}$ enhances awareness of the need to reserve water in order to sustain water quality in aquifers. To investigate the effects in aquifers from actual abstractions, we apply AGWaRe

(Gejl et al., 2018), i.e. a groundwater impact indicator that evaluates the stress of the groundwater resource compared to a reference. It is based on an absolute measure of how much water is available, $AMD_{aquifer}$ (**A**ailability-**M**inus-**D**emand), in an area and therefore gives an indication of absolute water remaining for other uses. AGWaRe is a normalised parameter by relating the $AMD_{aquifer}$ to a reference, AMD_{ref} :

$$AGWaRe = \frac{AMD_{ref}}{AMD_{aquifer}} \quad (3)$$

AGWaRe is cut-off below 0.1 and above 100, in line with the AWaRe methodology (Boulay et al., 2017). This implies that the greatest stress compares to 100 times the stress for the reference. $AMD_{aquifer}$ is availability-minus-demand in the aquifer:

$$AMD_{aquifer} = GWR_{sus} - WU_{actual} - EGWR_{wq} \quad (4)$$

where WU_{actual} is water abstraction for actual conditions. $AMD_{aquifer}$ represents the difference between groundwater recharge for the conditioned drawdown, actual human needs and the water needed to sustain water quality. AMD_{ref} is found with the same equation (4), but for a reference area. Thereby, AGWaRe relates local conditions to a reference condition (3). Replacing $EGWR_{wq}$ with (eq. 2), we obtain:

$$AMD_{aquifer} = GWR_{sus} - WU_{actual} - (GWR_{sus} - WU_{sus}) = WU_{sus} - WU_{actual} \quad (5)$$

and thereby $AMD_{aquifer}$ represents the *additional amount of water that could be abstracted without compromising water quality in aquifers*. If $AMD_{aquifer}$ is negative, actual water abstraction causes a greater drawdown than what one would expect to be sustainable and poses a risk of changing water quality. On the contrary, if $AMD_{aquifer}$ is positive, more water can be abstracted within the limits of the conditioned drawdown. With this definition, $AMD_{aquifer}$ can only be calculated for areas experiencing abstraction (Eq. 5), which is in accordance with our objective of evaluating impacts related to groundwater abstraction.

For an assessment of the distance to sustainable conditions (DSC), we also define the measure:

$$DSC = \frac{WU_{sus}}{(WU_{sus} - WU_{actual})} \quad (6)$$

where DSC is cut-off above 100 (negative values are given 100) and where 100 indicates that the actual water abstraction is close to or exceeds the sustainable limit.

2.3 Groundwater model and setup

We modelled the components of the water balance and the piezometric head, using the hydrological model for Zealand (a submodel of the national Danish hydrological model (DK-model), Højberg et al., 2015). The DK-model was constructed for assessing large-scale water balances and the interaction between groundwater and surface water on a regional scale (Højberg et al., 2013). The Zealand model simulates the fluxes to and from the groundwater resource and consists of 12 hydrostratigraphics, constituting the 12 calculation layers in the saturated zone. The DK-model includes daily precipitation, temperature and evapotranspiration (GEUS, 2009), and the results were validated against runoff measurements from more than 200 stream-gauging stations and more than 100,000 piezometric head observations from groundwater wells (Henriksen and Sonnenborg, 2003; Højberg et al., 2015). Groundwater abstraction was based on data from the national database JUPITER, which contains annual water statistics for utilities, industrial water use and model-simulated agricultural abstractions (Henriksen and Sonnenborg, 2003; Højberg et al., 2015). The geological model is discretised in a 100 x 100 m grid using layers and lenses for geological structures, whereas the hydrological model is discretised for a 500 x 500 m grid (Højberg et al., 2015; Klint et al., 2013). The model simulates an equivalent porous medium without any fracture flow, and we believe this assumption is valid despite limestone being a fractured medium, since the analysed groundwater drawdown represents long-term equilibrium conditions (Aisopou et al., 2014).

2.4 Conditioned drawdown

To model groundwater abstraction corresponding to a conditioned drawdown, we applied fixed head boundary conditions, correspondingly modelling a grid cell where water is drained until it reaches a certain piezometric head. The drained water constitutes the maximum water abstraction within the constrained drawdown. The fixed head was applied to all 1751 abstraction grid cells on Zeeland, and because well screens can extend across several layers, the drawdown was conditioned in seven confined layers, with limestone aquifers, sandy aquifers and aquitard layers covering them (the bottom seven layers in the model). The baseline scenario for our analysis refers to average actual abstraction for 2003 to 2012, which is 157 Mm³/year (million m³/year) for the groundwater system and 112 Mm³/year from confined aquifers.

In the simulations, drawdown was calculated as the mean difference in the piezometric head for each grid cell between pristine conditions (zero abstraction) and each scenario. Simulations were constrained as follows:

- We included water abstraction by public water utilities in the analysis, but we excluded agricultural and industrial wells, because they mainly abstract water from shallow, unconfined aquifers.
- To avoid saltwater intrusion, drawdowns were constrained to avoid piezometric heads below sea level.
- To avoid changing confined aquifers to unconfined, drawdowns were constrained to avoid piezometric heads below the bottom of the confining layer.

2.4.1 Scale

We wanted to include a consistent and relevant demarcation of groundwater bodies; however, this was a challenge because of the complexity and inconsistency of groundwater demarcations. Several well-defined scales exist for surface water where topography provides a simple and consistent demarcation of catchment areas. River basin, river catchment and sub- areas are all established

terms used in Danish River Basin Management plans (e.g. SVANA, 2016). For groundwater, however, there are no clear or stable demarcations on the local scale. Furthermore, the groundwater catchment area changes in line with abstraction and is therefore constant in neither time nor space. In Denmark, groundwater bodies (as referred to in the EU Water Framework Directive) are based partly on an aquifer delineation within the Danish geological model (GEUS, 2009) and partly by considering administrative and monitoring purposes (Trolborg et al., 2014). Groundwater bodies may merge aquifers located far apart, or divide continuous aquifers. Such mixed physical and administrative delineation impairs comparative assessments of groundwater impacts, and so with no obvious choice in terms of demarcation, we decided to explore five scales (Table 1).

Table 1: Five scale definitions for groundwater assessment and the associated applied scales with area range, advantages and disadvantages.

Generic scale definition	Applied scales	Area range (km ²)	Advantages	Disadvantages
Full groundwater system	Island of Zealand	7109	-Well defined groundwater demarcation	-Large-scale -Difficult to locate local stressors
Major administrative demarcations	River basins Topographic catchment areas for coastal water bodies (e.g. Roskilde Fjord and Køge Bay). This scale is used in the Water Framework Directive	440 – 2034 (Median 982)	-Well defined demarcation	-Large-scale -Difficult to locate local stressors -Demarcation based on surface water

Intermediate administrative demarcation	River catchments Topographic river catchment areas (e.g. Græse stream, Køge stream, etc. defined from downstream ID15 catchment areas in main river)	0.09 – 834 (Median 15)	-Well defined demarcation	-Demarcation based on surface water
Minor administrative demarcations	ID15. Sub-catchments (used in the Water Framework Directive)	0.07 – 105 (Median 11)	-High resolution -Easy to locate stressors	-Demarcation based on surface water
Minimum model discretization	500 x 500 m grid	0.25	-High resolution -Easy to locate stressors	-Not a hydrological demarcation

2.4.2 Defining conditioned drawdown

We considered three different conditioned drawdowns of 3, 6 and 9 m, as our earlier study of the connection between drawdown and changes in water quality parameters (Gejl et al. 2019) suggested a sustainable drawdown for Zealand between 3 and 9 m. For a discussion on the drawdown range, see Supporting Information (SI. 1). Following a conservative approach, we focused on a drawdown of 3 m, in order to enforce strict protection for the majority of aquifers. However, Gejl et al. (2019) showed that some aquifers were less vulnerable and could allow for a larger drawdown, without compromising water quality.

2.4.3 Defining groundwater recharge

We considered groundwater recharge as the vertical flow over a boundary and explored three definitions (Figure 2): 1) GWR_{inf} is the flow from an unsaturated to a saturated zone (also called 'net infiltration'), 2) $GWR_{net, aquifer}$ is the flow to and from the aquifer (both discharge and recharge, or net recharge) and 3) $GWR_{aquifer}$ is recharge to the aquifer.

Groundwater recharge is influenced by several factors such as pressure, hydraulic conductivity, depth to aquifer, artificial drainage, etc., and there can be several delineations, all of which would result in different quantifications of this process. Thorling et al. (2015a) used GWR_{inf} as a

Figure 2: Conceptual figure of delineations of groundwater recharge: 1) GWR_{inf} , the downward flow from an unsaturated zone to a saturated zone, 2) $GWR_{aquifer}$ groundwater recharge to the aquifer or 3) $GWR_{net, aquifer}$ the net groundwater recharge. GWD is groundwater discharge. The thickness of the arrows reflects the quantities of groundwater recharge in relative terms. AMD (availability-minus-demand) differs according to the boundaries.

265 demarcation, but it does not actually indicate groundwater availability in deeper parts of the
266 groundwater system, where the majority of abstraction occurs. Since the corresponding study (Gejl
267 et al., 2019) of the correlation between drawdown and water quality focused on confined aquifers,
268 we suggest $GWR_{net, aquifer}$ as the appropriate measure of groundwater recharge. $GWR_{aquifer}$ refer to
269 water flowing both upward and downward and is therefore smaller than $GWR_{net, aquifer}$ that only refer
270 to the recharge of the aquifer (downward). As expected, the quantities of the different delineations
271 of groundwater recharge show that GWR_{inf} is larger than $GWR_{aquifer}$ and $GWR_{net, aquifer}$ (Figure 3).

272

273 *Figure 3: Groundwater recharge for actual abstraction (2003-2012) for three delineations: a) GWR_{inf} is net*
 274 *infiltration (positive down), b) $GWR_{net, aquifer}$ is groundwater recharge to and discharge from limestone aquifers*
 275 *(positive down) and c) $GWR_{aquifer}$ is groundwater recharge to limestone aquifers. The groundwater recharge was*
 276 *modeled with the DK-model.*

3. Results

3.1 Actual abstraction and drawdown

The modelled distribution of the actual abstraction, $GWR_{net, aquifer}$, and drawdown in limestone aquifers from 2003-2012 showed that water abstraction and $GWR_{net, aquifer}$ were greatest in the eastern parts of Zealand (Figure 3b and Figure 4). Actual abstraction resulted in local drawdowns in limestone aquifers up to 24 m deep, while large areas had relatively small drawdowns (less than 2 m). Drawdowns less than 2 m even occurred in grid cells with an abstraction larger than 100,000 m³/year.

The correlation between the modelled and observed piezometric heads for 306 monitoring wells on Zealand (Sl. 3) showed a mean error (ME) of -0.14 m and an RMSE of 2.7 m. This was an excellent result especially considering that the top score is given to an RMSE < 4 in performance criteria when considering the goodness of hydrological models (Henriksen et al., 2003). We also plotted the difference in modelled and observed piezometric heads for the 28 well fields studied in Gejl et al. (2019) (Sl. 4), with the plot showing that generally the model is better at estimating relative changes than absolute levels. With the overall good model fit, we found that it was well-suited to our objective.

Figure 4: a) Actual average annual groundwater abstraction from 2003-2012 for private and public water supplies and b) modelled drawdown for actual abstraction on a grid size of 500 x 500 m.

3.2 Drawdown

For a conditioned drawdown of 3 m, the modelled abstraction was distributed more equally between the wells than for the actual abstraction (Figures 4 and SI. 5). Overall, the conditioned drawdown allowed for greater water abstraction than the actual abstraction from 2003-2012, but for some well fields, water abstraction would need to be reduced to obtain a maximum drawdown of 3 m. Similarly, $GWR_{net, aquifer}$ increased overall, but for some places with a former large drawdown, the $GWR_{net, aquifer}$ decreased.

3.3 Components of impact assessment and the influence of drawdown

The magnitude of the conditioned drawdown substantially influenced all components in $AGWaRe$, except GWR_{inf} (Table 2). Regarding the system as a whole, a conditioned drawdown of 3 m allowed for an overall increase in abstraction to 261 million m^3 /year, compared to the actual abstraction of 157 million m^3 /year. Regarding the aquifer the conditioned drawdown dominated the quantification

of $AMD_{aquifer}$, which ranged from 27 – 208 million m^3 /year for conditioned drawdowns of 3 and 9 m, respectively, meaning that the volumes of water available for other users greatly depend on the applied conditions. Similarly, the delineations of the groundwater system had a major influence on the components of AGWaRe (Table 2). As expected, considering the system as a whole, the components were larger than when considering the aquifer; for example, GWR_{inf} were larger compared to both $GWR_{net, aquifer}$ and $GWR_{aquifer}$. Furthermore, more water was needed for environmental purposes, $EGWR_{wq}$, when defining groundwater recharge as GWR_{inf} . For example, for a 3 m drawdown, and considering the entire groundwater system, 1344 million m^3 /year would have to be reserved to ensure water quality in the aquifers. $EGWR_{wq}$ poses a large share of the groundwater recharge (2.5 – 83%), depending on the boundaries and the sustainable drawdown. Considering $GWR_{net, aquifer}$ in the same regard, 26 million m^3 /year would be needed. The conditioned drawdown is the same, but the boundaries – and thereby the quantification of $EGWR_{wq}$ – are different. $AMD_{aquifer, ref}$ for a 3 m drawdown is 27 million m^3 /year, meaning that there is more water available within the sustainable limits.

Table 2: AGWaRe components for three system delineations of groundwater recharge (cf. Figure 2) for three conditioned drawdowns (3, 6 and 9 m) on Zealand. WU is abstraction for the conditioned drawdown, and actual abstraction is the average abstraction from 2003 to 2012. Depending on the boundaries, AMD is found for the aquifer or for the groundwater system (GWS) where GWR refer to either GWR_{inf} or $GWR_{aquifer}$.

System under assessment (delineation of GWR)	Drawdowns	WU (Mm ³)	GWR (Mm ³)	EGWR _{wq} (Mm ³)	AMD (Mm ³)	DSC
Groundwater system as a whole (GWS)	Actual abstraction (ref)	157	1593			
(GWR_{inf})	- 3m	261	1605	1344	104	2.5
	- 6m	444	1630	1186	287	1.5
	- 9m	588	1647	1060	431	1.4
Aquifer	Actual abstraction (ref)	112				
($GWR_{aquifer}$)	- 3m	139	208	69	27	5.2
	- 6m	196	239	43	84	2.3

	- 9m	320	355	35	208	1.5
Aquifer	Actual abstraction (ref)	112				
($GWR_{net, aquifer}$)	- 3m	139	165	26	27	5.2
	- 6m	196	213	16	84	2.3
	- 9m	320	328	8	208	1.5

3.4 Spatial resolution determines impact assessments

$AMD_{aquifer}$ was evaluated for different scales (Figure 5), and for the largest of these, namely Zealand as a whole (not shown), groundwater was still available and could be used within the definition of sustainable groundwater abstraction. For example, if a drawdown of 3 m were allowed, there would be 27 million m³/year available in limestone aquifers on Zealand. There were two river basins with larger actual abstraction than what was available within the conditioned 3 m drawdown (river basins 4 & 5 in Figure 5.a), indicating that actual drawdowns pose a risk of changing water quality over time in these river basins. Forty-nine out of 205 river catchment areas had a negative $AMD_{aquifer}$ (Figure 5.b). The scale of ID15 showed a large variation in $AMD_{aquifer}$ around Zealand, where 146 out of 610 ID15 catchment areas had a negative $AMD_{aquifer}$ (Figure 5.c). Evaluating the smallest scale of 500 x 500 m, more than half of the grid cells with abstraction had a negative $AMD_{aquifer}$ (Figure 5.d), and $AMD_{aquifer}$ was generally larger than AMD_{GWS} (SI. 5), which is expected, since absolute amounts of water available for use decrease downwards in the system. Scale is therefore a paramount choice in the assessment and comparison of water abstraction across regions.

Figure 5: $AMD_{aquifer}$ for limestone aquifers on Zealand, for four scales: a) river basin, b) river catchments, c) ID15 and d) model grid.

3.5 Spatial differences in the components of impact assessments and AGWaRe

There are large differences in AGWaRe components between the river basins (Table 3). For example, river basin 3 had the smallest $GWR_{aquifer, sus}$ of 2.5 million $m^3/year$, and actual abstraction was 1.0 million $m^3/year$. For a drawdown of 3 m, sustainable abstraction was 1.7 million $m^3/year$, indicating that there was 0.7 million $m^3/year$ water available for use in the river basin's limestone aquifers ($AMD_{aquifer}$). The difference between $GWR_{aquifer, sus}$ and WU_{sus} comprised the amount of water that should be left for sustaining water quality in the aquifer, $EGWR_{wq}$, which was 0.8 million $m^3/year$. River basin no. 6 had the largest $GWR_{aquifer, sus}$, the largest abstraction and, in spite of this, the largest $AMD_{aquifer}$ of 25.5 million $m^3/year$, thus indicating that this river basin was affected the least. Since AGWaRe and $AMD_{aquifer}$ is inversely related and it is based on absolute values, the river basin obtained the lowest score with the largest amount of water remaining available for other uses. This

is in contrast to relative assessments (e.g. WTA) that give the lowest score to the river basin with the largest share of water remaining for other use, in this case river basin no. 2. Two river basins (no. 4 and 5) obtained the largest scores possible of 100, and river basin no. 6 obtained the small score of 0.2 (Table 3). This indicates that groundwater stress on Zealand differs substantially, and the aquifers in two river basins were at least 100 times more stressed than aquifers on Zealand in general. Conversely, the aquifers in one river basin were five times less stressed than aquifers on Zealand in general. Similar tables for drawdowns of 6 and 9 m can be found in SI. 6. When comparing $AMD_{aquifer}$ for the different conditioned drawdowns, it is evident that $AMD_{aquifer}$ increased for all river basins when a larger drawdown is allowed. AMD_{ref} changes according to the conditioned drawdown definition, i.e. AMD_{ref} differs according to the constraints imposed on the model. Interestingly, DSC and $AGWaRe$ did not rank the river basins similarly (Table 3, SI. 6 and SI. 7), due to the difference in the numerator. In addition, DSC was relative and $AGWaRe$ represented absolute values. For River Basin no. 1, the GWR was smaller than the abstraction for sustainable conditions, resulting in a negative $EGWR_{wq}$. This does not mean that the water needed to sustain water quality is negative; rather, it shows that the definition of $GWR_{aquifer}$ applied in this study (as the flow over a vertical boundary) is simplified, since water will enter the aquifer horizontally from the sides in the model. Horizontal flow between the river basins can result in negative $EGWR_{wq}$, albeit this is not the case for the largest scale, i.e. Zealand (Table 2).

Table 3: AGWaRe parameters for limestone aquifers with a condition of a maximum 3 m drawdown for the river basins on Zealand, given per year. Actual abstraction refers to average abstraction from 2003 - 2012. $GWR_{aquifer, sus}$ is the groundwater recharge for a drawdown of 3 m, and WU is the corresponding water abstraction. Numbers have been rounded off.

River Basin	$GWR_{aquifer, sus}$	WU_{actual}	WU_{sus-3m}	$EGWR_{wq}$	$AMD_{aquifer}$	$AGWaRe$	DSC
no.	(million m ³)	(million m ³)	(million m ³)	(million m ³)	(million m ³)	(-)	(-)
1	21.7	16.3	22.3	-0.6	6.0	0.7	3.7

2	14.5	2.7	8.3	6.2	5.6	0.8	1.5
3	2.5	1.0	1.7	0.8	0.7	6.2	2.4
4	44.2	40.6	36.4	7.8	-4.2	100	100
5	17.5	19.5	12.5	5.0	-7.1	100	100
6	64.3	32.0	57.6	6.7	25.5	0.2	2.3

377

378 Considering the ID15 catchment areas, 60% of the abstracted water on Zealand comes from water-
379 scarce zones with an $AMD_{aquifer}$ smaller than zero (Figure 6.a). Approximately 30% comes from
380 aquifers in water-abundant ID15 catchment areas, with an $AMD > 10$ mm/year. Evaluating the
381 abstraction distribution and the corresponding impact on a river catchment scale or river basin scale
382 resulted in somewhat similar numbers. Comparing $AMD_{aquifer}$ for ID15 catchment areas to river
383 catchment scale and river basin scale, the aquifers in ID15 catchment areas had greater water
384 deficiency ($AMD_{aquifer} < 0$ mm/year) than river catchment areas or basins (Figure 6.b). Similarly, a
385 larger share of ID15 catchment areas had more water available for use. Thereby, evaluating
386 $AMD_{aquifer}$ for ID15 catchment areas suggested a larger redistribution of water abstraction than
387 evaluating for river basin areas. To obtain more sustainable groundwater abstraction, water
388 managers could use this knowledge when revising abstraction distribution and for planning new well
389 fields.

390

391 *Figure 6: a) Distribution of abstraction within different AMD ranges for ID15 catchments, river catchment areas*
392 *and river basins and b) the potential water abstraction that can be withdrawn before $AMD_{aquifer}$ reaches zero.*

4. Discussion

Our proposed method strives to assess groundwater sustainability, beyond the impacts on water balance, surface water systems and associated ecosystems, as covered in existing calculations for $EGWR_{flow}$ (GEUS, 2018). We attempt to evaluate potential impacts on aquifer water quality by evaluating the effects on drawdown rather than abstraction or percentages of GWR. It is a step towards obtaining the tools necessary to manage groundwater abstraction by considering groundwater quality and managing drawdown more systematically.

4.1. The influence of drawdown

Table 2 indicates that the conditioned drawdown greatly influences how much water can be withdrawn from groundwater, in that changing it from 3 to 9 m allows for doubling abstraction, thereby indicating that three times as much water is available for other uses, with an $AMD_{aquifer}$ of 104 and 431 million m^3 /year, respectively. Consequently, it is important that the implementation of a conditioned drawdown is in combination with an in-depth understanding of the impacts related to the conditioned drawdown in the studied area, e.g. changes in gradient and reduced run-off to groundwater-dependent ecosystems. Discussions on an appropriate conditioned drawdown can be found in Sl. 1 and Gejl et al. (2019).

4.2. Different indicators, different aims

Different indicators have different objectives. Whereas AWaRe, AGWaRe and AMD aim at comparing stress between different locations, *DSC* aims at indicating absolute stress at any given location. AWaRe and AGWaRe is developed for integration in LCA and therefore relates to current conditions and focuses on how they can be improved (Boulay et al., 2017; Gejl et al., 2018). All indicators evaluate the impact on the groundwater resource. In this study, we wanted to develop tools applicable in groundwater-based utilities, including for use in LCAs of their environmental impacts, and so in this regard we suggest AGWaRe and $AMD_{aquifer}$ as being particularly useful.

Current global freshwater use lies within a ‘safe operating space’ in terms of planetary boundaries, i.e. there is a low risk of human-induced destabilisation of the earth’s systems on a planetary scale (Steffen et al., 2015). Planetary boundaries constitute an important step towards sustainable use for some resources; however, for water, sustainable use differs around the world, and currently, there are many places with human-induced water scarcity (Haddeland et al., 2014; World Water Assessment Programme, 2009). *DSC* was developed to incorporate local, sustainable boundaries in the indicator.

The model iterates abstractions from confined aquifers, which for Zealand constitutes 93% of water abstraction, and the iteration does not consider water abstraction in upper aquifers. For our case, it is therefore reasonable to assume that the majority of actual impacts related to groundwater abstraction were included in our assessment. However, if a similar evaluation were conducted in areas where irrigation or other water abstractions from the top aquifers constitute a larger share of water abstraction, an analysis of the relevance of including all groundwater abstractions (including irrigation and industry) should be conducted.

4.3 Appropriate scale

The choice of scale greatly affected the results (Figure 5). The internationally recognised indicator AWaRe is an administrative demarcation consisting of a large grid too coarse for assessing relevant impacts of water utilities on Zealand (Figure 7). It indicates that the largest effects are seen in the outer parts of Zealand, and the area we would expect to be most stressed (close to Copenhagen) has the smallest impact on Zealand (Figure 7). Hence, it is difficult to use AWaRe in local assessments at the sub-national scale in the case of Denmark.

Figure 7: AwaRe characterisation factors for non-agricultural water use. Based on data from WULCA (2014).

The scales applied in this paper have their own individual advantages and disadvantages (Table 1). Between the five applied scales, the Zealand scale hides the groundwater stress that was found locally, and it cannot show which areas are more stressed than others. The river basins scale is coarser than AwaRe, but it does allow for the better representation than AwaRe of impacts in coastal areas (Figures 5 and 7); however, river catchments differ greatly in size, and so it is deemed too non-uniform for this assessment. ID15 catchment areas and model grid cells are small scales and can identify hotspots better than river basins or the AwaRe grid. Evaluating water balances within a grid scale measuring 500 x 500 m is a very detailed undertaking, and sometimes it is actually too detailed to represent realistic impacts. All of the studied scales have the disadvantage that they are based on hydrological catchments, rather than on groundwater catchments. For assessing local groundwater impacts, we suggest using a relatively small scale; for instance, in our case, the ID15 scale shows local differences and therefore can assist in the local planning, for example, of where to place a new well field, or redistributing abstraction by increasing it at one well field and reducing it at another. Using river catchment areas or ID15 catchment areas will help in indicating stress on a scale relevant for water supplies, whenever global assessments (e.g. Alcamo et al., 2003; Boulay et al., 2017; Gleeson et al., 2012) hide groundwater stress in terms of planning water withdrawal. Furthermore, ID15 and river catchment areas can be relevant in relation to gap analysis and river basin management plans (European Union, 2000). Eventually, a more homogenous subset of river

catchments more equal in size, consisting of clustered ID15 catchments in alignment with topographic and groundwater divides, could provide a stronger foundation for quantitative status assessments in future river basin management plans.

Future assessments should also consider $AMD_{aquifer}$ calculated for a 3D scale such as *groundwater bodies*. One disadvantage, though, is that groundwater bodies may overlap vertically, so it would be difficult to compare with the other scales applied. We recommend further work on delineating groundwater resources, to obtain relevant demarcations of groundwater resources and to ensure trustworthy assessments of impacts on the groundwater resource.

4.4 Defining groundwater recharge

The boundaries of groundwater recharge had a large influence on the results. Securing reliable impact assessments requires consistent and accepted delineation of groundwater processes, and for comparing independent evaluations, it is crucial to have similar delineations of the components in the impact assessment. Defining groundwater recharge as GWR_{inf} (c.f. Figure 2) gave an indication of water availability in the groundwater system but little information on availability in the aquifer. In reality, a large fraction will be drained or discharged into streams on Zealand. On the other hand, defining groundwater recharge as $GWR_{net, aquifer}$ provided concise information about water availability in the aquifer, albeit with little information of impacts on streams and surface water. Furthermore, this boundary resulted in large differences in $GWR_{net, aquifer}$ across Zealand, due, amongst other influencers, to variations in the elevation of the limestone aquifers (Figure 1). To ensure that the groundwater impact is reflecting groundwater stress, we used $GWR_{net, aquifer}$ in this study.

4.5 Vulnerability, contamination and other water quality issues

This study focused on including water quality affected by abstraction (Gejl et al., 2019). However, there could be other reasons for poor groundwater quality, such as saltwater intrusion (Ataie-Ashtiani et al., 2012), geogenic releases of nickel and arsenic (Miotliński et al., 2012; Polizzotto et al., 2008; Postma et al., 2012), the formation of chloroform compounds in forested areas (Laturnus et

al., 2000), impacts from the agricultural use of pesticides and fertilisers (Aisopou et al., 2014) and leaching from contaminated sites, for example spills at industrial production facilities, gasoline stations or old landfills (Locatelli et al., 2019). These water quality issues have not been considered in this study, but they will certainly influence where and how much water can be abstracted. Including these potential problems, we would expect that some vulnerable areas would be excluded from the analysis, due to poor water quality, and hence overall water availability would decrease. This is a relevant topic for further studies. To ensure holistic assessments, all local water quality issues or other relevant local aspects should be included, in order to obtain a complete picture.

4.6 Centralised vs. distributed groundwater abstraction

This study was based on the location of actual groundwater abstraction. If the wells were located differently, abstraction volumes corresponding to a conditioned drawdown would change, since the piezometric heads in the wells are interconnected. The approach applied in this study suggests that evenly distributed groundwater abstraction would be more sustainable concerning drawdown, because the lowering of the piezometric heads would be levelled out. This could indicate that centralised waterworks have significant impacts; however, the main point is to ensure a good distribution of abstraction between well fields. The actual physical placement of the waterworks is in reality a prioritisation between access to reliable water resources with stable water quality and costs for distribution pipes, pumping, etc. The method could encourage planners to distribute abstraction over larger areas while highlighting areas with excessive abstractions.

In theory, if we placed wells evenly distributed (in all grid cells in the model) the allowed abstraction would increase drastically within the conditioned drawdown. If we fully understand the impacts of drawdown, this would not be a problem, since the drawdown would be set to ensure stable groundwater quality; however, this is not actually feasible, due to groundwater infrastructure costs.

5. Conclusions

Our study developed and demonstrated a novel method for screening for sustainable groundwater abstraction, by managing groundwater abstraction according to a certain drawdown in confined aquifers' piezometric heads. This method has numerous application possibilities; for instance, it can be used to benchmark water utilities, it can be part of LCAs that focus on local water stress and it can be utilised by water utilities when planning for water security (European Union, 2000). The method was tested on Zealand, and it provides a viable contribution to the quantification of sustainable groundwater abstraction. Future developments should focus on the application of the method for different types of hydrogeological settings and test its use in other parts of Denmark and internationally.

We introduce a selection of new indicators for the groundwater impact assessment, e.g. 1) *DSC* that is relevant for evaluating how far groundwater abstraction is from sustainable conditions, 2) a spatially differentiated $EGWR_{wq}$ based on conditioned drawdown, which upon implementation in groundwater management will assist to ensure stable groundwater quality, and 3) *AGWaRe* and $AMD_{aquifer}$ for actual groundwater abstraction based on the novel component $EGWR_{wq}$ for use in environmental impact assessments such as LCAs.

We suggest that a spatial-scale definition similar to the Danish ID15 catchment area scale (0.07 – 105 km², median 11 km²) is currently the most practical and relevant scale for evaluating groundwater stress induced by water utilities.

Through the method applied for Zealand, it was found that for actual abstraction (2003-2012) there was an additional 27 million m³/year of water available, if a drawdown of 3 metres is allowed.

$EGWR_{wq}$ was quantified and amounted up to 83% of GWR and hence it is important to account for possible changes to groundwater quality when managing groundwater abstraction. However, actual abstraction should be redistributed in order to meet the conditioned drawdown in all the aquifers.

To achieve a positive $AMD_{aquifer}$ in all ID15 catchment areas, 41 million m³/year should be moved to

531 areas with more water availability. The methodology is a step towards obtaining tools to manage
532 groundwater abstraction, by considering groundwater quality, and to manage drawdown more
533 systematically, and it can be part of future holistic decision-making in water management scenarios,
534 in order to distribute water abstraction accordingly and thus minimise water stress.

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7. Supplementary information

SI. 1: Defining sustainable groundwater abstraction

From the study of correlations between water quality parameters and drawdown (Gejl et al., 2019), we can propose maximum allowed drawdown. The study showed that for the investigated well fields, changes in sulphate concentrations were connected to changes in water levels. There are some challenges involved in translating results from data into model characteristics, such as what constitutes pristine conditions in the real world, and how does a water level in a well translate into an average water level for a grid cell in the model? If we assume that the water level in the observation well (with a given distance away from the well field) represents the average water level for the grid cell with the abstraction well, and if we then also assume that the first measure of the water level represents pristine conditions, we can suggest, from a chosen acceptable change in sulphate concentration, a maximum lowering of the head. There are several ways to define an acceptable change in sulphate levels: 1) no change; assuming that an acceptable change in sulphate concentration is 50 mg/L, based on a correlation (in Figure 5 in Gejl et al., 2019) that indicates a maximum 3 m lowering of the head. 2) If we allow a change in sulphate concentration to 75 mg/L, we can change the piezometric head to approximately 6m. 3) If we allow for a change in sulphate concentration of 100 mg/L, we can change the piezometric head at approximately 8 m.

The well fields respond differently to abstraction, and therefore deciding on a common maximum drawdown suggests overly restrictive drawdowns for some well fields (where abstraction does not result in changing water quality) and excessively unrestricted drawdowns (in cases of well fields where water quality responds significantly to abstraction). For example, if a maximum drawdown is set to 6 m, four well fields will increase by more than 75 mg/L. One well field has a higher drawdown, without increasing the sulphate concentration to 75 mg/L. Besides the geological and hydrogeological differences between well fields, we also expect that there will be different

707 responses, depending on the degree of exploitation. We presume the effect from drawdown
708 increases, i.e. the sixth metre has a greater impact than the first metre.

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710 SI. 2: Modelled vs measured piezometric heads for 306 observation wells

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SI. 2: Modelled and observed head

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SI. 3: Comparison of observed and modelled heads for the 22 well fields

We compared the piezometric heads in 22 observation wells (Gejl et al., 2019) with the modelled head in the grid cell for the observation wells in the period 1990-2017 (Figure SI. 3.1). The model is designed to handle water balances on a catchment scale (Klint et al., 2013), therefore representing overall piezometric heads for an area or relative changes in piezometric heads rather than in a point. The piezometric head in a point is complex and can be influenced by local geological structures, and hence we expect some discrepancies between the modelled and observed piezometric heads, magnified further due to an observation well represents a point, where the model represents an area. Another aspect that can lead to discrepancies is that the DK-model is based on average abstractions per year, whereas in reality there can be large variations in actual abstractions at a well field during this time. To improve accuracy, abstraction should be improved or the model further calibrated. With the aim of understanding the broader picture, this model gives a good insight into water balances.

Figure SI. 3.1: Comparison of observed and modelled piezometric heads. The colours in the lower graph represent different well fields.

SI. 4: AMD_{GWS} for different scales

AMD_{GWS} was evaluated for different scales (SI. Figure SI. 4.1). For the largest scale, namely Zealand as a whole (not included in Figure SI. 4.1), there was still groundwater available that could be used within the definition of sustainable groundwater abstraction. For example, if a drawdown of 3 m were allowed, there was 15.6 mm/year available on Zealand (137 MmMm³). All river basins had water available, ranging from 6 to 48 mm/y (Figure SI. 4.1.a). Evaluating for river catchment, 35 out of 204 had a negative AMD_{GWS} (Figure SI. 4.1.b), which indicates that actual drawdowns pose a risk of changing water quality over time in these river basins (Gejl et al, 2019). The scale of ID15 showed a large variation in AMD_{GWS} around Zealand, with 102 out of 610 ID15 catchment areas with a negative AMD_{GWS} (Figure SI. 4.1.c). Evaluating the smallest scale of 500 x 500 m, almost half of the grid cells with abstraction had a negative AMD_{GWS} (Figure SI. 4.1.d).

Figure SI. 4.1: AMD_{GWS} for limestone aquifers on Zealand, for four scales: a) river basin, b) river catchments, c) ID15 and d) model grid.

SI. 5: The difference in groundwater abstraction and groundwater recharge between actual abstraction and the scenario of 3 m drawdown

Figure SI. 5.1: a) Abstraction and b) $GWR_{net, aquifer}$ for c) Abstraction and d) the difference in $GWR_{net, aquifer}$ between the two scenarios 3 m drawdown and actual abstraction.

SI. 6: AGWaRe parameters for the conditioned drawdown of 6 and 9 m for river basins

Table SI. 6.1: Yearly AGWaRe parameters evaluated for the aquifer with a condition of a maximum 6 m drawdown for the river basins on Zealand. Actual abstraction refers to average abstractions from 2003 – 2012.

River Basin no.	$GWR_{\text{aquifer, sus}}$ (Mm ³)	WU_{actual} (Mm ³)	WU_{sus} (Mm ³)	$EGWR_{\text{wq}}$ (Mm ³)	AMD_{aquifer} (Mm ³)	AGWaRe (-)	DSC (-)
1	40.1	16.3	41.1	-1.0	24.8	0.8	1.7
2	20.4	2.7	14.8	5.6	12.1	1.7	1.2
3	3.6	1.0	3.3	0.3	2.3	8.8	1.4
4	70.4	40.6	64.2	6.2	23.6	0.9	2.7
5	24.5	19.5	24.0	0.5	4.4	4.5	5.4
6	94.0	32.0	90.2	3.8	58.1	0.3	1.6

Table SI. 6.2: Yearly AGWaRe parameters for the aquifer with a condition of a maximum 9 m drawdown for the river basins on Zealand. Actual abstraction refers to average abstractions from 2003 - 2012.

River Basin no.	$GWR_{\text{aquifer, sus}}$ (Mm ³)	WU_{actual} (Mm ³)	WU_{sus} (Mm ³)	$EGWR_{\text{wq}}$ (Mm ³)	AMD_{aquifer} (Mm ³)	AGWaRe (-)	DSC
1	54.8	16.3	57.2	-2.5	41.0	0.8	1.4
2	24.8	2.7	20.5	4.4	17.8	2.0	1.2
3	4.7	1.0	4.9	-0.2	3.9	9.0	1.3
4	93.1	40.6	89.1	4.0	48.6	0.7	1.8
5	33.0	19.5	35.4	-2.4	15.8	2.2	2.2
6	117.3	32.0	113.1	4.2	81.1	0.4	1.4

761 SI. 7: DSC on different scales

762 DSC was evaluated for different scales (Figure SI. 7.1).

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764 Figure SI. 7.1: DSC for limestone aquifers on Zealand, for four scales: a) river basin, b) river catchments, c) ID15
765 and d) model grid.

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