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Water4Coasts
Nye metoder til integreret forvaltning og beskyttelse af kystnære vandressourcer

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Må citeres med kildeangivelse.

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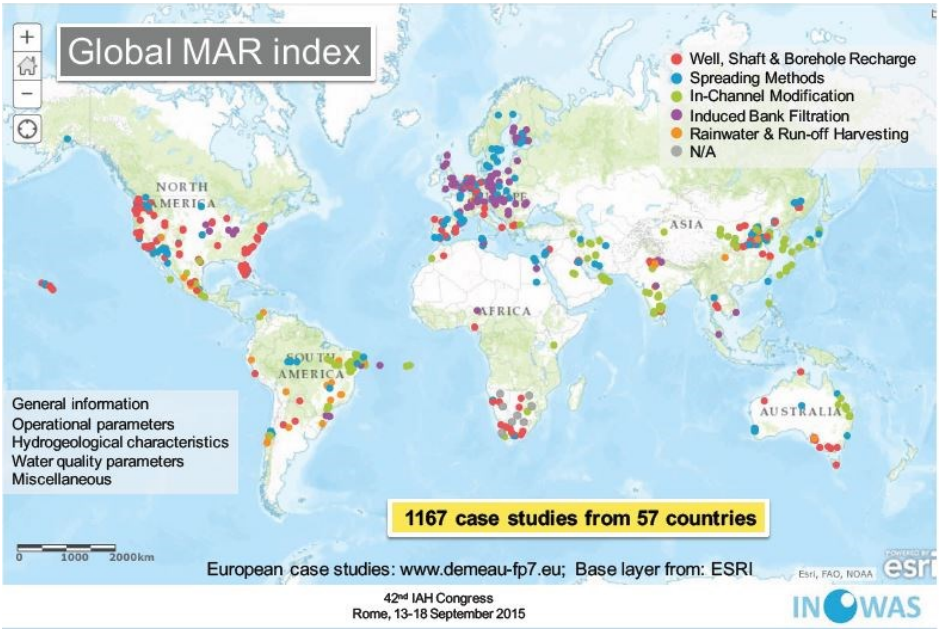
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Forord

Kystnære grundvandsmagasiner og vandressourcer er i stigende grad truet og påvirket af saltvandsindtrængning fra havet og fra gamle havaflejringer, især i tætbefolkede områder med relativ stor vandindvinding fra grundvandsmagasinerne (Post and Abarca, 2010; Hinsby et al., 2011; Tran et al., 2012).

Problemer med saltvandsindtrængning er især kendt fra grundvandsmagasiner langs kysterne i USA, Spanien, Holland og den sydlige Nordsøkyst generelt (Hinsby et al., 2011), men problemerne med saltvandsindtrængning, og anvendelse af nye tekniske løsninger til at kontrollerede den, er stigende globalt i alle klimazoner og inkluderer eksempelvis kystnære grundvandsmagasiner i Brasilien (da Silva et al., 2010), Kina (Xue et al., 2000) og Danmark (Rasmussen et al., 2013). Water4Coasts projektet indleder samarbejde mellem Kina, Brasilien og Danmark vedrørende udvikling og evaluering af nye metoder til at modvirke saltvandsindtrængning og sikre tilstrækkeligt ferskvand til drikkevand og markvanding i kystzoner.

Der er et stigende behov for at udvikle effektive og forbedrede metoder til at modvirke saltvands-indtrængning samt automatiserede metoder til at overvåge og kontrollere effekterne af disse globalt, ikke mindst i betragtning af de forudsete klimaændringer og et stigende havniveau (IPCC, 2014, Hansen, 2015). Figur 1 herunder viser den nuværende globale anvendelse af kontrolleret grundvandsdannelse (MAR = Managed Aquifer Recharge) blandt andet til kontrol af saltvandsindtrængning ved direkte injektion af ferskvand i boringer i kystnære grundvandsmagasiner (de røde cirkler på eksempelvis øst og vestkysten af USA og østkysten af Spanien).



Figur 1

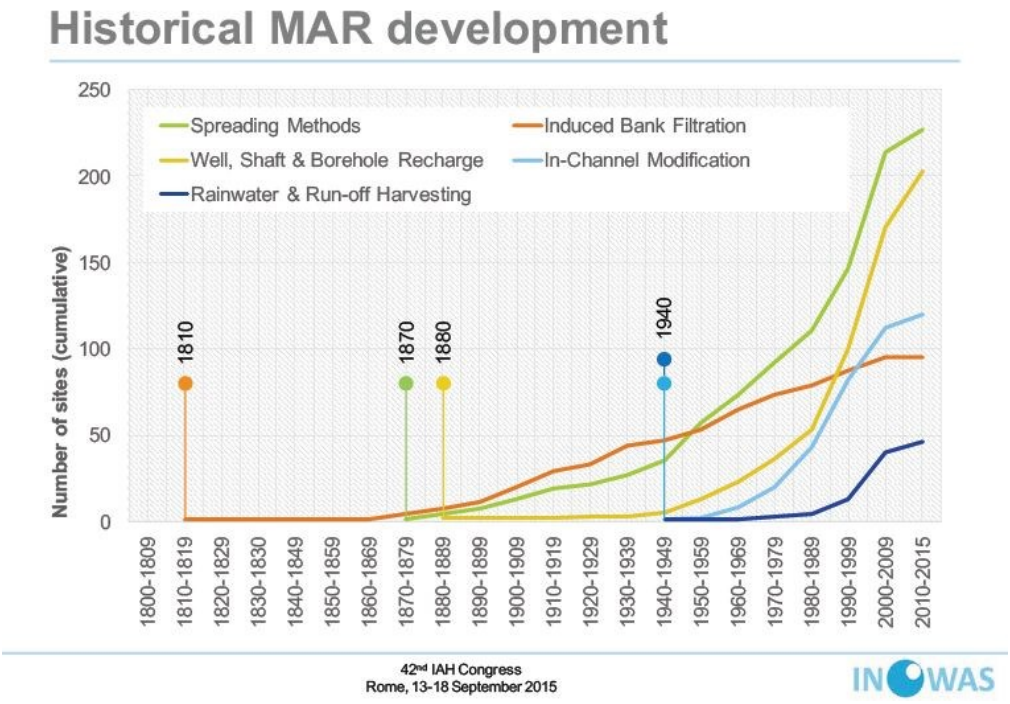
Global oversigt over anvendelsen af forskellige typer af kontrolleret grundvandsdannelse (Managed Aquifer Recharge / MAR). kilde: Stefan, C. (2015).

Sammenfatning

Kontrolleret grundvandsdannelse eller Managed Aquifer Recharge (herefter blot MAR) er i stærk stigning globalt (Figur 2), og der er et stort behov for at disse løsninger anvendes i stigende omfang mange steder i verden (de Marsily, 2015).

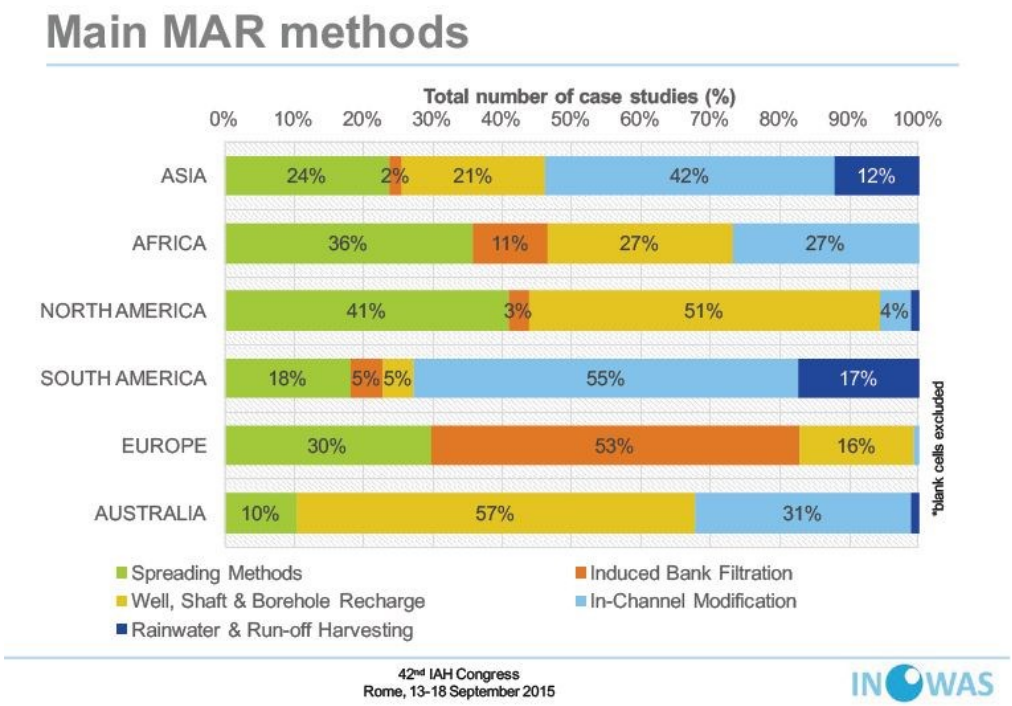
Der er meget begrænset erfaring med kontrolleret grundvandsdannelse i Danmark, og hvis dansk vandindustri, -forvaltning, - forskning og konsulentvirksomhed fortsat skal være blandt de førende i verden indenfor forvaltning af grundvandsressourcer generelt, er der behov for at danske virksomheder og forskningsinstitutioner også kan levere produkter, metoder, forskning og konsulentvirksomhed indenfor anvendelse af disse teknikker. Water4Coasts projektet søger at initiere denne proces.

Følgende figurer viser den stærkt stigende anvendelse af forskellige MAR-typer i globalt perspektiv, samt anvendelse af og formål hermed i forskellige verdensdele.



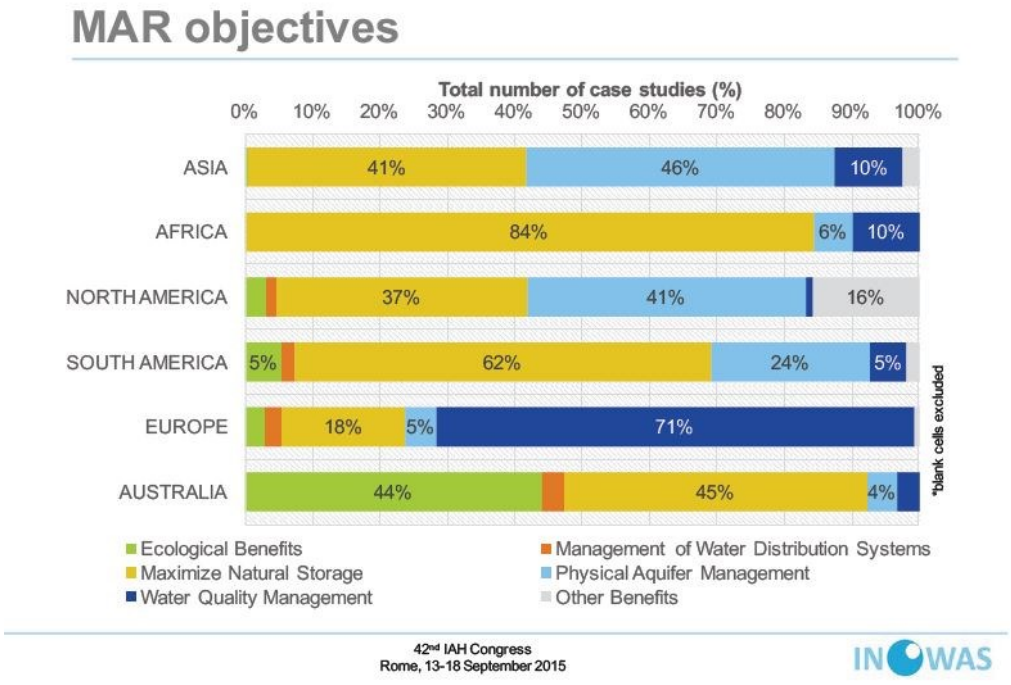
Figur 2

Den historiske globale udvikling i anvendelsen af de forskellige MAR metoder 1800-2015. kilde: Stefan, C. (2015).



Figur 3

Anvendelsen af MAR metoder i forskellige verdensdele. kilde: Stefan, c. (2015).



Figur 4

Formål med anvendte MAR løsninger i forskellige verdensdele. Kilde: Stefan, c. (2015).

MAR metoderne har som tidligere nævnt kun været undersøgt og anvendt i Danmark i meget begrænset omfang. Water4Coasts projektet søger at styrke dansk viden og metodeudvikling indenfor MAR, da der er behov for at gennemføre danske demonstrationsprojekter af MAR til kontrolleret grundvandsdannelse i områder, hvor der er eller kan blive brug for MAR løsninger til at sikre og beskytte vandindvinding og økosystemer; eksempelvis i storbyer (større byområder) og i kystområder med stærk sæsonbetinget indvinding til turisme eller landbrug og gartneri - både i og udenfor Danmark.

Behovet for at anvende disse metoder har hidtil ikke været påtrængende for Danske forhold, men med de forventede klimaændringer og kravene i EU's Vandramme- og Grundvandsdirektiver mm. om beskyttelse af grundvandets kvantitative og kemiske tilstand, herunder grundvandets saltholdighed, både i forhold til beskyttelse af drikkevandsressourcen og økosystemer (European Commission, 2015c), må det forventes at behovet herfor øges i fremtiden.

Water4Coasts projektet viser, at der eksempelvis på Marielyst, Sydfalster kan være gode muligheder for at anvende MAR løsninger til at beskytte drikkevandsressourcen mod saltvandsindtrængning. Der er dog en lang række problemstillinger af både teknisk, socio-økonomisk og politisk karakter, der skal undersøges nærmere og løses, før egentlige demonstrationsprojekter og fuld skala implementering kan gennemføres. Nogle af de væsentligste problemstillinger er belyst i Water4Coasts og beskrives kort herunder.

Resultater fra de gennemførte Water4Coasts undersøgelser viser:

1) Den kalibrerede grundvandsmodel for forsøgsområdet og modelkørsler hermed viser, at MAR løsninger kan være en relevant mulighed for at kontrollere saltvandindtrængning både fra kysten ("nyt havvand") og fra dybereliggende kridtlag ("gammelt havvand"). Modelsimuleringerne viser at nye Hollandske teknikker med anvendelse af horisontale injektions og pumpeboringer (Zuurbier et al., 2015), synes at være de mest effektive til kontrollering af saltvandsindtrængning (Rasmussen et al., 2015). Modelresultaterne og metodernes anvendelse og effekterne heraf, er dog meget afhængig af de hydrauliske og kemiske forhold i undergrunden og grundige og yderligere undersøgelser heraf er af afgørende betydning for vurdering af metodernes potentiale inden fuld-skala anlæg projekteres og installeres. Der er således stadig stor usikkerhed omkring de hydrauliske forhold i det delvist opsprækkede kridtmagasin på Falster, der varierer meget pga. påvirkninger fra isbevægelserne under sidste istid jfr. de glacialtektoniske processer, der blandt andet medførte dannelse af Møns Klint, og randmorænen i den vestlige del af Sydfalster (den vestlige del af forsøgsområdet på Marielyst). De hydrauliske og kemiske forhold af kridtbjergarten, der udgør grundvandsmagasinet på Marielyst, skal derfor undersøges nærmere før optimal design af et pilotanlæg kan udvikles. Ligesom behovet for vandrensning og kvalitetskravene til injektionsvand fra Nordkanalen eller andre relevante kilder (f.eks. andre indvindingsboringer) til injektion i og omkring indvindingsboringer skal fastlægges.

2) Undersøgelserne af vandkvalitet i Nordkanalen mellem rensningsanlæg og pumpestation på Marielyst / Bøtø på Sydfalster samt i grundvandsmagasiner i Kridtbjergarterne, hvorfra drikkevandet indvindes i dybder mellem 10 og 25 meter under terræn, viser at der er risiko for forurening og negative effekter på vandkvaliteten, hvis vand fra Nordkanalen injiceres i grundvandsmagasiner uden forudgående omfattende rensning af vandet fra kanalen. Forurening i kanalen omfatter udover pathogener både uorganiske og organiske forureninger fra landbrug (nitrat og pesticider mm.), industri (PFOS etc.) og husholdning (primært lægemidler). Den største gruppe af potentielle forurenende stoffer udgøres muligvis af lægemidlerne. I en screening af ca. 90 lægemidler på udledningen fra det lokale rensningsanlæg, er der fundet 35 lægemidler i målbare koncentrationer, og heraf er 19 over grænseværdien for pesticider på

0,1 µg/l i drikkevand (der findes p.t. ingen grænseværdi for lægemidler i drikkevand). Samlet set er der målt lægemidler i samlet koncentration før og under turistsæsonen på hhv. 17 og 25 µg/l. Udledningerne fra rensningsanlægget fortyndes, så der i Nordkanalen ikke måles lægemidler over grænseværdien for pesticider, men de udgør en risiko (om end lille) for vandkvaliteten i grundvands-magasinerne. Problemer med lægemidler i grundvand kendes især fra f.eks. Tyskland og Spanien, eksempelvis hvor grundvand indvindes fra boringer tæt på floder ved 'induced bank infiltration', der er meget anvendt i Europa (Figur 3).

Udover problemer med forureninger fra det injicerede overfladevand, kan der opstå problemer med vandkvaliteten ved frigørelse af sporelementer som f.eks. arsen og nikkel ved oxidation af pyrit ved ilt og nitrat i grundvandsmagasinerne. Oxidation af jernmineraler (f.eks. pyrit) i grundvandsmagasinet kan desuden skabe tekniske problemer ved tilstopning af filtre og sprækker i kildebjergarten, så boringernes ydelser sænkes markant. Begge problemstillinger har dog vist sig at kunne håndteres hensigtsmæssigt i flere eksisterende anlæg i U.S.A.

Simulering af kemiske processer ved injektion af ilt- og nitratholdigt vand, viser at effekten er meget afhængig af Kridtmagasinets hydrauliske egenskaber. Opfører kridtmagasinet sig primært som et dobbeltporøst sprækket magasin hindrer den langsomme diffusion fra kridtets matrix til sprækkerne effektiv adgang til de reducerende stoffer i bjergarten (primært organisk stof og pyrit), og dermed effektiv reduktion af ilt og nitrat. Foreløbige analyser af kridtbjergarten viser meget lave koncentrationer af pyrit (0,001 – 0,03 %) og organisk stof. Dette er en ulempe såfremt man ønsker at reducere nitrat i grundvandsmagasinet, men en fordel i forhold til at beskytte grundvandets mod sporelementer som nikkel og arsen, der kan frigøres fra pyrit, om end disse normalt sorberes på udfældede jernoxider.

Ovenstående problemstillinger omkring sundhedsmæssige og tekniske aspekter af påvirkning af vandkvaliteten ved injektion af potentielt forurenet ilt- og nitratholdigt overfladevand i iltfrie grundvandsmagasiner medfører, at der ofte vil være behov for effektiv vandbehandling inden injektion af overfladevand i grundvandsmagasinet.

Nye og mere omkostningseffektive rensningsmetoder gør dog i mange tilfælde metoderne rentable under mange forskellige forhold i globalt perspektiv. Således injiceres der mange steder renset overfladevand (eller endda renset spildevand) i grundvandsmagasiner eksempelvis i kystzoner i perioder med stor nedbør og lavt vandforbrug (vinterperioden) til oplagring og senere forbrug i perioder med lille nedbør og stort vandforbrug til f.eks. markvanding og turisme (sommerperioden). Grundig vurdering af fordele og ulemper (herunder cost-benefit analyser) ved kontrolleret grundvandsdannelse og eventuelle alternativer, bør altid gennemføres inden anlæg til kontrolleret grundvandsdannelse projekteres.

3) Operationelle MAR systemer som eksempelvis udviklet og anvendt af det Hollandske KWR Watercycle Research Institute (eg. Zuurbier et al, 2014, 2015) vurderes at have et stort globalt potentiale til beskyttelse af kystnære grundvandsressourcer herunder nogle kystnære danske vandressourcer. Metoderne udviklet af KWR stiller krav til effektiv og automatiseret kontrol af effekten af metoderne vha. automatiserede SRO anlæg for at driften af disse og kontrollen af saltvandsindtrængning kan optimeres – blandt andet ved at anvende 'on-line' klorid eller ledningsevne sonder til nær realtids måling og overførsel af data om grundvandets saltholdighed. Water4Coasts projektet initierede udviklingen af måle-systemer, der er fundamentet i sådanne anlæg, men mødte problemer med stabiliteten af både måle-elektroder og dataloggere, der ikke kunne løses endeligt indenfor projektet; problemerne er beskrevet nærmere i rappor-

ten ”Real time groundwater salinity monitoring” (Hinsby et al., 2015b). Løsninger herpå og systemer til styring og kontrol af saltvandsindtrængning undersøges yderligere i det netop initierede EU projekt ”SUBSOL” (European Comission, 2015a; KWR, 2015).

Udover formidling i de citerede rapporter og konference-bidrag nævnt i nærværende rapport, har Water4Coasts projektet bidraget til on-line videoformidling via
www.undergroundchannel.dk
videoerne kan ses på følgende to link:
http://www.undergroundchannel.dk/water4coasts-1 ,
http://www.undergroundchannel.dk/water4coasts-2

samt til en igangværende udarbejdelse af videobaseret undervisningsmateriale til folke- og gymnasieskoler om bæredygtig vandforvaltning i kystzoner, og en informationsvideo udarbejdet til organisationskomitéen bag ’Salt Water Intrusion Meetings’ fra det senest afholdte møde i Husum, Tyskland i 2014 - http://www.swim23.com/, der i øjeblikket udarbejdes af Alphafilm & Kommunikation (www.alphafilm.dk).

Summary

This report is a summary report in Danish of the results and conclusions of the “Water4Coasts” project– ‘New methods for integrated management of coastal water resources’. The project is part funded by the Danish Eco-innovation program of the Ministry of Environment and Food of Denmark (www.ecoinnovation.dk).

The project results and conclusions can be summarised as follows:

1) Simulations by the use of a density dependent groundwater flow model partly calibrated by airborne and borehole geophysical measurements (Rasmussen et al., 2013), indicate that injection of treated fresh surface waters into water wells by the use of MAR = managed aquifer recharge would be a feasible option to control saltwater intrusion and protect fresh groundwater resources on Southern Falster, Denmark. However, additional studies are required to ensure that the injected water does not jeopardize the high water quality in the exploited Chalk aquifer, and to improve the understanding of the hydraulic conditions in the Chalk. The simulations show that horizontal well configurations for both injection and abstraction could be the most efficient in controlling salt water intrusion (Rasmussen et al, 2015). However, the cost of these are still considerably higher than standard vertical wells, and the costs and benefits of both systems should be evaluated before finally deciding, which system to use.
2) Assessment of the water quality in drainage canals and groundwater show that highly efficient water treatment are needed in order to avoid the risk of contaminating the high quality groundwater in the investigated Chalk aquifer with contaminated surface waters from the drainage canal (Hinsby et al., 2015a). The required treatment probably has to include reverse osmosis in combination with other techniques in order to efficiently remove especially pathogens and organic contaminants. Furthermore, the impact of redox reactions occurring when injecting oxic groundwater into anoxic aquifers has to be evaluated thoroughly. Preliminary results show rather low concentrations of pyrite, organic matter and trace elements in the Chalk aquifer, indicating that the release and migration of trace elements probably are not a major concern for the investigated Chalk aquifer. However, this has to be confirmed by more detailed in situ studies of the aquifer.
3) Operational MAR systems such as those developed and applied by the Dutch KWR Watercycle Research Institute (e.g. Zuurbier et al., 2014, 2015) are considered to have a large potential globally, and they are also believed to be relevant solutions for protection of some Danish drinking water resources. Such solutions needs efficient automated pump monitoring and control systems (‘SCADA’) etc. using e.g. on-line electrical conductivity or chloride sensors. The Water4Coasts project initiated the development of such systems, but faced some technical problems with the long-term stability of the electrodes and data-logger systems. These issues are described in further detail in the report by Hinsby et al. (2015b) and the solution to these will be pursued in ongoing projects.

More details on the conducted investigations and the obtained results can be found in the three reports (Hinsby et al., 2015a,b, Rasmussen et al., 2015) presenting the preliminary assessment

of the use of managed aquifer recharge to control saltwater intrusion in the Marielyst area of the Falster island.

The investigations on the use of managed aquifer recharge and other subsurface solutions initiated on Falster, Denmark in the Water4Coasts project is currently continued in the 'SubSol' project: bringing coastal subsurface water solutions to the market (European Commission, 2015a) in 'Horizon 2020' the research and innovation program of the European Commission (European Commission, 2015b).

1. Introduktion

Udover saltvandsindtrængning står vandforvaltningen i kystzoner og kystnære vandløbsoplande overfor andre betydelige udfordringer. Udfordringerne varierer indenfor de forskellige klimazoner, men inkluderer: 1) Øget risiko for oversvømmelser (både fra havet samt vandløb og grundvand), 2) Risiko for øget udvaskning af næringsstoffer og andre forureninger til grundvandstilknyttede økosystemer og 3) Stigende antal tørker og faldende vandspejl især i aride og semi-aride områder.

Ad 1. Klimaændringer forventes at påvirke både vandets kvantitet og kvalitet i hele det hydrologiske kredsløb og øge risikoen for oversvømmelser ved kysten såvel som i baglandet (IPCC, 2015, Sonnenborg et al., 2012), samt øge udvaskning af næringsstoffer (Sonnenborg et al., 2012; Hinsby et al., 2012).

Ad 2. Kvælstofbelastningen af det hydrologiske kredsløb og akvatiske økosystemer (European Commission, 2015c) er et af de største miljøproblemer i globalt perspektiv (Rockström et al., 2009), og de indre danske farvande og Østersøen er blandt de mest belastede i verden (Diaz and Rosenberg, 2008; Conley, 2012). Blandt andet derfor har hovedparten af det øvre iltholdige grundvand i Danmark dårlig kemisk tilstand ifølge EU direktiver og hertil hørende tekniske retningslinjer (Hinsby et al., 2008, 2012). Det sætter Danmark under stort pres for at reducere udvaskningen af næringsstoffer betydeligt.

Water4Coasts projektet sigter mod at udvikle nye innovative metoder til integreret forvaltning af kystnære vandressourcer, som tager hensyn til tilstanden i såvel grundvand som overfladevand og tilknyttede økosystemer, baseret på kontrolleret grundvandsdannelse (”managed aquifer recharge”).

Water4Coasts projektet vurderer hvorvidt injektion af renset overfladevand fra f.eks. drænkanaler til anaerobe dele af grundvandsmagasiner via injektionsboringer kan forventes at være en effektiv metode til modvirkning og kontrol af saltvandindtrængning. Ifølge vurderingerne kan der ved metoden opnås flere gunstige effekter af MAR anvendelse i det danske forsøgsområde på Falster, hvor indvindingsboringer i et Kridt-magasin er påvirket af saltvandindtrængning (Rasmussen et al., 2013) og i områder med lignende problemstillinger.

Ved at injicere overfladevand i injektionsboringer kan der opnås flere positive effekter: 1) Der opbygges en hydraulisk barriere i grundvandsmagasinerne som modvirker yderligere saltvandsindtrængning 2) Et eventuelt nitrat-indhold i det injicerede vand reduceres til N2, og kvælstofbelastningen af kystvande og marine økosystemer mindskes (risikoen for eutrofiering reduceres) og 3) Risikoen for oversvømmelser fra drænkanaler reduceres, hvis der anvendes renset vand fra drænkanalerne til injektion i grundvandsmagasinerne.

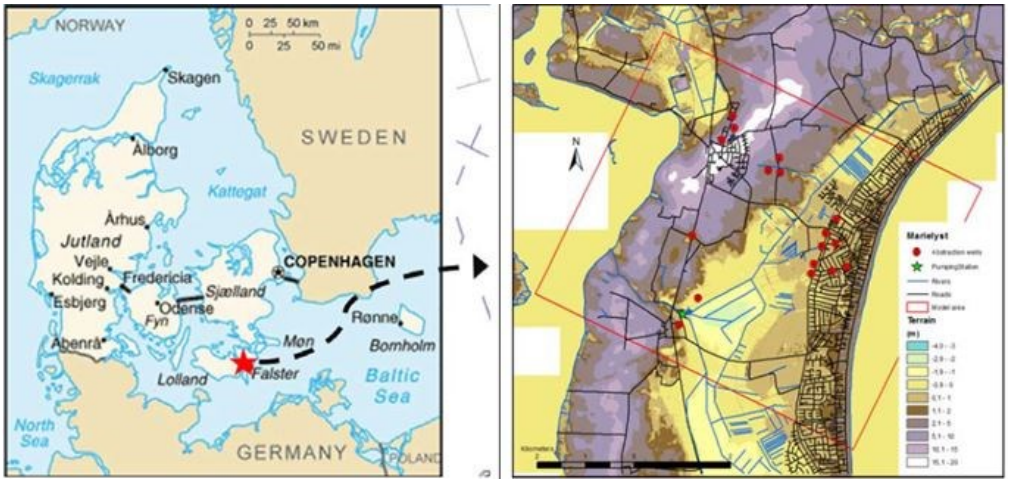
Undersøgelserne som er gennemført i Water4Coasts projektet indikerer at MAR løsninger er relevant for sikring af drikkevandsforsyning i kystnære områder og i områder med dybere liggende gammelt havvand i grundvandsmagasinerne, og at der er behov for at gennemføre yderligere undersøgelser og demonstrationsprojekter for at der kan opnås praktisk erfaring med forskellige MAR løsninger i Danmark til beskyttelse af drikkevandsressourcer i kystnære områder både inden- og udenfor landets grænser.

2. Grundvandsmodeller til evaluering af MAR og saltvandsindtrængning

Water4Coasts – aktivitet 1.1 og 2.2.

Densitetsafhængige grundvandsmodeller er et nødvendigt værktøj til evaluering af effekterne af MAR og hydrauliske barrierer mod saltvandsindtrængning både i forhold til risikoen for indtrængning fra havet i kystzonen og fra dybereliggende saltholdige grundvandsmagasiner ved forskellige konstellationer af pumpe- og injektionsboringer.

Water4Coasts projektet har opstillet og kalibreret en grundvandsmodel delvist på baggrund af geofysiske målinger (luftbåren SkyTEM, og borehulslogging) og geokemiske analyser af grundvandet indsamlet i Marielyst området på Falster (Rasmussen et al., 2013, 2015) og brugt denne model til at analysere problemstillingerne omkring og effekterne af MAR i projektområdet på Marielyst, Sydfalster (Figur 5, Rasmussen et al., 2015).



Figur 5

Water4coasts projektområdet på Marielyst, Sydfalster.

Detaljerede resultater og undersøgelser beskrives i hovedrapporten af Rasmussen et al. (2015). Hovedkonklusionerne af de gennemførte undersøgelser kan resumeres som følger:

Densitetsafhængige grundvandsmodeller er et vigtigt redskab til evaluering af effekten af MAR og etableringen af hydrauliske barrierer mod saltvandsindtrængning.

Luftbåren, overflade og borehuls geofysiske målinger og grundvandskemiske analyser i grundvandsmagasinerne kan forbedre modelopsætningerne betydeligt.

De gennemførte modelsimuleringer viser at horisontale boringer (Zuurbier, 2015) er mest effektive til at kontrollere saltvandsindtrængning. Simuleringerne viste eksempelvis at injektion af blot en tredjedel af vandmængden injiceret i tre vertikale boringer i en 150 m lang horisontal

boring var mere effektiv som hydraulisk barriere mod saltvandsindtrængning end barrieren etableret ved de tre vertikale boringer.

Simuleringer med forskellige boringskonstellationer viser at injektionsmængden har størst og mest afgørende betydning for effekten af injektionen.

Der bør etableres moniteringsboringer til kontrol af effekten af injektionerne med veldefinerede filtre i både ferske og salte dele af grundvandsmagasinerne, således at de enkelte filtre ikke placeres i både ferske og saltholdige dele af magasinet.

I Bilag 2 findes en mere detaljeret beskrivelse og illustration af modelopsætninger og modelresultater.

3. Vandkvalitetsvurderinger

Water4Coasts - aktivitet 1.3

Vandkvaliteten i grundvandsmagasinerne og det injicerede overflade vand er af afgørende betydning for kemiske processer, potentiel forurening og eventuel clogging (tilstopning) af filtre i injektions- og indvindingsboringer ved MAR løsninger (Martin, 2013).

Water4Coasts projektet evaluerede disse gennem prøvetagning og analyse af vandprøver indsamlet i drænkanaler og rensningsanlæg (figur 6, Hinsby et al., 2015a), og vurdering af grundvandskvalitet fra prøver indsamlet og analyseret i Marielyst Vandværks 10 indvindingsboringer i et tidligere projekt.



Figur 6

Luftfoto af undersøgelsesområde. de blå linier angiver gravede drænkanaler anlagt efter begyndelsen af landvindingsprojektet i 1863. Hoveddrænkanalen (”Nordkanalen”), som ses mellem renseanlægget (WWTP) i øverste del af fotoet og dræn pumpestationen i nederste venstre hjørne samt tilløb hertil (inkl. fra renseanlæg), blev prøvetaget til analyse af vandkvalitet før og under turist sæsonen i 2013.

Prøvetagningssteder til analyse af vandet i drænkanaler samt rensningsanlæg er vist herover. Resultaterne fra prøverne viser ingen forureninger af opløste organiske (på nær PFOS) eller uorganiske forureninger, der overstiger grænseværdier i hoveddrænkanalen. Analyser af det rensede vand i rensningsanlægget udtaget umiddelbart inden udledningen til Nordkanalen viser imidlertid relativt høje værdier for en lang række lægemidler (19 overstiger grænseværdien for pesticider på 0,1 µg/l). Herudover, er der målt et forhøjet indhold af fækale bakterier i to prøver i hovedkanalen nedstrøms renseanlægget muligvis fra overløb på rensningsanlægget eller sivebrønde i sommerhusområdet.

3.1 Uorganisk vandkemi i Nordkanalen mellem renseanlæg og pumpestation

Der blev udtaget vandprøver til analyse af uorganisk vandkemi i Nordkanalen på stationerne W4C-1, W4C-4 og W4C-6 hhv. ved renseanlægget, ca. midtvejs mellem renseanlægget og pumpestationen, og ca. 200 m før tilløbet til pumpestationen (se figur 6) i august og december 2013 for at illustrere årstidsvariationer i den uorganiske vandkemien i Nordkanalen

Tabel 1

ANALYSER AF VANDKEMI FRA PRØVER I NORDKANALEN UDTAGET MELLEM RENSEANLÆG OG PUMPESTATION UDTAGET I SOMMERPERIODEN (14. AUGUST 2013). ENHED MG/L

Station	Na	K	Mg	Ca	Cl	Br	NO3	PO4	SO4	HCO3	TOC	DOC
W4C-1	113	35	18	93	170	0,54	2,17	0,21	45	218	11	11
W4C-4	105	20	16	74	161	0,82	0,49	0,44	37	161	13	13
W4C-6	134	22	21	84	189	1,26	0,18	0,32	43	216	16	16
Middel	117	26	18	84	173	0,87	0,95	0,32	42	198	14	13

Tabel 2

ANALYSER AF VANDKEMI FRA PRØVER I NORDKANALEN UDTAGET MELLEM RENSEANLÆG OG PUMPESTATION UDTAGET I VINTERPERIODEN SAMT ET DRÆN LOKALISERET CA. 300 M OPSTRØMS W4C-6 (17. DECEMBER 2013). ENHED MG/L.

Station	Na	K	Mg	Ca	Cl	Br	NO3	PO4	SO4	HCO3	TOC	DOC
W4C-1	113	10	24	132	181	1,21	21	0,2	113	172	12	13
W4C-4	92	11	21	124	151	0,95	19	0,17	109	153	13	10
W4C-6	86	12	20	127	142	0,95	24	0,16	120	147	14	12
Middel	97	11	22	128	158	1,04	21	0,18	114	157	13	12
Dræn1	86	21	33	211	81	0,29	74	0,03	200	326	5,4	5,2

Den mest i øjnefaldende forskel på prøverne i Nordkanalen mellem sommer og vinter er de markant forhøjede nitrat og sulfat koncentrationer i vinterperioden. Forskellene skyldes primært landbrugsudvaskning i dræn i vinterperioden, hvor der er markant forhøjede Ca, nitrat (NO3), og sulfatkoncentrationer (SO4) i drænvandet (se analyserne for dræn1).

I vinterperioden har landbrugsaktiviteterne og vandkvaliteten i rørlagte dræn, der strømmer til Nordkanalen således en relativ stor betydning for vandkemien i Nordkanalen, hvorimod udstrømning fra rensningsanlægget har relativ stor betydning for vandkemien i sommerperioden, hvor det store antal sommerhusturister medfører øget tilstrømning til Nordkanalen af renset spildevand og forhøjet koncentration af phosphat (PO4) og kalium (K).

Tabel 3 herunder viser vandkemien fra 8 indvindingsboringer for Marielyst vandværk, hvoraf de to (242.319 og 242.320) er lokaliseret nord for Nordkanalen (kildeområde 3 etableret i 2005), mens de 6 andre er lokaliseret syd for Nordkanalen. Boring 242.44B og 242.178 hører til kildeområde 1, der blev etableret i 1950'erne mens boring 242.182, 242.212, 242.230 og 242.232 hører til kildeområde 2, der blev etableret i 1970'erne.

Analyserne viser at grundvandet har en høj koncentration af bikarbonat pga. kalkopløsning kridtmagasinet. Herudover har boringerne intet nitrat (og ilt) indhold, da der er tale om en reduceret kemisk miljø, hvor ilt og nitrat reduceres i nævnte rækkefølge af reducerende mineraler (pyrit) og organisk stof i kridtbjergarten.

Tabel 3

GRUNDVANDSKEMI FOR 8 AF MARIELYST VANDVÆRKS INDVINDINGSBORINGER. ENHED MG/L. BORINGERNE I KILDEFELT 1 LIGGER NÆRMEST ØSTERSØKYSTEN MOD ØST OG HAR DE HØJESTE KLORIDKONENTRATIONER

Boring	Na	K	Mg	Ca	Cl	Br	NO3	PO4	SO4	HCO3	Kildefelt
242.44B	165	7,3	31	127	270	1,16	<0,1	<0,05	7,8	420	1
242,178	147	7,2	37	124	257	1,00	<0,1	<0,05	10	425	1
242,182	20	4,3	21	104	31	0,125	<0,1	<0,05	6,6	378	2
242,212	82	5,9	30	127	185	0,66	<0,1	<0,05	10	396	2
242,230	51	5,0	28	122	104	0,37	<0,1	0,059	10	423	2
242,232	75	6,3	30	129	155	0,51	<0,1	<0,05	9,1	450	2
242,319	20	3,6	18	108	40	0,125	<0,1	<0,05	19	354	3
242,320	17	4,2	18	101	32	0,125	<0,1	<0,05	6,6	350	3
Middel	72	5	27	118	134	0,51	<0,1	<0,05	10	399	

3.2 Medicinrester i drænvand og renset spildevand.

Der er ikke målt forhøjede koncentrationer af lægemidler i drænvand. Derimod er der målt relativt høje koncentrationer af visse typer af hjertemedicin (f.eks metoprolol og Furosemide), psykofarmaka (f.eks Citalopram), smertestillere (f.eks Tramadol) og antibiotika (f.eks. Thimetoprim). Der er således 35 lægemidler med koncentrationer over detektionsgrænsen og en samlet koncentration på mere end 17 µg/l – eller 34 gange over grænseværdien for det samlede tilladte pesticid-indhold (Hinsby et al., 2015). Forurening af grundvandet med lægemidler er således en potentiel risiko ved injektion eller infiltration af overfladevand i grundvandsmagasiner.

3.3 Industrikemikalier

Der blev målt Bisphenol-A, D-n-butylphthalate og PFOS over detektionsgrænsen i prøver fra rensningsanlægget og/eller i Nordkanalen. Der er kun målt koncentrationer af PFOS over grænseværdien. PFOS blev og målt i koncentration op til 60 gange grænseværdien. Da PFOS er et stærkt problematisk stof indikerer dette fund alene, at overfladevand fra drænkanalerne ikke kan injiceres i grundvandet uden dette har gennemgået en effektiv og intensiv vandbehandling.

3.4 Pesticider

To herbicider (MCPA og 2,4-D) samt et nedbrydningsprodukt af Metribuzin (som kun må anvendes på kartofler) blev fundet over detektionsgrænsen. Imidlertid er kun MCPA fundet i koncentrationer over grænseværdien for pesticider. MCPA nedbrydes relativt let under iltholdige forhold, men er svært nedbrydelig under iltfrie forhold. Derfor vil MCPA også udgøre en risiko for det ilftrie kridtmagasin hvorfra grundvandet pumpes op til drikkevand.

3.5 Opløste uorganiske forureninger

Der er ikke målt nogen uorganiske forureninger i drænvandet, der udgør en umiddelbar forureningsrisiko for grundvandet. Nitrat måles i kun let forhøjede koncentrationer der typisk ligger på omkring halvdelen af drikkevandskravet på 50 mg/l. Der er dermed umiddelbart ingen risiko for drikkevandskvaliteten alene ud fra opblanding af grundvand med det injicerede overfladevand. Det ilt- og nitratholdige overfladevand vil dog oxidere (ilte) reducerede stoffer som pyrit og organisk materiale i grundvandsmagasinet, og derved kan der frigøres sporelementer som Ni, Zi og As fra især pyrit, der i hvert fald i en periode kan skabe problemer med forurening af drikkevandet. Amerikanske undersøgelser i lignende grundvandsmagasiner, indikerer dog at de forhøjede koncentrationer hurtigt vil aftage, og næppe vil medføre uoverkommelige problemer. Herudover kan ilt og nitrat oxidationen af pyrit medføre udfældning af jern-oxider, der kan stoppe porevolumenet i magasinbjergarten, gruskastning og filter, og dermed reducere tilstrømningen til boringen markant.

3.6 Konklusion af vurdering af vandkvalitetsaspekter

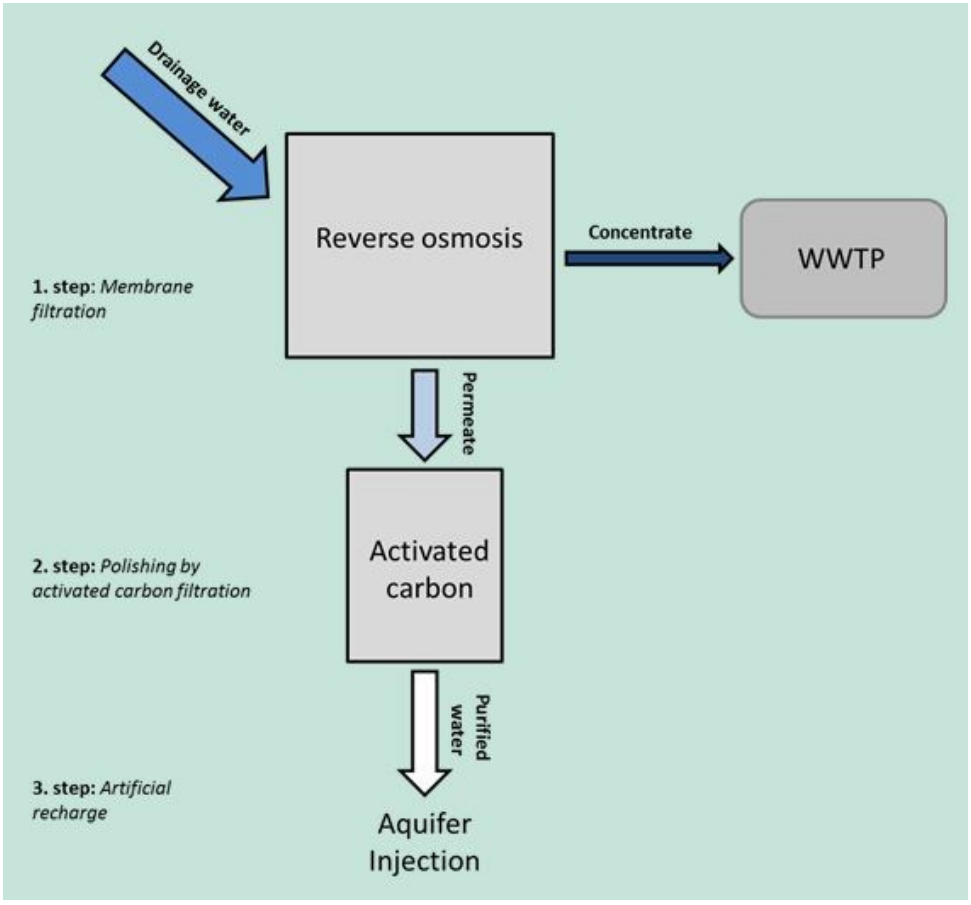
De gennemførte vandkvalitetsanalyser og –vurderinger viser, at der er behov for effektiv og intensiv rensning af vandet fra drænkanalerne før dette kan injiceres i grundvandsmagasinet uden risiko for forurening af grundvandet der indvindes til drikkevand. Der er brug for sikker fjernelse af både lægemidler, industrikemikalier, pesticider og pathogener før injektion af drænvand kan anbefales.
Effektive vandrensningsmetoder bliver dog mere og mere omkostningseffektive og rensning ved f.eks. omvendt osmose er konkurrencedygtig i mange situationer, og eksempler på injektion af overfladevand renset ved denne metode til et niveau, der ligger langt under drikkevandskravet, kendes bl.a. fra Belgien. En kort gennemgang af mulige rensningsteknikker gennemgås i næste afsnit og i Hinsby et al. (2015).

4. Vandrensning

Water4Coasts – aktivitet 1.4

Generelt er der behov for vandrensning før overfladevand kan injiceres direkte i grundvandsmagasiner gennem injektionsboringer for at undgå forurening med pathogener, organiske og uorganiske stoffer (Hinsby et al., 2015a) og uhensigtsmæssige reaktioner i grundvandsmagasinet, der kan frigive problematiske stoffer eller generelt skabe problemer for vandindvindingen.

Water4Coasts projektet vurderer, at der generelt er behov for en effektiv vandrensning, der sikrer at grundvandsressourcen ikke forurenes. Vandrensning ved hjælp af omvendt osmose eventuelt efterfulgt af filtrering på aktivt kul eller lignende højeffektive metoder anses for at være nødvendig såfremt, der er risiko for forurening af værdifulde grundvandsressourcer ved injektionen.



Figur 7

ILLUSTRATION AF MULIG VANDRESNINGSSTRATEGI FOR RENSNING AF OVERFLADEVAND FRA DRÆNKANALER FØR DIREKTE INJEKTION I GRUNDVANDMAGASINER. TRIN 1: MEMBRAN FILTRERING VED OMVENDT OSMOSE (KONCENTRATET FRA OMVENDT OSMOSE FILTRERINGEN BLIVER SENDT TIL ALMENT RENSEANLÆG), TRIN 2: FJERNELSE AF LAVMOLEKYLÆRE ORGANISKE FORURENINGER PÅ AKTIVERET KUL FILTRE, TRIN 3: INJEKTION AF FILTRATET I GRUNDVANDSMAGANIET.

Rensning, herunder afsaltning, ved omvendt osmose har tidligere været relativt dyrt, men nye og forbedrede metoder har gjort teknikken mere udbredt og mere anvendelig (Hinsby et al. 2015a).

5. Overvågning og kontrol af saltvandindtrængning

Water4Coasts – aktivitet 1.5, 1.6 og 3.2

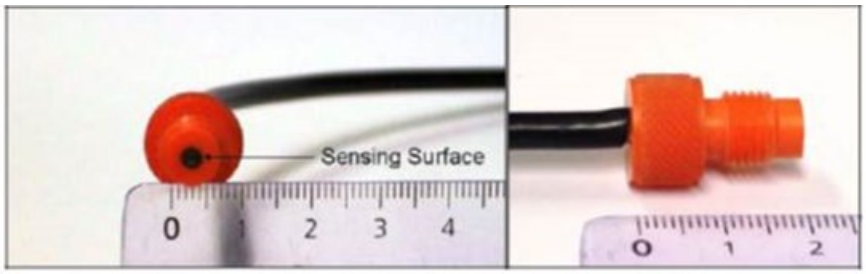
Optimeret og effektiv styring, regulering og overvågning (SRO) er afgørende for velfungerende MAR systemer og SRO systemerne har brug for effektiv dataindsamling og -overførsel til automatiseret styring af systemerne, der kontrollerer pumperne i indvindings- og injektionsboringer baseret på tryk-, klorid- og/eller ledningsevnemålinger i boringerne (Hinsby et al., 2015b).

Projektaktiviteterne 1.5, 1.6 og 3.2 har omfattet udvikling af både hardware (sensorer) og software til styring af SRO systemer til kontrol af saltvandsindtrængning. Hovedresultater og konklusioner præsenteres kort i dette kapitel. Der henvises til Hinsby et al. 2015b for yderligere og mere detaljerede oplysninger om projektaktiviteter og –resultater.

5.1 Udvikling af kloridsensorer

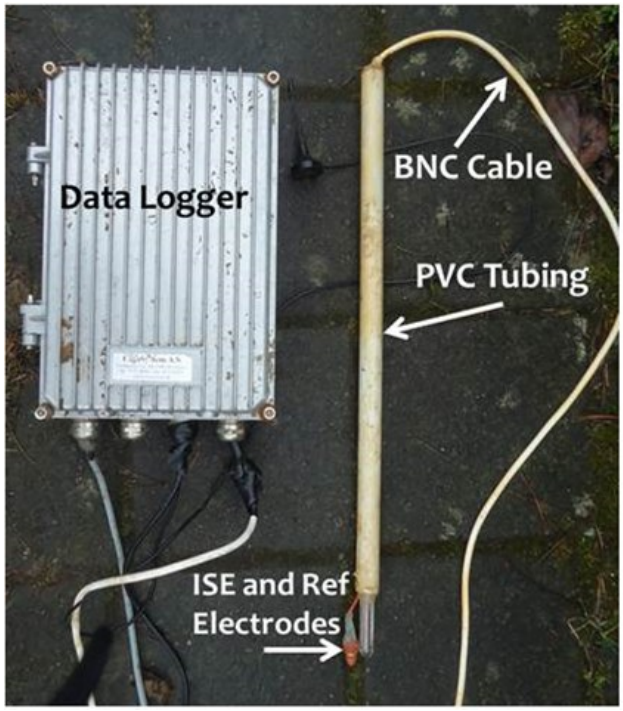
Tidligere undersøgelser og forsøg med udvikling af simple klorid elektroder til kontinuert monitering af saltvands-/kloridindhold (Thorn and Mortensen, 2012; Thorn and Urish, 2013), har vist et stort potentiale for anvendelse af disse til kontinuert overvågning af saltvandsindtrængning ved lavere kloridkoncentrationer end standard elektroder til måling af elektrisk ledningsevne. Undersøgelserne har vist at klorid elektroderne kan måle ned til koncentrationer på ca. 5 mg/l.
De tidligere undersøgelser blev gennemført ved terræn efter at vand fra boringerne var pumpet op til elektroderne. I Water4Coasts projektet er elektroderne videreudviklet, således at de kan placeres permanent i boringerne.

Det primære formål med sensor-udviklingen er at konstruere en eller flere elektroder, der kan placeres i en boring, der kan forbindes til en datalogger med GPRS forbindelse, som kan sende ”real-time” data til servere og programmer til styring af pumper og kontrol af saltvandsindtrængning, samt til on-line visualisering af den historiske udvikling i klorid-og nitratkoncentrationer.



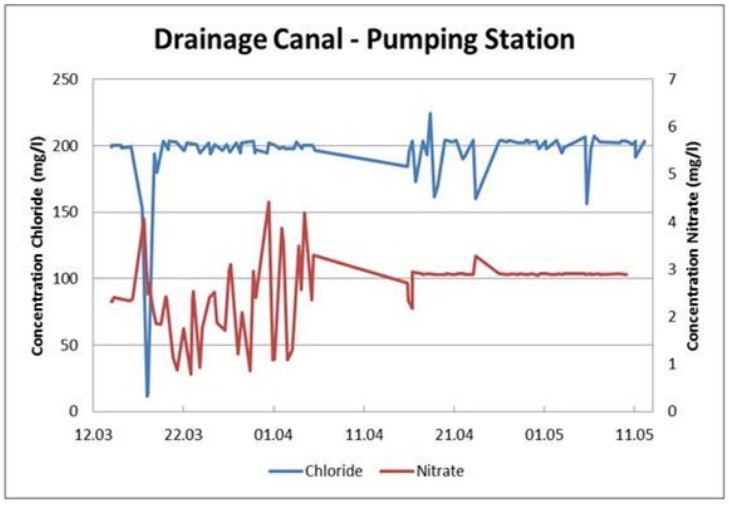
Figur 8

Foto af udviklede kloridelektrode til måling af kloridkoncentrationer i grundvandsboringer



Figur 9

Billede af datalogger samt tilkoblede klorid- og reference-elektroder



Figur 10

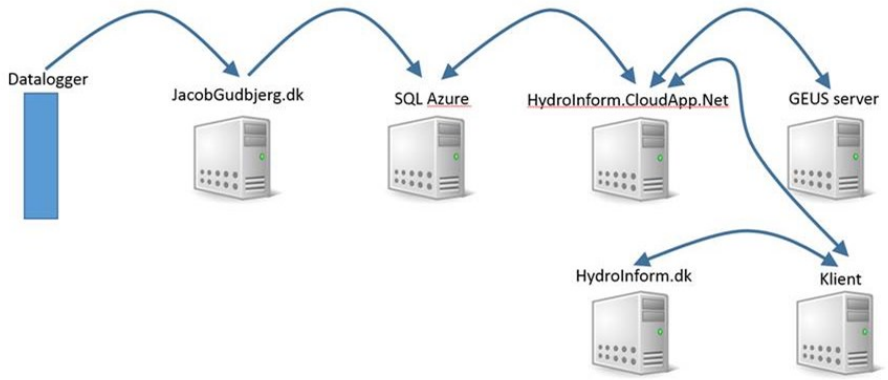
Illustration af on-line målinger af klorid og nitrat målinger ved hjælp af ion selektive elektroder (målesonder) installeret på udløb fra Nordkanalen ved Bøtø Nor pumpestation på Sydfalster i den sydlige del af Water4Coasts projektområdet.

Både klorid- og nitratsensorer blev udviklet og testet i projektet, men der var problemer med især driftssikkerheden af nitratsensoren og data-loggeren. Der skal arbejdes videre med disse i nye projekter for at stabiliteten af disse kan forbedres markant. Der er dog tale om problemer, som det vurderes, at det er muligt at løse med en begrænset indsats i tid og penge, og projekt-deltagerne fortsætter at arbejde hermed i forskellige sammenhænge. Eksempelvis i det nye EU Horizon 2020 projekt ”SubSol” (Bringing SUBsurface water SOLutions to the market, www.subsol.org).

5.2 Datahåndtering og informationsteknologi

Klorid og nitrat sensorernes måledata opsamledes på datalogger hvorfra de via GPRS blev sendt til en server. Dataene blev sendt ved hjælp af TCP-protokollen. ComSystem A/S (www.comsystem.dk) leverede dataloggerne og et medfølgende program, der kan "lytte" efter data og skrive dem til en text-fil, når de modtages. Hvis programmet ikke kører på serveren mistes data dog. Programmet skal startes manuelt og kan ikke køres som en service, hvilket ikke er optimalt.

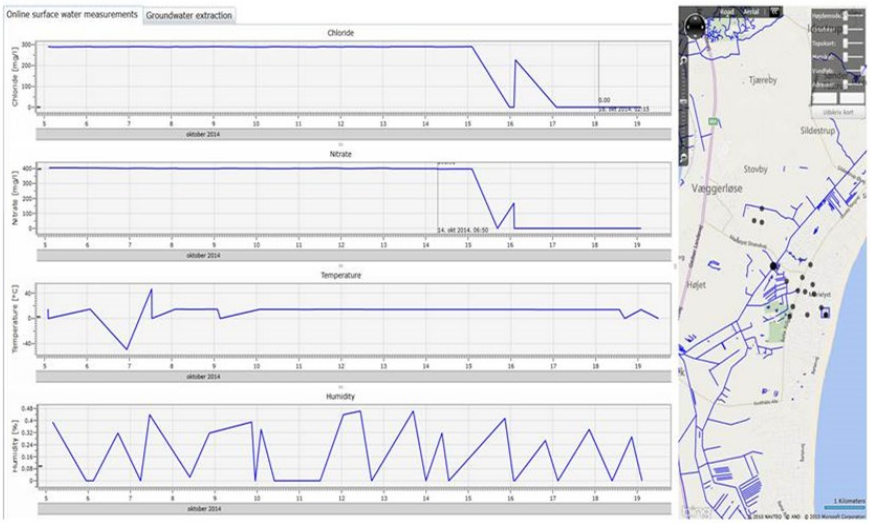
I Water4Coasts har HydroInform (www.hydroinform.dk) udviklet en procedure til at håndtere og videresende data til en "cloud" database (SQLAzure). På toppen af databasen ligger en HydroInform service (HydroInform.CloudApp.Net) gennem hvilken data kan tilgås fra diverse web-klienter. Der er ingen direkte adgang til databasen udefra. Der er udviklet endnu en service, der kan sende data herfra til SiteFx databaser (se figur 11 og afsnit 5.3).



Figur 11

Dataflow fra data-logger til webklient

Datahåndteringen illustreret i figur 11 kan virke kompliceret, men med den viste struktur er de enkelte dele uafhængige, hvilket betyder, at der kan foretages ændringer i databasen uden at resten af systemet påvirkes. Eksempelvis er det relativt enkelt at tilføje data fra andre dataloggere. Bemærk også at der ikke er nogen betydelig forsinkelse af data i systemet. Data der sendes fra dataloggeren er synlig indenfor et minut på hjemmesiden (figur 12). Der er udviklet et effektivt system til visualisering af data på hjemmesiden www.hydroinform.dk, der kan håndtere store mængder data effektivt, og hvor det er muligt at downloade data direkte fra de viste grafer (Hinsby et al., 2015b).



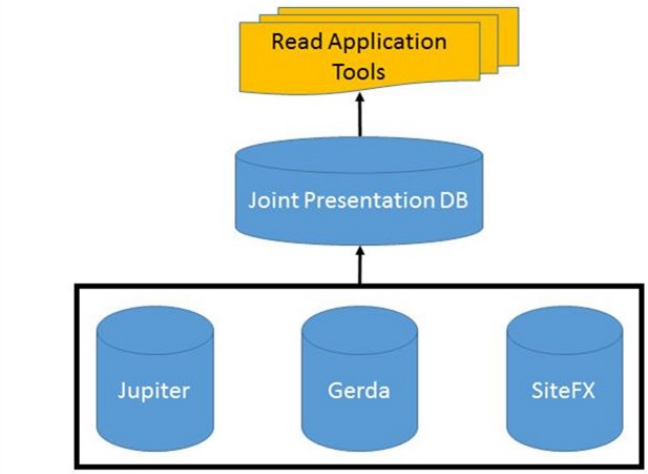
Figur 12

Illustration af real-time datavisualisering på HydroInforms' hjemmeside (www.hydroinform.dk).

5.3 Dataintegration og visualisering på interaktive webservere

Den endelige præsentation og integration af data på en fælles platform er af stor betydning for effektiv evaluering af saltvandsindtrængning og andre komplicerede fænomener, der kræver sammenstilling af en lang række data ofte fra forskellige databaser. I Water4Coasts projektet er data inddelt i "regular relational databases" og "Web Map Services" (WMS). Databaserne udgøres af SiteFX, Gerda og Jupiter. Den første er udviklet af et Canadisk geosoftware firma (www.earthfx.com) medens de to sidste er udviklet på GEUS til håndtering af Geofysik data (GERDA) og Geologi og grundvandsdata (Jupiter). SiteFX kan lagre alle typer geologi og miljødata i både tid og rum. Der er udviklet en procedure til at sende data fra klorid og nitratsensorerne til en SiteFX database (se beskrivelse i forrige afsnit), der er fuldt operationel, når data "uploades" som angivet.

I samme øjeblik data er tilgængelige i databaserne, kan "read application tools"(Figur 13) anvendes til at tilgå relevante data fra alle databaser og institutioner på forskellig vis. Det interaktive web baserede værktøj "EacoWeb", der er udviklet af EarthFX i Canada (www.earthfx.com) i samarbejde med COWI og GEUS, kan anvendes til at præsentere og visualisere data der defineres efter brugerens behov indenfor et givent område. Eksempler herpå kan findes på nedenstående link til hjemmesiden. Bemærk at hjemmesiden kræver anvendelse af det angivne brugernavn (UN) og kode (PW);
https://eacoweb.cowi.com/water4coasts/
UN: water4coasts
PW: w4c-14



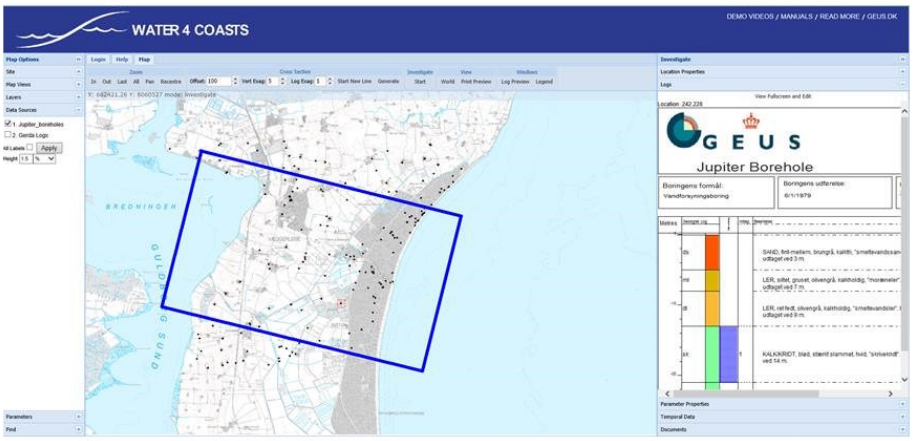
Figur 13

Illustrering af kildedata integration fra forskellige databaser på fælles platform

Data der kan tilgås og håndteres på den fælles platform kan integreres og visualiseres på et uendeligt antal måder i dette Eacoweb værktøj. Water4Coasts forsøgsområdet på Marielyst, Sydfalster kan eksempelvis illustreres som vist i figur 14.

Figuren er et screen dump fra Water4Coasts webserveren, som det kan se ud i det udviklede Eacoweb web værktøj. Udover forsøgsområdet, der er markeret med en rektangel, vises også en borehulslog fra Jupiter databasen på GEUS, samt placeringer af alle boringer i området, der ligeledes er indeholdt i Jupiter.

Brugeren kan arbejde interaktivt med data, der ligger i alle tilknyttede databaser, og kan eksempelvis få udtegnet geologiske tværprofiler langs linjer, der defineres af brugeren eller vist tidsserier af relevante vandkvalitetsparametre – eksempelvis klorid – i Marielyst Vandværks indvindingsboringer. Mulighederne er mangfoldige.



Figur 14

Screen dump fra den udviklede water4coasts hjemmeside på Eacoweb. Figuren viser Water4Coasts forsøgsområdet (blå rektangel) ved Marielyst og Bøtø på Sydfalster, samt til højre i billedet et geologisk boreprofil fra en boring in området (Hinsby et al., 2015b)



Figur 15

Screen dump fra Water4Coasts hjemmesiden på Eacoweb med den tidslige udvikling i koncentrationer af udvalgte parametre i en boring i undersøgelsesområdet (Hinsby et al., 2015b).

6. Konklusion og vision

Hovedkonklusioner og visioner baseret på resultaterne fra Water4Coasts projektet kan resumeres som følger:

1) Den kalibrerede grundvandsmodel for forsøgsområdet og modelkørsler hermed viser, at MAR løsninger kan være en relevant mulighed for at kontrollere saltvandindtrængning både fra kysten (”nyt havvand”) og fra dybereliggende kridtlag (”gammelt havvand”). Modelsimuleringerne viser at nye Hollandske teknikker med anvendelse af horisontale injektions og pumpeboringer (Zuurbier et al., 2015), synes at være de mest effektive til kontrollering af saltvandsindtrængning (Rasmussen et al., 2015).

   Metodernes anvendelse og effekterne heraf, er dog meget afhængig af de hydrauliske og kemiske forhold i undergrunden, og grundige og yderligere undersøgelser heraf på lokal skala omkring indvindingsboringer er af afgørende betydning for vurdering af metodernes potentiale inden fuld-skala anlæg omkring indvindingsboringerne projekteres og installeres.

   Der er således stadig stor usikkerhed omkring de hydrauliske forhold i det delvist opsprækkede kridtmagasin på Falster, der er stærkt påvirket af isbevægelser under sidste istid jfr. de glacialtektoniske processer, der blandt andet medførte dannelse af Møns Klint, og randmorænen i den vestlige del af Sydfalster (den vestlige del af forsøgsområdet på Marielyst).

   De hydrauliske og kemiske forhold af kridtbjergarten, der udgør grundvandsmagasinet på Marielyst, skal derfor undersøges nærmere før optimal design af et pilotanlæg kan udvikles. Ligesom behovet for vandrensning og kravene til injektionsvand fra Nordkanalen eller andre relevante kilder til injektionsvand (f.eks. andre indvindingsboringer) til injektion i og omkring indvindingsboringer skal fastlægges.

2) Undersøgelserne af vandkvalitet i Nordkanalen mellem rensningsanlæg og pumpestation på Marielyst / Bøtø på Sydfalster samt i grundvandsmagasiner i Kridtbjergarterne, hvorfra drikkevandet indvindes i dybder mellem 10 og 25 meter under terræn, viser at der er risiko for forurening og negativ effekter på vandkvaliteten, hvis vand fra Nordkanalen injiceres i grundvandsmagasiner uden forudgående omfattende rensning af vandet fra kanalen.

   Forurening i kanalen omfatter udover pathogener både uorganiske og organiske forureninger fra landbrug (nitrat og pesticider mm.), industri (PFOS etc.) og husholdning (primært lægemidler). Den største gruppe af potentielle forurenende stoffer udgøres muligvis af lægemidlerne. I en screening af ca. 90 lægemidler på udledningen fra det lokale rensningsanlæg, er der fundet 35 lægemidler i målbare koncentrationer, og heraf er 19 over grænseværdien for pesticider på 0,1 µg/l i drikkevand (der findes p.t. ingen grænseværdi for lægemidler i drikkevand). Samlet set er der målt lægemidler i samlet koncentration før og under turistsæsonen på hhv. 17 og 25 µg/l.

Udledningerne fra rensningsanlægget fortyndes, så der i Nordkanalen ikke kan måles lægemidler over grænseværdien, men de udgør en risiko (om end lille) for vandkvaliteten i grundvandsmagasinerne. Problemer med lægemidler i grundvand kendes især fra f.eks. Tyskland og Spanien, eksempelvis hvor grundvand indvindes fra boringer tæt på floder ved 'induced bank infiltration', der er meget anvendt i Europa (Figur 3).

Udover problemer med forureninger fra det injicerede overfladevand, kan der opstå problemer med vandkvaliteten ved frigørelse af sporelementer som f.eks. arsen og nikkel ved oxidation af pyrit ved ilt og nitrat i grundvandsmagasinerne. Oxidation af jernmineraler (f.eks. pyrit) i grundvandsmagasinet kan desuden skabe tekniske problemer ved tilstopning af filtre og sprækker i kildebjergarten, så boringernes ydelser sænkes markant. Begge problemstillinger har dog vist sig at kunne håndteres hensigtsmæssigt i flere eksisterende anlæg i U.S.A.

Modelsimuleringer af kemiske processer ved injektion af ilt og nitratholdigt vand, viser at effekten er meget afhængig af Kridtmagasinets hydrauliske egenskaber. Opfører kridtmagasinet sig primært som et dobbeltporøst sprækket magasin hindrer den langsomme diffusion fra kridtets matrix til sprækkerne effektiv adgang til de reducerende stoffer i bjergarten (primært organisk stof og pyrit), og dermed effektiv reduktion af ilt og nitrat. Foreløbige analyser af kridtbjergarten viser meget lave koncentrationer af pyrit (0,001 – 0,03 %) og organisk stof. Dette er en ulempe såfremt man ønsker at reducere nitrat i grundvandsmagasinet, men en fordel i forhold til at beskytte grundvandets mod sporelementer som nikkel og arsen, der kan frigøres fra pyrit, om end disse normalt sorberes på udfældede jernoxider.

Ovenstående problemstillinger omkring både sundhedsmæssige og tekniske aspekter af påvirkning af vandkvaliteten ved injektion af potentielt forurenet ilt- og nitratholdigt overfladevand i iltfrie grundvandsmagasiner medfører, at der ofte vil være behov for effektiv vandbehandling inden injektion af overfladevand i grundvandsmagasinet. Nye og mere omkostningseffektive rensningsmetoder gør dog i mange tilfælde metoderne rentable under mange forskellige forhold i globalt perspektiv. Således injiceres der mange steder renset overfladevand (eller endda renset spildevand) i grundvandsmagasiner eksempelvis i kystzoner i perioder med stor nedbør og lavt vandforbrug (vinterperioden) til oplagring og senere forbrug i perioder med lille nedbør og stort vandforbrug til f.eks. markvanding og turisme (sommerperioden).

3) Teknologier til overvågning og kontrol af saltvandsindtrængning er tilgængelige i et vist omfang, men der er behov for yderligere udvikling af bedre og mere effektiv teknologi og software til overvågning og kontrol (SRO-anlæg) af saltvandsindtrængning i grundvandsmagasiner. De nødvendige teknologier og det nødvendige databehov er generelt kendt, men der mangler målrettet udvikling af SRO anlæg, lige fra stabile kloridsensorer til effektiv software, der kan kontrollere og minimere saltvandsindtrængning på vandværker der er truet af saltvandsindtrængning.

Danmark har den nødvendige erfaring og knowhow til at gennemføre nødvendige hydrogeologiske og vandkemiske undersøgelser og udvikle innovative og effektive værktøjer (vandrensning, sensorer og software til SRO systemer osv.) til vandværker, der er truet af saltvandsindtrængning. Værktøjerne kan finde anvendelse især i kystnære områder, hvor en meget stor del af verdens befolkning befinder sig i dag, og hvor stigende havniveau og befolkningstal vil øge presset på ressourcerne. Det kan give Danmark mulighed for eksport af instrumenter, software og knowhow til kontrol af saltvandsindtrængning og beskyttelse af ferskvandsressourcer i kystområder til et globalt marked med stigende behov.

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Bilag 1. GEUS rapport 2015/ 92 - Modelling the effects of hydraulic barriers to control saltwater intrusion in a coastal chalk aquifer

DANMARKS OG GRØNLANDS GEOLOGISKE UNDERSØGELSE RAPPORT 2015/92

Water4Coast

Modelling the effects of hydraulic barriers to control saltwater intrusion in a coastal chalk aquifer

Water4Coast delrapport for aktivitet 1.1 og 2.2

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& Klaus Hinsby

GEOLOGICAL SURVEY OF DENMARK AND GREENLAND
DANISH MINISTRY OF ENERGY,UTILITIES AND CLIMATE



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1. Summary and Conclusions

Coastal aquifers are at risk of saltwater intrusion. This is a well described problem from many coastal areas and from coastal areas with limited access to other freshwater resources. Hydraulic barriers are one technique to protect the coastal aquifers from saltwater intrusion, e.g. with injection wells creating an artificial groundwater divide between the seawater and the groundwater aquifer by injection of water into the subsurface.

Groundwater models including variable density flow can be used to evaluate the effects of different types and configurations of hydraulic barriers. Geophysical data like borehole logging and geophysical surveys can be used to improve the calibration of groundwater models.

The possible effects of using injection wells as hydraulic barrier against salt water intrusion have been tested with an existing groundwater model for the Marielyst area on Falster, and with generic models.

Geophysical data have been interpreted and implemented in the groundwater model in order to improve the model calibration. The pilot point calibration technique has also been tested for the study area in Marielyst, Falster.

The results of the pilot point test calibration suggest that it could be valuable to implement the more heterogenetic distribution of hydraulic conductivities found in the pilot point calibration.

Various configurations and pumping rates for injection wells have been tested. The model studies have shown that e.g. a horizontal well with a length of 150 m injecting 1/3 of the water injected by three vertical wells with horizontal spacing of 300 m have a larger effect on preventing seawater intrusion. With a large vertical spacing between vertical injection wells a lot of injected water may be wasted.

The test of different injection well setups with generic models indicates that the injection rate is of most importance in heterogeneous aquifers.

Monitoring wells should be deep enough to be able to measure the changes in seawater intrusion below the freshwater lens. Monitoring wells should have monitoring points at different depth in the aquifer.

The modelling results show a large potential of hydraulic barriers and injection wells for controlling salt water intrusion and protection of coastal freshwater resources. However, further and more detailed investigations of the hydraulic parameters and geochemistry of the Chalk aquifer on Southern Falster is needed in order to further explore and define the possible and most efficient subsurface solutions for controlling salt water intrusion around the water supply wells of the Marielyst water works.

The upper part of the exploited Chalk aquifer has been strongly affected by several glaciations and glaciotectonics, and the hydraulic properties vary significantly. Hence, focused and detailed investigations at the water supply wells are needed in order to identify the best and most efficient options for controlling salt water intrusion and protecting the freshwater resources including the design and location of potential injection and abstraction wells.

The investigations on the use of managed aquifer recharge and other subsurface solutions on Falster, Denmark initiated in the Water4Coasts project is currently continued in the 'SubSol' project: 'Bringing coastal SUBsurface water SOLutions to the market' (European Commission, 2015a) in 'Horizon 2020' - the research and innovation program of the European Commission (European Commission, 2015b). A project flyer, results and updates on the progress of this project can be found on the project website: www.subsol.org.

2. Introduction

Background
Coastal aquifers are at risk of saltwater intrusion which may reduce or even preclude groundwater abstraction for water supply from these aquifers. This is a well described problem from many densely populated coastal areas and from coastal areas with limited access to other freshwater resources, e.g. minor islands. Climate change is likely to result in increasing sea levels which again may increase the risk of saltwater intrusion to coastal groundwater aquifers.

Hydraulic barriers are one technique to protect the coastal aquifers from (increasing) saltwater intrusion. Hydraulic barriers can be established in several ways, e.g. with injection wells creating an artificial groundwater divide between the seawater and the groundwater aquifer by injection of water into the subsoil. Other methods include multiple wells abstraction controlling the freshwater-saltwater interface (Zuurbier et al. 2014), negative hydraulic barriers (Pool and Carrera 2010), or installing impermeable walls between the saltwater and the fresh groundwater aquifer (Kaleris and Ziogas 2013).

One method to evaluate the effects of different types and configurations of hydraulic barriers is to use groundwater models which include transient and variable density flow domains. Besides measurements of groundwater heads and stream discharges, geophysical borehole logging, geophysical surveys and groundwater quality data can be used to improve the calibration of the variable density groundwater model.

Previous studies
Several studies have recommended the use of geophysical data to improve the calibration of variable density groundwater models. Geo-electrical inversion models of the sub-surface and variable density groundwater models can be cross-validated and thereby improve the modelling of saltwater intrusion (Comte and Banton, 2007). Carrera et al. (2010) stated that model calibration can be improved significantly by including the airborne geophysical measurements in the calibration process.

Another method of improving the model calibration is by using the pilot point technique where the spatial variations in hydraulic conductivities can be determined (Christensen and Doherty, 2008; Dausman et al., 2010; Langevin and Zygnerski, 2013; Maneta and Wallender, 2013).

Injection wells used as hydraulic barriers to prevent (further) saltwater intrusion may (also) act as an ASR (Artificial Storage and Recovery) system where part of the injected water (may possibly) end up in the abstraction wells (David and Pyne 2005). Ward et al. (2008 and 2009) analyses the effects of lateral flow, density-driven flow, dispersive mixing, discontinuities in pumping, and anisotropic and layered heterogeneous aquifers on ASR systems.

Misut and Voss (2007) found through model studies that freshwater storage by injection followed by phases of pause and recovery by extraction may benefit coastal water users if less water is recovered than injected and thereby establishing a hydraulic saltwater intrusion barrier.

This study
WATER4COASTS Activity 1.2 includes saltwater intrusion and freshwater injection modelling. A calibrated saltwater intrusion model (SWI) already exists for the Danish study site on the island of Falster (Rasmussen et al., 2013). New airborne geophysical data collected for the

Danish Nature Agency has become available since the calibration of the existing model. Also a new investigation borehole has provided new geological and geophysical data.

The first phase of this activity includes the recalibration of the existing model based on new airborne geophysical data and new data from geophysical borehole logging. Pilot point calibration is tested to see if it improves the model performance. In the second phase the recalibrated model are used to analyse the effect of different configurations and management strategies of injections wells.

The chemical, biological and pollution aspects of injecting surface water into the groundwater aquifer are dealt with under Activity 1.3 and 1.4.

Activity 1.2 includes 2 milestones, M1: Falster model calibrated and validated and M3: Required well injection in different injection designs and scenarios simulated by model.

WATER4COASTS Activity 2.2 includes the development of generic saltwater / freshwater models to illustrate the general effects of various types of hydraulic barriers under various hydrogeological and climate scenarios.

Generic models can be applied to assess different designs of hydraulic barriers for relevant hydrogeological and climatological settings representing current and future climate settings for different global regions including projected climate change. Based on existing systems and literature different injection designs will be developed and evaluated by the generic models. Monitoring system designs will be proposed for monitoring the future evolution of groundwater and surface water quantity and quality during artificial aquifer recharge. This will include new technical solutions for monitoring groundwater and surface water chloride and nitrate concentrations (e.g. Thorn and Mortensen, 2012; Sulzbacher et al., 2012).

Activity 2.2 includes 1 milestone, M6: Generic model simulations completed.

The results of Activity 1.2 and 2.2 are both described in the present report.

The SWIM23 conference held in Husum in June 2014 offered an opportunity to present WATER4COASTS project Activity 1.2. A scenario with the existing model showing the effect of three injection wells in a changing climate on the water quality in two groundwater abstraction wells was presented (Appendix 1 and 2).

3. Methods

3.1 Study area

The study site is located in the south-eastern part of Denmark on the island of Falster. Towards the east the study area is bounded by the Baltic Sea and towards the west of the strait of Guldborgsund. The local waterworks are abstracting groundwater from the shallow chalk aquifer overlain by quaternary and post glacial sediments of mainly clayey tills and sands. The well fields located closest to the Baltic Sea coast have seen an increasing chloride concentration in the abstracted groundwater during the last decades (Figure 3-1).

The mean annual precipitation is around 700 mm/year, and the groundwater recharge has been estimated to 250 mm/year. The total groundwater abstraction in the area is 400,000 m^3/year or less than 5 % of the recharge. The main discharge from the area is through the pumping station downstream at the Marrebæk canal (Figure 3-1).

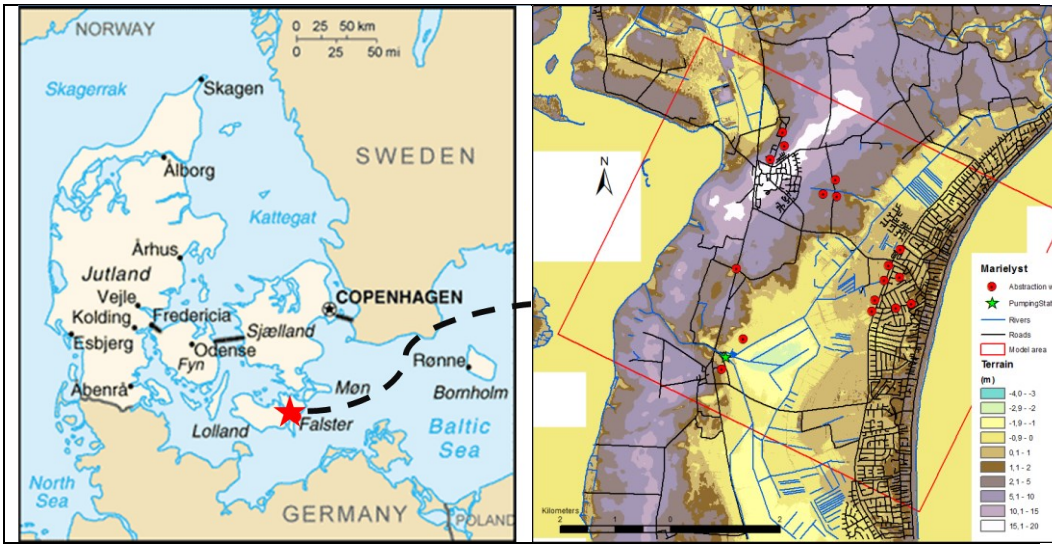


Figure 3-1. Location of study area, abstraction wells, drainage canal and pumping station

3.2 Data collection

Existing hydrogeological data from the study area are retrieved from the National well database (Jupiter), the National geophysical database (GERDA), the Groundwater reports database, and the Groundwater model database. During the last 15 years a special effort has been done to file existing data and to collect new data as part of the Danish Groundwater Mapping.

3.2.1 Groundwater chemistry

In the study area it is assumed that there are three potential sources for the salinity measured in groundwater abstraction wells; diffusion and/or upconing from deeper layers, interglacial/post glacial saltwater trapped in glacial/post glacial deposits, and seawater intrusion.

Groundwater chemistry data from selected existing groundwater abstraction wells were obtained from Marielyst Waterworks, the National well database (Jupiter), and from a previous project in the area, BaltCICA (http://www.baltcica.org/).

The molar ratio of sodium to chloride [Na]/[Cl] is an indication the origin of the salinization in a well. If the ratio is below 0.9 the salinization is most likely to come from seawater intrusion, whereas if the ratio is above 0.9 diffusion of connate saline water is the most like source of chloride (Appelo and Postma 2005, Thorn 2011).

Measurements of groundwater salinity and geophysical measurements can be linked and converted through well documented relationships and conversion factors, e.g. Archie's law (Larsen et al., 2006, Klitten and Wittrup, 2006, Rasmussen et al. 2013).

The most common parameters when dealing with the groundwater chemistry and geophysics are chloride concentrations, total dissolved solids concentrations (TDS), fluid resistivity and conductivity, bulk resistivity and conductivity, and the formation factor.

3.2.2 New research borehole, geological data

A new 45 m deep investigation borehole was established between the waterworks old well field and the Baltic Sea 150 m from the coast line, borehole no. 242.375 (Figure 3-2). Cores and soil samples were taken for a detailed lithological and stratigraphical description of the hydrogeological setting. Soil samples were collected for each one meter and detailed described in the soil lab. Also a soil core was collected from a minor section of the borehole (Appendix 3).

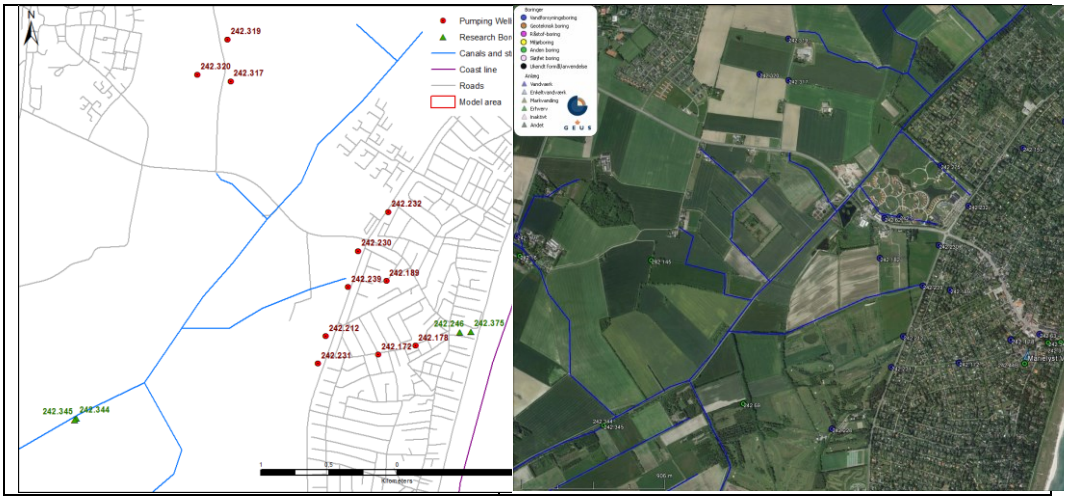


Figure 3-2. Location of reach boreholes and groundwater abstraction wells

The borehole was completed with a PVC screen from approximately 2 m above the top of the chalk formation to the bottom of the borehole. The reason for installing a screen in the chalk borehole was to be able to analyse groundwater flow conditions in the uppermost part of the chalk and in the zone between the chalk and the overlaying quaternary deposits.

3.2.3 Borehole geophysics, wireline logging

Borehole geophysics using wireline logging was carried out in the new investigation borehole with the purpose of getting more information of the groundwater flow in the fractured chalk aquifer with focus on the zone between the chalk aquifer and the overlaying quaternary deposit. The existing wells in the area, mainly groundwater abstractions wells, are all completed with an iron casing finishing 2-5 meters below the top of the chalk and are open boreholes 10 – 15 meters into the chalk. It is expected that the uppermost parts of the chalk is highly fractured or crushed resulting in a high hydraulic conductivity. The second focus point of the borehole logging is in the deepest part of the borehole with the transition zone above which advective groundwater flow is dominating and the saltwater is flushed out. Below the advective zone diffusion is becoming a more dominant process and saltwater is present in higher concentrations in the formation.

Borehole geophysics from the 100 m deep research borehole (242.344) is reported in Rasmussen et al. (2013).The following wireline logging data are collected from the research boreholes: Natural gamma, formation conductivity (induction log), fluid temperature and conductivity, and propeller flow log. Temperature/conductivity and flow logs are measured under two pumping conditions, no pumping and during constant groundwater abstraction.

3.2.4 Airborne geophysics and MEP

During the recent national groundwater mapping activities part of the study area was covered with airborne electromagnetics (AEM) by the Danish SkyTEM company. The Danish Nature Agency supported the data collection. The Aarhus Workbench software was used for handling, processing, and inversion of the resistivity measurements (Naturstyrelsen Roskilde, 2011).

One limitation of the AEM is that it cannot record resistivities in built-up areas due to power lines and cables. In the study area there is a high density of summerhouses along the east coast, which mean that no SkyTEM data are available from this area. Two of the well fields are located in the area with high density of summerhouses. The SkyTEM data was mainly collected in the central part of the study area, but not around two well fields and along the eastern coast line. 50 km of flight lines with a distance of 150-200 m were carried out. 17 km^2 has been covered with SkyTEM surveys out of the 44 km^2 study area (Figure 3-3).

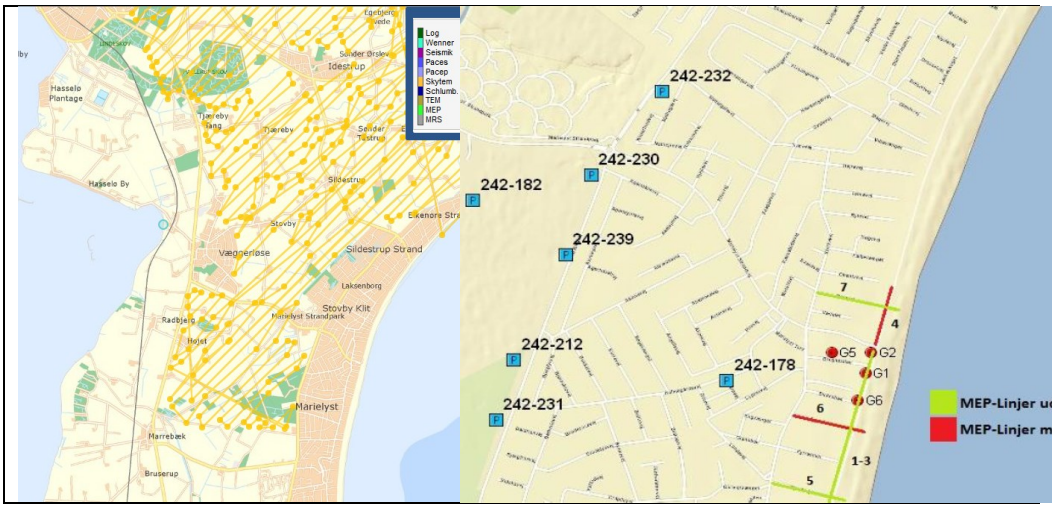


Figure 3-3. SkyTEM flight lines (left) and MEP profile lines (right)

SkyTEM surveys provide a good areal 3D coverage of resistivity measurements down to 100-150 meters below ground surface. The depth of investigation (DOI) varies depending of the vertical succession of layers and formation resistivities of these layers. The measured resistivities were inverted to both 5-layer models and 19 layer models. The conversion of SkyTEM resistivities to 3D lithological units for hydrogeological groundwater modelling is associated with uncertainties (He et al. 2014). The horizontal and vertical resolution of SkyTEM resistivity inversions depends among other things on the applied SkyTEM technology, the flight line density, and the sequence and properties of the local geology.

Five Multi Electrode Profiling (MEP) lines along and perpendicular to the eastern coastline, were collected in the area where it was not possible to use AEM methods (Figure 3-3). The MEP profiles give 2D resistivity information along the profile lines down to a depth of 60 meters (Steiness, 2011).

3.2.5 Data management and visualisation using GeoScene3D

The Danish software GeoScene3D (GS3D) is a software programme for visualizing, interpreting, editing and publishing geological data for e.g. groundwater modelling projects. In GS3D it is possible to visualize many different types of geological data, from for example boreholes, geophysics, terrain models, geological layers and water chemistry (http://i-gis.dk/).

SkyTEM resistivity profiles (flight lines) are interpolated with GS3D into grids values, which are converted to TDS values. Iso-resistivity maps and TDS values are interpolated and exported for later possible use in model calibration process.

3.3 Existing local groundwater model

3.3.1 Local model setup

As part of the BaltCICA project a variable density groundwater model was established for the study area to analyse possible effects of sea level rise, changed groundwater recharge and changed drainage stage on saltwater intrusion and groundwater quality (Rasmussen et al. 2013).

The numerical modelling complex MODFLOW-2000/MT3DMS/SEAWAT was used for simulating 3-D variable density groundwater flow and solute transport. Groundwater Vistas version 6 was used as the pre- and post-processing tool.

The Hydrogeologic-Unit Flow (HUF) Package for MODFLOW2000 was used to define hydrogeological units independently from the numerical layers. Hydraulic properties are assigned to the hydrogeological units in the HUF-package and the HUF-package calculates the effective hydraulic properties for the numerical layers of the flow model. The advantage of using the HUF-package is the ability to represent hydrological units of variable thickness and distribution independently of model layers in the flow model (Figure 3.4).

The model area is 44 km^2 of which 12 km^2 is sea. The horizontal grid size is 50 m by 50 m. The hydro-stratigraphical model implemented with the HUF-package has 15 geological layers, while the flow model consists of 32 numerical layers with thicknesses varying from 2 to 12 m. The thickness of the numerical layers gradually increases with depth down to −200 m a.s.l. A more

detailed description of the model setup and climate scenarios is found in Rasmussen et al. (2013).

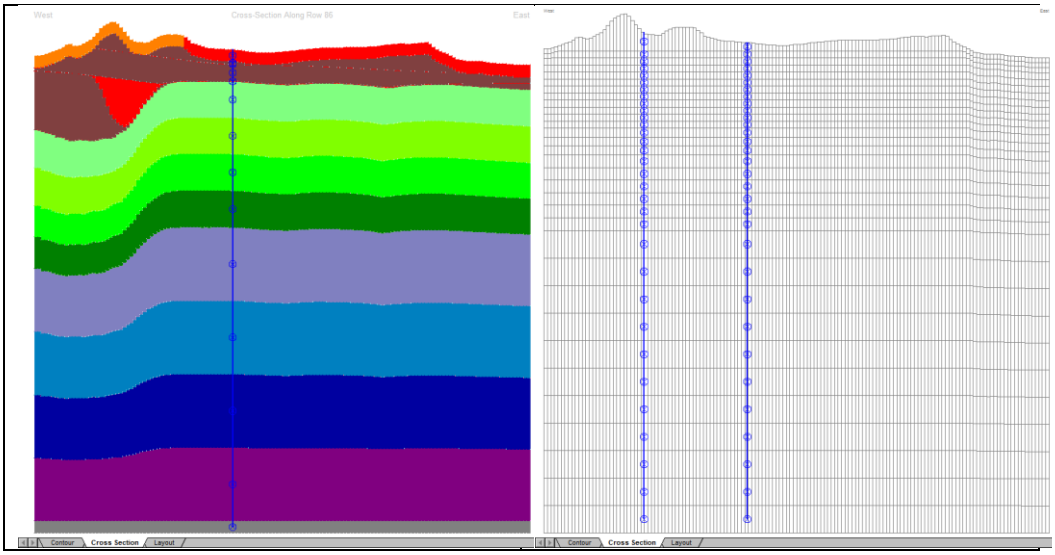


Figure 3-4. Cross section of the hydrogeological layering in the HUF-package setup (left) and the model layers in the groundwater flow model (right)

The hydraulic conductivities of the existing model are seen in Table 3-1, definition of hydraulic zones in Table 3-2, storage and porosity values in Table 3-3, and diffusion and dispersivity values in Table 3-4.

Table 3-1 Hydraulic conductivities of hydro-stratigraphical units (HUF-package). For explanation of blue and yellow colours, see Table 3-2 and text below Table 3-1

Layer	Lithology	Kx[m/d]	Kx[m/s]	Kz[m/d]	Kz[m/s]	Kz/Kx
1	fractured clay	0.43200	5.00E-06	0.04320	5.00E-07	0.1
1-6	sand	43.20000	5.00E-04	4.32000	5.00E-05	0.1
2-6	clayey till	0.08600	9.95E-07	0.00860	9.95E-08	0.1
7	chalk	4.32000	5.00E-05	0.43200	5.00E-06	0.1
8	chalk	0.86400	1.00E-05	0.08640	1.00E-06	0.1
9	chalk	0.43200	5.00E-06	0.04320	5.00E-07	0.1
10	chalk	0.08640	1.00E-06	0.02160	2.50E-07	0.25
11	chalk	0.00864	1.00E-07	0.00864	1.00E-07	1
12	chalk	0.00600	6.94E-08	0.00600	6.94E-08	1
13	chalk	0.00400	4.63E-08	0.00400	4.63E-08	1
14	chalk	0.00200	2.31E-08	0.00200	2.31E-08	1
15	chalk	0.00086	1.00E-08	0.00086	1.00E-08	1

The variations in hydraulic conductivities reflects the proposed zonation of salinity with depth from an advective zone in the upper part of the chalk formation, a transition zone below which the diffusion processes are dominating (Larsen et al. 2006; Bonnesen et al., 2009). The hydraulic conductivity of the drain bottom is 0.86 m/d.

Table 3-2 Hydraulic zones in the existing model

Hydraulic zones (chalk)	Zone thickness (m)	Layer (no.)
Quaternary	5-35	1-6
Advective	15	7
Transition	30	8-9
Diffusion	130	10-15

Table 3-3 Storage and porosity values

Layer	Lithology	Ss (m^{-1})	Sy	Eff.Porosity
1	fractured clay	0.2	0.2	0.2
1	sand	0.25	0.25	0.25
2-6	sand	1.00E-04	0.2	0.2
2-6	clayey till	1.00E-04	0.05	0.05
7	chalk	1.00E-05	0.15	0.15
8-15	chalk	1,00E-05	0,35	0,35

Table 3-4 Diffusion and dispersivity values

Layer	Property	Value	Value
1-15	Diffusion	4.10E-05 (m^2/d)	4.75E-10 (m^2/s)
1-15	Dispersivity, Longit_D	8 (m)	
1-15	Dispersivity, Transv_D	0.05 (m)	
1-15	Dispersivity, Transv_V	0.001 (m)	

I order to minimize numerical dispersion a fine model discretization may be necessary. A rule of thumb is that the Peclet number should not be larger than 4. The Peclet number is the ratio between the length of the model grid cell and the longitudinal dispersivity. For models with higher Peclet number than 4, is has been recommend to use the TVD solver (Total Variation Diminishing) for the model simulations (Langevin et al. 2010). The grid size of the used flow and transport models are 50 m by 50 m.

The model simulations was divided into six phases with four separate but consecutive model setups and simulations with different boundary conditions to represent the historical changes in the area (Figure 3-5). The groundwater head and salinity distribution at the end of each model phase was used as initial heads and concentrations for the next model phase (Rasmussen et al., 2013).

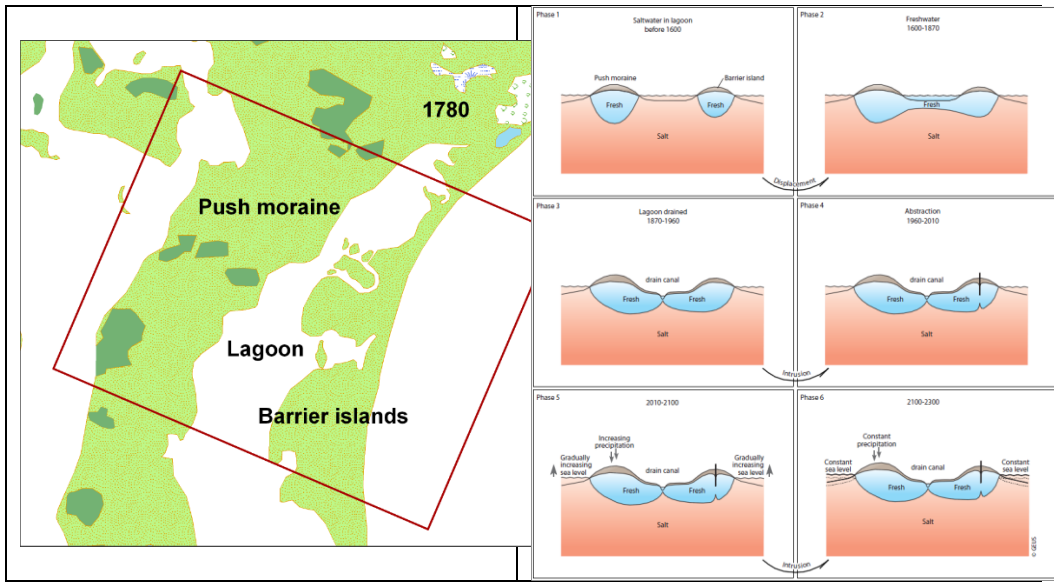
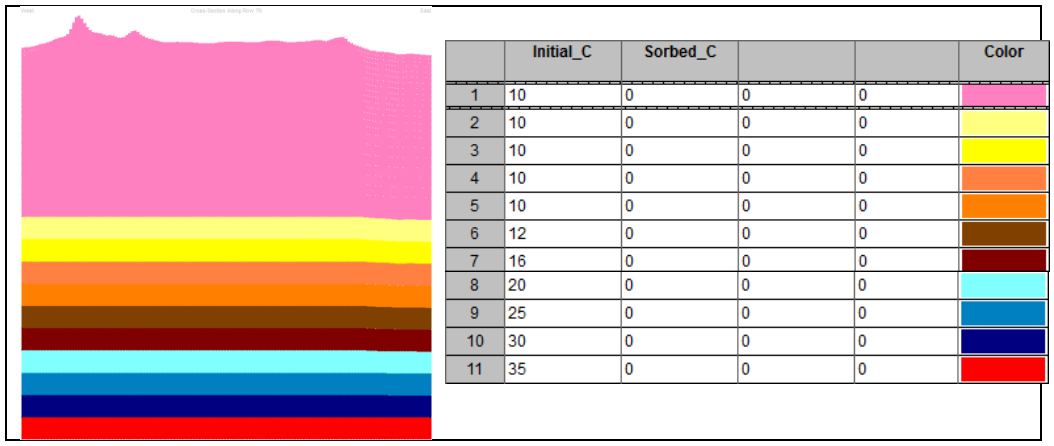


Figure 3-5. Landscape of study area around 1780 (left), and modelled transition phases of the area from a lagoon with barrier islands to reclaimed land with groundwater abstraction and climate change impacts (right), (Rasmussen et al., 2013)

The initial saltwater concentrations for modelling Phase 1 are seen in Figure 3-6.



	Initial_C	Sorbed_C			Color
1	10	0	0	0	
2	10	0	0	0	
3	10	0	0	0	
4	10	0	0	0	
5	10	0	0	0	
6	12	0	0	0	
7	16	0	0	0	
8	20	0	0	0	
9	25	0	0	0	
10	30	0	0	0	
11	35	0	0	0	

Figure 3-6. The initial saltwater concentrations (Initial_C) for modelling Phase 1, model cross section (left), concentrations for each zone (the colours) in g TDS/l (right) from terrain to -200 m a.s.l.

3.3.2 Local model recalibration

Two methods are used to explore the possibilities of improving the calibration of the existing local groundwater model. The first method includes newly collected SkyTEM data and the borehole geophysics from the study area. The second method is a test of the pilot point method using groundwater heads.

Using geophysics

In the study area information from wireline logging in boreholes deep enough to show the upper part of the transition zone from the flushed advective zone with fresh water down towards the diffusion zone is only available from one well. The salinity gradient from this well is

compared with logging results from other boreholes in chalk formations in the eastern Denmark.

The SkyTEM data have provided a good areal coverage of geological formation resistivities from large parts of the study area. Observed variations in the measured resistivities can be result of variations in either or both salinity and lithology. SkyTEM data can provide valuable information about depth to saltwater and the resistivity of geological formations. The accuracy of the SkyTEM measurements is not (so) high compared to wireline logging (GeofysikSamarbejdet, 2011). The SkyTEM measurements are limited to a certain depth, the "depth of investigations" (DOI) depending on the geology in the area, but usually it is between 100 and 200 meter below the ground surface.

A recalibration of the existing groundwater model where the detailed variations in geology and salinity obtained from the SkyTEM measurements are interpreted and used for model calibration lies outside the framework of this project. The approach using SkyTEM data for calibration will be to look at the possibility of interpolate an iso-resistivity surface corresponding to the upper boundary of the transition zone based on the corresponding resistivity and depth found with wireline borehole logging. This surface will define top of the specific model layer below which the transition zone is present and decreasing hydraulic conductivities are introduced in the layers below.

Using pilot points

The aim of using pilot points technique in the model calibration is to explore the effects of (small scale) geological variability. The distribution of hydraulic conductivity within the model domain or a geological unit can be described by a set of pilot points. Individual pilot points can be assigned to different hydraulic conductivity zones/geological units/model layers within the model domain. Only those points assigned to a particular zone or geological unit can be used in calculating hydraulic conductivities values throughout that zone using the kriging interpolation procedure. The variogram upon which kriging is based can be different in each zone, reflecting differences in the geology, or in the level of heterogeneity, expected within each geological unit. The parameter estimation software PEST is used to estimate the hydraulic conductivity of the zone/aquifer at each such pilot point. These point hydraulic conductivities will then be spatially interpolated to all of the active cells within the model domain using kriging (Rumbaugh, 2013).

In the study area the main part of the groundwater head observations are from the upper 10-20 meters of the chalk. The model layer containing the most head observations are chosen for the pilot point calibration, for other layers the manual calibration procedure is used.

A number of pilot points are introduced to the model layer 7, where most groundwater head observations are found (Figure 3-8). Each pilot point is defined by a number, xy coordinates, layer number, hydraulic conductivity target value Kx (Kz), minimum Kx (Kz), and maximum Kx (Kz). The location of pilot points are subjective, but conventional wisdom tells that the number of parameters involved in a parameter estimation should be kept to a minimum. Using pilot points in conjunction with PEST's regularization functionality prevents the onset of numerical instability. The regularization brings a high degree of numerical stability to a parameter estimation problem if regularization constraints are appropriately defined. The pilot point procedure includes a calibration constraint "homogeneous unless proven otherwise", so heterogeneity is only introduced in the calibration process "because it has to be there". A set of "kriging factors" for each of the cells of the finite difference grid can be calculated by Groundwater Vistas in advance of the actual interpolation process (Rumbaugh, 2013).

Rule of thumb for pilot point calibration (Rumbaugh, 2013):

- The number of pilot points should be 2 – 3 times the numbers of observations (targets)
    - For horizontal Kx use target triangulation for location of pilot points and afterwards fill the gaps with pilot points

    - For vertical Kz place pilot points at target locations and afterwards fill the gaps
- Combine ‘truncated SVD’ and ‘SVD assist’
- Use ‘super pilot points’. The number of super pilot points should be 10 – 20 % of the number of pilot points
- Use ‘preferred homogeneity’
- If there are no observations in a zone PEST will not be able to do a good estimate of the parameters
- Variogram: α could be 20 % of the maximum dimension of the zone
- Kriging search radius: large enough so that every cell within the zone can have an inter-polated value.

The pilot point calibration is performed as a non-variable (uniform) density modelling. Areas with high salinity in the existing local model are converted to no-flow cells (Figure 3-9).

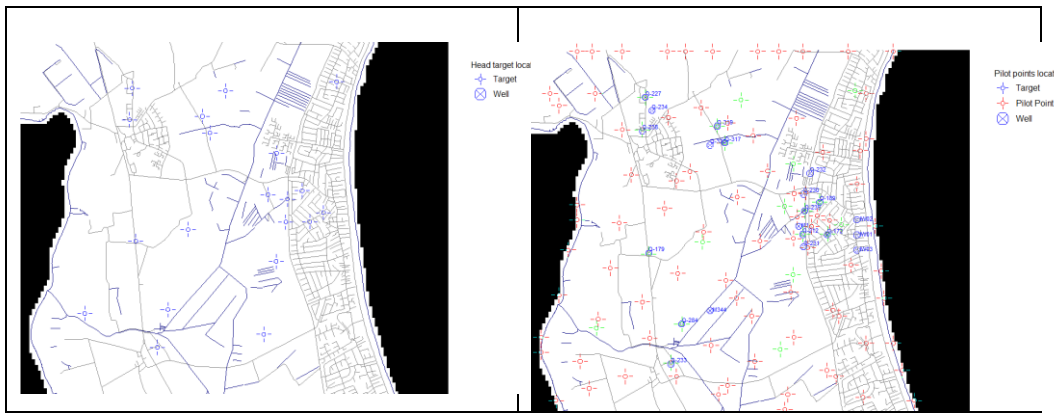


Figure 3-7. Groundwater head observations in model layer 7 (left). Location of pilot points, Kx red cross, Kz green cross (right)

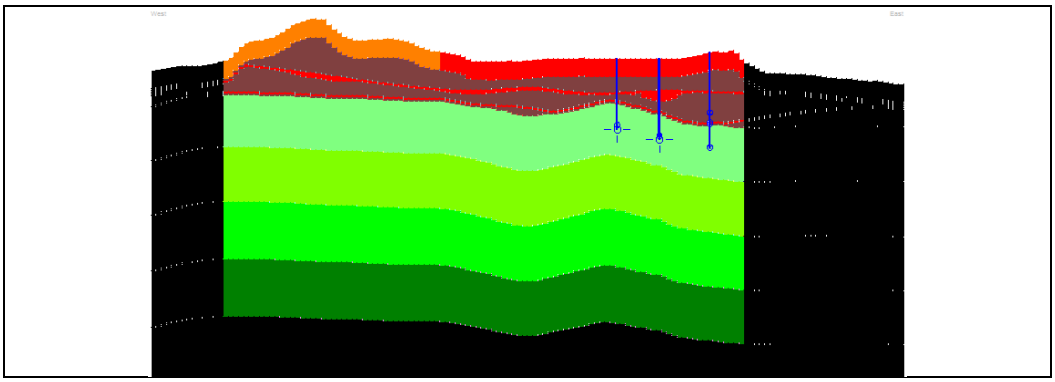


Figure 3-8. Areas with high salinity in existing model are converted to no-flow cells (black) for the pilot point tests

3.3.3 Injection well setup

Four out of a number of suggested possible scenarios of injection well designs and pumping rates were analysed with the variable density groundwater model (Table 3-7).

Table 3-5 Injection wells scenarios, the ones marked with yellow were completed

Injection scenario	Number of wells	Well Direction[1,5]	Pumping schedule[2,3]	Pumping rate (%)	Model recalibration	Climate scenario[4]
1	3	Vertical	Constant	100	Before	3
2b	3	Vertical	Constant	100	After	3
2e	3	Vertical	Constant	100	After	2
	2	Vertical	Constant	100	After	
	2	Vertical	Recirculating	100	After	
	2	Vertical	Recirculating	50	After	
	3	Vertical	Alternating	50/0	After	
2c	3	Vertical	Alternating	100/0	After	3
	3	Vertical	Constant	50	After	
	3	Vertical	Constant	200	After	
2d	1	Horizontal	Constant	100	After	3
	1	Horizontal	Constant	50	After	
	1	Horizontal	Alternating	100/0	After	

1. Wells placed parallel to coastline, except recirculation wells
2. Recirculation of groundwater between one pumping and one injection well (ATES system) perpendicular to coastline
3. Alternating pumping schedule: half year pumping / half year pause
4. Scenario 3: Sea level rise +0.75 m and decrease in groundwater recharge by 15%. Scenario 2: Sea level rise +0.75 m and no change in groundwater recharge (Rasmussen et al., 2013)
5. Length of horizontal well: 150 m (length of three model cells).

The three synthetic injection wells are located 300 m from the coast with 250 m spacing. An injection rate equal to the average abstraction rate from each of the waterworks abstraction wells, 68 m^3/d (24,820 m^3/y), is used as point of reference (100%). The well screens are placed in the upper fractured chalk -12 to -30 m a.s.l. (meter above sea level). An alternating pumping schedule is applied in one scenario: ½ year pumping and ½ year pause (Figure 3-9).

For the horizontal well a screen length of 150 m is used, which is the length of three model grid cells. The horizontal well is located -18 to -21 m a.s.l. and 300 m from the coast line. The hori-

zontal well is implemented in MODFLOW in as a standard boundary condition (Figure 3-9).

Year 2300 marks the end of the test of the climate change scenarios in the existing model, therefore the injection of water with the injections wells is started in year 2300 and the simulation is continued for additional 100 years (Rasmussen et al. 2013).

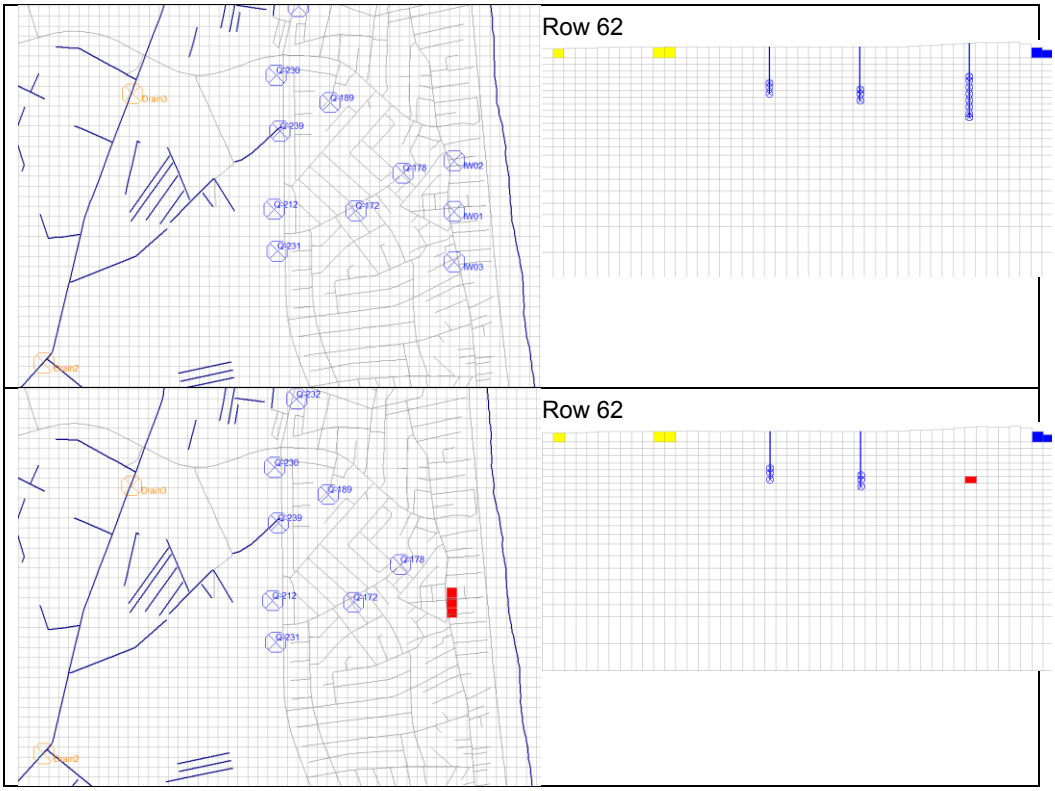


Figure 3-9. Location of injection wells, plan view left, cross section right. Vertical injection wells (upper) and horizontal well (lower)

3.3.4 Climate change scenarios

In the existing model eight climate change scenarios were simulated with different combinations of sea level rise, groundwater recharge, and drainage canal stage. Scenario 0 represents a situation where no changes occur. Scenario 1 represents an estimate of the most likely future with an increase in recharge of 15% and sea level rise of 0.75 m. In scenario 2–8 a sensitivity analysis of the most likely scenario is carried out, where other realistic changes in sea level and recharge have been implemented. The drainage canals play an important role in the modelled groundwater system. To evaluate the sensitivity of the drains, two simulations (scenarios 7 and 8) are performed with changed stage, +30 and −30 cm, respectively (Rasmussen et al. 2013).

For the present study two climate change scenarios were selected: Scenario 3: Sea level rise +0.75 m and decrease in groundwater recharge by 15%, and Scenario 2: Sea level rise +0.75 m and no change in groundwater recharge (Table 3-5).

3.4 Generic groundwater models

3.4.1 Model setup

The generic models are simplified model setup of the existing groundwater model for the Marielyst area. Similar model boundary conditions and parameters are to some extent used as a starting point for the generic models. But the generic model area is smaller, the geological model is simpler, and the generic models consist of fewer calculation layers (Table 3-6, 3-7, 3-8, 3-9, 3-10, and Figure 3-10).

Table 3-6. Basic data for setup of generic models

Model setup	Value	Unit
rows	40	
columns	40	
layers	12	
grid x	50	m
grid y	50	m
top elevation.	10	m
bottom elevation	-70	m
layer average thickness	6.67	m
total active cells	192,000	
total area	4,000.000	m2
Stress periods		
SP-1	36,500	d
SP-2	328,500	d
No. time steps		
SP-1	10	
SP-2	100	

Table 3-7. Hydraulic conductivities of the standard generic model

Layer	Lithology	Kx[m/d]	Kx[m/s]	Kz[m/d]	Kz[m/s]	Kz/Kx
1-2	sand	43.2	5.00E-04	4.32	5.00E-05	0.1
3	clayey till	0.0864	1.00E-06	0.00864	1.00E-07	0.1
4-7	chalk1	4.32	5.00E-05	0.432	5.00E-06	0.1
8-11	chalk2	4.32	5.00E-05	0.432	5.00E-06	0.1
12	chalk3	0.000864	1.00E-08	0.000864	1.00E-08	1

Table 3-8. Storage and porosity values for the standard generic model

Layer	Lithology	Ss (m-1)	Sy	Effective Porosity
1-2	sand	0.25	0.25	0.25
3	clayey till	1.00E-04	0.05	0.05
4-7	chalk1	1.00E-05	0.15	0.15
8-11	chalk2	1.00E-05	0.15	0.15
12	chalk3	1.00E-05	0.35	0.35

Table 3-9. Diffusion and dispersivity values for the standard generic model

Layer	Property	Value	Value
1-12	Diffusion	4.10E-05 (m^2/d)	4.75E-10 (m^2/s)
1-13	Dispersivity, Longit_D	8 (m)	
1-14	Dispersivity, Transv_D	0,05 (m)	
1-15	Dispersivity, Transv_V	0,001 (m)	

Table 3-10. Boundary conditions for generic models

Boundary conditions		Value	Unit	Note
Recharge				
		0.0007	m/d	
		0.00081	m/d	climate scenario +15%
	concentration	0.5	g TDS/l	
Pumping, 1 well: P1				
	SP-1	0		
	SP-2	68	m3/d	model layer 5
Drain				
	Stage of drain	-1	m	
	Width of drain	2	m	
	Length of drain in cell	50	m	
	Thickness drain bed	0.5	m	
	Hydraulic conductivity	0.172	m/d	
	Conductance	34.4	m2/d	
Fixed Head				
		0	m	
		+0.5	m	sea level rise
Sea concentration				
		10.5	g TDS/l	
Bottom layer concentration				
		10.5	g TDS/l	

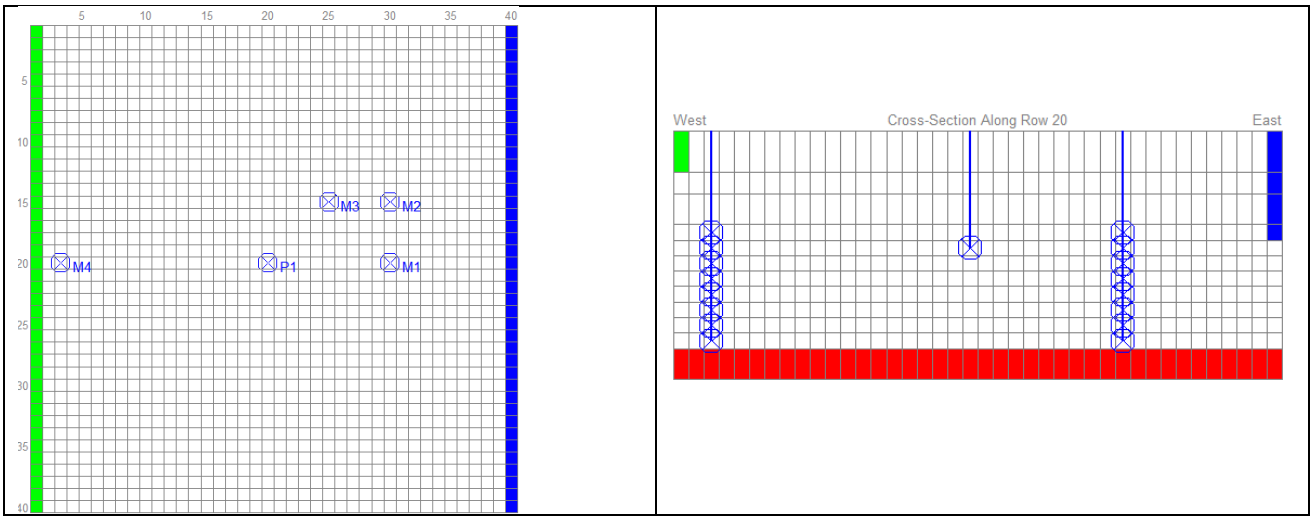


Figure 3-10. Horizontal (left) and vertical (right) view of standard generic model. P1 is the pumping well, M1-M4 are monitoring wells

3.4.2 Sensitivity analysis

The generic models are used to test and compare the effect of changes in different model boundary conditions (BC) and parameters like bottom layer concentrations, diffusion coefficients and hydraulic conductivity of aquitard. Scenarios 1-4 are generic models including saltwater diffusion only, where all other scenarios include saltwater intrusion (Table 3-11).

Table 3-11. Scenarios with generic models (BC: boundary condition 1-4, H: Head, C: concentration, avg.: average value, Q: pumping rate, SWI: saltwater intrusion)

Sce. no	Chalk model	BC1	BC2	BC3	BC4	Diff. coef	Aquit. Kx,z	Rech	Remarks
1	2 layer	Const.H	Q avg.	Const.C		avg.	avg.	avg.	Diffusion only
2	2 layer					high			
3	2 layer						low		
4	2 layer			C high					
5	2 layer	Drain	Q avg.	Const.C	Const. H.C.4layers	avg.	avg.	avg.	SWI and cana
6	2 layer				Const. H.C.3lyers				
7	3 layer				Const. H.C.4layers				K1 > K2 > K3,
8	3 layer		Q high						
9	2 layer				+1.0m				SLR 1.0 m
9b	2 layer				+0.5m				SLR 0.5 m
10	2 layer							+15%	Recharge Increase 15%
11	2 layer						low		

In scenario 6 the seawater intrusion boundary condition with constant head and concentration is assigned to layer 1-3, whereas in other scenarios the same boundary condition is assigned to layer 1-4.

In scenarios 8-10 the effect of projected climate change are illustrated by modelling sea level rise and changed groundwater recharge.

3.4.3 Hydraulic barrier setup

Five scenarios with injection of water into the aquifer with the purpose of creating a hydraulic barrier to protect the groundwater abstraction wells from saltwater is analysed with the generic models. The scenarios focus on injection rates and well configurations, numbers of wells, vertical or horizontal wells (Table 3-12 and Figure 3-11).

Table 3-12. Injection wells scenarios with generic models (BC: boundary condition 1-4, H: Head, C: concentration, avg.: average value, Q: pumping rate)

Sce. no	Chalk model	BC1	BC2	BC3	BC4	Inject. well	Remarks
12	2 layer	Drain	Q avg.	Const.C	Const. H.C.4layer	1 vert.	Q = 68 m3/d Qconc.= 0 g/l
12b	2 layer		Q avg.			1 vert.	Q = 68 m3/d Qconc.= 0,5 g/l
13	2 layer		1/4 Q avg.			1 vert.	Q = 17 m3/d
14	2 layer		Q avg.			3 vert.	Q = 22.7 m3/d for each well
15	2 layer		1/4 Q avg.			1 horiz.	1 layer, length: 3 cells = 150 m

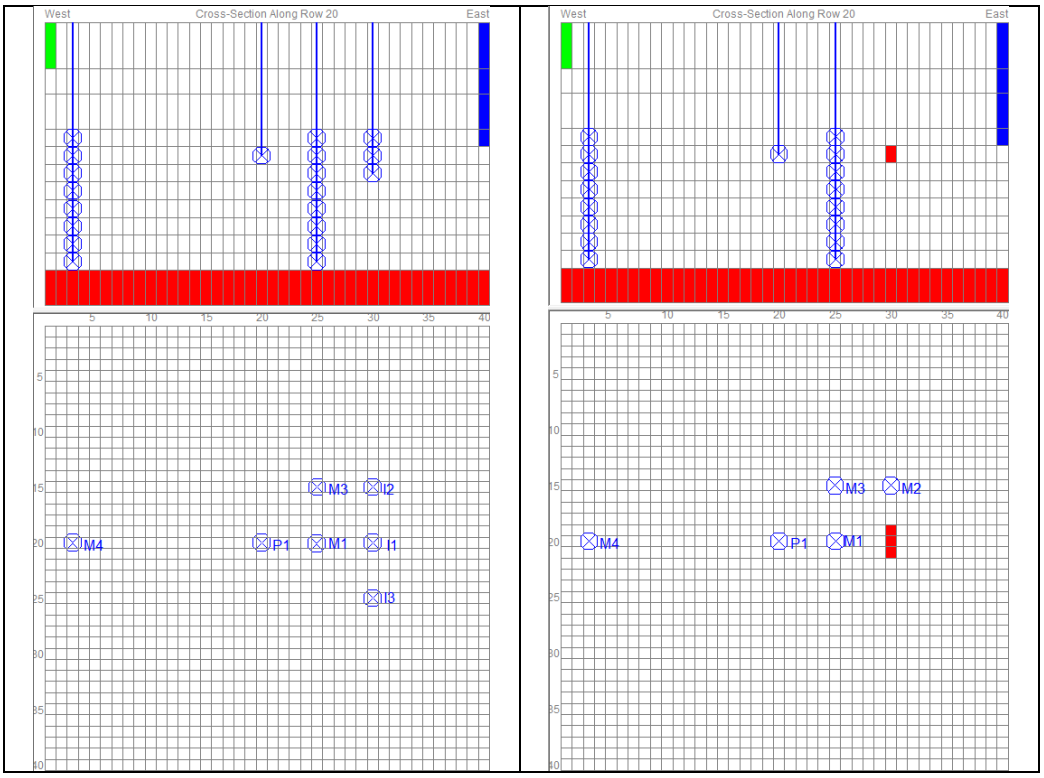


Figure 3-11. Generic model setup for scenario 14 with 3 injection wells (left) and scenario 15 with 1 horizontal well (right). P1 pumping well, M1-M4 monitoring wells.

3.4.4 Monitoring system designs

Depending on the generic model setup 3-4 monitoring wells are placed between the pumping well and the coast, between the pumping well and the injection well, and near the canal. The monitoring wells have a monitoring point in each model layer in the chalk aquifer (Figure 3-10 and 3-11).

4. Results

4.1 Sources of salinity

If the molar ratio of sodium to chloride [Na]/[Cl] is below 0.9 the salinization is most likely to come from seawater intrusion. If the ratio is above 0.9 diffusion of connate saline water is the most like source of chloride. Data from the deeper groundwater abstraction wells in the area, marked with light green in Table 4-1, all have a ratio below 0.9, with an average ratio of 0.8.

Table 4-1. Groundwater [Na]/[Cl] ratio

Borehole	Depth (mbgs)	Date	Na[mg/l]	Na[mmol/l]	Cl[mg/l]	Cl[mmol/l]	Na/Cl
242.44B	17,00	10-04-1970	74,00	3,22	148,00	4,17	0,77
		26-06-1989	130,00	5,65	228,00	6,43	0,88
		15-01-2004	143,00	6,22	293,00	8,27	0,75
242.178	23,00	10-04-1970	30,00	1,30	53,00	1,50	0,87
		26-06-1989	110,00	4,78	202,00	5,70	0,84
		29-01-1996	135,00	5,87	255,00	7,19	0,82
		24-01-2013	130,00	5,65	270,00	7,62	0,74
242.172	18,50	26-06-1989	47,00	2,04	101,00	2,85	0,72
		23-01-2015	110,00	4,78	208,00	5,87	0,82
242.190	19,00	10-04-1970	29,00	1,26	50,00	1,41	0,89
		12-06-1995	128,00	5,57	270,00	7,62	0,73
		21-01-2003	109,00	4,74	262,00	7,39	0,64
242.212	15,60	26-06-1989	55,00	2,39	113,00	3,19	0,75
		07-01-2013	88,00	3,83	200,00	5,64	0,68
242.230	15,00	26-06-1989	29,00	1,26	57,00	1,61	0,78
		23-01-2015	58,00	2,52	120,00	3,39	0,75
242.232	17,00	26-06-1989	60,00	2,61	118,00	3,33	0,78
		23-01-2015	86,00	3,74	162,00	4,57	0,82
242.317	13,00	11-01-2006	16,00	0,70	35,00	0,99	0,70
		23-01-2015	23,00	1,00	36,00	1,02	0,99
242.319	16,00	26-05-2005	17,00	0,74	29,00	0,82	0,90
		16-01-2012	21,00	0,91	41,00	1,16	0,79
242.320	14,00	26-05-2005	21,00	0,91	37,00	1,04	0,88
		24-01-2013	17,00	0,74	31,00	0,87	0,85

There is a tendency to decreasing values of the [Na]/[Cl] ratio as the sodium and chloride concentration increase, which indicate that sea water intrusion is taking place.

4.2 Geophysics and saltwater distribution

Wireline logging from the 100 m deep borehole in the study area has provided data indicating that the advective zone is deeper than assumed in the old model setup, approximately 20 m (Figure 4-1).

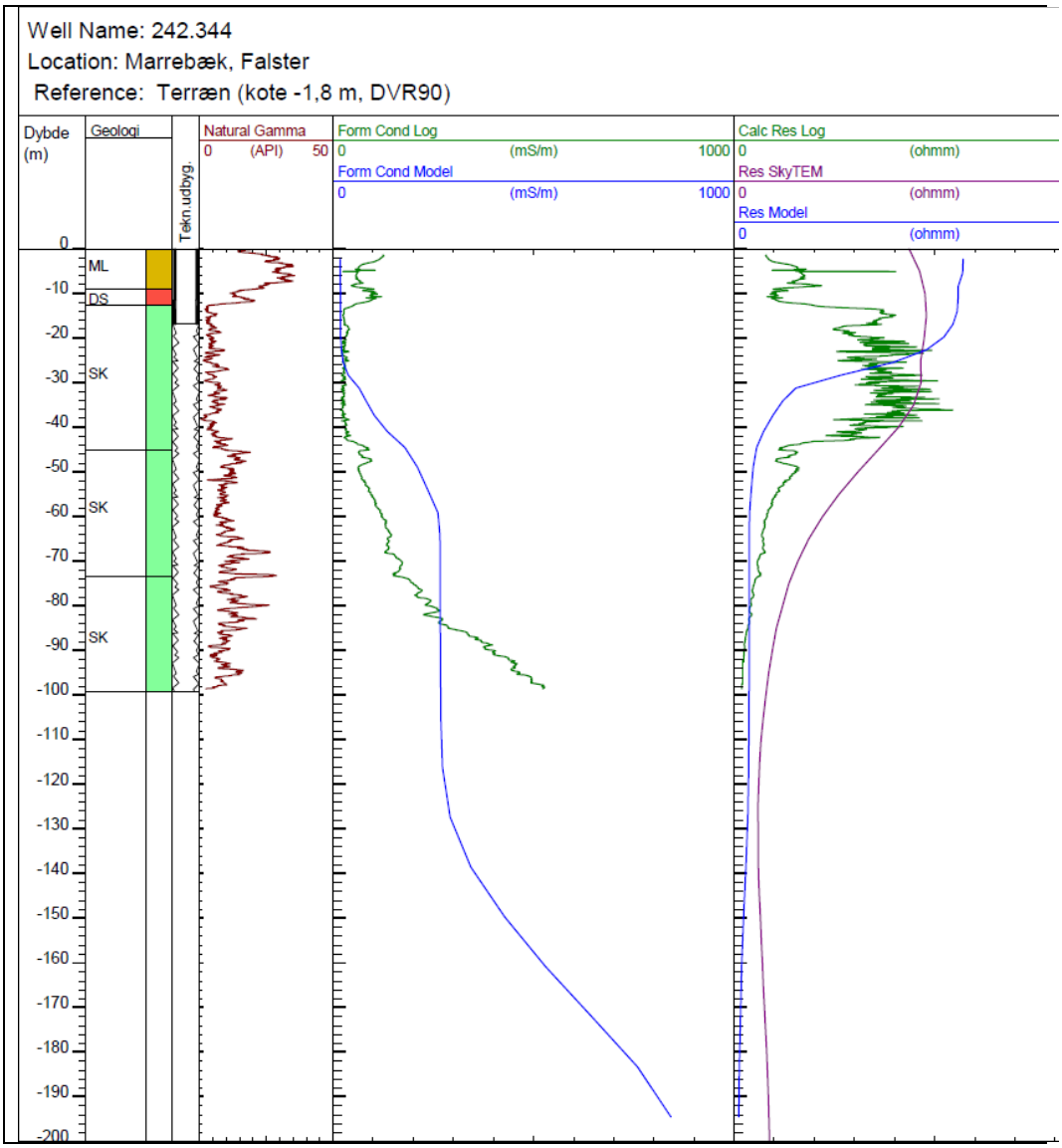


Figure 4-1. Centre column showing formation conductivity from wireline logging (green) and modelled formation conductivity calculated from TDS concentrations (blue). Right column shows the formation resistivities including the SkyTEM measurements (magenta). The left column shows the natural gamma radiation

In the study area there is only information from wireline logging from one borehole that is deep enough to capture the upper part of the transition zone from the flushed advective zone with fresh water down through the upper part of the transition zone. The salinity gradient from this well is in good accordance with logging results from other boreholes in chalk formations in the eastern part of Denmark (Figure 4-2). This salinity gradient is to some extent modelled and reproduced with the existing model using a gradually decreasing hydraulic conductivity with depth in the transition zone and appropriate initial conditions (Section 3.3.1.).

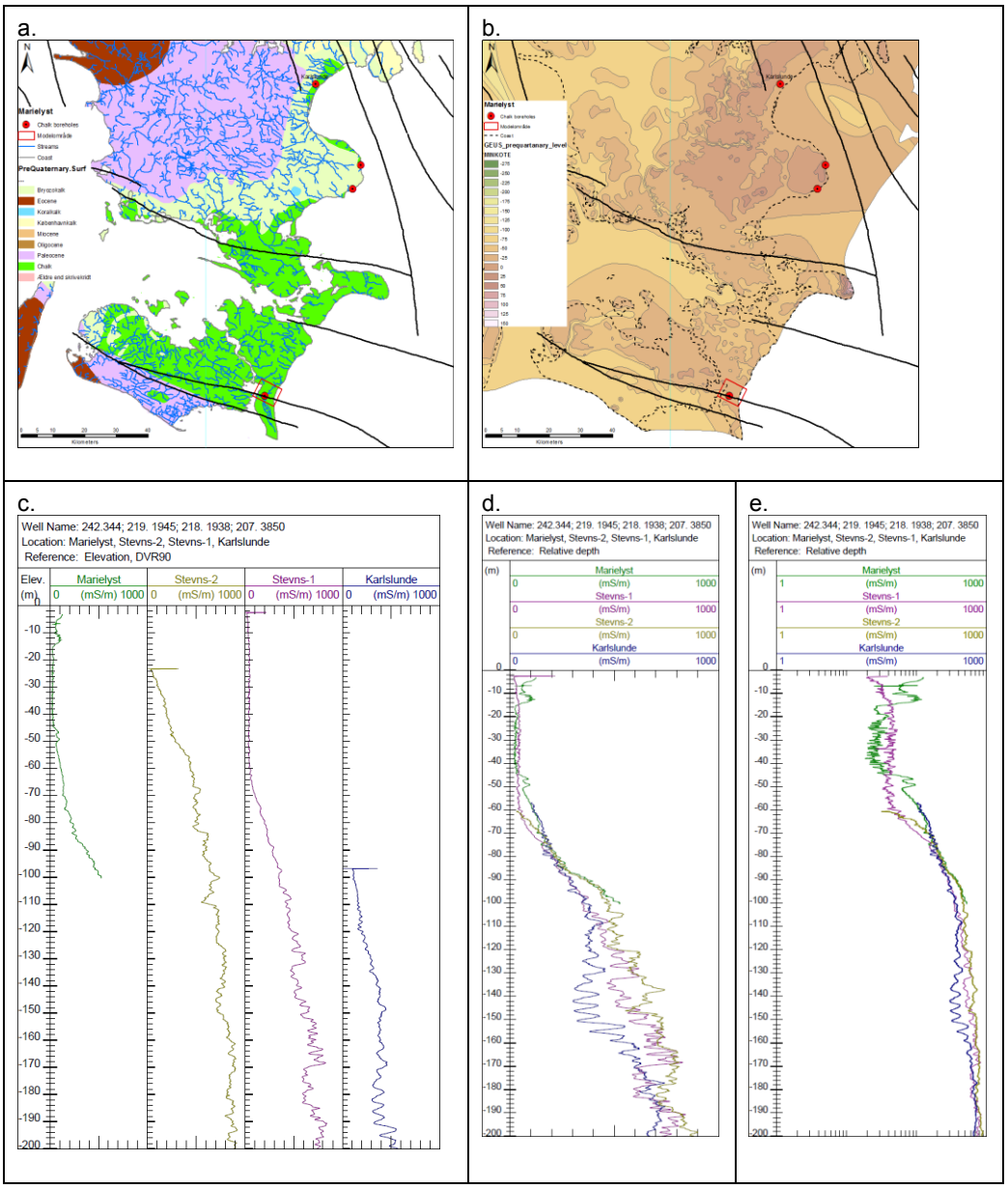


Figure 4-2. a. Location of deeper research boreholes in chalk and top pre-quaternary geology, b. Location of deeper research boreholes in chalk and top pre-quaternary elevation, c. Formation conductivity from wireline logging of the four boreholes, left to right corresponds to south to north location of boreholes in panel a. and b., d. The four induction logs depth shifted to match similar gradients, normal scale, e. The four induction logs depth shifted to match similar gradients, log scale

It is observed that the gradient (the increase of formations conductivity with depth) is of the same magnitude when comparing the deeper borehole in the study area with other boreholes in chalk settings in the eastern part of Denmark. In Figure 4-2 a comparison of wireline induction logs from four chalk boreholes in eastern Denmark (Study area Marielyst, Stevns-2, Stevns-1, and Karlslunde, order of well names correspond to south to north location on maps).

Even though the elevation where the transition from advective dominated zone to the diffusion dominated zone varies from location to location, the logging results indicate that the proposed conceptual model with a well-defined advective zone above a well-defined transition zone can be generalised to the chalk aquifer in the study area. The salinity gradients from Marielyst and Karlslunde are very similar, and the gradients from Stevns-2 and Stevns-1 are very similar.

With SkyTEM measurements a good areal coverage of geological formation resistivities has been obtained from large parts of the study area (Figure 4-3).

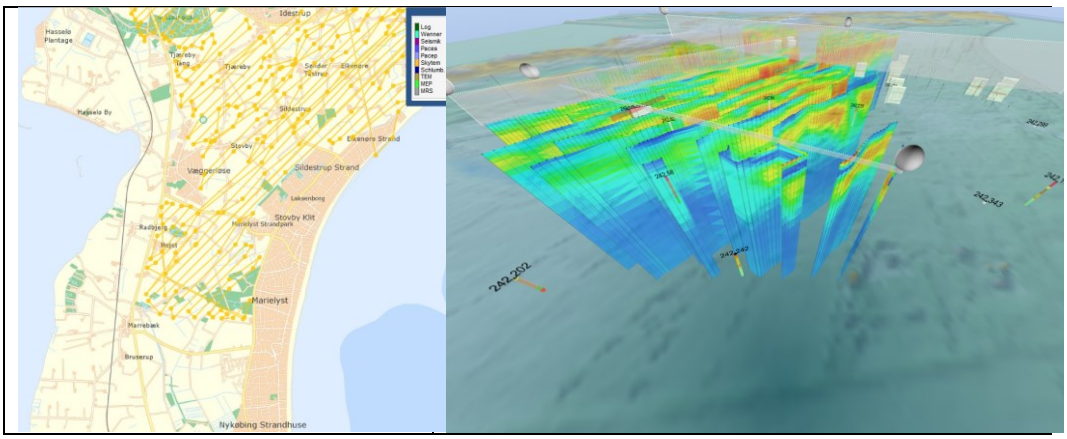


Figure 4-3. SkyTEM flight lines (left). SkyTEM resistivities displayed in 3D (right)

Observed variations in the measured resistivities can be result of variations in either or both salinity and lithology. SkyTEM data can provide valuable information about depth to saltwater and the resistivity of geological formations. The accuracy of the AEM measurements is not (so) high compared to e.g. wireline logging (GeofysikSamarbejdet, 2011). The SkyTEM measurements are limited to a certain depth, the “depth of investigations” (DOI) depending on the geology in the area, but usually between 60 and 120 meter below the ground surface (Figure 4-4).

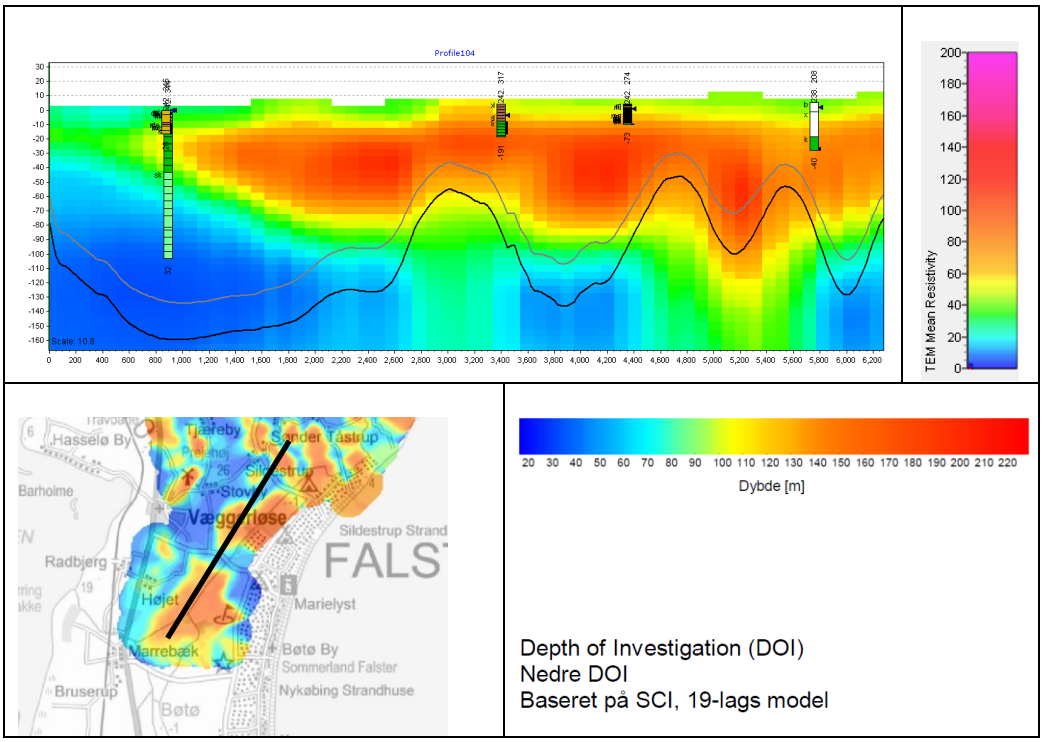


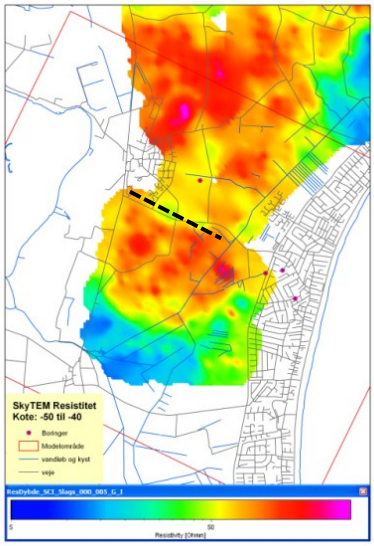
Figure 4-4. Cross section showing the upper (grey) and lower (black) Depth Of Investigation (DOI), and SkyTEM resistivities (upper panel). Map showing lower DOI and location of cross section (lower panel)

A recalibration of the existing groundwater model where the detailed variations in geology and salinity obtained from the SkyTEM measurements are interpreted and used for model calibration lies outside the framework of this project. The original idea was to interpolate an iso-resistivity surface corresponding to the upper boundary of the transition zone based on the corresponding resistivity and depth found with wireline borehole logging. This surface would define the top of the specific model layer below which the transition zone is present and decreasing hydraulic conductivities are introduced in the layers below.

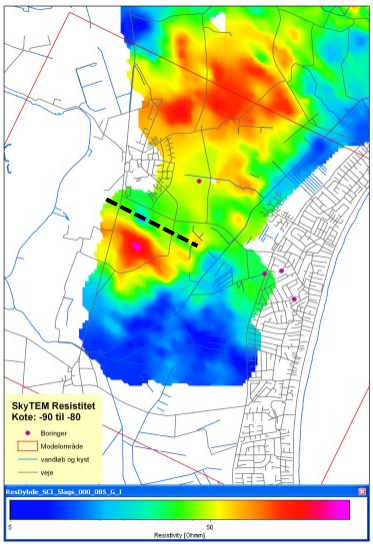
As described above the SkyTEM survey did not cover the entire model area, so it is not possible to use the SkyTEM data to define the bottom of the advective zone for the whole model area. Figure 4-4 shows how the transition zone extends deeper north of the borehole (242.344), from about -60 m at the borehole to -100 m around 1000 m north of the borehole where it levels out. The grey line in fig Figure 4-4 shows the upper DOI (Depth Of Investigation) and the black line the lower DOI. In areas where the SkyTEM data shows deeper zones of higher resistivities the DOI is at higher elevations in most but not all cases. In general the DOI varies between -60 m to -160 m a.s.l., with an average investigation depth of -100 to -120 m a.s.l.

Figure 4-5 shows SkyTEM resistivities for different 10 m depth intervals. In the central part of the area a west-east section is seen with lower resistivities compared to the surrounding areas. This area coincides with an area where saltwater is located deeper based on TEM soundings (Storstrøms Amt, 2000).

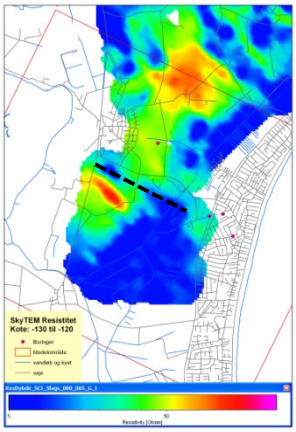
a.
Resistivities: -40 - -50 m



b.
Resistivities: -80 - -90 m



c.
Resistivities: - 120- -130m



d.
Depth to saltwater

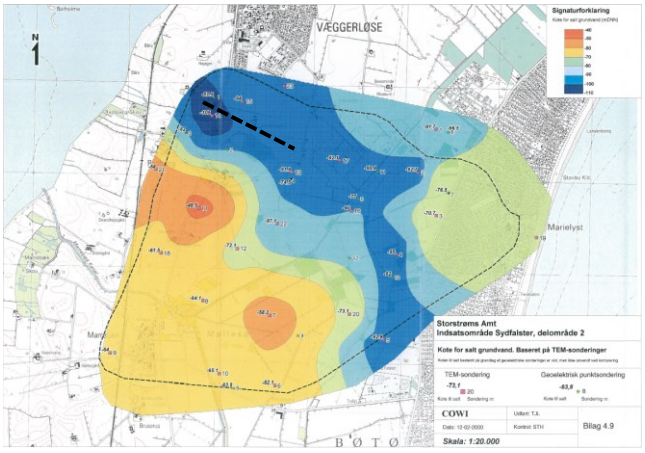


Figure 4-5. a., b., c. show SkyTEM for different 10 m depth intervals and d. is depth to saltwater based on TEM soundings (Storstrøms Amt, 2000). Black dotted line showing area with low resistivities and high depth to saltwater compared to surrounding areas.

Figure 4-6 shows a combined plot of formation conductivity from wireline logging and SkyTEM resistivities, a good overall agreement in the changes with depth of the conductivity/resistivity is seen.

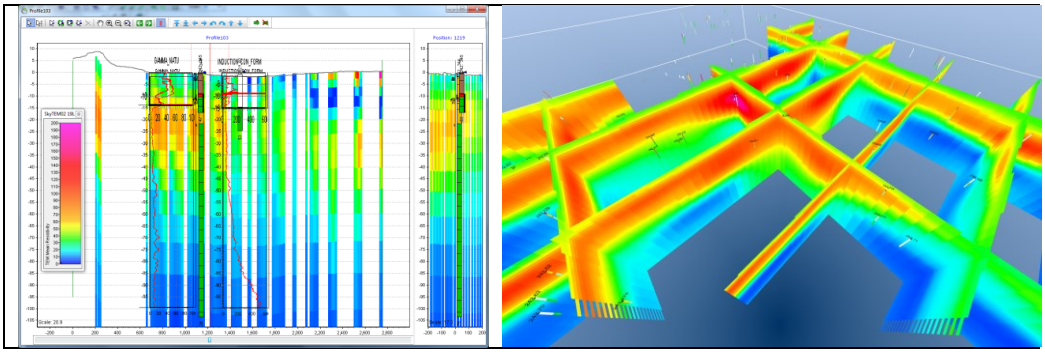


Figure 4-6. Combined plot of formation conductivity from wireline logging and SkyTEM resistivities (left), and example of 3D-interploation of SkyTEM resistivities (right)

With GeoScene3D it is possible to make a 3D-interploation of SkyTEM resistivities (Figure 4-6). With the 3D-grid of SkyTEM resistivities it is possible to draw 3D interpolated iso-surface of e.g. 20 ohm-m resistivity and compare it with the DOI (Figure 4-7).

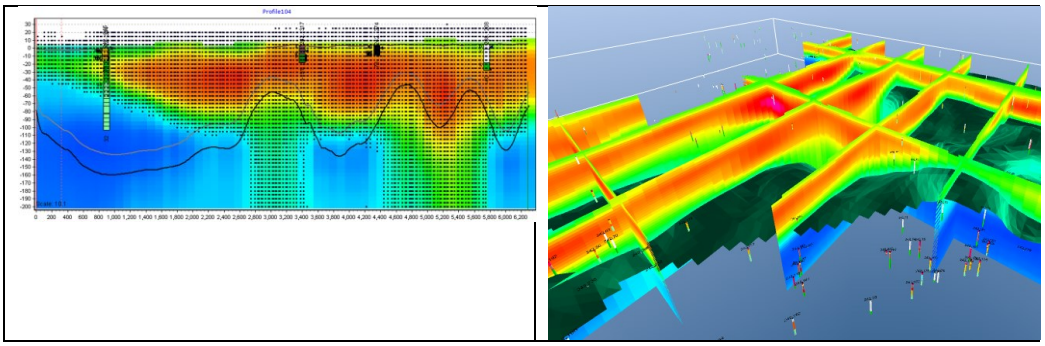


Figure 4-7. Combined plot of upper (grey) and lower (black) SkyTEM DOI and SkyTEM resistivities greater than 20 ohm-m (black dots) (left), and 3D interpolated iso-surface of 20 ohm-m resistivity (right)

The DOI (upper and lower) limits in certain areas the possibility to use SkyTEM data to estimate the areal distribution of the depth to the bottom of the advective zone, the top of the transition zone. The variations in the location of the DOI are related to the succession of soil resistivities.

Due to the partial coverage of SkyTEM data and the limitations from the SkyTEM DOI it was decided to limit the changes in the model setup for the recalibration to a change in the hydraulic conductivities for the advective zone base on the borehole wire-line logging and keep the elevation of the different conductivity zones as they are, it means keep the model layers as they were in the old model.

The results of MEP soundings in eastern part of the built-up area around the waterworks oldest well field indicates the existence of layers of mud and peat with higher formation conductivity (wireline logging data) and low formations resistivity (MEP soundings) between 5 to 9 meters depth compared to the clayey layers 9 – 15 m depth, top of chalk (Figure 4-8, Appendix 3) (Steiness, 2011). The mud and peat layers may contain trapped interglacial/post glacial saltwater as the conductivity is higher than seen in the clay layers.

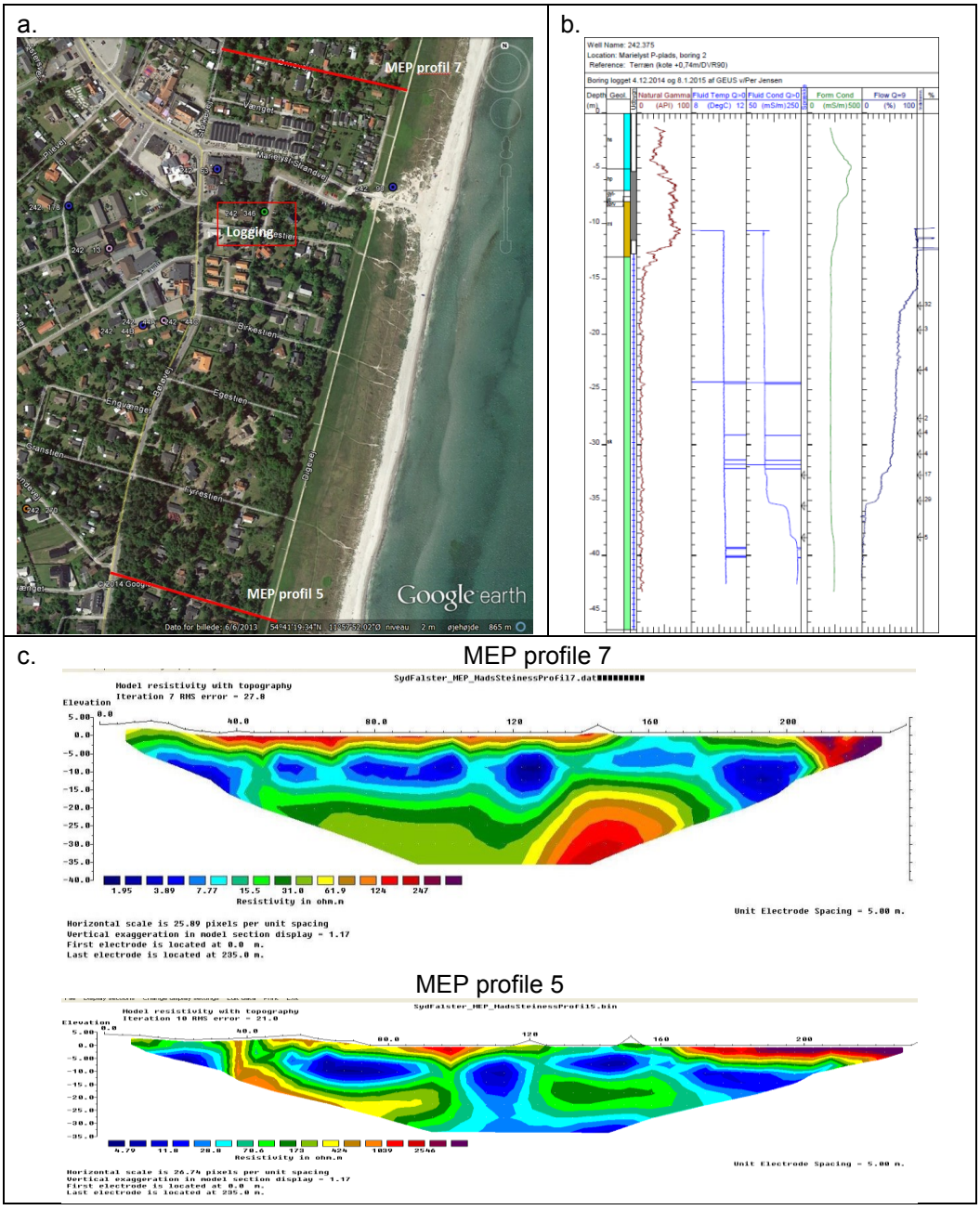


Figure 4-8. a. Location of borehole for wireline logging and MEP profile 5 and 7, b. wireline logging new borehole (242.375), c. MEP resistivity profiles no. 7 and 5

4.3 Recalibration of existing local model

4.3.1 Using geophysics

Based on the borehole geophysics the hydraulic conductivities of the hydro-stratigraphical model (HUF-model) layer 9 were increased to the same parameter values as layer 8 in order to increase the advective zone. To maintain a gradual decrease of hydraulic conductivities with depth all K-values were moved one layer up compared to the old model from layer 9 down to layer 14 (HUF model). The transition zone is now mainly HUF-model layer 9 and 10 (Table 4-2, 4-3, and 4-5).

Table 4-2. Hydraulic zones in the new adjusted model

Hydraulic zones (chalk)	Zone thickness	Layer	Hydraulic Conductivity
Quaternary	5-35	1-6	K_{sand}, K_{clay}
Advective	30	7-8	K_A
Transition	30	9-10	K_A -> K_T
Diffusion	120	11-15	K_D

Table 4-3. Hydraulic conductivities in the new adjusted model (HUF-model)

Layer	Lithology	Kx[m/d]	Kx[m/s]	Kz[m/d]	Kz[m/s]	Kz/Kx
1	fractured clay	0.43200	5.00E-06	0.04320	5.00E-07	0.1
1-6	sand	43.20000	5.00E-04	4.32000	5.00E-05	0.1
2-6	clayey till	0.08600	9.95E-07	0.00860	9.95E-08	0.1
7	chalk	4.32000	5.00E-05	0.43200	5.00E-06	0.1
8	chalk	4.32000	5.00E-05	0.43200	5.00E-06	0.1
9	chalk	0.86400	1.00E-05	0.08640	1.00E-06	0.1
10	chalk	0.43200	5.00E-06	0.04320	5.00E-07	0.1
11	chalk	0.08640	1.00E-06	0.02160	2.50E-07	0.25
12	chalk	0.00864	1.00E-07	0.00864	1.00E-07	1
13	chalk	0.00600	6.94E-08	0.00600	6.94E-08	1
14	chalk	0.00400	4.63E-08	0.00400	4.63E-08	1
15	chalk	0.00200	2.31E-08	0.00200	2.31E-08	1

The variations in hydraulic conductivities reflects the proposed zonation of salinity with depth from an advective zone in the upper part of the chalk formation, a transition zone below which the diffusion processes are dominating (Larsen et al. 2006; Bonnesen et al., 2009).

The gradual change (with depth) in hydraulic conductivities from layer to layer follows the recommendations for making the variable density modelling process more smooth.

The initial concentrations in the new model setup were adjusted according to measurements from wire line logging of the formation electrical conductivity (Table 4-4).

Table 4-4. Initial concentrations in old and new adjusted model (Flow-model)

Model layer	Conc. zone	Measured EC-bulk (mS/m)	Calc. conc.* (mg TDS/l)	Top layer elev. (m)	Bottom layer elev. (m)	Init. Conc. new model (mg TDS/l)	Init. Conc. old model (mg TDS/l)
1-14	1					10,000	10,000
15	1	(25)	1,000	-43	-46	10,000	10,000
16	1	80	3,200	-46	-51	10,000	10,000
17-22	1					10,000	10,000
23	2	350	14,000	-90	-101	12,000	10,000
24	3	550	22,000	-101	-112	14,000	10,000
25	4			-112	-123	16,000	10,000
26	5			-123	-134	18,000	10,000
27	6			-134	-145	20,000	12,000
28	7			-145	-156	22,000	16,000
29	8			-156	-167	24,000	20,000
30	9			-167	-178	26,000	25,000

* $C_i = 10*EC_{bulk}*F$, formation factor F = 4
* C_i layer 15 is based on concentrations from abstraction wells not affected by saltwater (well 242.320)

The adjusted and recalibrated model shows a slightly better model performance (Table 4-6). Model simulation with the adjusted (re-calibrated) model indicates that the advective zone extends even deeper. It is also seen that for this study area and model setup the depth of the saltwater interface changed significantly from after 50 years of simulation to after 340 years of simulations (Figure 4-9). This model phase starts in 1960 with the initiation of the major groundwater abstraction in the area in 1960.

It should be further analysed if steady state conditions has been achieved after 340 years of simulation, and how the transition and initial conditions from one model phase to the other affect the model results. Model results indicate that the salinity gradient of the transition zone is modelled quite well, but it is not located deep enough, about 15 meters too high for this specific location around borehole 242.344 (Figure 4-9). In the deeper part of the diffusion zone the modelled salinity has not changed from the initial concentrations, because the total model simulation period of 3.700 years is very short compared to the timescale of the diffusion processes (Bonnesen et al., 2009).

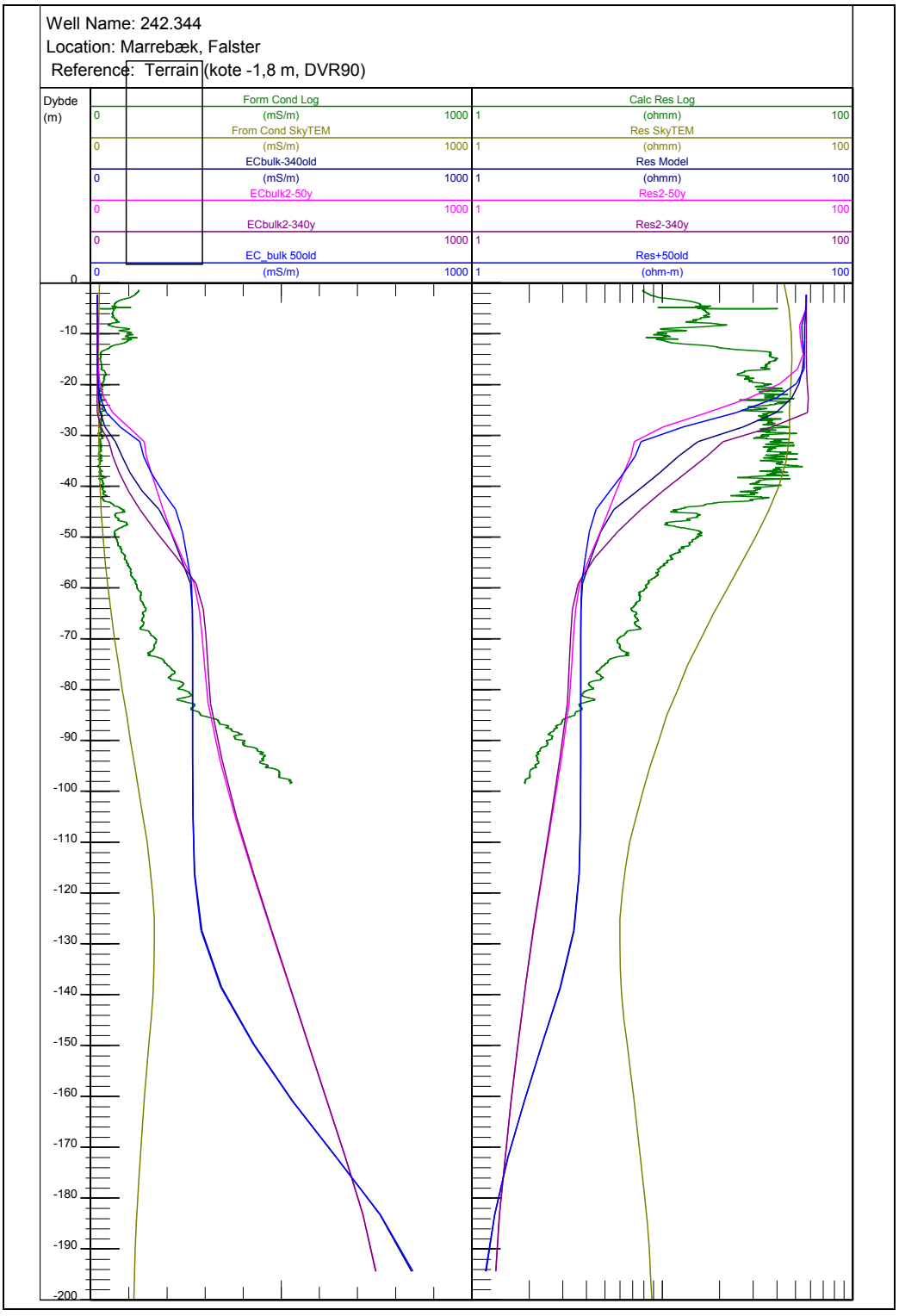


Figure 4-9. Modelled formation conductivities (left) and resistivities (right) compared to wireline logging and SkyTEM survey: 1) before recalibration: blue "… 50old", 2) after recalibration, at 50 years model simulations: red "…50y", and at 340 years model simulations: purple "…340y"

4.3.2 Using pilot points

Calibration using pilot points was tested in six different combinations of model parameters.

Test 1 and 2 were performed with hydraulic conductivities from the old model setup. Tests 3 to 6 were performed on the adjusted recalibration model (Table 4-5). 'Pre-pp07-3tr' (text marked in grey) shows the calibration statistics after implementation of the changed distribution of hydraulic conductivities to the new model. The best calibration statistics has been achieved with the smaller variogram alpha (α) (Table 4-6).

Table 4-5. Model recalibration setup (in grey 'Pre-pp07-3tr' see text above)

Calibration no.	Remarks
Manual calibration	
MarHUF008	old calibrated setup
MarHUF009	k-values layer 9 = k-values layer 8 (larger advective zone); layer 9-13: k-values moved 1 layer down
Pilot points calibration	
pp07-1tr	pilot points layer 7, 33 kx PP
pp07-2tr	change PP min, max, start value
pre-pp07-3tr	before PP run, changed Kx zone 5 and below, larger adv zone
pp07-3tr	after PP run, changed Kx zone 5 and below, larger adv zone
pp07-4tr	as -3; 67 kx PP, 39 kz PP, keep kx/kz ratio, 24 super PP
pp07-5tr	as -4; variogram α=750 (prev. run α=1500 m), 24 super PP
pp07-6tr	as -4; variogram α=2000 (prev. run α=1500 m), 24 super PP

Table 4-6. Calibration statistics for recalibration of model

Calibration no.	Residual Mean	RMS Error	dHmax	Sobs	ME/ dHmax	RMS/ Sobs	RMS/ dHmax
Manual calibration							
MarHUF008	-0.15	1.53	7.44	1.52	-0.02	1.01	0.206
MarHUF009	-0.10	1.50	7.44	1.50	-0.01	1.00	0.202
Pilot points calibration							
pp07-1tr	0.42	1.50	7.44	1.44	0.06	1.04	0.202
pp07-2tr	0.33	1.39	7.44	1.29	0.04	1.08	0.187
pre-pp07-3tr	0.50	1.64	7.44	1.56	0.07	1.05	0.220
pp07-3tr	0.44	1.45	7.44	1.38	0.06	1.05	0.195
pp07-4tr	0.50	1.64	7.44	1.56	0.07	1.05	0.220
pp07-5tr	0.39	1.30	7.44	1.23	0.05	1.06	0.175
pp07-6tr	0.50	1.43	7.44	1.33	0.07	1.08	0.192
Target crit.	-	-	-	-	0.010	1.650	0.025

The results of pilot calibration of horizontal hydraulic conductivities of model layer 7 are seen in Figure 4-10.

The calibration statistics from the pilot point calibration is similar to the results of the manual calibration based on the results of the geophysics (Table 4-6). For the injection model simulation tests the model setup from the manual recalibrated groundwater model is used.

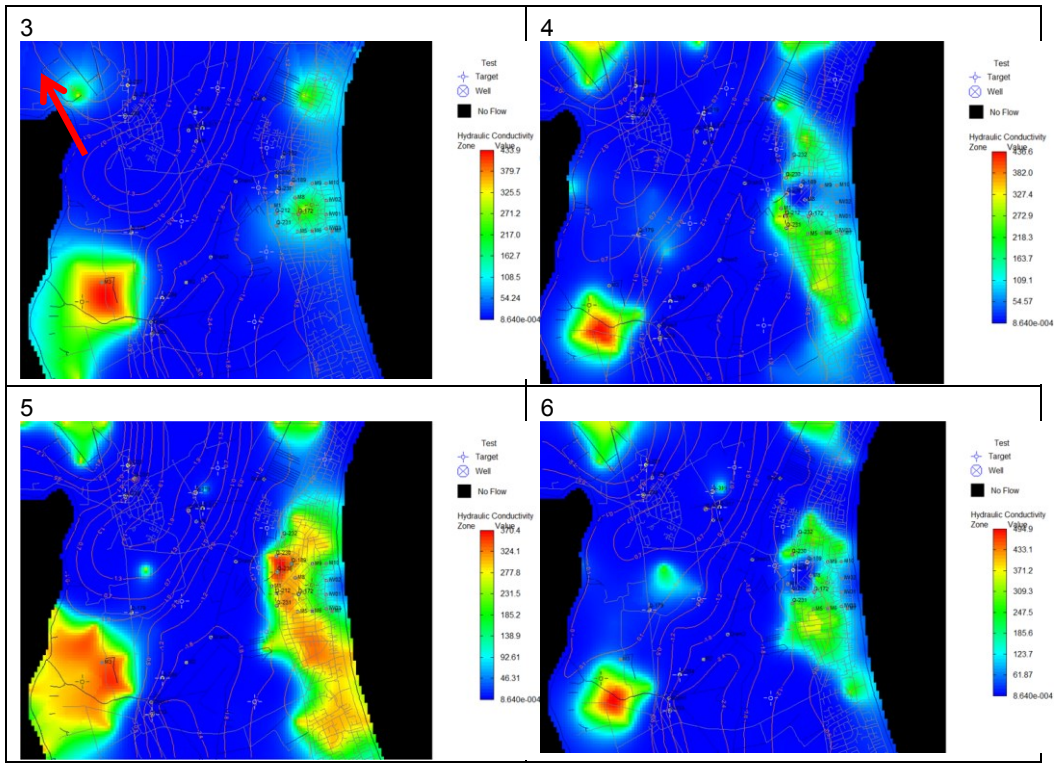


Figure 4-10. Results of pilot calibration of horizontal hydraulic conductivities of model layer 7, the number refers to the calibration setup Table 4-5. Red arrow direction is north.

The results of the pilot point calibration tests indicate that a smaller correlations length results in larger heterogeneity (Figure 4-10).

4.4 Injection wells in local model

Figure 4-11 shows the results of the injection tests for two groundwater abstraction wells in the area, well no. 172 nearest the injection wells and well no. 212 further away (Figure 3-9). Table 4-6 gives an explanation for the legend in Figure 4-11. An explanation of the model setup of the injection simulations is given in section 3.3.3.

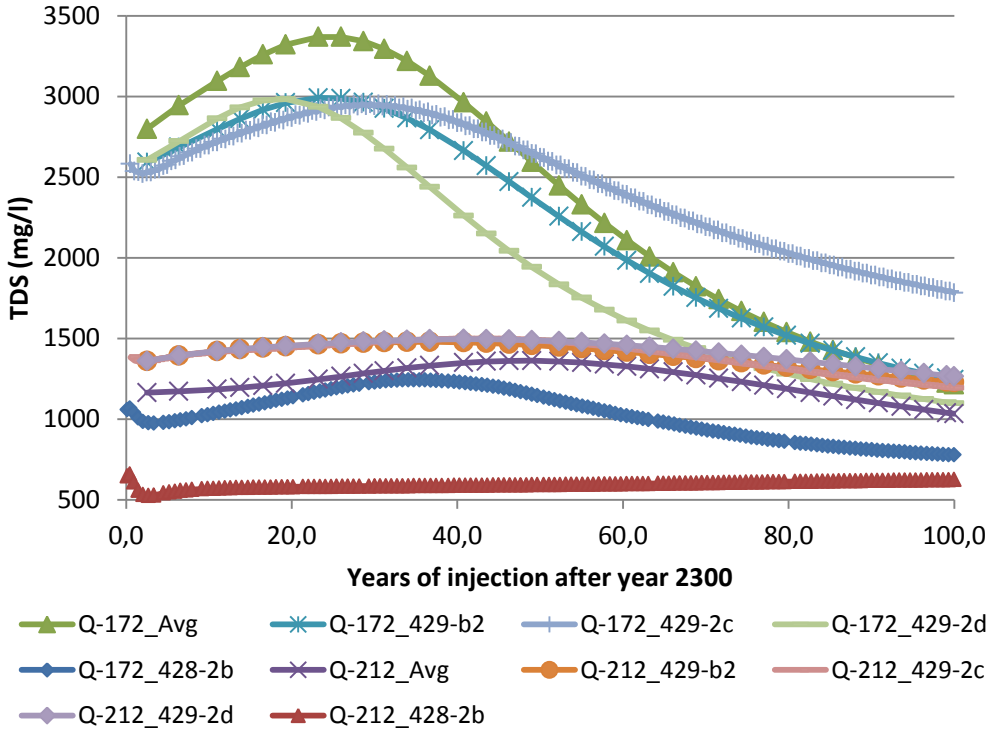


Figure 4-11. TDS concentrations in well 172 and 212 during 100 years of injection, explanation of legend see Table 4-7

Table 4-7. Explanation of the legend to Fig. 4-11 (above)

Figure 4-11 Legend	Well	Climate Scenario	Injection (m^3/d)	Remarks
Q-172_Avg	242.172	3	3 x 68	Before recalibration
Q-172_429-b2	242.172	3	3 x 68	After recalibration
Q-172_429-c2	242.172	3	½ x 3 x 68	Alternating pumping
Q-172_429-d2	242.172	3	1 x 68	Horizontal well
Q-172_428-b2	242.172	2	3 x 68	Climate scenario 2
Q-212_Avg	242.212	3	3 x 68	Before recalibration
Q-212_429-b2	242.212	3	3 x 68	After recalibration
Q-212_429-c2	242.212	3	(½ x) 3 x 68	Alternating pumping
Q-212_429-d2	242.212	3	1 x 68	Horizontal well
Q-212_428-b2	242.212	3	3 x 68	Climate scenario 2

After the model recalibration the TDS concentration in well 172 is lower than for the old model before the start of the injection period. At the end of the injection period the TDS concentration is the same for the two models. For both models the maximum concentration is seen after 25

years of injection where after the concentration decreases to around half of the initial TDS concentration. For well 212 the TDS concentration is higher in the recalibrated model than the old model before start of injection, and it continues to be so throughout the whole injection period. The maximum concentration is seen after 35 years in the recalibrated model, and 15 years later in the old model.

For the alternating injection a similar pattern is seen, but with a later maximum concentration and a higher end-concentration after 100 years of injection, the total injection is 50% of the continuous injection. With the horizontal injection well with a continuous injection of 33% of the total vertical well injection volume the increasing TDS concentration is reversed at an earlier point in time, after 20 years of injection, and the end-concentration after 100 years of injection is the lowest of the shown scenarios for well 172. Figure 4-12 shows the differences in freshwater distribution around the injection wells comparing vertical and horizontal injection wells.

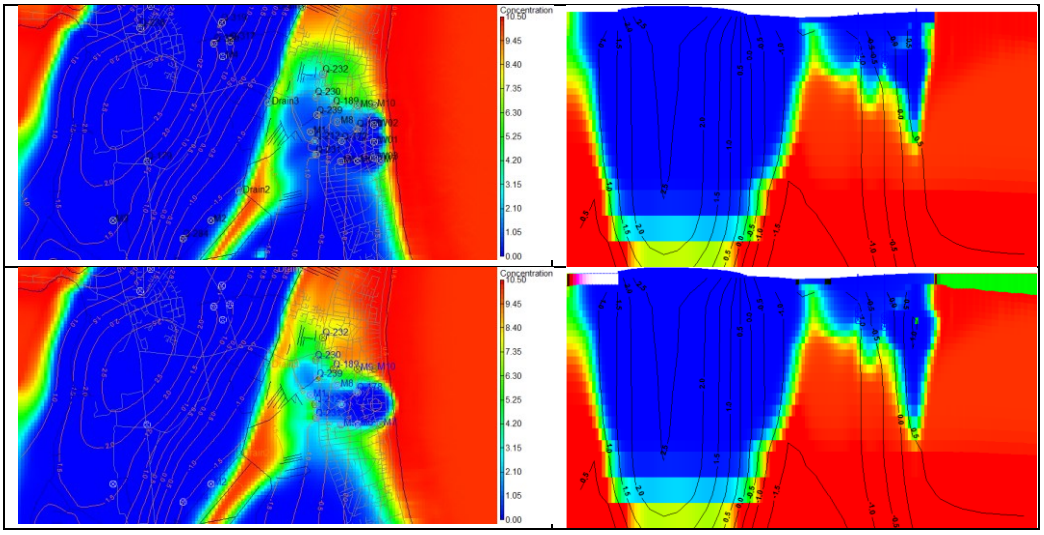


Figure 4-12. Plan view and cross section of freshwater distribution around the vertical (upper) and horizontal injection well(s) (lower)

For the less severe climate scenario 2 the TDS concentrations are lower before the start of the injection compared to scenario 3. The increasing TDS concentration in well 172 is reversed after 35 years, and ends at 50% of the start concentration after 100 years of injection. The injection has no effect on well 212, where the TDS concentration continues to increase during the whole injection period.

Notice that the TDS concentrations are the additional seawater TDS contribution to the freshwater. In the model the freshwater TDS concentration is set to be zero.

Water from the injection wells will end up in the groundwater abstraction wells. A tracer simulation indicates that after 100 years of injection half of the groundwater abstracted from the waterworks abstraction well closest to the injection well origins from the injection well. It suggests that the injection wells act more like storage, recovery and dilution of saltwater than as a hydraulic barrier against saltwater intrusion (Figure 4-13).

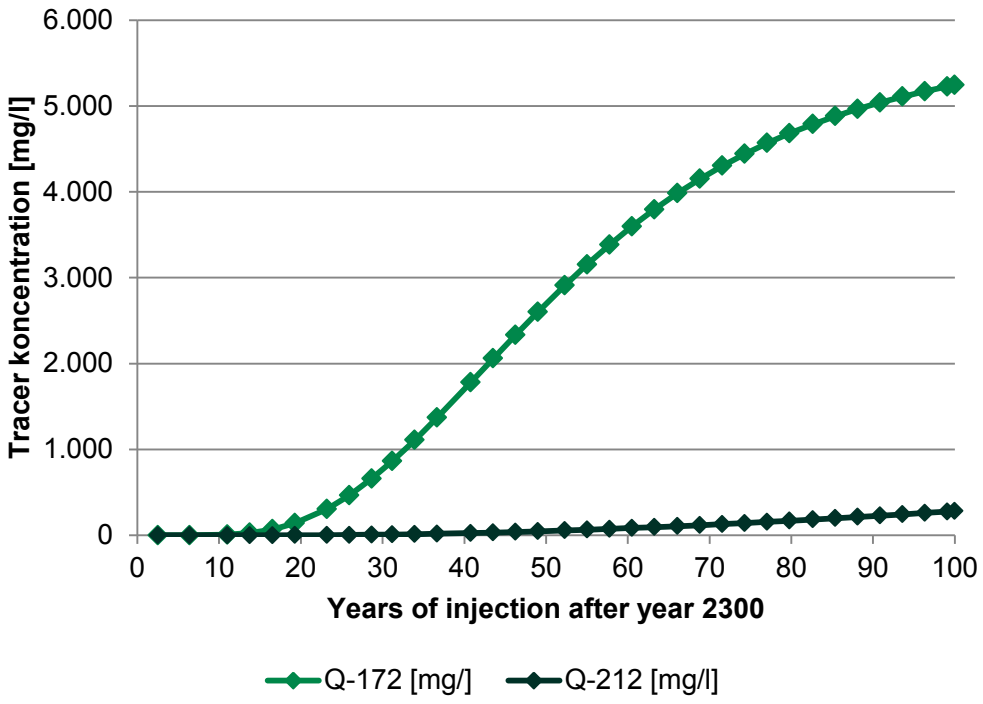


Figure 4-13. Tracer-concentration in two abstraction wells from 100 years of injection (scenario 3-b2). Concentration of injected water: 10.500 mg TDS/l.

4.5 Generic models

The two generic models, the one with diffusion only and the one also with seawater intrusion, see Table 3-11, shows significant differences in saltwater transport towards the pumping well (Figure 4-14). In the generic model only including the diffusion processes all water are flowing towards the drain and the pumping well causing the saltwater concentration in the pumping well to stabilize at a lower level compared to the generic model also including seawater intrusion (Figure 4-15).

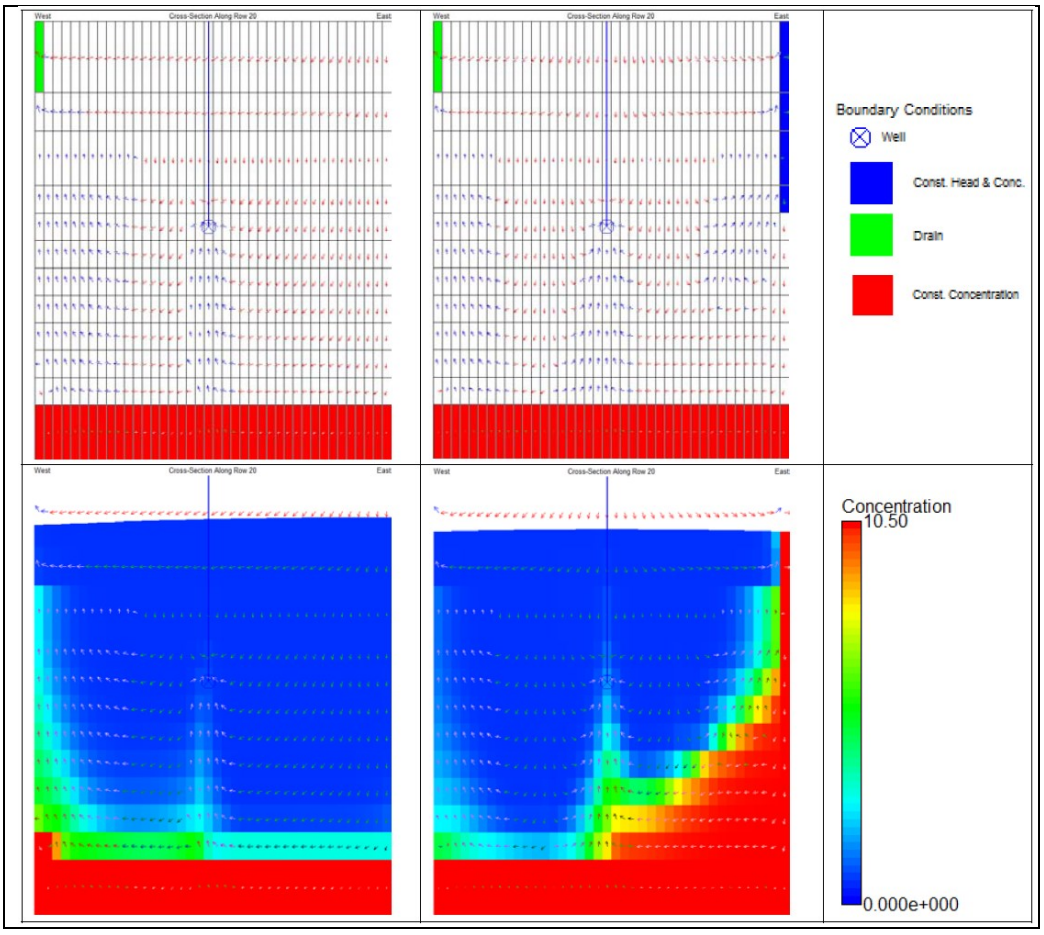


Figure 4-14. Cross sections showing flow vectors (upper) and TDS concentrations (lower) for generic model scenario 4 (left) and scenario 5 (right)

Figure 4-15 also shows that a higher diffusion coefficient, a lower hydraulic conductivity of the clay layer (the aquitard) covering the chalk aquifer, or a higher saltwater concentration in the bottom layer all results in at higher TDS concentration in the pumping well compared to the standard values (Table 3-11, scenario 1-4). In the generic models it takes more than 400 years after the start of the pumping before the TDS concentration in the well reach a constant level. The pumping well in the generic model with seawater intrusion reaches a much higher saltwater concentration after 400 years of pumping compared to the genetic models with only diffusion of saltwater from the bottom layer. It is also seen in Figure 4-15 (scenario P1-gm05) that it takes about 250 years before saltwater originating from the seawater intrusion starts to affect the TDS concentration in the pumping well. This generic model setup does not represent a natural situation where a saltwater wedge normally has developed inland from the sea before the start of a groundwater abstraction. But the generic models illustrate the magnitude of saltwater origination from the two different sources, diffusion and seawater intrusion.

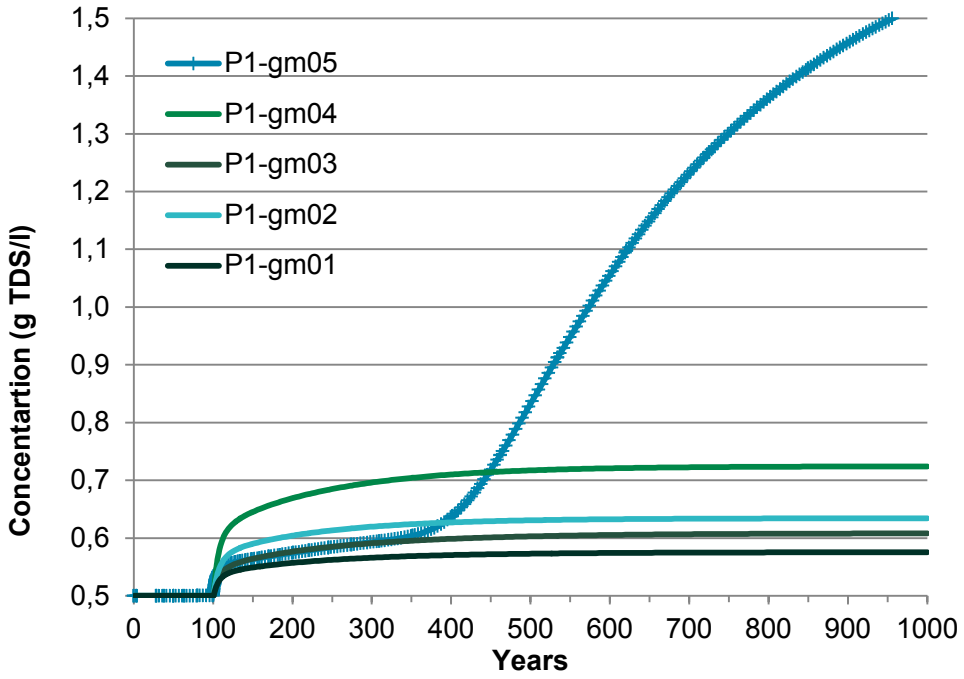


Figure 4-15. Concentration of TDS in pumping well P1 for the generic models scenario 1-5. Pumping from P1 started after 100 years (gmXY refers to the scenario number Table 3-11)

For the generic models scenarios 5-11 (Table 3-11) the results are shown as TDS concentration for the pumping well (Figure 4-16).

For scenario 6, where the seawater boundary condition is removed from the top layer of the chalk aquifer, little or no effect is seen of the seawater intrusion. In scenario 7 and 8, where a lower hydraulic conductivity is applied to the lower part of the aquifer (Figure 4-17), we also see little or no effect of seawater intrusion. The main groundwater flow towards the pumping well is taking place in the upper high conductivity layers, whereas low-conductivity layers is holding back the intruding saltwater (Figure 4-17).

Scenario 11, with a lower hydraulic conductivity of the clay aquitard, shows a higher TDS concentration in the pumping well compared to the standard scenario 5. More groundwater is drained away in the upper sand layers and less freshwater is recharging the chalk aquifer.

For the two climate change scenarios 9b and 10 (sea level rise of 0,5 m and 15% increase in groundwater recharge), the sea level rise results in an large increase in saltwater in the pumping well. An increase in recharge will for a period result in a small reduction of the TDS concentration in the pumping well, but after a longer period of pumping the concentration is the same as for the standard generic model, scenario 5 (Figure 4-16).

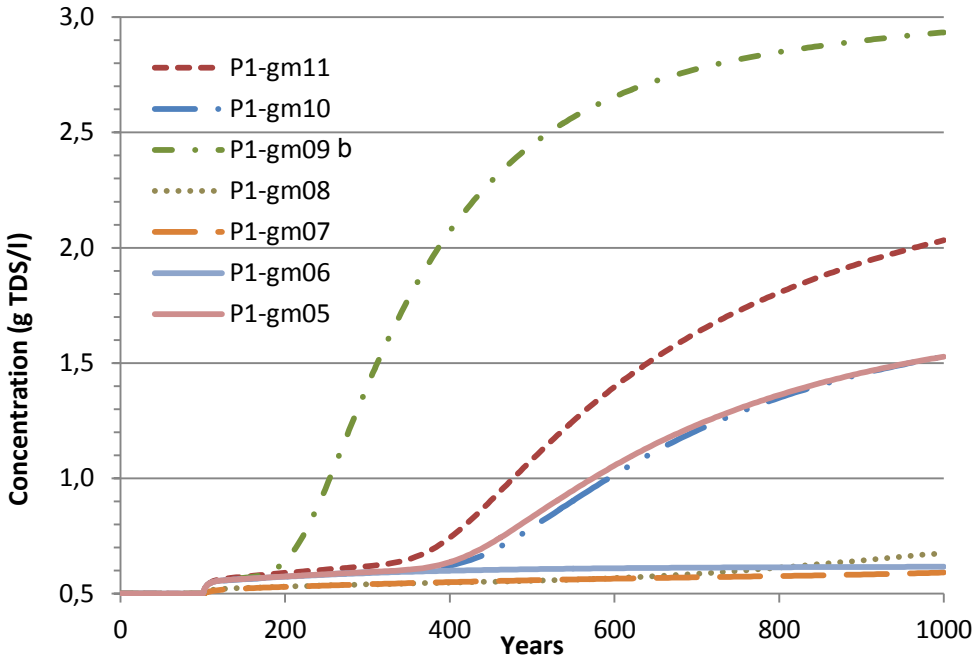


Figure 4-16. Concentration of TDS in pumping well P1 for the generic models scenario 5-11. Pumping from P1 started after 100 years.

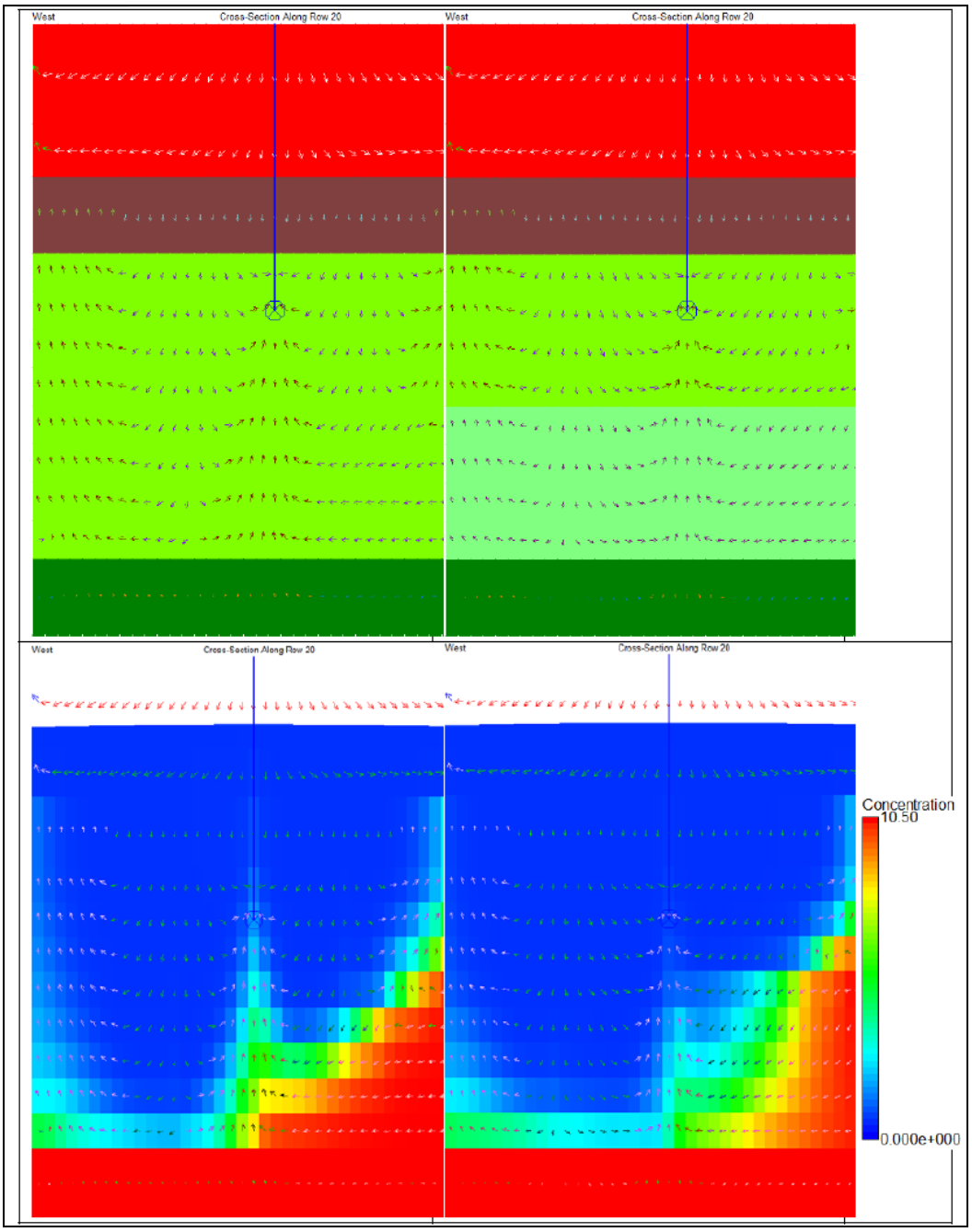


Figure 4-17. Cross sections showing flow vectors and geological layering (upper) and TDS concentrations (lower) for generic model scenario 5 (left) and scenario 8 (right)

4.5.1 Injection wells in generic models

Injection of water with no or very little concentration of salt will dilute and reduce the TDS concentration in the pumping well slightly, scenario 12 (Figure 4-18). Focusing the injection of water in one well compared to three wells (scenario 12b and 14) seems to give a slightly lower concentration for these geologically homogeneous generic models (Figure 4-18). The high injection rates used in scenario 12, 12b and 14 keeps the TDS concentration in the pumping well at low levels comparable to the concentrations found in the generic diffusion models (Figure 4-15).

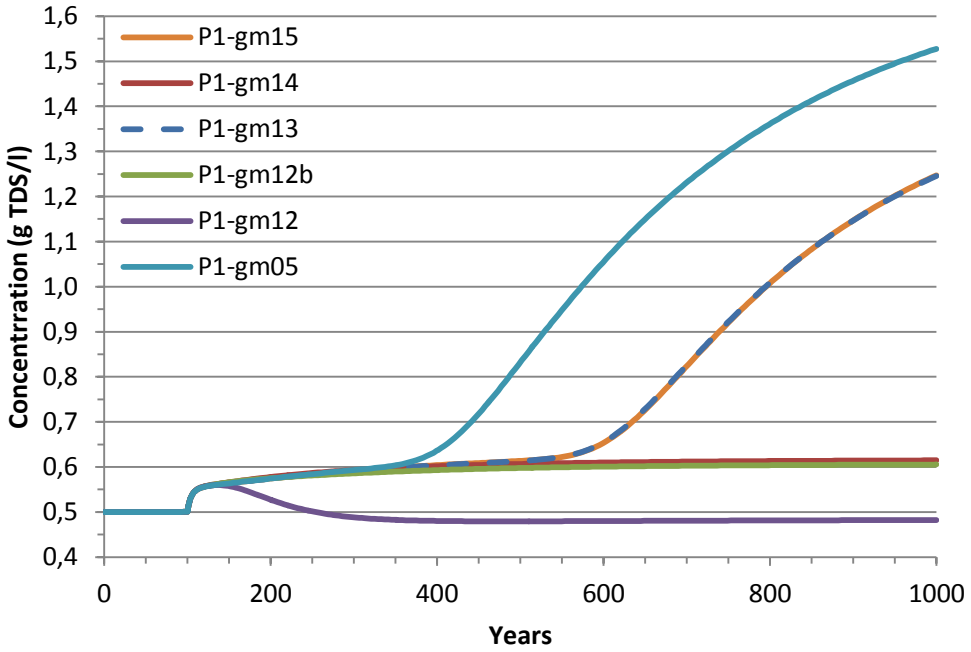


Figure 4-18. Concentration of TDS in pumping well P1 for the generic models scenario 12-15. Pumping from P1 started after 100 years.

A reduced injection rate to 25% (scenario 13 and 15) is not enough to prevent the TDS concentration in the pumping well to increase after 500 years of injection. Compared to the standard generic model scenario 5 the reduced injection rate will delay the increase in TDS concentration in the pumping well by 200 years (Figure 4-18).

Using a vertical injection well (scenario 13) or a horizontal injection well (scenario 15) results for these generic models in the same TDS concentration in the pumping well (Figure 4-18). Only very minor differences in concentration distribution are seen between the vertical and the horizontal well (Figure 4-19).

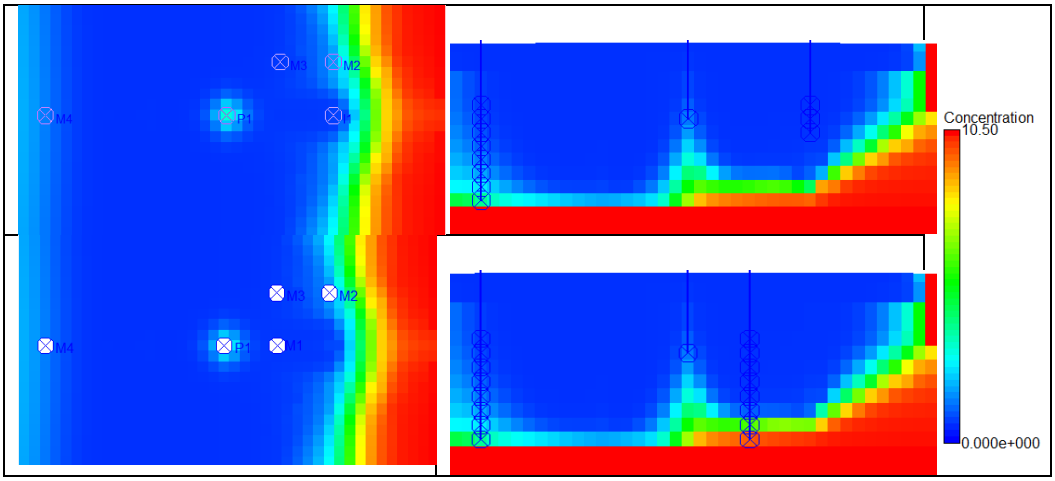


Figure 4-19. Plan view and cross section of freshwater distribution around the vertical injection well scenario 13 (upper) and horizontal injection well scenario 15 (lower)

4.5.2 Monitoring system designs

Results from the monitoring well, M1, placed between the pumping well and the sea or the injection well (Figure 3-10, 3-11) are shown in this section.

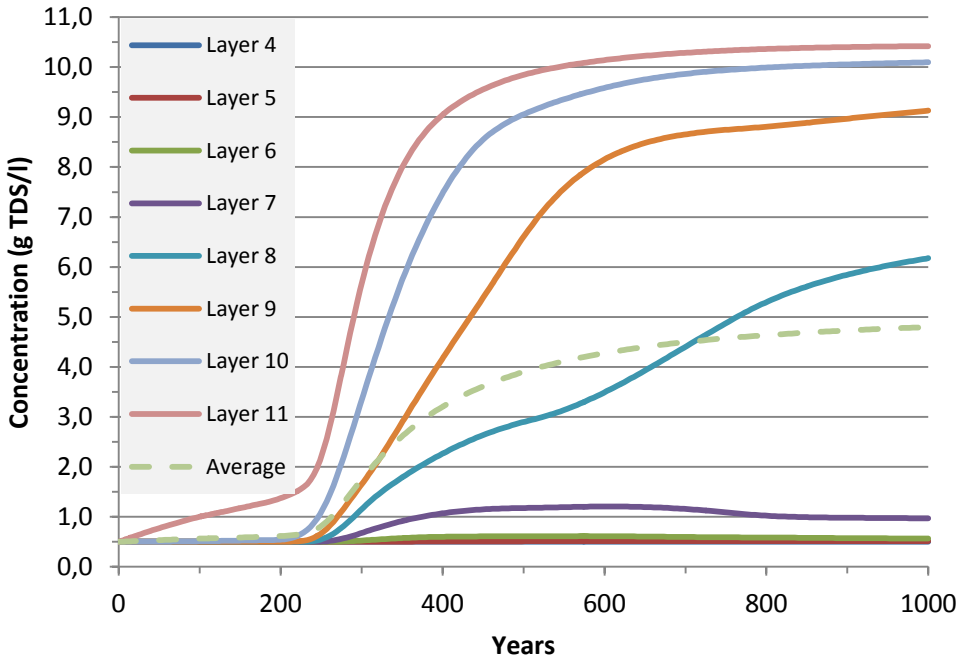


Figure 4-20. Concentration of TDS in monitoring well M1 for different model layers, generic model scenario 5. Pumping from P1 started after 100 years

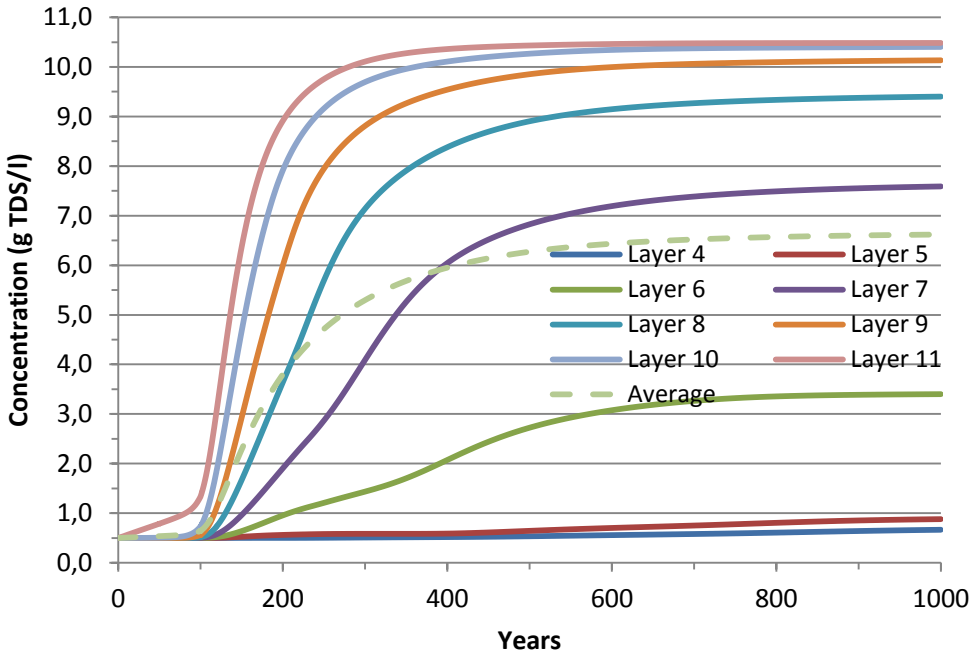


Figure 4-21. Concentration of TDS in monitoring well M1, generic model scenario 9. Pumping from P1 started after 100 years

Results from the monitoring well M1 shows that concentrations are higher in deeper layers, and that increases in TDS concentrations are seen at an earlier point in time in the deeper layers compared to the upper layers. Some upper layers are unaffected by saltwater intrusion. This picture also applies to the sea level rise scenario 9, where more layers are affected compared to scenario 5 (Figure 4-20 and 4-21). As mentioned earlier the generic model setup does not represent a natural situation where saltwater wedges normally has developed inland from the sea before the start of a groundwater abstraction, but are used to illustrated general cases.

Though the effect of a vertical injection well and a horizontal well seems to be the same for the TDS concentration in the pumping well (Figure 4-18), the saltwater concentration at different depths shows differences for the two scenarios. For the vertical injection well the seawater intrusion seems to arrive later to the monitoring well and only the three deeper layers are affected, whereas for the horizontal well more layers are affected and at an earlier time. But the horizontal well seems to prevent the saltwater from diffusion in the deepest layer 11 to reach the monitoring well (Figure 4-22 and 4-23).

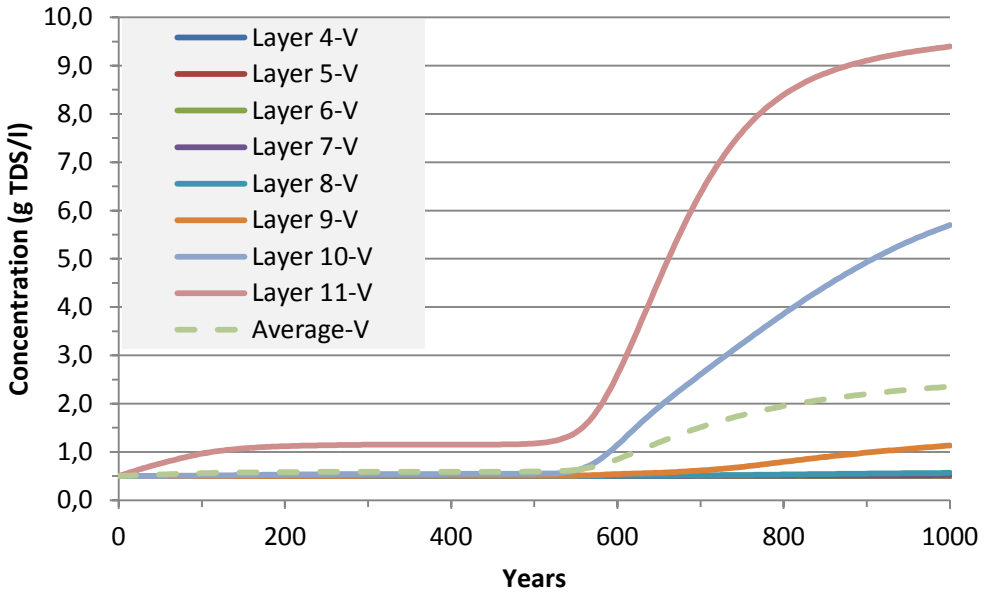


Figure 4-22. Concentration of TDS in monitoring well M1, generic model scenario 13 with 1 vertical injection well, I1. Injection was started at year 0

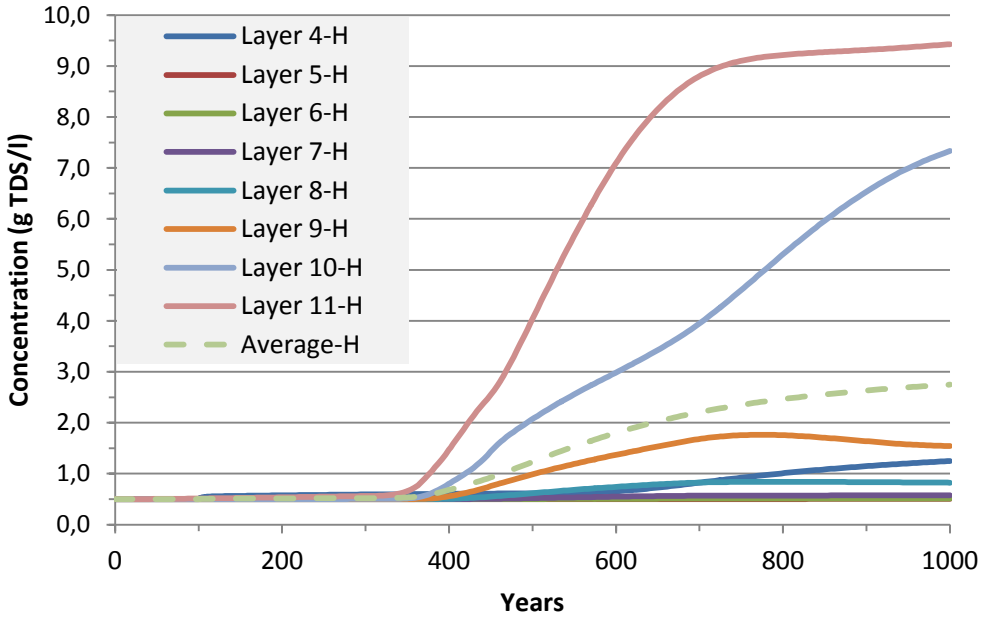


Figure 4-23. Concentration of TDS in monitoring well M1, generic model scenario 15 with 1 horizontal injection well. Injection was started at year 0

5. Discussion

5.1 Using geophysics for model calibration

Model results indicate that salinity gradient of the transition zone is modelled quite well, but it is not located deep enough, 15 meters too high. In the deeper diffusion zone the modelled salinity has not changed from the initial concentrations, because the total model simulation period of 3.700 years is very short compared to the timescale of the diffusion processes (Bonnesen et al., 2009).

It is also seen that for this study area and model setup the depth of the saltwater interface changed significantly from after 50 years of simulation to after 340 years of simulations, starting with the initiation of the major groundwater abstraction in the area in 1960. It should be further analysed if steady state conditions has been achieve and how the transition and initial conditions from one model phase to the other affect the model results.

5.2 Pilot point calibration

As most of the groundwater head observations in the study area are from the upper most part of the chalk aquifer, corresponding to one layer in the model, it has only been possible to perform a regular pilot point calibration for this model layer, layer 7. The calibration statistics did not show better model performance than for the manual calibration. Only a few tests were performed and the following suggestions for future work with the pilot point calibration might improve the calibration with the pilot point calibration techniques:

- The number and distribution of pilot points, both horizontally and vertically
- Analyse the effect of bounds / constraints on the pilot point parameters
- Including other hydraulic conductivity zones (model layers) in the calibration, i.e. the possible effect of changes in hydraulic conductivities for one zone (pilot point calibrated zone) might affect the hydraulic conductivity of other zones
- The used model setup was a freshwater-only sub-model of the variable density model of the whole study area.
    - A pilot point calibration of the full variable density model for the study area is too computer time demanding that is it not realistic in the present project (Langevin and Zygnerski, 2013)
    - A more accurate delineation of the transition zone between freshwater and saltwater to define no-flow zones in the freshwater model used for pilot point calibration.

A general pattern in the pilot point calibration tests is that the hydraulic conductivities are higher in the part of the aquifer located below the barrier island in the eastern part of area. It is also in this area that two of the water works three well fields are located. The estimated hydraulic conductivities for that area varies between 100 and 300 m/d ($1.2*10^{-3}$ – $3.5*10^{-3}$ m/s), which are relatively high values.

Around the third well field in the central/northern part of the area the hydraulic conductivities are the same as in the major part of the model area.

The interpretation of the results of the pilot point should be linked to a plausible geological structural explanation. An east – west structure is seen in the central part of the barrier island where a lowering in the surface of the chalk aquifer may coincide with a fault structure (Mathiesen et al. 2009), and a lower hydraulic conductivity according to the pilot point calibra-

tion. The high conductivity values seen in the southwest of the area is most probably related to presence of an overlaying thick clay layer and an associated dip of the top chalk layer influencing the pilot point calibration (Figure 5-1).

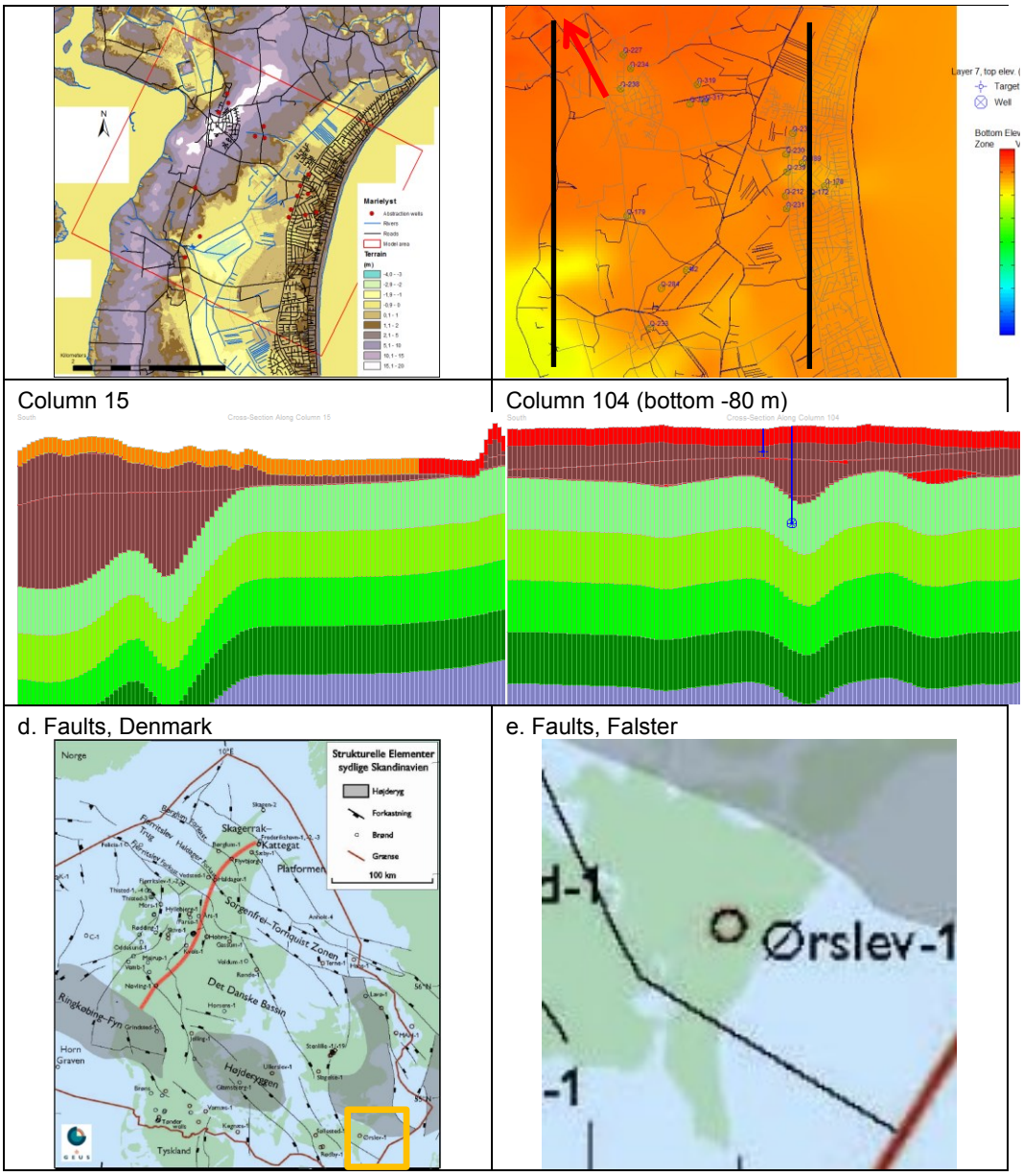


Figure 5-1. Topography, geological layering and faults in the study area

5.3 Injection wells as hydraulic barriers, local models

The present model studies have shown that a horizontal well with a length of 150 m injecting 1/3 of the water injected by three vertical wells with horizontal spacing of 300 m have a larger effect on preventing seawater intrusion. With a large vertical spacing between vertical injection wells a lot of injected water may be wasted. One other aspect that might affect the effectiveness is the vertical injection interval. For the horizontal well the injection is limited to one model layer

of 3 meters thickness with high hydraulic conductivity, whereas the vertical injection wells inject water over at depth interval of 18 meters including model layers both with high and lower hydraulic conductivity.

Geological heterogeneities was only included in the injection tests to the extent of using the concept of Hydrogeologic-Unit Flow (HUF-package) transforming geological layers of varying thickness but each with a uniform hydraulic conductivity to (nearly) vertical flow model layers. The results of the pilot point test calibration suggest that it could be valuable to implement the more heterogenetic distribution of hydraulic conductivities found in the pilot point calibration in order to optimize water injection.

Alternating water injection was tested by applying an injection scheme of half year injection and half year pause. For the applied model and injection setup is seems to have very little effect on the efficiency as a hydraulic barrier. The injection rate seems to have the major effect.

Water from the injection wells will end up in the groundwater abstraction wells. A tracer simulation indicates that after 100 years of injection half of the groundwater abstracted from the waterworks abstraction well closest to the injection well origins from the injection well. It suggests that the injection wells act more like storage, recovery and dilution of saltwater than a hydraulic barrier against saltwater intrusion.

5.4 Generic models

The used generic models were based on the local model setup for the Marielyst area. The generic models have shown that saltwater from diffusion from deeper layers may increase the TDS concentrations in pumping wells by 20-40%. Depending on the aquifers contact to the sea (the boundary conditions), the effect of seawater intrusion may vary from a contribution like what was seen from diffusion to triple the TDS concentrations in pumping wells.

A lower hydraulic conductivity of the deeper part of the chalk aquifer is seen to reduce the effect of seawater intrusion in the pumping well.

Sea level rise is seen to have a much greater effect on the TDS concentration in the pumping well compared to an increase in groundwater recharge.

The test of different injection well setups with generic models indicates that the injection rate is of most importance. The relative insignificance of using 1 or 3 injections wells, or a horizontal versus a vertical well, may be because the generic model has a very homogenous geological setup.

Monitoring wells should be deep enough to be able to measure the (changes in) seawater intrusion below the freshwater lens. Monitoring wells should have monitoring points at different depth in the aquifer. Even in the geological homogenous generic models show major differences is in TDS concentrations at different depths.

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Appendices

Appendix 1: SWIM23 Extended abstract

Appendix 2: SWIM23 Poster

Appendix 3: Well record for borehole DGU nr. 242.375

Appendix 1: SWIM23 Extended abstract

Evaluating hydraulic barriers for reducing and controlling saltwater intrusion in a changing climate

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ABSTRACT

Groundwater abstraction from coastal aquifers is vulnerable to sea level rise, increasing groundwater abstraction and drainage because they may potentially impact saltwater intrusion and hence groundwater quality depending on the hydrogeological setting. In the present study the impacts of sea level rise, drainage systems and changes in groundwater abstraction are quantified for an island located in the Western Baltic Sea using the modeling packages MODFLOW/MT3DMS/SEAWAT. Increasing chloride concentrations have been observed in several abstraction wells indicating that saltwater intrusion is ongoing. The water resources on the island are abstracted from a confined chalk aquifer. In order to prevent saltwater intrusion a hydraulic barrier is established. The effectiveness of the barrier consisting of injection wells is examined for a projected climate change scenario using variable density modeling.

INTRODUCTION

Groundwater abstracted from coastal aquifers is at risk of increased saltwater intrusion as an effect of the projected climate changes. This problem is studied for an area located in the southeastern part of Denmark on the island of Falster in the Baltic Sea (Figure 1). The local waterworks abstract groundwater from a shallow chalk aquifer. Part of the aquifer is located near the coast where an increasing chloride concentration has been monitored in groundwater abstraction wells over the last decades (Rasmussen et al. 2013).

In some coastal areas where saltwater intrusion has been threatening groundwater well fields, injection wells have been installed to generate a hydraulic barrier that prevents further saltwater intrusion, e.g. in Spain and USA. Climate changes might cause, among other things, sea level rise and changes in groundwater recharge. The objective of the present study is to conduct a model analysis of the effect of injecting freshwater into the groundwater aquifer to act as a hydraulic barrier to prevent further saltwater intrusion in a climate change scenario with increasing sea level and reduced groundwater recharge.

METHODS

The groundwater abstraction wells in focus are located on a barrier island between the Baltic Sea to the east and a low laying drained area to the west (Figure 1). A previous model study has examined possibly effects of increasing sea level, changes groundwater recharge, and canal stage on groundwater quality (Rasmussen et al. 2013). The model area of 44km^2 is discretized using a grid size of 50m by 50m, and 32 model layers varying in thickness from 2m to 12m down to a depth of -200m.a.s.l. The main aquifer consists of fractured and crushed calk overlain by up to 45m of clayey till and sand.

In order to model the historical changes the study area has undergone from an area with saltwater lagoon and barrier islands to reclaimed and drained land with groundwater abstraction, a modeling period of more than 3000 years were carried out in order to reach a steady-state situation for the freshwater-saltwater distribution. From 1960 groundwater abstraction was implemented in the model. The modeling packages MODFLOW/MT3DMS/ SEAWAT were used for the variable density modeling.

Eight combinations of sea level rise (0.5m, 0,75m, and 1m), changed groundwater recharge (decrease of 15%, increase of 15%), and changed canal stage (-30cm, +30cm) were analyzed. The climate change effects of sea level rise and changed recharge was gradually implemented in the model form 2010 to 2100. The model simulations were continued for additionally 200 years to year 2300.

It was found that the most severe scenario concerning the chloride concentration in groundwater abstraction wells were the scenario with a 0.75m sea level rise and a decrease in groundwater recharge of 15%. This "worst case" scenario has been used for the present study.

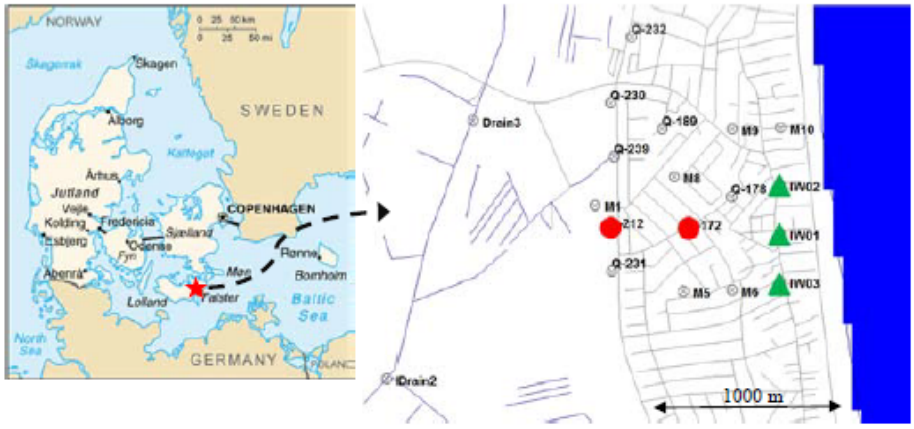


Figure 1. Location of study area (red star on left panel). Right panel shows part of the study area with two abstraction wells (red dots) and three injection wells (green triangles).

Three injection wells were added in the area between the well field and the cost line (Figure 1). The injection wells were located so far from the coast that they were not penetrating the saltwater wedge. The wells were screened in the upper fractured chalk from an elevation of -12m to -30m. The injection rates were the same as the average abstraction rates for the waterworks wells, 68 m^3/d. The injection wells are located 300m from the coast with spacing of 250m. The abstraction wells of interest for this study, Q-172 and Q-212, are located 750m and 1150m, respectively, from the coastline (Figure 1).

The injection of water in the three injections wells was started in year 2300 and the simulation was continued for additional 100 years. The salinity concentrations were monitored in the two abstraction wells, Q-172 and Q-212 (Figure 1).

RESULTS

100 years of freshwater injection has a significant effect on the extension of the freshwater lens in both horizontal and vertical direction. Figure 2 upper panels show the freshwater-saltwater distribution in the year 2300 before start of the injection. Figure 2 lower panels show the freshwater-saltwater distribution after 100 years of freshwater injection.

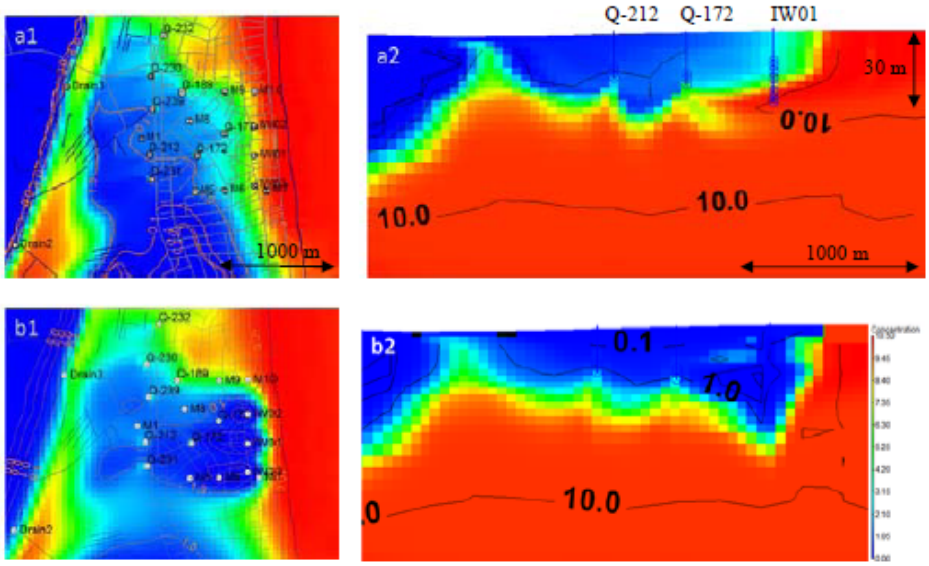


Figure 2. Saltwater distribution, plan view and cross section. Plan view: model layer 8, elevation -22m. Cross sections through wells Q-212, Q-172, and IW01. Panel a1 and a2: before injection. Panel b1 and b2: after 100 years injection. Contours: TDS (g/l).

The spacing between the three injection wells is seen to be sufficiently close to preventing saltwater intrusion between the injection wells.

Figure 3 shows the effect of the freshwater injection on TDS concentration in the two abstraction wells, Q-172 and Q-212. In the left panel the effects of the climate scenario implemented from year 2011 to 2300 is shown. The right panel shows the effect on TDS concentrations during 100 years of water injection starting in the year 2300. The effect of the injection well is first observed in the abstraction well closest to the injection wells and the coast, Q-172, after 25 years. In the abstraction well located further from the injection wells and the coast, Q-212, the effect is seen after 50 years. In well Q-172 a significant reduction in TDS concentration is found through the rest of the injection period. After 100 years of injection, the TDS concentrations are reduced to a level that is close to the concentrations found before the effects of the climate changes commenced.

However, the negative effects of climate changes could have been avoided if the injection wells had been installed before the onset of sea level rise and reduction in groundwater recharge.

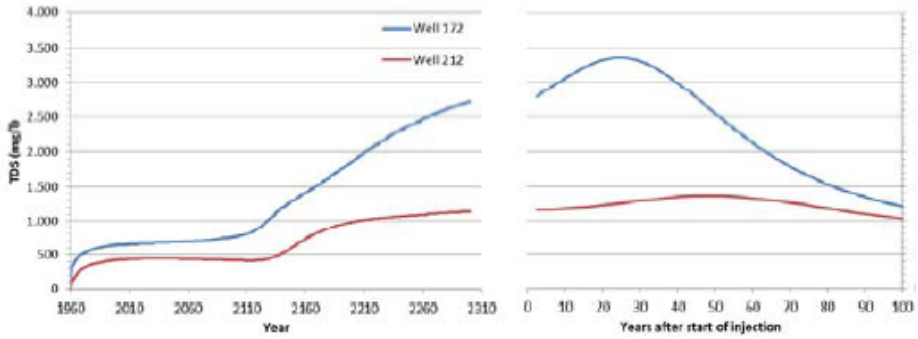


Figure 3. Effect of injection wells on TDS concentration in two groundwater abstraction wells. Location of wells, see figure 1. Left panel shows effects of a climate scenario implemented from year 2011. Right panel shows effect on TDS concentrations of 100 years water injection starting in the year 2300.

DISCUSSION AND CONCLUSIONS

Injection wells are tested as a hydraulic barrier to alleviate increased saltwater intrusion into coastal aquifers due to effects of projected climate change effects. Variable density modeling studies shows that fresh water injected into a coastal aquifer can prevent further increase in chloride concentrations in abstracted groundwater. The effectiveness of the hydraulic barrier in reversing the increasing trend in chloride concentrations in groundwater abstraction wells depends among other things on the timing of the injection scheme installation relative to the timing of climate changes.

Further studies in the area are planned to characterize the variation in hydraulic conductivity by calibrating the variable density model against available data on hydraulic head and chloride concentration using the pilot points method. Results from an airborne electromagnetic survey (SkyTEM) will be used to estimate the chloride concentration in the aquifers and it will be tested if these data can be used as targets in the calibration process. Also different designs of the hydraulic barrier including the number of injection wells, the injection rate and the location and orientation of the wells will be examined.

REFERENCES

P. Rasmussen, T. O. Sonnenborg, and K. Hinsby. 2013. Assessing impacts of climate change, sea level rise, and drainage canals on saltwater intrusion to coastal aquifer. Hydrol. Earth Syst. Sci., 17, 421–443, 2013. www.hydrol-earth-syst-sci.net/17/421/2013/. doi:10.5194/hess-17-421-2013.

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Appendix 2: SWIM23 Poster

WATER4COASTS

23rd Saltwater Intrusion Meeting, 2014
Husum, Germany

GEUS

Evaluating hydraulic barriers for reducing and controlling saltwater intrusion in a changing climate

Per Rasmussen[1], Torben O. Sonnenborg[1] and Klaus Hinsby[1]
[1]Dept. of Hydrology, Geological Survey of Denmark and Greenland
Copenhagen, Denmark

1. Introduction

Groundwater abstracted from coastal aquifers is at risk of increased saltwater intrusion as an effect of projected climate changes.

A variably density model analysis is used to analyze the effect of injecting freshwater: using groundwater wells a hydraulic barriers.

Looking at climate change scenarios with increasing sea level and changing groundwater recharge.



2. Methodology

The modeling packages MODFLOW/ MT3DMS/ SEAWAT are used for the variable density modeling of transition phases in study area (Fig. 1).

Phase five includes projected climate change scenarios of sea level rise and changed groundwater recharge.

A phase seven includes a later hundred years period of freshwater injections.

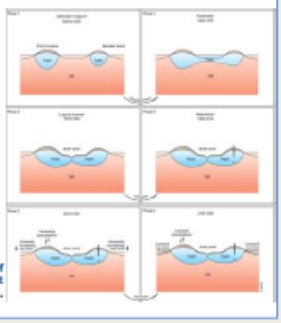


Fig 1. Modeled changes of study area (Rasmussen et al., 2013).

3. Field site

The study area is located in the south-eastern part of Denmark on the island of Falster in the Baltic Sea (Fig. 2).

The local waterworks abstract groundwater from a shallow coastal chalk aquifer.

Three synthetic injection wells are added to the groundwater model in the area between the well field and the cost line.



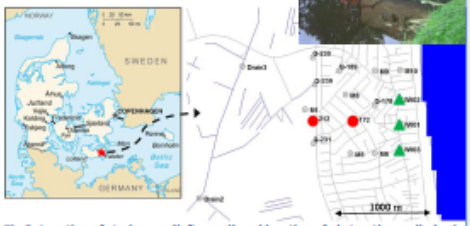


Fig 2. Location of study area (left panel) and location of abstraction wells (red dots, right panel) and synthetic injections wells (green triangles, right panel).

4. Model setup

The three synthetic injection wells: 300m from coast, 250m spacing, injection rates as waterworks abstraction rates (68 m³/d), well screens in the upper fractured chalk -12 to -30 masl.

The injection of water in the three modeled injections wells is started in year 2300 and the simulation is continued for additional 100 years.

5. Results

100 years of fresh-water injection has significant effect on the extension of the freshwater lens in both horizontal and vertical direction (Fig. 3).

Negative effects of climate changes can be avoided if injection wells are installed before the onset of sea level rise and reduction in ground-water recharge (Fig. 4).

The climate change scenario used:

- 0.75m sea level rise
- -15% groundwater recharge

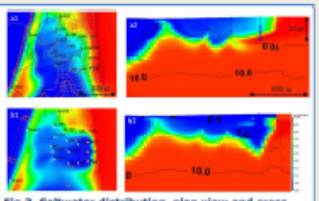


Fig 3. Saltwater distribution, plan view and cross section. Plan view: model layer 8, elevation -22m. Panel a1 and a2: before injection. Panel b1 and b2: after 100 years injection. Contours: TDS (g/l).

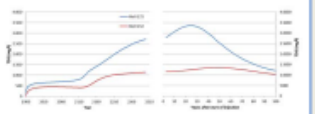


Fig 4. Effect of injection wells on TDS concentration in two groundwater abstraction wells. Left panel: effects of a climate scenario implemented 2011 to 2100. Right panel: effect on TDS concentrations of 100 years water injection starting in the year 2300.

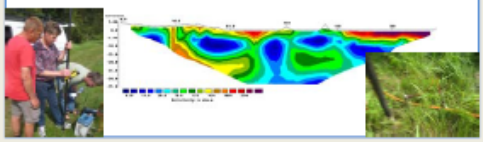
6. Conclusions and future plans

Variable density modeling studies shows that fresh water injected into a coastal aquifer can prevent further increase in chloride concentrations in abstracted groundwater in a changing climate.

The effectiveness of the hydraulic barrier depends on e.g. the timing of the injection relative to the timing of climate changes.

Future studies:

- Characterize the hydraulic conductivity by calibrating model against hydraulic head and chloride concentration using the pilot points method
- Use results from geophysical surveys (MEP, SkyTEM, logging) to estimate the chloride concentrations for targets in the PP calibration
- Effect of different designs of hydraulic barriers.



7. References

P. Rasmussen, T. O. Sonnenborg, and K. Hinsby. 2013. Assessing impacts of climate change, sea level rise, and drainage canals on saltwater intrusion to coastal aquifer. Hydrol. Earth Syst. Sci., 17, 421–443, 2013. www.hydrol-earth-syst-sci.net/17/421/2013/. doi:10.5194/hess-17-421-2013.

GEOLOGICAL SURVEY OF DENMARK AND GREENDLAND WWW.GEUS.DK

Appendix 3: Well record for borehole DGU nr. 242.375

De Nationale Geologiske Undersøgelser for Danmark og Grønland — Udskrevet 16/6 2015 Side 1

BORERAPPORT — **DGU arkivnr: 242. 375**

Borested : Marielyst Strandvej
4873 Væggerløse

Kommune : Guldborgsund
Region : Sjælland

Boringsdato :

Boringsdybde : 46 meter

Terrænkote : 0.65 meter o. DNN

Brøndborer : GEO
MOB-nr :
BB-journr :
BB-bornr :

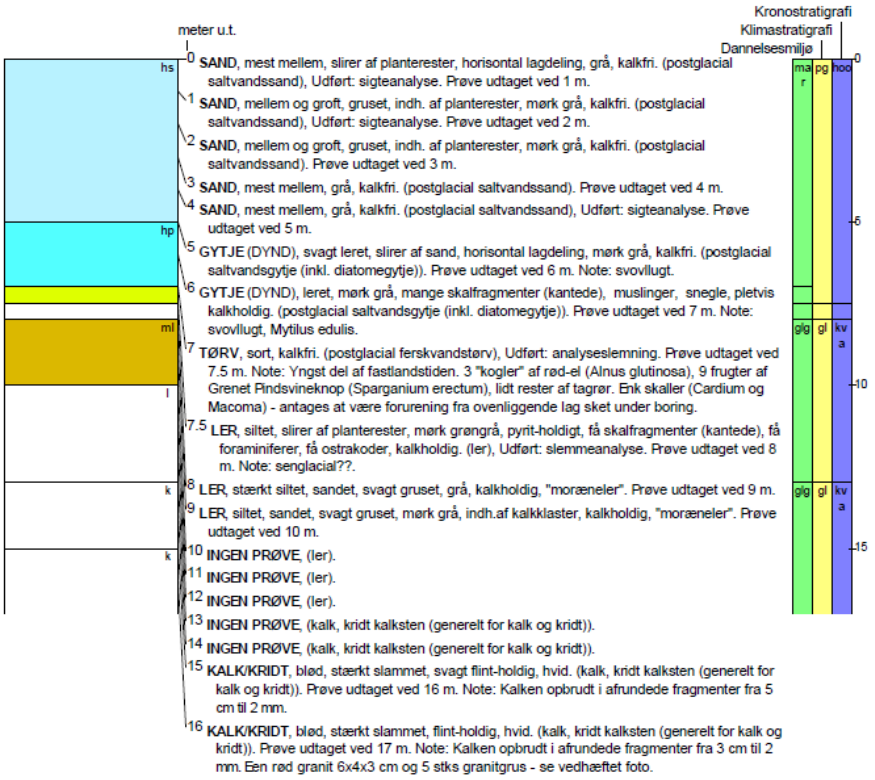
Prøver
- **modtaget** : 31/10 2014 **antal** : 42
- **beskrevet** :
- **antal gemt** : 0

Formål : Undersøg./videnskab
Anvendelse :
Boremetode :

Kortblad : 1511IIINØ
UTM-zone : 32
UTM-koord. : 691320, 6064344

Datum : WGS84
Koordinatkilde : GEUS
Koordinatmetode : Ortofoto

Notater : Beskrives pt.
Afventer borehulslogging og indmåling af moræne/kalkgrænse.
10-15 meter: Kernet interval af overgang ml. moræne og kalk.
15-27 meter: Granitsten i 17, 24 og 27 meter antyder, at kalken er glacialt forstyrret. Eller dinosaur kråsesten!



0 **SAND**, mest mellem, slirer af planterester, horisontal lagdeling, grå, kalkfri. (postglacial saltvandssand), Udført: sigteanalyse. Prøve udtaget ved 1 m.

1 **SAND**, mellem og groft, gruset, indh. af planterester, mørk grå, kalkfri. (postglacial saltvandssand), Udført: sigteanalyse. Prøve udtaget ved 2 m.

2 **SAND**, mellem og groft, gruset, indh. af planterester, mørk grå, kalkfri. (postglacial saltvandssand). Prøve udtaget ved 3 m.

3 **SAND**, mest mellem, grå, kalkfri. (postglacial saltvandssand). Prøve udtaget ved 4 m.

4 **SAND**, mest mellem, grå, kalkfri. (postglacial saltvandssand), Udført: sigteanalyse. Prøve udtaget ved 5 m.

5 **GYTJE** (DYND), svagt leret, slirer af sand, horisontal lagdeling, mørk grå, kalkfri. (postglacial saltvandsgytje (inkl. diatomegytje)). Prøve udtaget ved 6 m. Note: svovllugt.

6 **GYTJE** (DYND), leret, mørk grå, mange skalfragmenter (kantede), muslinger, snegle, pletvis kalkholdig. (postglacial saltvandsgytje (inkl. diatomegytje)). Prøve udtaget ved 7 m. Note: svovllugt, Mytilus edulis.

7 **TØRV**, sort, kalkfri. (postglacial ferskvandstørv), Udført: analyseslemning. Prøve udtaget ved 7.5 m. Note: Yngst del af fastlandstiden. 3 "kogler" af rød-el (Alnus glutinosa), 9 frugter af Grenet Pindsvineknop (Sparganium erectum), lidt rester af tagrør. Enk skaller (Cardium og Macoma) - antages at være forurening fra ovenliggende lag sket under boring.

7.5 **LER**, siltet, slirer af planterester, mørk grøngrå, pyrit-holdigt, få skalfragmenter (kantede), få foraminiferer, få ostrakoder, kalkholdig. (ler), Udført: slemmeanalyse. Prøve udtaget ved 8 m. Note: senglacial??.

8 **LER**, stærkt siltet, sandet, svagt gruset, grå, kalkholdig, "moræneler". Prøve udtaget ved 9 m.

9 **LER**, siltet, sandet, svagt gruset, mørk grå, indh.af kalkklaster, kalkholdig, "moræneler". Prøve udtaget ved 10 m.

10 **INGEN PRØVE**, (ler).

11 **INGEN PRØVE**, (ler).

12 **INGEN PRØVE**, (ler).

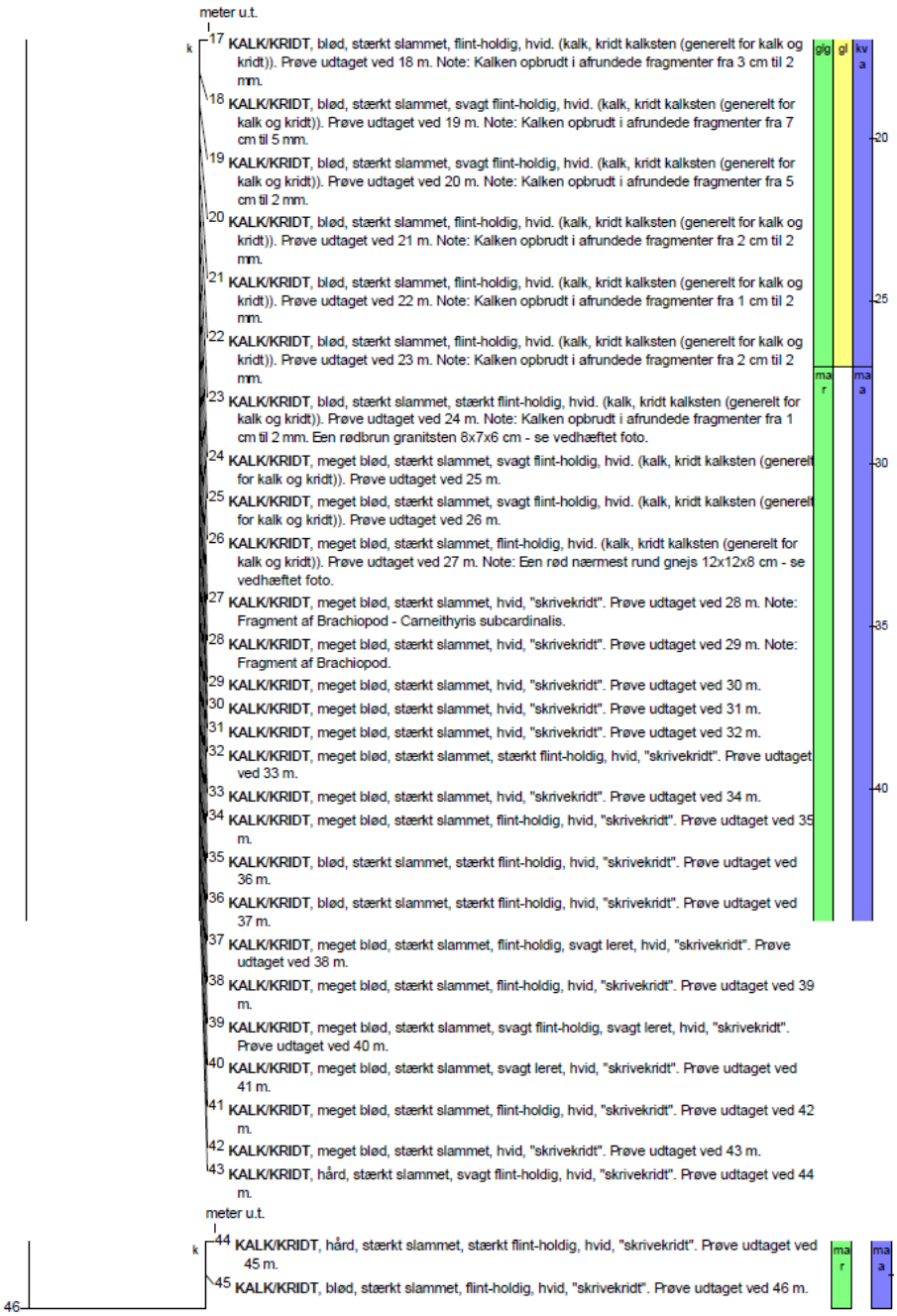
13 **INGEN PRØVE**, (kalk, kridt kalksten (generelt for kalk og kridt)).

14 **INGEN PRØVE**, (kalk, kridt kalksten (generelt for kalk og kridt)).

15 **KALK/KRIDT**, blød, stærkt slammet, svagt flint-holdig, hvid. (kalk, kridt kalksten (generelt for kalk og kridt)). Prøve udtaget ved 16 m. Note: Kalken opbrudt i afrundede fragmenter fra 5 cm til 2 mm.

16 **KALK/KRIDT**, blød, stærkt slammet, flint-holdig, hvid. (kalk, kridt kalksten (generelt for kalk og kridt)). Prøve udtaget ved 17 m. Note: Kalken opbrudt i afrundede fragmenter fra 3 cm til 2 mm. Een rød granit 6x4x3 cm og 5 stks granitgrus - se vedhæftet foto.

BORERAPPORT **DGU arkivnr: 242. 375**



Aflejringsmiljø - Alder (klima-, krono-, litho-, biostratigrafi)

meter u.t.			
0	-	7	marin - postglacial - holocæn
7	-	7.5	lakustrin - postglacial - holocæn
7.5	-	8	marin? - senglacial? - kvartær?
8	-	13	glacigen - glacial - kvartær
13	-	27	ant. glacigen - ant. glacial - ant. kvartær
27	-	46	marin - maastrichtien

Bilag 2. GEUS rapport 2015/93 - Evaluation of water quality issues before application of managed aquifer recharge in a coastal chalk aquifer

WATER4COASTS

Evaluation of water quality issues before application of managed aquifer recharge in a coastal chalk aquifer

Hinsby, K., Johnsen, A.R., Sørensen, S.R. and Postma D.

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1. Summary

This report is one out of three reports assessing the potential for application of managed aquifer recharge to control salt water intrusion on the Southern part of the Falster Island, Denmark. The two other reports by Rasmussen et al., 2015 and Hinsby et al. 2015 consider the model simulations of injection into the Chalk aquifer and efficient data acquisition, transfer and management for the development of advanced Supervisory Control and Data Aquisition (SCADA) systems, respectively. This report considers water quality issues related to aquifer storage and recovery e.g. by using either groundwater or treated surface water (drainage water) for e.g. injection during the wet wintertime and retrieval again in the dryer summer period in order to reduce and control salt water intrusion in the region and protect subsurface drinking water resources. The report evaluates the risks of using aerobic and anaerobic groundwater as well as aerobic drainage water for injection into the Chalk aquifer.

It is concluded that anaerobic groundwater from the water works connected pipelines may be used for injection without any treatment. Analyses from all the water supply wells show that they generally contain very similar anaerobic groundwater older than 75 years without any measurable human impact or contamination. Hence, this water type can be infiltrated without initiating problematic geochemical reactions and jeopardizing the quality of the high quality groundwater in the Chalk aquifer.

In contrast the drainage waters in the drainage canals contain elevated concentrations of both pesticides (MCPA-), industrial chemicals (PFOS), and at one point also high numbers of faecal bacteria (e.coli). Furthermore, a wide range of pharmaceuticals used for a variety of diseases have been found in rather high concentrations in the effluent from a waste water treatment plant discharging to the drainage canal. Hence, if these waters have to be used for injection, strict quality control and/or an intensive treatment and dilution e.g. with pristine groundwater pumped from the Chalk aquifer, will be required to ensure that contamination is not introduced at unacceptable levels into the pristine high quality groundwater of the Chalk aquifer in the area.

Furthermore, the injection of aerobic water containing oxygen and/or nitrate into the anaerobic groundwater will induce oxidation of reduced minerals (Iron sulphides) and organic matter, which may lead to on one hand both technical problems with clogging of wells screens by iron oxides and water quality issues with elevated concentrations of some trace elements and metals, but on the other hand also positive side effects such as removal of nitrate in the subsurface and reduced nitrogen loadings to the coastal waters of the western Baltic Sea.

The results from the preliminary investigations and assessments clearly show that further investigations of the geochemistry and the hydraulics of the Chalk aquifer are required in order to better assess positive and negative effects and the feasibility of injecting anaerobic groundwater or treated aerobic drainage water into the fractured Chalk aquifer to protect subsurface water resources used for water supply on Southern Falster.

2. Background

Coastal aquifers are affected by saltwater intrusion globally, and have been under pressure for many decades especially in densely populated areas with significant groundwater abstraction for water supply (Post and Abarca, 2010; Hinsby et al., 2011, da Silva and Montenegro, 2012). Such problems are especially known and managed (controlled) in aquifer systems along coastlines of U.S.A. and Northern Spain (Sheahan, 1977; Iribar et al., 1997; Misut and Voss, 2007; Custodio, 2010), but problems appear in an increasing number of coastal aquifers in different climate zones, globally, including many coastal aquifers in northeastern Brazil (Bocanegra et al, 2010; da Silva et al., 2010) , China (Wu et al., 1993; Xue et al., 2000, Liu et al., 2001; Han et al., 2011) and Denmark (Andersen et al., 2005; Jørgensen et al, 2008; 2012, Thorn, 2011; Rasmussen et al., 2013), and measures need to be implemented in order to ensure freshwater supplies in these regions in the future.

Climate change is projected to affect precipitation patterns and increase sea levels globally and thereby increasing the risk of saltwater intrusion and deterioration of coastal water resources (Kundzewicz et al., 2007; Hinsby et al., 2011; Loaiciga et al., 2012). Although the extend of the saltwater intrusion highly depends on the geological setting (Werner and Simmons, 2009; Chang et al. 2011), and also may stem from other sources than recent seawater such as old connate waters in former marine sediments (Buckley et al., 2001; Bonnesen et al., 2009; Tran et al., 2012) and even may include highly saline brines (Xue et al., 2000; Han et al., 2011). Especially low-lying areas and areas that receive less precipitation, and experience falling water tables, will be under severe pressure (Essink et al., 2010).

Hence, salt water intrusion due to climate change, sea level rise and increasing demands
Is becoming an increasing challenge for many water supply companies in coastal regions, globally, and calls for the development of new adaptive and innovative solutions to prevent salt water intrusion, monitor the development and ensure sustainable and high quality water supply in the future (Essink, 2001; Post, 2005; Post and Abarca, 2010; Hinsby et al., 2011, Thorn and Mortensen, 2012; Sulzbacher et al., 2012, Rasmussen et al., 2013). Managed aquifer recharge through basins in phreatic aquifers or injection wells in artesian aquifers are both possible solutions, which have been applied and operated at large scales for decades mainly around large cities such as Los Angeles (since the early 1950's) (http://dpw.lacounty.gov/wrd/barriers/historical.cfm), and Barcelona; where treated wastewater are injected to avoid saltwater intrusion and ensure water supply for the city (Hernandez et al., 2011). Freshwater injection and aquifer storage and recovery is furthermore considered for emergency water supply of New York City (Misut and Voss, 2007). It is foreseen that managed aquifer recharge will be a cost-efficient way to protect water resources globally in the future (Megdal and Dillon, 2014).

Besides the salt water intrusion problem coastal catchments often face other important problems that are expected to become more severe due to climate change and its impact on the hydrological cycle, water tables and sea level.

The problems vary between regions and climatological settings, but include:

1) Increased flooding risks (Sonnenborg et al., 2012),
2) Increased nutrient loadings to rivers, lakes and coastal marine waters in wet temperate regions (Hinsby et al., 2008, 2012; Sonnenborg et al., 2012; Wikner and Andersson, 2012)
3) Droughts, overabstraction and declining water tables in arid and semi-arid regions (Ragab and Prudhomme, 2002; Montenegro and Ragab, 2012), which in some re-

gions lead to severe land subsidence, a phenomenon known or antipicated for many overabstracted aquifers around the world including Europe (e.g. Boni et al., 2015; Ruiz-Constan et al, 2016), Brazil (e.g. dos Santos et al., 2012) and China (eg. Wu et al., 2010; Shi et al, 2016).

Re 1) The study area on Southern Falster have experienced several floods from rising groundwater tables and drainage canals during the past 10 years. The latest significant flood in 2011 resulted in the broadening of the North canal by up to 50 % along several kilometers as a meaure to more efficiently handle and discharge excess precipitation to the coastal waters via the drainage canals. The extension of the drainage canals are believed to efficiently control and avoid future floods in the area, but injection of drainage waters directly to the Chalk aquifer could support the drainage system and decrease the risk of flooding even further.

Re 2) Nitrogen loadings to ecosystems and resulting effects is currently considered to be one of the most severe environmental problems on a global scale (Steffen et al. 2015) by e.g. increasing the risk of eutrophication, harmful algal blooms, hypoxia, fish kills and acidification in many coastal waters (Diaz and Rosenberg, 2008; Rabalais et al., 2009; O´Neill et al., 2011; Paerl and Paul, 2012) including coastal waters in Brazil, China and Denmark (Diaz and Rosenberg, 2008; Cai et al., 2011; Hinsby et al., 2012).

The Baltic Sea and it's coastal waters most probably exhibits the largest problems at present with oxygen depletion and hypoxia due to excessive nutrient loading, even in a global perspective (Diaz and Rosenberg, 2008; Conley, 2012). Partly for that reason the status of the major part of shallow oxic groundwater in Denmark (located in the Western Baltic Sea) is of poor status and does not comply with EU directives as groundwater threshold values for total nitrogen ("nitrate"), which have been established according to the EU Water Framework and Groundwater Directives to protect associated aquatic ecosystems, are breached (Hinsby et al., 2008, 2012). This puts a strong pressure on Denmark to reduce nutrient loadings to coastal waters in Denmark and the Western Baltic Sea, and to find new innovative and cost-efficient ways for this purpose. The coastal waters of Falster is currently not considered to receive excess loadings of nutrients, but considering the general status of the Baltic Sea, and the increasing pressure and vulnerability of the Baltic Sea and coastal waters in general due to climate change (O'Neill et al., 2011), it is worth exploring measures to decrease nutrient loadings from agricultural catchments to coastal waters.

Re 3) Overabstraction and land subsidence is not an issue of concern on Southern Falster. Hence, MAR will not be studied as a means of avoiding or reducing land subsidence. However, it is worth noticing that MAR is relevant for mitigating or avoiding land subsidence in other coastal aquifers around the world, and in the longer perspective it may also be relevant for other coastal aquifers in Denmark.

3. Introduction

The WATER4COASTS project seeks to assess, test and developed new and efficient solutions for integrated,sustainable and adaptive management of groundwater and surface water that take into account both surface and subsurface water quantity and quality aspects and the protection of water resources and ecosystems in coastal regions. By injecting water from other water supply wells with lower salinity or waste water treatment plants and drainage canals into for instance artesian anaerobic aquifers by managed aquifer recharge the hydraulic pressure increase and reduce the risk of saltwater intrusion (Misut and Voss, 2007; Hernandez et al., 2011).

Furthermore, any nitrate in the injected water and the stage in drainage canals are reduced, and hence the risk of eutrophication and flooding.

The main objective of WATER4COASTS is to develop and demonstrate new efficient tools for sustainable and adaptive water management in coastal aquifers and catchments based on artificial recharge to coastal aquifers to ensure good status of water resources and coastal ecosystems in order to:

1) Prevent salt water intrusion towards coastal well fields
2) Reduce nitrogen loads from coastal catchments to marine waters
3) Reduce the risk of flooding in the hinterland from e.g. drainage canals and Groundwater and
4) Develop efficient monitoring systems

This report describes the current organic (incl. pathoghens) and inorganic quality of groundwater, drainage and waste water, discuss the treatment needs before injection of oxic drainage water into an anoxic confined chalk aquifer, and negative and positive effects and the cost-efficiency of different potential water treatment technologies.

As described in the background section (section 2) and indicated above MAR may be a relevant measure for mitigating different types of negative climate change impacts on water resources, ecosystems and the built environment in coastal regions. For the case of Southern Falster it is the protection of the coastal groundwater resources against both lateral salt water intrusion from the Baltic Sea and vertical salt water intrusion from old seawater in the deeper part of the Chalk aquifer, which is the main concern, and hence will be the main focus and topic for this and future studies on the protection of water resources on Southern Falster.

4. Water quality assessment

4.1 Groundwater quality and age

Analyses of groundwater sampled in water supply wells of the Marielyst Water Works show that the sampled groundwater in the Chalk aquifer are relatively old (> 75 yr) as they contain no Tritium (^{3}H) and no tritiogenic Helium (trit-^{3}He, table 4.1) as described in Rasmussen et al. (2013). In accordance with this no other human impacts and contamination are found in the Chalk aquifer in the national groundwater monitoring program as expected (Hinsby et al., 2001).

Table 4.1. Measured 3H/3He values and estimated 3H/3He groundwater ages in water supply wells of Marielyst waterworks compared to model simulated travel times.

Well no.[1]	He^{4}radio nml/kg	^{3}H TU	err-^{3}H TU	trit-^{3}He TU	$^{3}H/^{3}He$ age yr	simulated travel times yr
242.178 (1)	8.97E-04	0.01	0.01	0.0	> 75	113
242.182 (2)	1.02E-04	0.06	0.12	0.0	> 75	-
242.212 (2)	3.83E-04	-0.01	0.01	0.0	> 75	67
242.230 (2)	5.39E-04	0.09	1.06	0.0	> 75	42
242.231 (2)	3.28E-04	0.01	0.15	0.0	> 75	70
242.232 (2)	7.21E-04	0.01	0.02	0.0	> 75	92
242.239 (2)	2.11E-04	0.01	0.01	0.0	> 75	95
242.317 (3)	1.30E-05	0.05	0.02	3.3	75	85
242.319 (3)	1.12E-05	0.02	0.02	0.5	> 75	48
242.320 (3)	1.38E-05	0.00	0.02	0.9	> 75	45

[1] Numbers in parentheses indicate the well field each well belongs to.

Hence, the only known groundwater quality issue in the Chalk aquifer is elevated chloride concentrations due to salt water intrusion from three natural sources (marine Holocene sediments, fossil seawater in the deeper part of the Chalk, the Baltic Sea), which have increased due to water abstraction for water supply and drainage pumping. This result in increasing salt (chloride) concentrations in water supply wells to an extent where the chloride concentrations have breached the WHO and EU drinking water standard of 250 mg/L in two water supply wells (242.44B and 242.178) in well field one closest to the Baltic Sea (Table 4.2, Figure 4.1, Rasmussen et al., 2013).

If it is ensured that groundwater in the pipelines of the water supply company does not contain groundwater with chloride concentrations above 250 mg/l and it is anaerobic (groundwater from the Chalk aquifer is anaerobic i.e. with oxygen and nitrate concentrations below detection limits) groundwater can be re-injected into the aquifer by managed aquifer recharge without any concern of salinization, pollution or problematic reactions in the aquifer around the injection well.

Continuous pumping from the water supply wells and the drainage canals will result in continuously increasing chloride concentrations unless effective countermeasures such as subsurface water solutions that include managed aquifer recharge e.g. in combination with remediation pumping are introduced. Such solutions have shown promising results in e.g. the Netherlands (Zuurbier, et al., 2014, 2015).

Recharge of groundwater by injection of anaerobic fresh groundwater into more saline groundwater wells during the rainy winter season e.g. using the ASR coastal solution (Zuurbier et al.,

2014) to sustain continuous abstraction of freshwater in Marielyst on Southern Falster seems to be a potential and highly relevant solution for the protection of the coastal water resources in the area.

Table 4.2. Concentrations of major cations and anions in groundwater samples collected from water supply wells of marielyst water works. The water supply wells of well field 1 is located closest to the Baltic sea coast line have the highest chloride (Cl) concentrations breaching the WHO and EU drinking water standard of 250 mg/l.(well 242.44b and 242.178). Unit = mg/l.

Well	Na	K	Mg	Ca	Cl	Br	NO3	PO4	SO4	HCO3	Well field no.
242.44B	165	7,3	31	127	270	1,16	<0,1	<0,05	7,8	420	1
242,178	147	7,2	37	124	257	1,00	<0,1	<0,05	10	425	1
242,182	20	4,3	21	104	31	0,125	<0,1	<0,05	6,6	378	2
242,212	82	5,9	30	127	185	0,66	<0,1	<0,05	10	396	2
242,230	51	5,0	28	122	104	0,37	<0,1	0,059	10	423	2
242,232	75	6,3	30	129	155	0,51	<0,1	<0,05	9,1	450	2
242,319	20	3,6	18	108	40	0,125	<0,1	<0,05	19	354	3
242,320	17	4,2	18	101	32	0,125	<0,1	<0,05	6,6	350	3
Average	72	5	27	118	134	0,51	<0,1	<0,05	10	399	

4.2 Drainage water and discharge from waste water treatment plant (WWTP)

Water from drainage canals draining the area and ensuring dry farm lands and summer housing areas would be another source for injection waters to control salt water intrusion. A study on the quality of these was therefore conducted in order to assess the need for water treatment of the drainage waters before these are injected.

The studies on the quality of the drainage waters in drain canals and pipes focused on the area between the most upstream sampling site W4C-3 located approximately 100 m upstream the waste water treatment plant (WWTP) andW4C-6 located approximately 150 m upstream the pumping station. All sampling sites are located between well field two and three of the water works (Figure 4.1). This part of the drainage system seems to be the most relevant potential source for injection into the groundwater system after required treatment. If water for injection was collected upstream the WWTP it would contain less pharmaceutical contaminants, but it would then reduce the flow in the canal at the WWTP and hence reduce the dilution of the contaminants from the WWPT. Using waters from the canal between the WWTP and the pumping station for injection is therefore considered the best option as long as the water is treated properly and it is ensured that only pure water is injected into the pristine high quality groundwater of the Chalk aquifer.

4.2.1 Drainage canal and WWTP sampling sites.

For the chemical analyses, we sampled water from Marrebæk Kanal (or the “North Canal”), a canal that drains an agricultural area west of the town of Marielyst (Fig. 1). The elevation of the water surface in the canal is typically 1-2 meters below sea level. The canal receives water from smaller side canals and drain pipes that drains the agricultural fields, and from the discharge of water from Marielyst Waste Water Treatment Plant (WWTP). The water in the main drainage canal runs in the direction from the WWTP (north-east) towards the pumping station

(south-west). The upstream part of the canal is on the fringe of the town and therefore might also receive some water directly from private housings, either as run off or as non-intentional leaching.

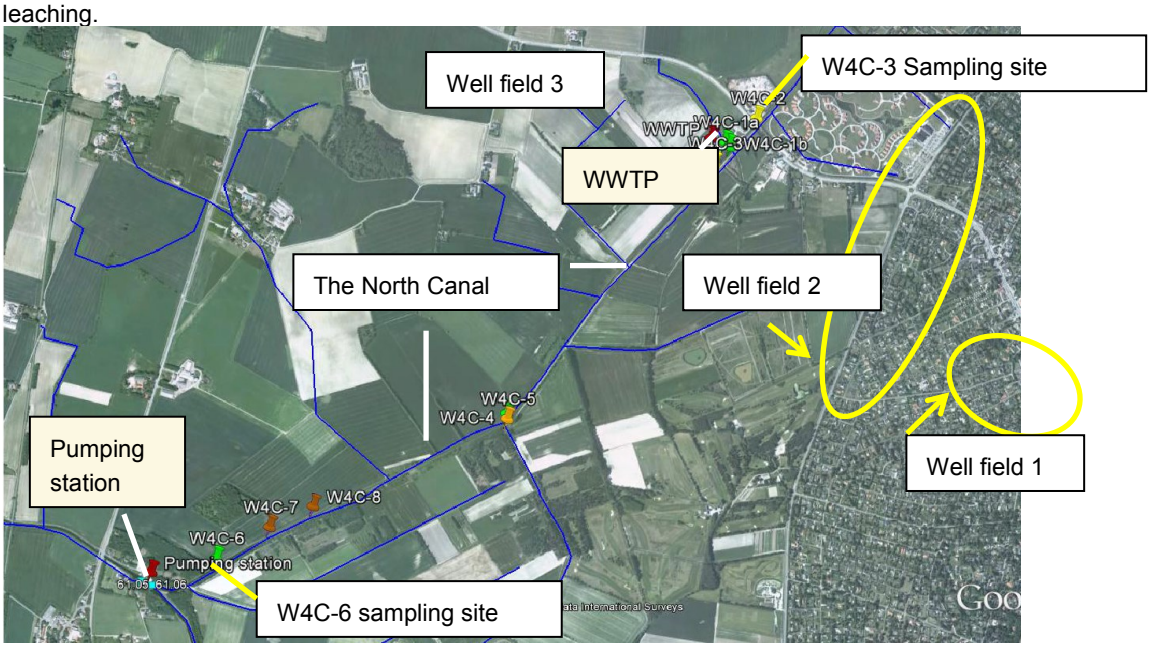


Fig. 4.1. Overview of the Marrebæk Kanal and the positions where water was sampled for chemical and microbial analyses.

The town of Marielyst is a center for tourism with a permanent population of approximately 700 persons. During the summer months, however, the population may approach 50.000 persons. This means that the treatment plant during the dry summer months may provide a significant proportion of the water in the drainage canal. The use of pesticides, chemicals and pharmaceuticals may vary over the year. Two sampling times therefore does not give a full screening of the composition of the water, but should rather be seen a snapshot examples of types and levels of compounds that may be present in the water. The two sampling dates were chosen so that they represented a spring scenario with a small population (24.04.2013) as well as a summer scenario with tourists (14.08.2013).
To cover most sources of pollution, we sampled water both from the main canal, from a side canal, from tile drains and directly from the outlet of the WWTP. All samples were sampled directly into flasks provided by the analytical laboratories and treated as recommended by these laboratories with respect to storage and transport. Samples were collected from below the surface of the canal to avoid the surface slick and always upstream of the person who carried out the sampling. Four types of analyses were applied to the samples: content of industrial chemicals, content of pharmaceutical compounds, content of pesticides and content of faecal bacteria. Chemical analyses were carried out by Eurofins, whereas the microbial analyses were done by Amphe-Bac Aps. The samples were prioritized so that the most expensive analyses were carried out only for the most important samples. The locations of the sampling sites and the specific analyses were as follows:

W4C-1: Samples taken directly in the discharge pipe from the wastewater treatment plant, analysed for industrial chemicals, pharmaceutical compounds and faecal bacteria.
W4C-2: Main drainage canal approximately 80 meters downstream of the waste water tratment plant, analysed for pesticides and faecal bacteria.
W4C-3: Main drainage canal approximately 100 meters upstream of the waste water tratment plant, analysed for faecal bacteria.
W4C-4: Main drainage canal downstream of the waste water tratment plant half way to the pumping station , analysed for industrial chemicals and faecal bacteria.

W4C-5: A small side-canal approximately 10 meters downstream of W4C-4, the canal receives water only from agricultural land, analysed for pesticides and faecal bacteria.
W4C-6. Main drainage canal 150 meters upstream of the pumping station, analysed for pesticides and faecal bacteria.
W4C-7: Samples taken directly in a mall drain pipe 400 meters upstream of the pumping station, analysed for faecal bacteria.
W4C- 8: Samples taken directly in a large drain pipe 600 meters upstream of the pumping station, analysed for faecal bacteria.

The results from the different sites and analyses are provided in the following sections.

4.2.2 Pesticides in drainage water

Many different pesticides have been used over the years in Denmark in agriculture and private households. It was therefore important to prioritize which pesticides to look for since the economy did not allow an extensive screening for a large number of compounds. We therefore selected the prioritized pesticides on the basis of results from the Danish Pesticide Leaching Assessment Program (Brüsh et al., 2012) and the Danish Groundwater Monitoring Program (Thorling et al., 2012), i.e., we analysed for many of the compounds that are most likely to leach from agricultural fields.

The prioritized pesticides covered one fungicide (Fenpropimorph), one insecticide (Pirimicarb) and 12 herbicides (2,4-D; Atrazine; Bentazon; Bromoxynil; Ethofumesat; Fluazifop-P-butyl; Ioxynil; MCPA; Metamitron; Metribuzine; Propyzamide; Terbutylazine) and 7 herbicide degradation products (2-hydroxy-terbutylazine; Deethyl-atrazine; Deethyl-terbutylazine; Deisopropyl-atrazine; Metribuzine-deamino; Metribuzine-deamino-diketo; Metribuzine-diketo).

Two phenoxyalcanoic acid herbicides (MCPA and 2,4-D) and one degradation product (metribuzin-deamino-diketo) was detected (table xx). MCPA was found both upstream (W4C-2) and downstream (W4C-6) in the main canal and in the spring sample also in the small side canal. The MCPA concentrations in the main canal in the summer samples exceeded the drinking water threshold limit with 400% at the upstream location and 600% at the downstream location. 2,4-D was found only upstream and only in the summer sample. MCPA and 2,4-D are marketed both for agricultural use and for private use on lawns; the sources therefore may be both leaching from the agricultural fields and from the private gardens upstream of the waste water treatment plant. Metribuzin, on the other hand, has only been approved for use on potato fields in Denmark. The presence of the metribuzin degradation product metribuzin-deamino-diketo in water from the small side canal (W4C-5) at both sampling times, suggests leaching from potato fields in the area drained by the side canal.

Table 4.3. Pesticides detected in water samples from the main canal and a sidecanal (analysed by Eurofins).

µg/ L	Canal at WWTP 24/04-2013	Canal at WWTP 14/08-2013	Small side-canal 14/08-2013	Small side-canal 14/08-2013	Main canal at pumping station 24/04-2013	Main canal at pumping station 14/08-2013
MCPA	0,036	0,40	0,054	<0,03	0,027	0,59
2,4-D	<0,01	0,011	<0,01	<0,03	<0,01	<0,01
Metribuzin-deamino-diketo	<0,01	<0,01	0,029	0,040	<0,01	<0,01

4.2.3 Industrial chemicals in effluent from WWTP and drainage water

The major waste water treatment plants in the Copenhagen area have carried out a large screening of industrial and pharmaceutical compounds in the discharge water and in sludge (Madsen 2011). A number of focus compounds have been selected as important compounds to monitor since these compounds exceeded the acceptable environmental quality standards (EQS) for surface waters. We screened discharge water from the Marielyst WWTP (W4C-1) and water from the canal half way to the pumping station (W4C-4) for the presence of these industrial focus chemicals (Table 4.3) and compared the levels with EQS limits. It should be noted that most of the EQSs reported below are annual averages for rivers and lakes, not maximum concentrations. This parameter was chosen since we cannot calculate reliable averages from only two time points. Annual average EQSs for other inland waters are often lower than the values for rivers and lakes whereas EQSs for maximum allowed concentrations typically are higher.
Only three of the chemicals were detected, bisphenol A, butylphthalate and PFOS (perfluoroctansulfonsyre). Bisphenol A exhibits hormone-like properties and is used in the manufacture of epoxy resins, some types of plastic and in the lining of food cans. Bisphenol A was detected only in the discharge water from the treatment plant, but it was not detectable half way downstream the canal which suggest degradation, sorption or high dilution of the compound in the canal. The national EQS for bisphenol A is 0,1 for inland waters (Miljøministeriet, 2010) which was not exceeded in the treatment plant discharge. Di(2-ethylhexyl) phthalate (DEHP) is the most common phthalate that is used as plasticizers. We did not detect DEHP in any sample, but we did detect another phthalate plasticizer dibutylphthalate in one sample from the canal, but at a concentration of 0,10 µg/L which is below the inland waters EQS of 2,3 µg/L (Miljøministeriet, 2010) .

PFOS is a representative of a large group of surface active perfluorinated compounds that are used in firefighting foam and in textile impregnation products. PFOS is persistent, bioaccumulative and toxic to mammals including humans (OECD 2012). The PFOS concentrations were low, but the compound was detected in all four samples which raises some concern. The EQS for PFOS is 0,00013 µg/L (Madsen 2012) indicating that the acceptable level was exceeded by 60 fold in the treatment plant discharge and by 40-60 fold in the canal water half way to the pumping station. This means that extensive degradation or dilution would be needed to meet environmentally acceptable concentrations.

Octylphenol and 4-n-nonylphenol are examples of alkylphenols that are used as precusors for example in the synthesis of detergents. None of them were detected. Hexabromocyclododecane (HBCD or HBCDD) and brominated diphenyl ethers (BDE) are brominated flame retardants whereas triphenylphosphat (TPP) is an organo-phosphate flame retardant which is also used in lubricants. None of the flame retardants were detected, but the quantification limit of HBCD and BDE were above their EQS.

Table 4.4. Content of industrial focus chemicals in discharge water from Marielyst waste water treatment plant (analysed by Eurofins).

µg/ L	WWTP outlet 24/04-2013	WWTP outlet 14/08-2013	Down stream 24/04-2013	Downstream 14/08-2013
Bisphenol A	0,02	0,02	<0,001	<0,001
HBCD	<0,01	<0,01	<0,01	<0,01
SUM 6 BDE	<0,01	<0,01	<0,01	<0,01
DEHP	<0,1	<0,1	<0,1	<0,1
Di-n-butylphthalat	<0,1	<0,1	0,10	<0,1
Octylphenol	<0,1	<0,1	<0,1	<0,1
4-n-Nonylphenol	<0,01	<0,01	<0,01	<0,01
Triphenylphosphat	<0,02	<0,02	<0,02	<0,02
PFOS	0,006	0,006	0,004	0,006

4.2.4 Pharmaceuticals in effluent from WWTP and drainage canal

We screened water from the outlet of the Marielyst WWTP (W4C-1) for the presence of pharmaceutical compounds since pharmaceuticals in high concentrations have been reported to be present in Danish wastewater after waste water treatment (Madsen 2012). Samples taken directly from the WWTP discharge pipe were analysed for the following 93 pharmaceutical compounds: Amilorid, Amiodaron, Amlodipin, Atenolol, Atorvastatin, Azithromycin, Bendroflumetiazid, Bromhexin, Bromokriptin, Budesonid, Cefuroxim, Cetirizine, Ciprofloxacin, Citalopram, Cyklofosfamid, Desloratadin, Dextropropoxifen, Diazepam, Diklofenak, Enalapril , Enrofloxacin, Erytromycin, Etinylöstradiol, Febantel, Felodipin, Fentanyl, Flunitrazepam, Fluoxetin, Fluvastatin, Fluvoxamin, Furosemid, Gemfibrozil, Glibenklamid, Hydroklortiazid, Hydrokortison, Ibuprofen, Ifosfamid, Ipratropium, Isosorbidmononitrat, Karbamazepin, Karvedilol, Ketokonazol, Ketoprofen, Klozapin, Kodein, Lansoprazol, Loratadin, Losartan, Metoprolol, Metronidazol, Mianserin, Mirtazapin, Mometasonfuroat, Naproxen, Nelfinavir, Nitenpyram, Noretisteron, Norfloxacin, Norgestrel, Ofloxacin, Omeprazol, Oxazepam, Oximetazolin, Oxitetracyklin, Paracetamol, Paroxetin , Prazikvantel, Prednisolon, Propranolol, Pyrantel, Raloxifen, Ramipril, Ranitidin, Risperidon, Salbutamol (=Albuterol), Salmeterol, Sertralin, Simvastatin, Sulfametoxazol, Tamoxifen, Terbutalin, Tetracyklin, Tioridazin, Tramadol, Trimetoprim, Tylosin, Warfarin, Xylometazolin, Zolpidem, Zopiklon, Oestradiol, Oestriol, Oestron.

Table 4.5. Pharmaceutical compounds (µg/l) detected in discharge water from Marielyst waste water treatment plant (analysed by Eurofins).

Pharmaceutical compound	Compound class	Sampling date 18/4-2013	Sampling date 14/8-2013
Atenolol	Cardiovascular diseases	120	150
Amlodipin	Cardiovascular diseases	<10	10
Losartan	Cardiovascular diseases	420	710
Metoprolol	Cardiovascular diseases	2500	6900
Warfarin	Cardiovascular diseases	17	56
Propranolol	Cardiovascular diseases	810	150
Ramipril	Cardiovascular diseases	21	51
Felodipin	Cardiovascular diseases	17	11
Carvedilol	Cardiovascular diseases	<10	13
Gemfibrozil	Cardiovascular disease (cholesterol)	340	130
Furosemide	Cardiovascular diseases, diuretic	2800	2000
Hydrochlortiazide	Cardiovascular disease, diuretic	780	2000
Carbamazepine	Psychoactive	230	na
Citalopram	Psychoactive	2300	2600
Chlozapin	Psychoactive	13	13
Mirtazapin	Psychoactive	120	150
Oxazepam	Psychoactive	190	520
Sertraline	Psychoactive	27	19
Zopiklon	Psychoactive	27	<10
Diklofenak	Painkiller, NSAID	240	400
Ibuprofen	Pain killer, NSAID	27	98
Ketoprofen	Pain killer, NSAID	20	85
Naproxen	Pain killer, NSAID	130	130
Codeine	Pain killer	680	1700
Tramadol	Pain killer	3600	6400
Erythromycin	Antibiotic	24	32
Trimethoprim	Antibiotic	1200	47
Azithromycin	Antibiotic	230	61
Sulfametoxazole	Antibiotic	160	160
Terbutaline	Asthma , premature labor	12	46
Ranitidine	Inhibits stomach acid production	31	150
Omeprazol	Inhibits stomach acid production	<10	11
Cetirizine	Antihistamine (allergy)	190	560
Xylometazolin	Nasal decongestant	10	20
Cyclophosphamid	Chemotherapy (cancer)	<10	22
Sum		17.300	25.400

35 out of the 93 compounds were detected above their detection limits. The concentration varied orders of magnitude from close to the detection limit (10 ng/L) to more than 1000 ng/L. The compounds found in highest concentration were associated with cardiovascular disease (furosemide, metropolol, hydrochlortiazide), depression (citalopram), pain relief (codeine, tramadol) and infections (trimethoprim).

EQS values for some of the pharmaceutical focus compounds were reviewed by Madsen et al (2011): dichlofenac 10 ng/L, propranolol 5 ng/L, amiodaron 9 ng/L, ciprofloxacin 5 ng/L, azithromycin 9,4 ng/L, sulfamethoxazol 26,8 ng/L and citalopram 510 ng/L. Some of these compounds were in our study found in concentrations that exceeded the EQS by a factor of up to 40 times (dichlofenac), 162x for propranolol, 24x for azithromycin, 6x for sulfamethoxazol and 5x for citalopram. The focus compounds ciprofloxacin and amiodaron were not detected.

Conclusions
The analyzed industrial priority chemicals generally would be of little concern for artificial groundwater injection. One exception was PFOS, which has a very small EQS that was exceeded. PFOS and other perfluorinated chemicals are furthermore resistant to degradation in the environment, which suggests that the surface water in Marrebæk Kanal should either be diluted or treated before injection.

The level of pesticides was generally low, except for MCPA. MCPA is degradable under aerobic condition in most environments and therefore would probably pose a minor risk for aerobic groundwater injection. MCPA, however, might be persistent in many anoxic environments suggesting that aerobic treatment of surface water might be advisable before injection into anoxic aquifers.

The high load of pharmaceutical compounds in the WWTP discharge (total concentrations of 17,3 µg/L and 25,4 µg/L) is highly problematic. The WWTP is optimized for cleaning the waste water, and still the pharmaceutical compounds are present in the discharge, indicating a low degradability of many of these compounds. Furthermore, much of the water in the drainage canal would during the dry summer months originate from the WWTP resulting in little dilution of the pharmaceutical compounds. Also, these compounds are designed to have an effect in humans. We might be able to estimate the effect of single compounds, but we have little knowledge on possible cocktail effects, that is, the combined effect of the 35 different pharmaceuticals.

4.2.5 Pathogens

E. coli.

Water samples were analysed for the levels of E. coli by the IDEXX Quantitray most probable number (MPN) method. The analyses were carried out by Amphi-Bac Aps. Strains of E. coli are major inhabitant of the gastrointestinal tract in warm-blooded animals; total E coli MPN is thus an indicator of faecal contamination. E coli were detected in highest concentrations in the WWTP discharge followed by lower concentrations 80 meters downstream of the WWTP where the discharge had been mixed with surface water in the canal. In the spring samples, concentrations were further reduced downstream of the WWTP and E. coli were not detectable in water from tile drains or water from the small side-canal. The result was somewhat different for the summer sampling. There was no water running in the tile drains and analyses of these were therefore impossible. The side-canal, however, showed high concentrations (3448 per 100 mL) of E. coli. Aerial photos of the side-canal did not indicate any sources of domestic wastewater. Another source could have been liquid pigmanure from the fields. The

quantitray was therefore analysed for the presence of the STII gene, which is a specific indicator of E. coli strains from pigs (Khatib 2003). The results were negative indicating that the source was not pigs, but might instead have been cattle or birds. Samples from the main canal upstream of the WWTP surprisingly also showed presence of E. coli which probably originated from domestic waste water leaching into the canal from the houses neighboring the upstream part of the canal. This might either be from leaking sewerage systems or from percolation pipes from septic tanks. For comparison, the EQS defining good quality of inland waters is 1000 E. coli per 100 mL (EU directive, 2006) whereas E. coli is not allowed at all in drinking water.

Table 4.6. Concentration of E. coli in water samples (analyzed by Amphi-Bac APs). MPN: most probable number.

	MPN/100 ml 24/04-2013	MPN/100 ml 14/08-2013
w4c-1: WWTP discharge	7300	8820
w4c-2: main canal 80 m downstream of WWTP	1300	3448
w4c-3: main canal 100 m upstream of WWTP	120	95
w4c-4: main canal between WWTP and pumping station	400	52
w4c-5: side-canal between WWTP and pumping station	<10	4107
w4c-6: kanal ved pumpestationen	330	120
w4c-7: small drain pipe 400 m upstream of pumping station	<10	--
w4c-8: large drain pipe 600 m upstream of pumping station	<1	--

The microbial EQS defining good quality of inland waters is 1000 E. coli per 100 mL (EU directive, 2006). The EQS was exceeded in two samples from the main canal close to the WWTP, but the load was reduced downstream. This suggests that injection of water from the canal would be less problematic with respect to faecal contamination as long as the water would have a retention time in the aquifer that is long enough for faecal microorganisms to be removed. It should be noted, however, that E. coli are not allowed at all in drinking water.

5. Assessment of processes and risks when injecting aerobic water into anaerobic chalk aquifer

The injection of water types (groundwater, surface water or rain water) different from the water types occurring in the aquifer at the injection wells will potentially have negative effects on the water quality at the well and result in water types breaching drinking water standards and/or create technical problems (clogging of screens and pumps of the water supply wells). Preferably the injected water type should be identical or nearly identical to the water type at the well. On the other hand biogeochemical reactions in the aquifer could be a benefit to the aquatic environment if aerobic water with nitrate and/or other contaminants are injected into the aquifer as especially nitrate will be removed efficiently from the injected waters in the anaerobic aquifer by reactions with reduced minerals such as pyrite.

5.1 Inorganic chemistry

A preliminary assessment of the consequences of injection of oxygen and nitrate containing (aerobic) groundwater was therefore conducted as part of the Water4Coasts studies assuming average concentrations of oxygen and nitrate in drainage canal waters and typical contents of pyrite and organic matter in the Chalk aquifer.

As the Chalk aquifer of Southern Falster is impacted by glaciotectonics (the movement of glaciers during the last ice age) as other Chalk deposits in the area e.g. Møns Klint about 50 km northeast of Marielyst clearly demonstrates, significantly affects the area, and as geological logs, borehole flow logs and Geoprobe hydraulic profiling tests also indicate, the hydraulics of the Chalk aquifer is highly varying and complex. Some flow logs indicate dual porosity systems with dominating fracture flow while other flow logs and HPT hydraulic profiles at other locations indicate single porosity behaviour in the region.

It was therefore decided to assess how pyrite in the Chalk would behave and affect water quality in both dual porosity fracture flow and single porosity systems. It should be noted that the behaviour of these two porosity “end-members” is quite different, and that the Chalk aquifer most probably would behave as a mixture of these two systems. Hence it is expected that the properties vary significantly both laterally and vertically i.e. that dual porosity behaviour with fracture flow would be dominant in some areas and at some levels (30 – 80 m below surface?), while single porosity behaviour would be dominant at other locations and depths (< 30 and and > 80 m below surface?). At all sites both porosity types may affect the hydraulics significantly depending on the depths. Only very detailed investigations will be able to decide, which of the porosity types dominate at a specific location and depth, and what is the situation at one well most probably will not describe the situation at the next.

In the following sections it is demonstrated how injection of aerobic water types into the anaerobic Chalk aquifer will affect the evolution of the concentrations of some key inorganic hydrochemical parameters dissolved in groundwater.

The first model, “Advective flow” described in section 6.1.1 on the following page is for Darcy flow with pyrite homogeneous distributed in the sediment. The second “Stagnant zone” model described in section 6.1.2 is for a heterogeneous flow system like dual porosity Chalk with hydraulic active fractures. The models are designed to be conceptual to illustrate the characteristic properties and differences between the systems.

Both models are for a breakthrough in a borehole with the filter at 20 m depth.
In both models there is Pyrite in the sediment to reduce the incoming oxygen and nitrate of the injected aerobic water type e.g. representing drainage water of the North Canal in the investigated area.

5.1.1 Simulating pyrite oxidation in single porosity media with advective flow

Figure 5.1 below illustrate the changes over time in oxygen, nitrate, sulphate, hydrogen (pH), alkalinity (HCO3-) and calcium concentrations.

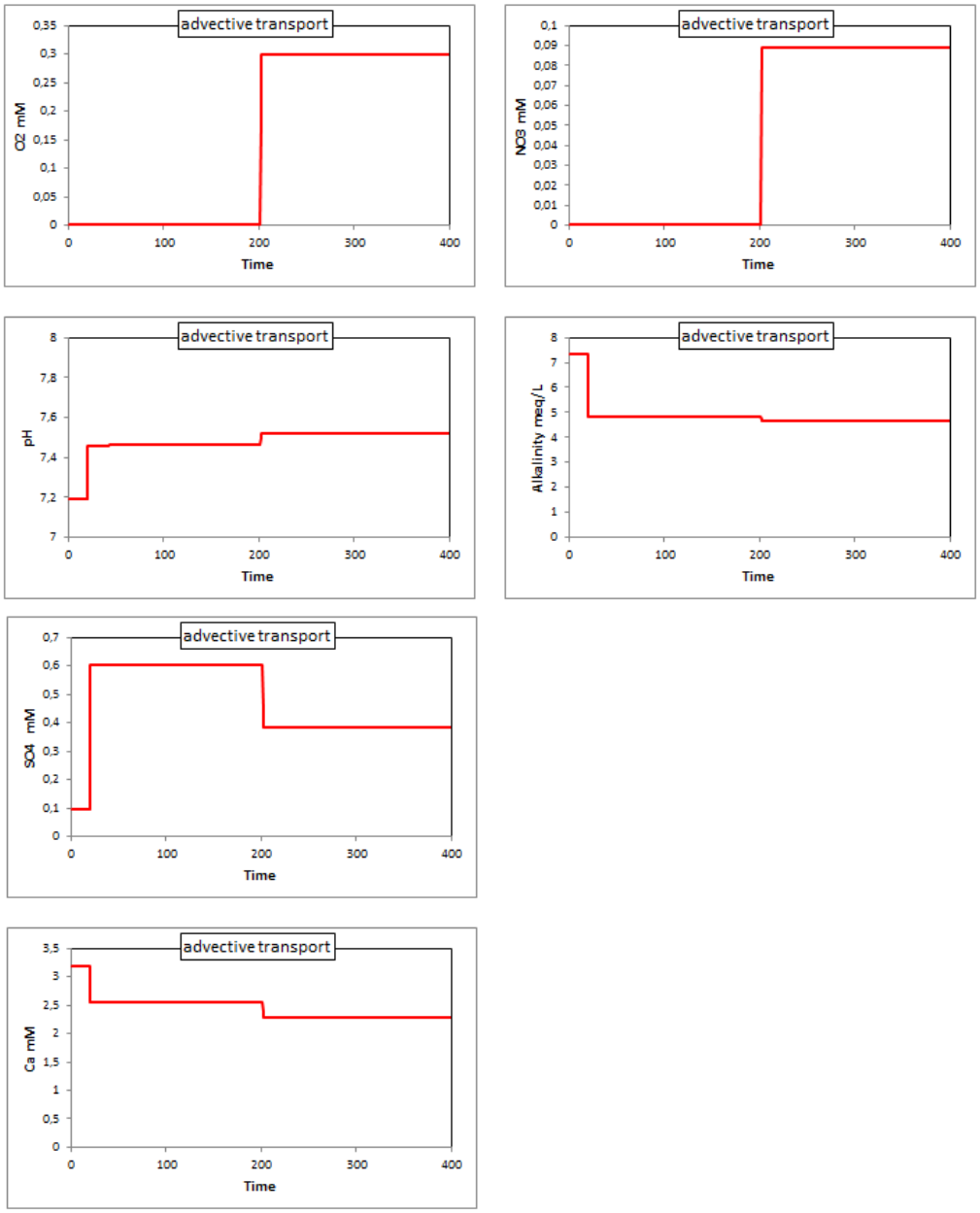


Figure 5.1. Evolution of O2, NO3, pH, Alkalinity, sulfate and calcium concentrations over time when injecting aerobic (oxygen and nind trate containing) groundwater into anaerobic pyrite containing single porosity aquifer.

Both depth and time are arbitrary because we have currently little information on the pyrite content of the sediment. In this example we have made the amount of pyrite present in the sediment small to illustrate what happens when the system becomes flushed.

The SO4 graph has three plateaus. The first is the original groundwater content. Next is channel water with sulfate added from pyrite oxidation and the final one reflects sulfate in channel water. The consumption of pyrite along the flow path is marked by the O2 and NO3 breakthrough curves. All Fe2+ released from pyrite is precipitated as goethite. However, the resulting Fe2+ content in the groundwater depends on the stability chosen for the Fe-oxide. If a less stable Fe-oxide is chosen, sulfide oxidation becomes energetically more favorable and less Fe2+ is oxidized (Increasing SI goethite from 0 to 2 increases Fe2+ to 0.05 mM).

In the channel; O2 = 0.3 mM and NO3 0.09 mM. Accordingly O2 is 2.6 times as important as pyrite oxidant compared to nitrate

The system is buffered by equilibrium with calcite. With respect to the carbonate system, the original groundwater has a lower pH and higher Ca and alkalinity than the channel water. It is caused by a higher P_co2 when the groundwater infiltrates through the soil. Less CO2 will be added when the channel water with the lower Ca, alkalinity and P_co2 is injected directly into the aquifer. This can be expected to be a general feature when injecting surface water into aquifers. The acid production from pyrite oxidation will be buffered by calcite dissolution, temporarily raising the Ca and alkalinity concentrations.

5.1.2 Simulating pyrite oxidation in dual porosity media with fracture flow

This situation corresponds to heterogeneous flow systems like fracture flow through dual porosity Chalk aquifers or alternating sand and clay layers. The reductant, in this case pyrite, will typically be located in the limestone matrix or clay layers where there is no advective flow and the only solute transport is by diffusion. PHREEQC can model this as exchange between a mobile zone and a stagnant, immobile, zone. For this example we have chosen the geometry of alternating sand and clay layers as described by Appelo and Postma (2005, p 508), and the results are shown in Figure 5.2.

In the stagnant model the oxidant (O2 or NO3) has to diffuse from the mobile zone (fracture) into the immobile zone (matrix) where the pyrite is located, with which it can react. The amount of pyrite in the immobile zone is set to be infinite as will be approximately the case in most natural systems at human time scales. Again the system is buffered by calcite and goethite may precipitate. The amount of oxidant (O2 or NO3) that becomes reduced from the water on its way to the screen at 20 m depth will depend on the flow velocity of the water. The faster the flow the shorter the residence time to diffuse into the immobile zone and less oxidant is reduced as well as less pyrite oxidized as is reflected by the sulfate concentration,

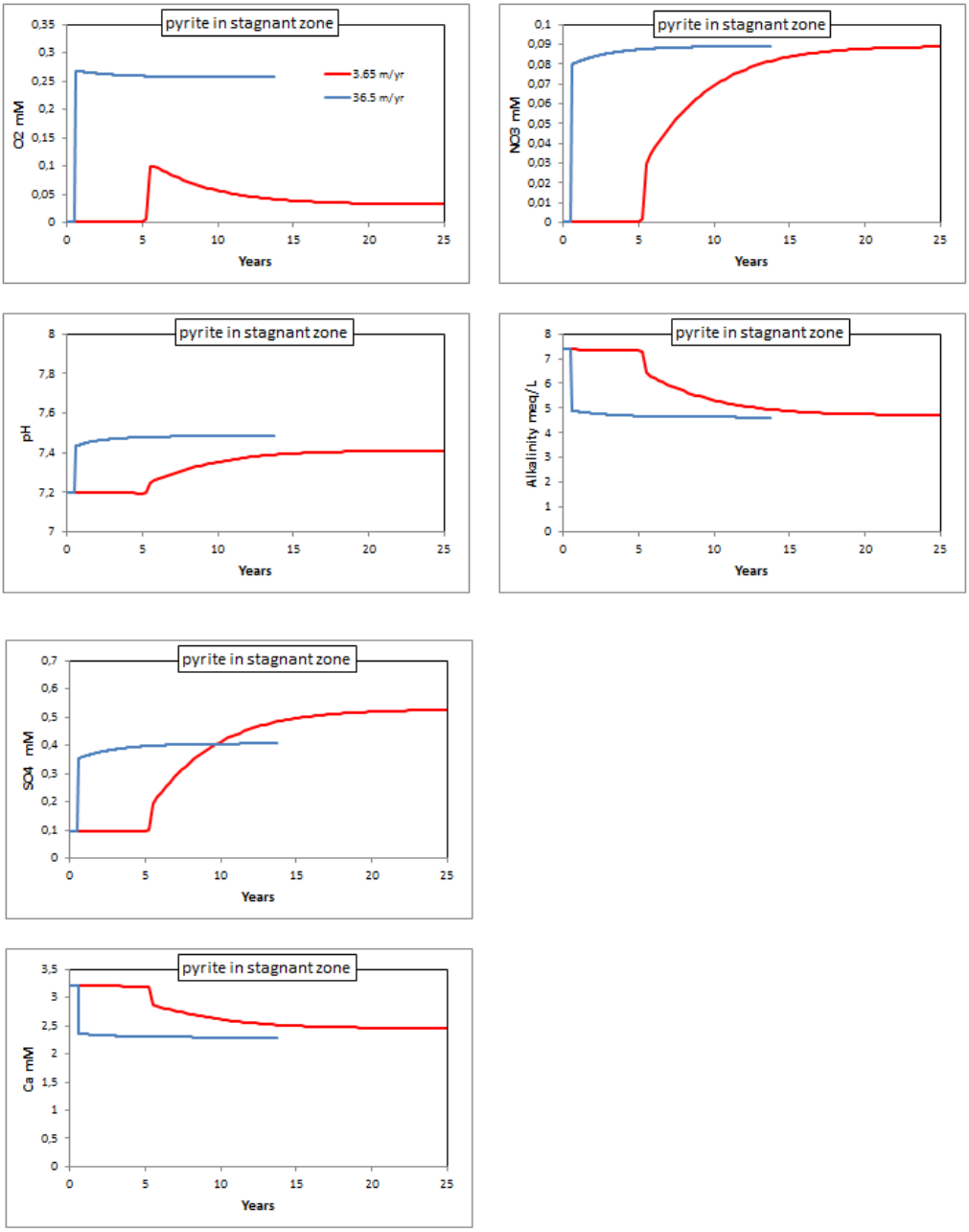


Figure 5.2. Evolution of O2, NO3, pH, Alkalinity, sulfate and calcium concentrations over time when injecting aerobic (oxygen and n trate containing) groundwater into an anaerobic pyrite containing dual porosity aquifer.

According to Fick's law the flux of oxidant diffusing into the matrix will depend on the concentration gradient. Since the concentrations of O2 and NO3 always will be zero in the immobile zone in the presence of pyrite (in an equilibrium model) the fluxes of O2 ad NO3 are directly proportional with the concentrations. In this case the O2 conc is much higher than the NO3 conc and therefore much more O2 is consumed. If the NO3 conc was much higher, the reverse could be the case. In reality it is a complex interplay between diffusive and advective transport and thermodynamics that controls the relative consumption of O2 and NO3. This is quite different from the advective case where thermodynamics are controlling and basically both are consumed at the same time.

As mentioned previously the Chalk aquifer on Southern Falster may be behave as a mixture of both systems with single porosity and advective flow being dominant locally in some areas while dual porosity may be dominating or significant in others. Only detailed investigation of the hydraulics of the Chalk aquifer by well-designed pumping and tracer tests will be able to identify how the hydraulics of the Chalk aquifer works at a given location. Future studies are expected to reveal this and to illustrate how efficient pyrite oxidation would be for nitrate removal in the aquifer and how precipitation of iron oxides may affect the efficiency of water withdrawal from the Chalk aquifer at wells with injection of aerobic water types.

5.2 Organic chemistry

The efficiency of anaerobic aquifers for removal of organic contaminants varies considerably depending on the type of contaminant, but many organic contaminants do not degrade or degrade only very slowly (are recalcitrant) under anaerobic conditions. There is a wide range of possible organic contaminants, which potentially could be found in the drainage canals besides the contaminants found in the North Canal and in the effluent from the waste water treatment plant, at the two sampling campaigns in the project, which may behave very differently, and it was not possible to assess the occurrence and behaviour of these within the scope of the project.

It should be noted that more than 35 pharmaceuticals were found in the discharge from the waste water treatment plant, and that many of these are very resistant to degradation in the environment. Although the effluent from the waste water treatment plant is diluted significantly in the drainage canal, that no pharmaceuticals were found above detection limits in the canal, and that some of the pharmaceuticals would be degraded in the aquifer, there may be a risk that the sum of the concentrations of the pharmaceutical would be a problem for the water works and water supply of the area if injected into the aquifer (Lopez-Serna et al., 2013). Hence, injection of water from the drainage canals into the Chalk aquifer cannot be recommended without an extensive treatment of these waters that efficiently remove the contaminants in combination with an extensive quality control on the injected water.

5.3 Pathogens

Like pharmaceuticals, pesticides and other organic contaminants pathogens is another relevant concern, when considering the injection of surface waters into fractured Chalk aquifers. E.coli was found in high numbers at one location during the project sampling (see section 4.2.5) clearly indicating that it cannot be recommended to inject surface waters into the aquifer without extensive treatment that will also remove any potential pathogens in the drainage waters.

Pathogens are generally removed quite efficiently in single porosity aquifers, but they may be transported fast over rather long distances in fractured aquifer systems (e.g. Masciopinto et a., 2008). Hence, efficient treatment is needed before injection of drainage water into the fractured Chalk aquifer on Falster.

6. Water pre-treatment options before injection, - efficiency and costs

Due to the detection of fecal bacteria and organic micro-pollutants in the drainage water different treatment methods will need to be implemented prior to injection into the groundwater aquifer. The broad spectra of contaminants, including potentially pathogenic microorganisms, industrial and household chemicals, pharmaceuticals and pesticide residues detected in the drainage waters may require a treatment train composed of different successive techniques. The most relevant approaches will be discussed in the following section in the light of their efficiency and costs.

6.1 Water quality requirements

The European Drinking Water Directive requires that drinking water, and water used in households and in food products, should uphold a specified quality level. The purity requirements for example include threshold limits for single pesticide residues at 0.1 µg L-1, and a limit for the sum of all residues at 0.5 µg L-1. The threshold for industrial and household chemicals are generally specified for each compound individually, where for example the threshold limit for nonyl-phenols are 20 µg L-1 and for acetone it is 2 mg L-1. In contrast to this, there are currently no threshold limits for pharmaceutical compounds. The occurrence of fecal bacteria, expressed a coliform bacteria, must not exceed zero in a volume of 100 ml water.

6.2 Removal of fecal bacteria

Several methods for disinfection of drinking water exist and the most commonly used are ultraviolet light (UV) irradiation and chemical disinfection using for example free chlorine, ozone or monochloramine.

6.2.1 UV and chlorination.

Irradiation with UV and chemical disinfection by e.g. free chlorine is effective at killing a broad range of microorganisms in natural waters. Some specific types of microorganisms may however elude inactivation in such treatments, where for example fractions of different spore-forming bacteria appear resistant (Rose and Rice 2014).

Common for both irradiation and chemical disinfection is the possibility for creation of degradation products from the organic micro-pollutants present in the drainage water. This could potentially broaden the range of organic contaminants making none of these options ideal for disinfection as a standalone technique in the current situation. A scenario where these processes are included in a multi-barrier treatment train could however be a possibility.

6.3 Removal of organic micro-pollutants

Industrial and household chemicals, pharmaceuticals and pesticide residues are more or less comparable organic compounds and their individual removal efficiency in a given treatment strategy will depend on their physical and chemical properties. This means that a chosen treatment approach in theory should be able to cover them all.

It has previously been found that conventional approaches such as chemical oxidation by free chlorine only is capable of removing approximately 50% of the investigated pharmaceuticals (WHO 2011). More promising, however, are advanced water treatment processes including for example activated carbon or membranes (e.g. reverse osmosis or micro- and nano-filtration), where removal efficiencies of above 99% have been reported for pharmaceuticals (WHO, 2011; Watkinson et al., 2007). Similar efficiencies can be expected for organic contaminants from the groups of industrial and household chemicals and pesticides.

6.3.1. Sorption on activated carbon.

Granulated activated carbon (GAC) or powdered activated carbon (PAC) are in some cases used in drinking water production for removal of pesticides, for examples – but it has also been implemented for removing organic compounds that negatively affects taste and odor of the drinking water. Activated carbon treatment is especially efficient against hydrophobic compounds and it has previously been shown, that for example the pharmaceuticals diclofenac and carbamazepine (Table 4.5) are removed efficiently by GAC in waterworks (Ternes et al. 2002). In another study with simulated drinking water treatment using PAC, the removal of for example the painkiller ibuprofen (Table 4.5) and a broad range of other pharmaceuticals and personal care products were less efficient (Westerhoff et al. 2005). A similar drawback of activated carbon filtration is the reduced efficiency against polar substances such as MTBE and X-ray contrast agents (Lipp et al. 2010).

Even though activated carbon could be an efficient approach for removing many organic contaminants it appears more relevant for purification of contaminated groundwater, where the particulate and organic matter contents typically is low. One might anticipate that treatment of water from the present study would lead to a frequent saturation of the activated carbon with natural organic matter from the drainage water. This would block the adsorption sites and restrict and eventually stop the removal of organic contaminants (Lipp et al. 2010). Operation of such a system under these conditions could pose a significant economic burden due to the need for periodically removing and regenerating the activated carbon filter.

6.3.2. Membrane filtration

Membrane filtration is based on retaining molecules due to size exclusion, electrostatic repulsion and/or hydrophobic adsorption (Yoon et al. 2006). Although conventional filtration can be efficient at removing a range of different organic molecules such methods can embed impurities in the membranes and accumulate a cake that must be back-flushed periodically to restore productivity (Stover 2014). Advanced membrane processes involving for example nanofiltration or reverse osmosis technology has been shown to efficiently remove a broad spectrum of pesticide, pharmaceuticals and personal care products from contaminated water (Yoon et al. 2006; Lipp et al. 2010; Bodzek et al. 2011).

Reverse osmosis can remove > 99% of large organic molecules, such as most pharmaceutical compounds and pesticides (Lipp et al. 2010; WHO 2011; Bodzek et al. 2011) for example. To prevent clogging of the filters reverse osmosis uses a crossflow to keep the membrane surface clear (Stover 2014). This produces a concentrate containing higher levels of the contaminants

and discharge of this fraction need to be addressed (Lipp et al. 2010). Besides the organic pollutants, reverse osmosis has also been used to remove inorganic compounds as nitrate, fluoride ions, boron, arsenic, chromium and different heavy metals (Bodzek et al. 2011). Additionally, microorganisms including e.g. bacteria, protozoa and viruses are also removed during the filtration (Bodzek et al. 2011).

6.4 Suggested treatment strategy

The drainage water attended for injection into the aquifer is affected by fecal bacteria and a range of different organic micro-pollutants – both the ones detected within the Water4Coasts project (Tables 4.3 – 4,6), and without a doubt also a broad range of new emerging compounds not included in our analytical survey. This is obviously a challenge when selecting an efficient treatment strategy.

Composing a multi-barrier approach for water purification with successive techniques such as activated carbon filtration followed by chlorination or UV disinfection could be a strategy. As mentioned above however, activated carbon filters will likely be saturated with natural organic carbon from the drainage water and a frequent and expensive removal and regeneration of the filters need to be implemented.

Technological improvement with advanced membrane systems has however reduced the cost by e.g. reverse osmosis by a factor of four or more (Stover 2014). Ongoing research and development will likely reduce the price even further. Reverse osmosis technology will target all the detected compounds and bacteria and such a versatile technology appears to be the most relevant choice for treatment of the drainage water. Additionally, such a strategy will likely also cover new organic pollutants detected in future monitoring programs.

Initial reverse osmosis purification could be connected to a secondary activated carbon filter, with the aim of removing micro-pollutants having a low molecular weight, as these might pass through the reverse osmosis step (Lipp et al. 2010). A reverse osmosis strategy leaves a concentrate behind from the crossflow of membranes, and this fraction could be transported to the nearby waste water treatment plants. The flow in the suggested treatment strategy is illustrated in figure 6.2.

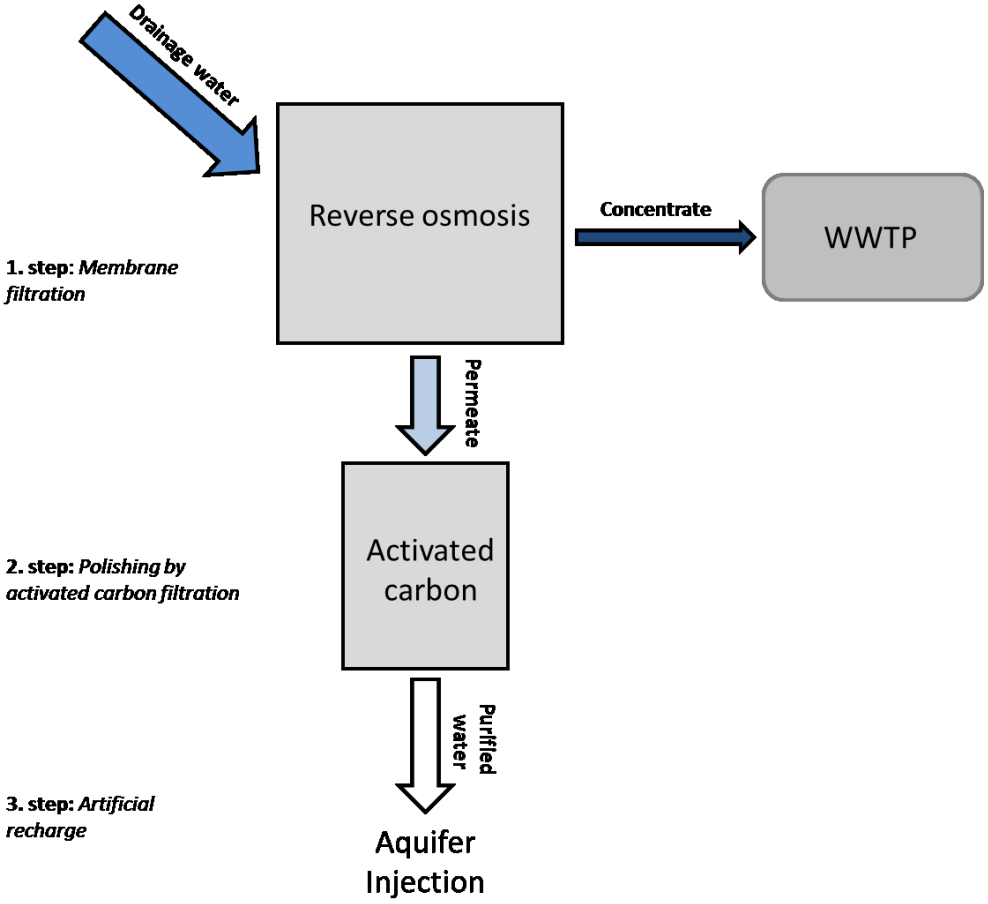


Figure 6.2: A possible treatment strategy for the drainage water prior to injection into the groundwater aquifer. 1. Step: membrane filtration with reverse osmosis; 2. Step: removal of low molecular weight organic compounds from the permeate by activated carbon sorption; 3. Step: artificial recharge of the aquifer. Concentrate produced from the reverse osmosis will separately be treated in a nearby wastewater treatment plant (WWTP).

7. Discussion and conclusion

Managed aquifer recharge and water banking (storage of water for later use) are techniques of increasing importance (Megdal and Dillon, 2015; Zhao and Wang, 2015), which have been increasingly studied and applied for protection, storage and recovery of water resources, globally, during the past decade (Figure 7.1), and there's a strong need to gain experience with the efficiency, positive and negative effects of the different MAR designs in different climatological and geological settings as well as to develop sound policies and risk management frameworks (Nandha et al, 2015).

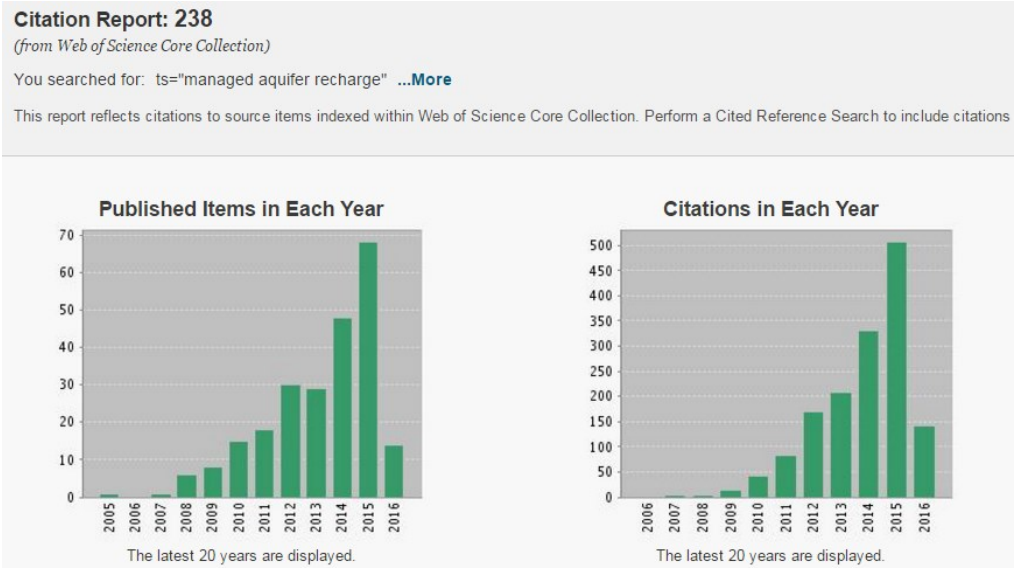


Figure 7.1. Plot from the Citation report from Web of Science (Thomson Reuters) for a search on the keyword "managed aquifer recharge" in the Web of Science Core Collection.

The application and experience with MAR is still limited in Denmark, but the technique potentially have several beneficial effects that may be applied for the protection of both water resources, ecosystems and the built environment especially in densely populated coastal areas with large groundwater abstraction and drawdown that increase salt water intrusion to coastal aquifers. Hence, there's a need for developing experience and policies for MAR in Denmark, and this report together with the related reports on modelling and monitoring (Rasmussen et al., 2015; Hinsby et al. 2015) provide initial background information of relevance for the further assessment of the application of MAR in Denmark.

The application of MAR is still in its infancy, and the development and application of sound principles for MAR is strongly needed for sustainable water resources management in many areas of the world with excess groundwater abstraction. The application of MAR introduces risks to the quality of the groundwater resources unless proper policies and risk management frameworks for MAR are developed (Nandha et al., 2015) in collaboration between research organisations and water authorities.

This report describes initial assessment of some water quality aspects of MAR that need to be addressed before injection of aerobic groundwater or surface water into the investigated dual-

porosity Chalk aquifer. The assessment clearly show that efficient treatment and water quality control is required both of waters injected to and recovered from the aquifer in order to ensure that the MAR procedures do not jeopardize the quality of the groundwater resources abstracted for drinking or irrigation water or for sustaining good ecological status of groundwater dependent terrestrial or associated aquatic ecosystems.

Substances of concern include both inorganic trace elements such as arsenic and nickel, micro-organic contaminants such as pharmaceuticals, industrial chemicals and pesticides and it includes the possible occurrence of pathogens in surface waters. Hence, efficient pretreatment before injection of such water is required in order to ensure that no harm is done to the groundwater quality at and around the injection wells, and in addition efficient quality control of recovered waters are also required in order to demonstrate that the injection has not induced geochemical reactions and e.g. release of harmful trace elements etc.
Concerns associated with application of MAR in the Marielyst area of southern Falster and in general, are mainly related to drinking water quality i.e. human health and the efficiency of the drinking water abstraction wells (clogging risks)

Whether extensive pretreatment and quality control is economically feasible depends on what other options may be available at the investigated site, and this have to be compared and evaluated in cost-benefit analyses. Potential other options and the costs of the treatment compared to these strongly depend on local conditions, and hence they have to be evaluated from case to case. MAR in combination with pretreatment in constructed wetlands may be an interesting option for some areas (Hamadeh et al, 2014).

Managed aquifer recharge may not be the first choice for sustainable water management at a given site, but at some sites it may be the only option, and without any doubt it will be of increasing importance globally for sustainable water resources management in the future considering projected population growth and climate change impacts.

8. Recommendations

This report (and Hinsby et al., 2016) briefly describes the fast development of studies on the application of MAR during the past decade reflecting the need for the development of new techniques for sustainable management and conjunctive use of water resources, globally. However, it also indicates that there are many water quality and technical issues that have to be resolved and efficiently tackled in both policies and technical guidances in order to ensure good practices for MAR under different hydrogeological conditions.

The authors recommend and encourage that regional and national authorities and research institutions as well as the European Commission collaborate on the development of policies, risk management frameworks and technical knowhow and guidances related to proper development of MAR practices. Water resources are under severe pressure due to increasing abstraction, population growth, climate change and sea level rise and there is an increasing need for development of sustainable water resources management and conjunctive use of groundwater and surface water, globally. Managed aquifer recharge may be the most promising technique for sustainable water resources management in coastal regions and other regions with e.g. rapidly declining water tables and salt water intrusion, and new tools and innovative approaches using different designs of MAR such as the techniques described by Zuurbier et al. (2014, 2015) should be supported and encouraged.

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Bilag 3. GEUS rapport 2015/94 – Real time groundwater salinity monitoring - data transfer, management and visualisation on interactive web servers

WATER4COASTS

Real-time groundwater salinity monitoring - data transfer, management and visualisation on interactive web servers

Hinsby, K., Thorn, P., Scharling, P.B. and Gudbjerg, J.

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1. Summary

Saltwater intrusion in coastal aquifers is of increasing concern for drinking water supply in coastal regions globally and efficient monitoring and control systems to assess and control salt water intrusion in coastal aquifers is of increasing importance.

In the Water4Coasts project we initiated work on developing and testing new monitoring sensors for measurements of chloride concentrations and efficient data upload systems to provide the necessary data for the development of a future real-time SCADA (Supervisory Control And Data Acquisition) framework for control of salt water intrusion.

The obtained data and the developed SCADA system are mainly required for controlling salt water intrusion and protecting fresh groundwater resources at coastal water wells and water works. However, it will also be well suited for sustainable management of aquifers further inland, which are affected by intrusion of saline waters from either old marine sediments or evaporation (in arid or semi-arid areas).

In this technical report we describe the outcome and main recommendations and conclusions of the conducted work towards the development of new SCADA systems for efficient control of freshwater abstraction and salt water intrusion at water works based on the monitoring of chloride concentrations in groundwater. We believe that we have developed an efficient approach for monitoring, data handling and visualisation of chloride concentrations etc., which will be key parameters for the development of future efficient SCADA systems to control salt water intrusion in aquifers. We have encountered some technical problems with the long term stability of field sensors, when they are located in surface water, and with the performance of data loggers, which have to be improved before a fully operating system is in place. However, we believe that it is possible to overcome the technical problems in the near future, and the project group will continue to develop the acquisition of data required for SCADA solutions etc. for controlling salt water intrusion in existing and future related projects.

2. Background

Coastal aquifers are affected by saltwater intrusion globally, and have been under pressure for many decades especially in densely populated areas with significant groundwater abstraction for water supply (Post and Abarca, 2010; da Silva and Montenegro, 2012). Such problems are especially known and managed (controlled) in aquifer systems along coastlines of U.S.A. and Northern Spain (Sheahan, 1977; Iribar et al., 1997; Misut and Voss, 2007; Barlow and Reichard, 2010; Custodio, 2010), but problems appear in an increasing number of coastal aquifers in different climate zones, globally, including many coastal aquifers in northeastern Brazil (Bocanegra et al, 2010; da Silva et al., 2010) , China (Wu et al., 1993; Xue et al., 2000, Liu et al., 2001; Han et al., 2011) and Denmark (Andersen et al., 2005; Jørgensen et al, 2008; 2012, Thorn, 2011; Rasmussen et al., 2013), and measures need to be implemented in order to ensure freshwater supplies in these regions in the future.

Climate change is projected to affect precipitation patterns and increase sea levels globally and thereby increasing the risk of saltwater intrusion and deterioration of coastal water resources (Kundzewicz et al., 2007; Hinsby et al., 2012; Loaiciga et al., 2012). Although the extend of the saltwater intrusion highly depends on the geological setting (Werner and Simmons, 2009; Chang et al. 2010), and also may stem from other sources than recent seawater such as old connate waters in former marine sediments (Buckley et al., 2001; Bonnesen et al., 2009; Tran et al., 2012) and even may include highly saline brines (Xue et al., 2000; Han et al., 2011). Especially low-lying areas and areas that receive less precipitation, and experience falling water tables, will be under severe pressure (Essink et al., 2010).

Hence, salt water intrusion due to climate change, sea level rise and increasing demands constitute an increasing challenge for many water supply companies in coastal regions, globally, and calls for the development of new adaptive and innovative solutions to prevent salt water intrusion, monitor the development and ensure sustainable and high quality water supply in the future (Essink, 2001; Post, 2005; Post and Abarca, 2010; Hinsby et al., 2012, Thorn and Mortensen, 2012; Sulzbacher et al., 2012). Managed aquifer recharge through basins in phreatic aquifers or injection wells in artesian aquifers are both possible solutions, which have been applied and operated at large scales for decades mainly around large cities such as Los Angeles (since the early 1950's) (http://dpw.lacounty.gov/wrd/barriers/historical.cfm), and Barcelona where treated wastewater are injected to avoid saltwater intrusion and ensure water supply for the city (Hernandez et al., 2011; http://portail2.reseauconcept.net/Upload/ciheam/fichiers/OM_A_88.pdf). Freshwater injection and aquifer storage and recovery is furthermore considered for emergency water supply of in New York City (Misut and Voss, 2007). Managed aquifer recharge will most probably be a cost-efficient way to protect water resources globally in the future (Megdal and Dillon, 2014).

Besides the salt water intrusion problem coastal catchments face other important problems that are expected to become more severe due to climate change and its impact on the hydrological cycle, water tables and sea level.
The problems vary between regions and climatological settings, but include 1) Increased flooding risks (Thodsen 2007; Dankers and Feyen, 2008; Sonnenborg et al., 2012) and 2) increased nutrient loadings to rivers, lakes and coastal marine waters in wet temperate regions (Hinsby et al., 2008, 2012; Sonnenborg et al., 2012; Wikner and Andersson, 2012) and 3) drought and declining water resources in arid and semi-arid regions (Ragab and Prudhomme, 2002; Montenegro and Ragab, 2012).

Nitrogen loadings and resulting effects is currently considered to be one of the most severe environmental problems on a global scale (Rockström et al. 2009) by e.g. increasing the risk of eutrophication, harmful algal blooms, hypoxia, fish kills and acidification in many coastal waters (Diaz and Rosenberg, 2008; Rabalais et al., 2009; Cai et al., 2011; Paerl and Paul, 2012) including coastal waters in Brazil, China and Denmark (Diaz and Rosenberg, 2008; Cai et al., 2011; Hinsby et al., 2012). The Baltic Sea most probably exhibits the largest problems at present with oxygen depletion and hypoxia due to excessive nutrient loading, even in a global perspective (Diaz and Rosenberg, 2008; Conley, 2012). Partly for that reason the status of the major part of shallow oxic groundwater in Denmark (located in the Western Baltic Sea) is of poor status and does not comply with EU directives as groundwater threshold values for total nitrogen ("nitrate"), which have been established according to the EU Water Framework and Groundwater Directives to protect associated aquatic ecosystems are breached (Hinsby et al., 2008, 2012). This puts a strong pressure on Denmark to reduce nutrient loadings to coastal waters in Denmark and the Western Baltic Sea, and to find new innovative and cost-efficient ways for this purpose.

3. Introduction

The WATER4COASTS project seeks to developed new and efficient solutions for integrated, sustainable and adaptive management of groundwater and surface water that take into account both surface and subsurface water quantity and quality aspects for protection of water resources and ecosystems in coastal regions. By injecting (managed aquifer recharge) water from waste water treatment plants and drainage canals into for instance artesian anaerobic aquifers the hydraulic pressure increase and reduce the risk of saltwater intrusion, Misut and Voss, 2007; Hernandez et al., 2011), while at the same time nitrate in the injected water and the stage in drainage canals are reduced, as well as the risk of eutrophication and flooding.

The main objective of WATER4COASTS is to develop and demonstrate new efficient tools for sustainable and adaptive water management in coastal aquifers and catchments based on managed aquifer recharge to coastal aquifers in order to ensure good status of water resources and coastal ecosystems and:
1) Prevent salt water intrusion towards coastal well fields
2) Reduce nitrogen loads from coastal catchments to marine waters
3) Reduce the risk of flooding in the hinterland from e.g. drainage canals and groundwater, and
4) Monitor the evolution of groundwater chemical and quantitative status

This technical report describes the outcome of activity 1.5: *Development of cheap chloride and nitrate sensors,* 1.6: *Data handling and water information technology, and* 3.2: *On-line dissemination on interactive webserver.*

4. Sensor development, testing and data transfer

4.1 Sensor development

Sensor Development Background and Goals:

Previous studies by Thorn and Mortensen (2012) and Thorn and Urish (2013) have illustrated the potential for the use of a simple chloride electrode for continuous monitoring of groundwater salinity at concentrations lower than what conductivity measurements can provide. The sensors consisted of a simple oxidized silver rod in a plastic casing. Combined with a reference electrode, the sensor can provide an inexpensive system for continuous measurement of chloride concentrations as low as 5 mg/l. However, these measurement were conducted after the sample water was pumped up from the well and not in-situ.

Bendikov (2005) described the potential for producing relatively inexpensive, solid-state sensor, similar to those that can be found on the market for approximately $150. However, these sensors use an organic based film as its sensing surface, which have not been tested outside of laboratory conditions. Thorn (2010) found difficulties in using similar organic films for sodium sensors in the field, and therefore more testing on field application needs to be done.

The primary goal for sensor development is to construct a simple array that can be placed either in a groundwater monitoring well or surface water body, which then can be connected to a data-logger with GPRS connection to send the signal real-time to a server for further data presentation and analysis. In order to achieve this the specific objectives are:

- Construct a sensor array with chloride and nitrate sensors, assessing signal degradation between the sensors and data logger;
- Assess the durability of the sensors for continuous, long-term application in both groundwater and surface water;
- Assess the possibility for continuous, real-time data transmission from the data-logger to project servers.

Theoretical Background for Ion Selective Electrodes (ISE):

Ion selective electrodes use the principles of potentiometry in order to determine the specific ion concentration being measured. Potentiometry, in its basic sense, is the measurement of voltage between two electrodes in an electrochemical cell, in order to measure concentrations of dissolved ions in a solution (Harris 1996, Manahan 2001). The Nernst equation is used to account for the effect of the different ion activity in this electrochemical cell. Since activity is essentially equivalent to ion concentration, the Nernst equation states:

$$E = Eo - \frac{2.303RT}{nF}\log[a] \quad (1)$$

where E is the measured potential, Eo is the standard electrode potential, both measured in volts, R is the molar gas constant, T is the absolute temperature, n is the ion valence, F is the Faraday constant and [a] is the concentration of the ion. Using a standard reference temperature of 25ºC, the Nernst equation simplifies to:

$$E = Eo - \frac{0.059}{n}\log[a] \quad (2)$$

From this equation, it is apparent that the potential is proportional to the log of the concentration for the measured ion. Therefore, under ideal conditions, there is expected a change of 59 mV per every decade (10-fold) increase in concentration for the chloride and nitrate sensors. This slope, of course is an ideal “Nernstian” slope. When dealing with ion-selective electrodes, the actual response of the electrodes will often be less than that of the ideal Nernstian slope (Brown et al. 1976, Bühlmann et al. 1998, Hobby et al. 1983, Sutter et al. 2004). This can be accommodated through calibration of the electrode through standards of changing ionic concentrations. Through the calibration of the sensor using known concentrations, both the slope and Eo can then be calculated for the ISE. Thus, the calibrated slope can be used in place of the ideal slope, and the ISE can then be used to calculate (measure) the concentrations in unknown samples.

4.2 Sensor Array Construction

Chloride sensors**:**
The chloride sensors were constructed using silver rods, 2.5mm in diameter and 2.5cm long. They were encased in customized plastic housing for installation, with one end of the rod flush with the end of the plastic housing (Fig. 1). The other end was attached to a BNC cable electric cable, which is connected to the data logger.

Nitrate sensors:
The nitrate sensors used in this study were solid film ISE sensors from Vernier, which use a similar sensing surface to that developed by Bendikov (2005). Like the chloride sensor, the nitrate electrode was attached to a BNC cable, which was then connected to the data logger. The Vernier sensors were tested in order to see if the technology can work in the field before further development of the technology described by Bendikov (2005) was applied.

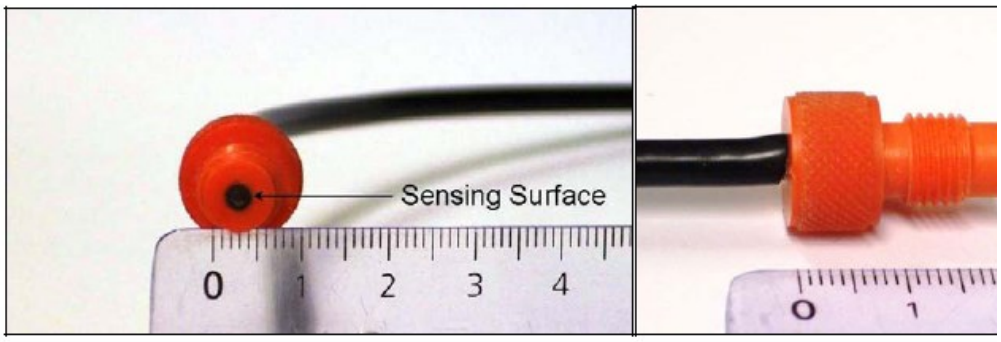


Figure 1. Chloride sensor used in the constructed sensor array (scale in cm).

Reference electrode:
The reference electrode used in the sensor array was a double-junction glass electrode produced by Sigma-Aldrich. The reference electrode was attached to a BNC cable, which then was connected to the data-logger.

Sensor array:
The sensors were sealed in 2,5 cm diameter PVC tube. The sealed tube was necessary to keep the contact between the sensors and the BNC cable dry. The tubing was also weighted with dried fine gravel in order to allow the sensor array to sink in water column, and provide

more stability for application in surface waters. Length of the BNC cable between the sensors and the data-logger varied from 5m to 25m.

Data-Logger:
The data-logger used in the study was produced by ComSystem A/S (www.comsystem.dk). The data-logger was able to receive potential signals at +/- 500 mV as well as temperature. The data-logger came with GPRS capabilities, which then sent the collected potential and temperature data to the receiving server. Data was logged once every 15 minutes, and then sent via GPRS once every hour.

Figure 2 shows the constructed sensor array and data logger.

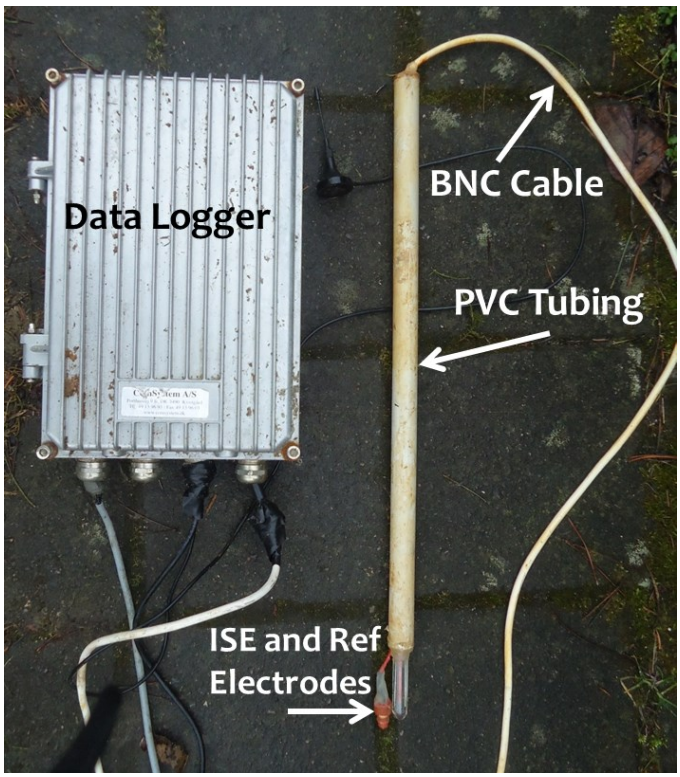


Figure 2. Photograph showing the sensor array and the data logger. The sensor array in this picture contains the chloride sensor and reference electrode at the end of the PVC tubing connected by a BNC cable to the data-logger.

4.3 Sensor testing

Sensor Calibration:

The first step is determining how well the sensor array/data-logger used in the field compares to ideal application in laboratory conditions. The purpose is to determine and quantify the signal degradation between the sensor array and data-logger. For ideal conditions, both the chloride and nitrate sensors were connected directly to a laboratory voltmeter. The potential was measured in solutions with nitrate and chloride concentrations between 10^{-5} and 10^{-1} mg/l, producing the ideal calibration curves for the sensors. The sensor array, connected to the data-logger to be used in the field was then tested on the same solution concentration ranges, and compared to the ideal conditions. The testing was conducted in the laboratory.

The results of the sensor calibration are shown in Figure 3. In this case, there was a 5 m cable between the sensor array and data-logger. For the chloride sensor, there was very little difference between the calibration curve for the sensors connected directly to the laboratory voltmeter and the sensor array connected to the data-logger. In both cases, the minimum concentration for detection was about 5 mg/l chloride, and the relationship of approximately 45 mV decrease in potential for every decadal increase in chloride concentration. This is slightly less than the Nernst ideal of 59 mV decrease per decade concentration. The chloride sensor was also tested with a 15 m and 25 m cable between the sensors and data-logger with similar calibrating results.

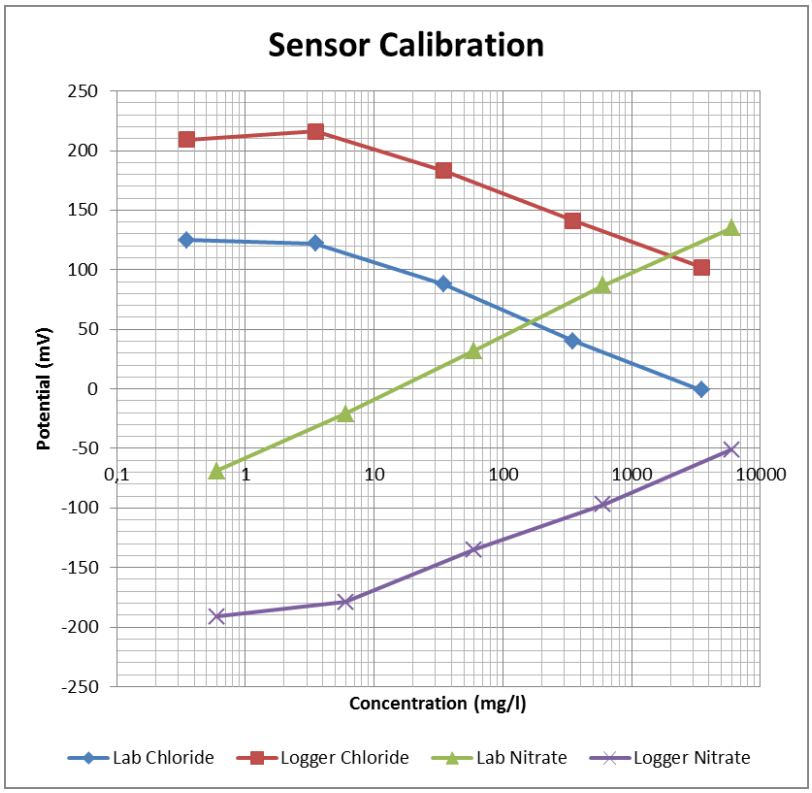


Figure 3. Calibration curves for the sensors before field application. Lab Chloride and Lab Nitrate show the curves for the sensors when connected directly to the voltmeter in the laboratory. Logger Chloride and Logger Nitrate show the curves for the sensors in the sensor array connected to the data-logger to be used in the field tests.

The nitrate sensor revealed a slightly degraded signal between the measurements from the laboratory voltmeter and the sensor array connected to the data-logger (Figure 3). This is, again, with a 5-meter cable. In this case, calibration curve for the nitrate sensor connected to the voltmeter had a response of just over 50 mV/decade concentration change, with a minimum concentration detection of under 0.6 mg/l. In contrast, calibration curve for the nitrate sensor on the sensor array showed a relationship of approximately 45 mV/decade, with a minimum concentration detection limit of 6 mg/l. In spite of the slightly degraded signal in the sensor array/data-logger setup, it is determined that the response is still good enough so that the sensor can be tested in the field.

Field Application – Surface Waters:

The sensor array with both nitrate and chloride sensors were tested in the field in two different sites. The first site tested the nitrate and chloride concentrations of treated wastewater right before it was released into a drainage canal. The test was conducted between August 31 and October 1, 2013 with a shorter follow-up two week test in November 2013. The second site

was in a drainage canal near a pumping station, where flow was maintained by the pumping of water from the canal to the sea. Here the test was conducted between March 10 and May 14, 2014. A calibration of the sensors was conducted both before and after each test.

The results of the calibration of the sensors before and after the tests are shown in Figure 4. Before the testing in both the treated wastewater (Aug. 2013) and in the drainage canal (Mar. 2014), both the chloride and nitrate sensors had good responses at the same level as the calibration tests described above. However, at the end of the tests, neither sensor had any sort of response. Therefore at some point in both tests, the sensors became no longer sensitive to changes in either chloride or nitrate. It was observed at the wastewater treatment plant that there were snails attached to the sensors which could be a source of fouling and reduced sensitivity. Therefore a fine net was put over the sensor array for all subsequent surface water tests in order to help protect the sensor surfaces, but yet allow water to flow freely through the net. The second short test in November, 2013, however, fouling of the sensor surfaces still occurred, with the sensors unresponsive after just two weeks.

At the end of the tests, both the chloride sensor and reference electrode were able to be cleaned and reconditioned for use. The nitrate sensor, on the other hand, was not able to be reconditioned and had to be replaced after each test.

Because the sensors failed at some unknown point during the tests, it is impossible to evaluate the drift or noise from the sensors applied in the surface waters. Figure 5 shows a plot of the signal received from the sensors in the test near the pumping station in March to May 2014. The signal shows an unstable nitrate signal, however, the sensitivity for nitrate is higher because the concentration is so low (measured at about 3 mg/l). This is also at the lower end of, if not below, the sensitivity range of the sensors as determined in the initial calibration. Therefore the nitrate sensor, because concentrations are so low, is likely not responding to changes in concentration. This was also the case at the wastewater treatment plant, where nitrate concentrations were measured from samples to be 1 mg/l. Because the nitrate concentrations were very low contrary to expectations prior to monitoring, it is not possible to determine when the nitrate sensors stopped working.

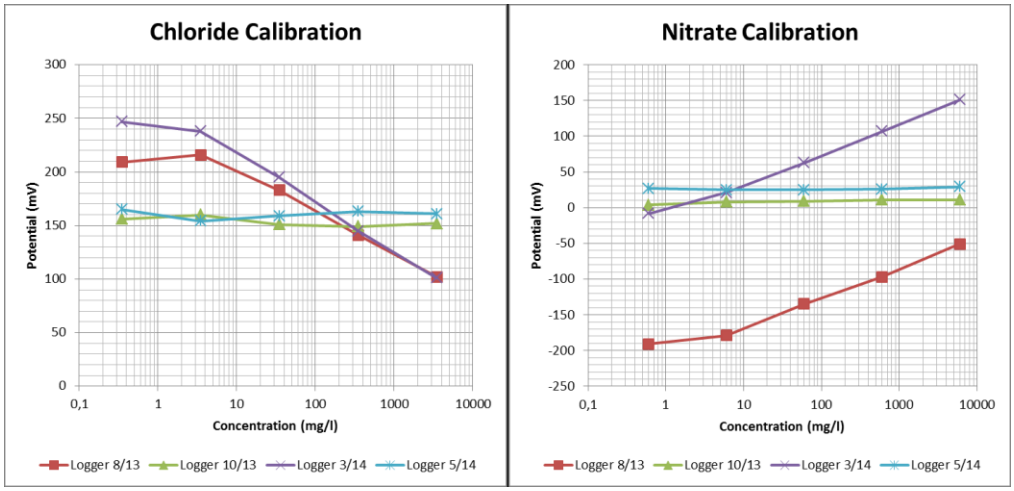


Figure 4. Calibration of the chloride and nitrate sensors before and after the surface water tests. Logger 8/13 shows the calibration before the wastewater treatment plant test (August 2013), with Logger 10/13 showing the calibration after the test (October 2013). Logger 3/14 shows the calibration before the pumping station test (March 2014) and the calibration after the test (May 2014).

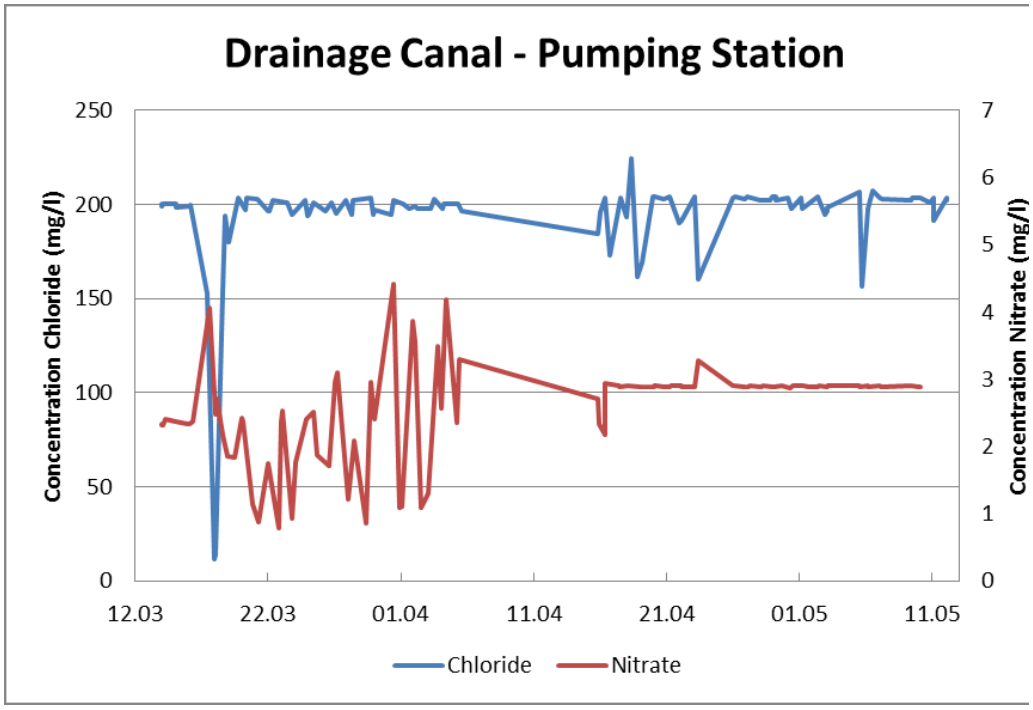


Figure 5. Chloride and nitrate measured with the sensor array and data-logger, as measured at the drainage canal near the pumping station between 12 March, 2014 and 11 May, 2014. Note that there is a 10-day period in the middle of the monitoring period where power to the data-logger was lost with the battery needing to be repaired.

In both tests, the chloride sensors had a more stable signal than the nitrate when applied in the surface waters. However, in both tests, as the monitoring progressed the chloride sensor became more unstable. This is illustrated in Figure 5, where the sensor was relatively stable before April 11, whereas afterwards it became much more unstable. Although it is impossible to determine, this could be the point when fouling made the chloride sensor unstable an unresponsive to the actual chloride concentrations in the surface water.

In summary for the surface water tests, it is clear that fouling is a problem. In all tests, both the chloride and nitrate became unresponsive to changes in concentrations. This is likely due to biological fouling of the sensing surfaces. It is not possible to determine how long the sensors remained responsive, but it is likely that the lifespan of the sensors in the surface waters is under two weeks (under the testing conditions of this study).

Field Application – Groundwater Monitoring Well:

The chloride sensor was applied in a monitoring well at Marielyst Waterworks. In this set-up there was a 15m cable between the sensor array and data-logger, with the sensors deployed at 14 m under the water table – close to the screen interval in the well. The sensors were deployed from the 26 September, 2014 to the 22 November, 2014. The sensors were calibrated both before and after the study, with very little difference between the two calibration curves (Figure 6). Thus it is clear that, unlike in the surface water, the chloride sensor remained responsive through the entire deployment period.

The signal received from the sensors was also stable (Figure 7). The chloride concentrations as measured from the sensors was seen to decrease from 272 mg/l on September 26 to 263 mg/l on October 15; a decrease of 9 mg/l. Noise in the signal was also low, less than the equivalent of 2 mg/l chloride concentration. Samples taken from the well at the start of the deployment and on October 17 showed a chloride concentration of 272 mg/l and 275 mg/l

respectively. A third sample taken on November 22 had a concentration of 266 mg/l showing that there was very little change in chloride concentration in the monitoring well. Assuming a stable chloride concentration throughout the test, the sensors had a drift of 12 mg/l over the period between September 26 and October 15, with an average drift of -0,6 mg/l per day.

It should be noted that after October 15, there were technical problems with the data-logger. The power and signal from the logger became unstable, even after repairs were made. The signal was received sporadically after October 15, with the data-logger ceasing to function at all after November 17. Therefore, there is no logging data for the sensors after October 15 which could be used to assess the drift and stability of the sensors after October 15, 2014.

In summary, the chloride sensor deployed in the groundwater well performed very well. The signal received when the data-logger was working (over 20-days) was stable, with drift a relatively steady -0.6 mg/l per day. Calibration tests showed that sensors were just as responsive after the 2-month deployment as they were before deployment.

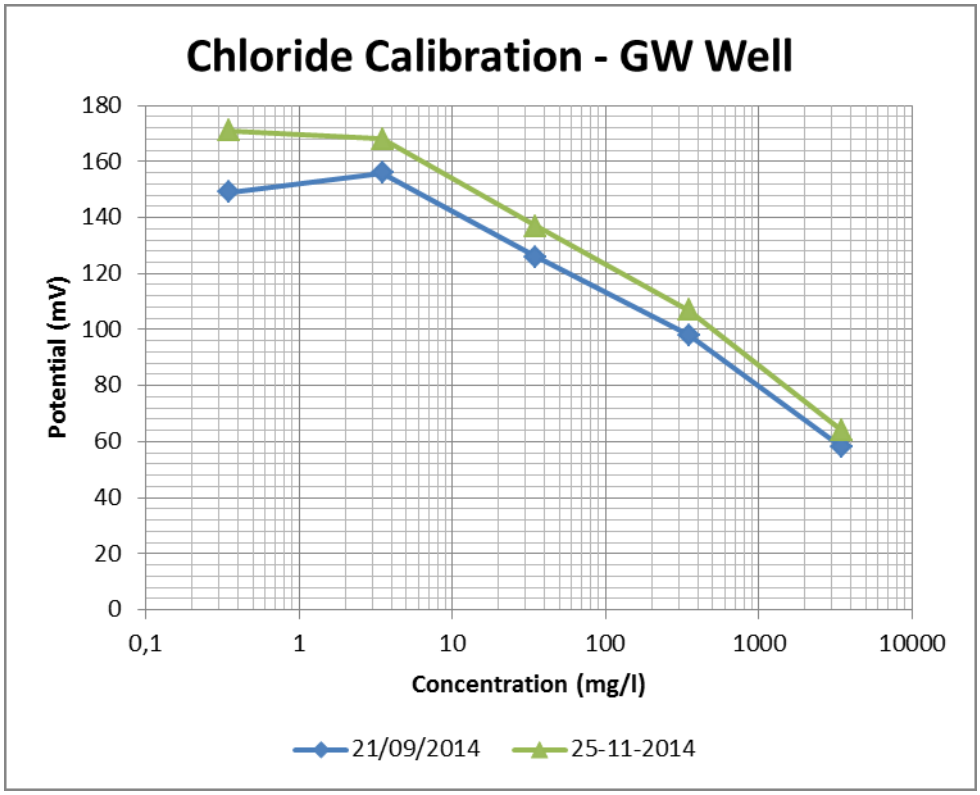


Figure 6. Calibration of the chloride sensor before and after deployment in the groundwater monitoring well at Marielyst Waterworks.

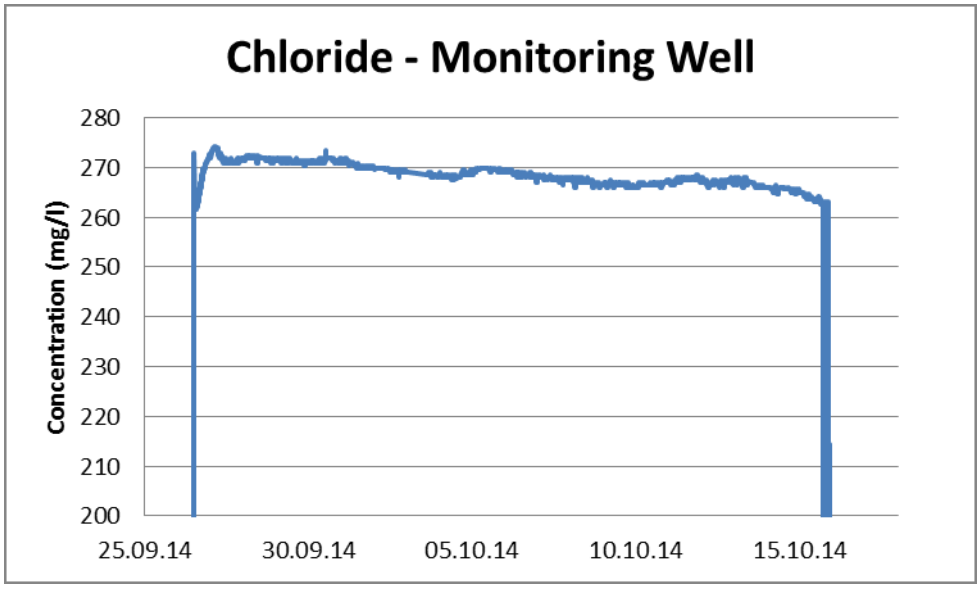


Figure 7. Signal response from the chloride sensor. Note the low measurement in the beginning shows when the sensors were deployed and turned on. The low measurements just after 15 October shows the time when the data-logger started to malfunction.

4.4 Data collection and transfer

Data was gathered and transferred with the data-logger from Comsystem (www.comsystem.dk) with GPRS capabilities. The data transfer is relatively simple. The data-logger registers the measurements from the sensors, being both temperature and potential in mV. The measurements are digitized and stored in the data-logger memory. At one-hour intervals, the data logger connects to a GSM network and sends the data to a specific IP address which receives the data and processes the data (the data processing is presented in chapter 5).

For the study, mobile internet capabilities were provided by Telenor, which had very good GSM coverage at all three test sites. There was maintained a strong mobile signal, providing stable data transfer and connection with the data-logger, when it was in operation.

The study is a field-based study, and the data-logger was located next to the stream/well which was being monitored. The data-logger was not connected to an electrical net, with batteries supplying the needed power to run the system. The data-logger and batteries were sealed in tight aluminium casing allowing the electronic equipment to withstand the differing weather conditions, including rain and low temperatures – according to the manufacturer, the logger should be able to continue operation at temperatures below -10° C.

The operation of the data-logger at the test sites turned out to be less stable than advertised. The longest period of continuous data transfer was just under one month. Difficulties with the data-logger included failure of the battery and shorting out of the electronics in the system. Particularly sensitive was the system measuring potential from the sensors. The battery supporting the sensors proved to be extremely sensitive to the humidity, often shorting out in spite of efforts to remove humidity from the casing. The electronic system also shorted out twice, prompting the need for repairs of the data-logger. The reasons for this are unknown, but likely due to sensitivity to changing atmospheric conditions (cold and humid conditions). In the end, the data-logger proved to be unstable in the Danish test site, causing only sporadic data col-

lection and monitoring over shorter periods than what was originally planned. Further work in future projects are needed in order to sort out the problems, and develop systems, which does not require maintenance for a period of at least one month.

5. Data handling and data information technology

5.1 Data upload and handling

The data loggers send data via GPRS to a server. The server is identified with an IP-number and a port-number, which has to be programmed into the data logger. Data are send using the TCP-protocol and ComSystem A/S (www.comsystem.dk) delivers a client program that can listen for data and write them to a text-file when they are received. If this program is not running on the server, the data is lost. The program has to be started manually and cannot run as a service, which is not optimal. Whenever the server reboots due to automatic updates the programme needs to be restarted.

We developed a small service that parsed the text-files whenever new data arrived and forwarded the data to a cloud database (SQL Azure). On top of the database we had a service layer (HydroInform.CloudApp.Net) through which data could be accessed from web-clients. Thus, there is no direct access from outside to the database. Furthermore we have developed a small service that can get data via the service layer and transfer it to a SiteFx database (see section 6.1). The data flow is illustrated on figure 8 below.

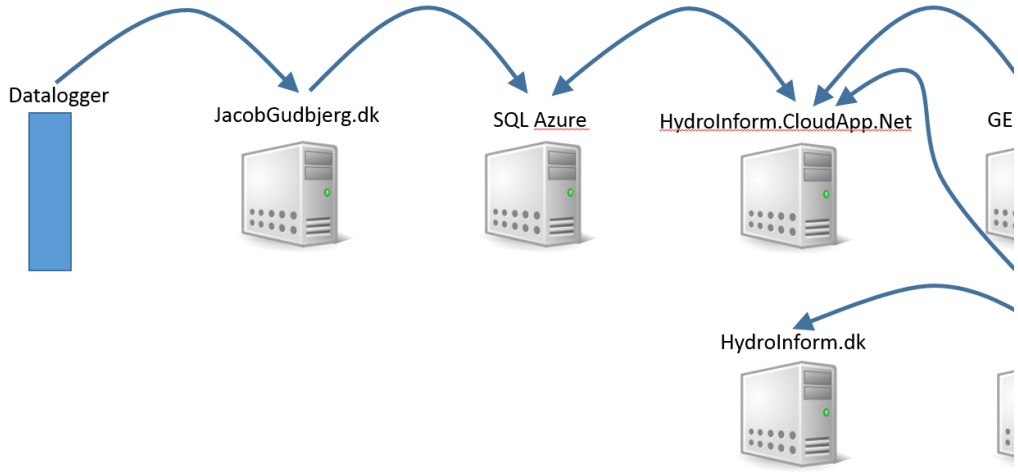


Figure 8. The data flow from logger to web clients.

The data flow Figure 5.1 may seem overly complicated; however, with this architecture the individual components are not interdependent. This means that we can for instance make changes to the database without changing the rest of the system. Adding new types of data from other dataloggers would also just require that a parser was made that could send data into the underlying database.

Note also that there is no significant time delay in this system. Within the same minute data are send from the logger it is available on the web page.

5.2 Data information technology

We have developed a website where all measured sensor data are available, and can be visualised. A screen dump is shown on Figure 9 below. In the graphs it is possible to zoom in on selected time intervals. When zooming in more data is displayed. Data are only send from the server when necessary. This advanced technology is necessary because the loggers may run for several years collecting data every minute. Downloading all data every time the web page is accessed would make the system increasingly slow. Thus, we have developed a system that can display huge amount of data very fast. From the graph it is possible to download the data directly. Examples can be seen on www.hydroinform.dk.

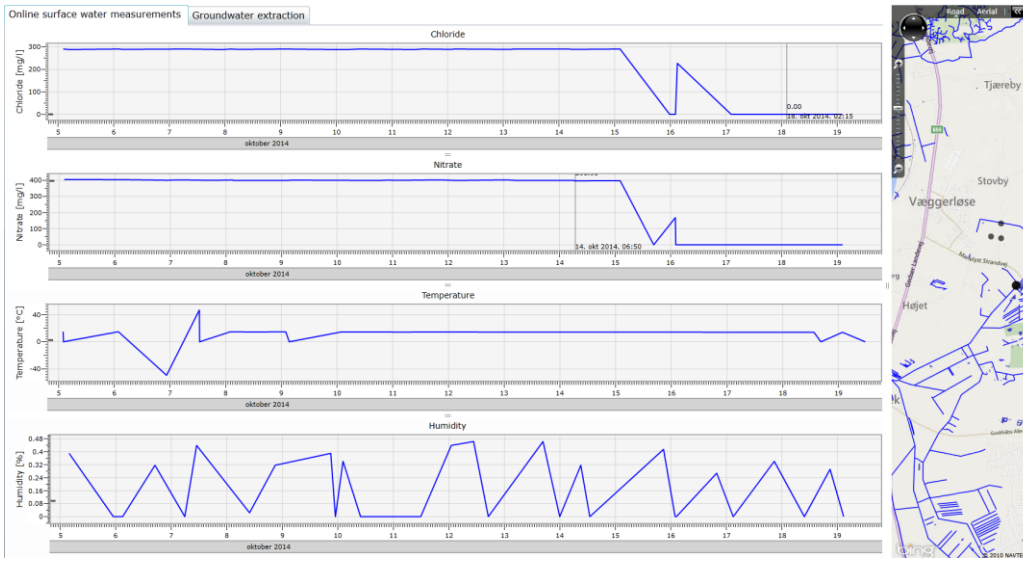


Figure 9. Screen dump from webpage displaying online data from sensors and selected data-logger.

6. Data integration and interaktive webserver

6.1 Data integration and databases

The final presentation of data relies on multiple data sources that are brought together for joint presentation. The source data are divided into regular relational databases and a few Web Map Services (WMS). The available databases are SiteFX (see section 5.1 for input instructions), Gerda and Jupiter.

SiteFX store all types of geological and environmental data, either static or time variant. A procedure for populating data from the sensors described in section 4 was developed as a part of this project (see section 5.1 for input instructions) and is fully operating when sensors upload data as described.

Jupiter is the Danish national well database storing information regarding well construction, geology, water levels, water chemistry and water quality from water works. The brand new tool, DBSync, for continuous update of the entire Jupiter database for external users (consulting companies, drillers, authorities etc.), has been applied in this project in order to keep the foundation continuously updated without manual interference.

Gerda is the Danish national geophysical database containing many types of geophysical data (airborne and surface em, wireline logs etc.) . The entire database serves as the source data foundation but only results of the down-hole logs are presented on-line for the end users in fixed pdf plots. Gerda is updated manually on a monthly basis.
Background maps using the WMS technology include, public maps from the Danish Cadastre (the Danish Geodata Agency), soil maps from GEUS (1:200´000) and Orthophotos (corrected aerial photographs) from COWI.

6.2 Data visualization on interactive webserver

Once the source data foundation is secured (section 6.1), read application tools (Figure 10) can be used to access the relevant data from different databases and organisations in various ways. The interactive web based tool EacoWeb developed by EarthFX Inc. (www.earthfx.com) is used to present relevant data defined by the user from within the project area. A homepage has been developed and can be accessed from:

https://eacoweb.cowi.com/water4coasts/
UN: water4coasts
PW: w4c-14

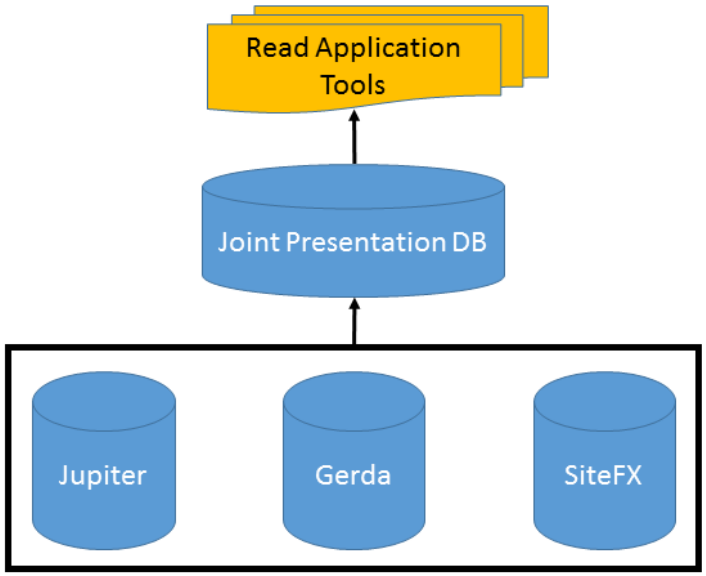


Figure 10. Illustration of source data integration from different databases.

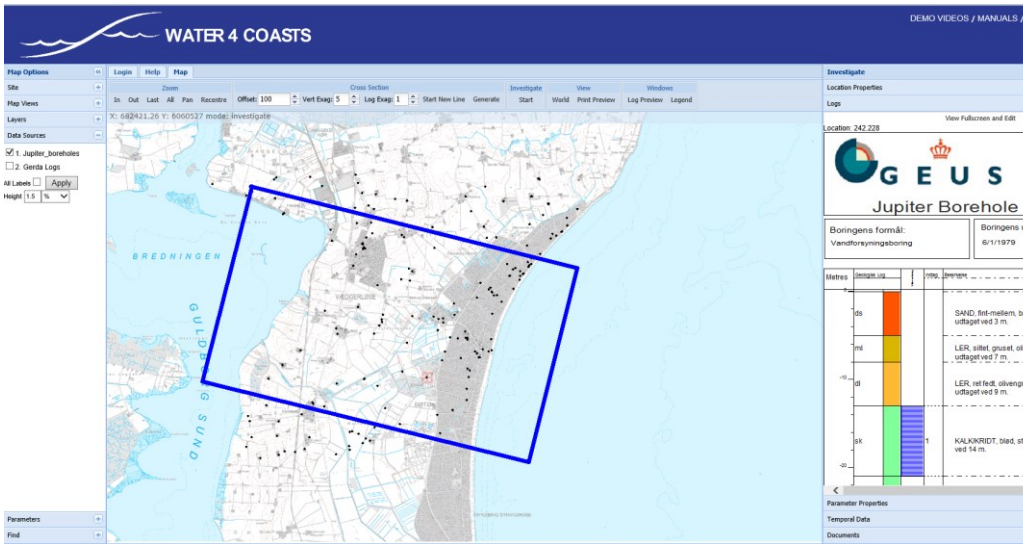


Figure 11. Screen dump from Water4Coasts interactive webserver displaying the project area as well as a geological log from a selected well in the Jupiter database.

It is a project specific homepage for the Water4Coasts project only and can contain multiple sites even though only the Falster (Denmark) case is available at this time.
Data outside the project area is available but merely filtered out. All data presented through EacoWeb are always up-to-date because it is extracting data from a data foundation that is continuously updated.

6.2.1 Functionality

At the left site under Map Options, it is possible to browse and enable different data.
Site and *Map view* is given, as there is only one.
Layers contain three different background maps, Orthophoto, soil map and a color scale presentation of wells having water level and chemical sampling.

Data sources contain Jupiter boreholes and Gerda Logs. It is possible to enable a label for active Data sources from the tick mark box and manually scale the size of the label.

From the top bar at the left, it is possible to zoom and pan in various ways

In the centre of the top bar, it is possible to draw cross-sections through wells for further inspection of the geological logs.

For further investigation of a desired borehole (Location) use the Investigate→Start button from the top menu that will activate the Location properties Investigation menu at the very right of the homepage.

Investigate Functionality:
Location Properties display some selection information regarding the selected borehole.
Logs Display a log of the borehole together with screen position. For Gerda logs, it will display the selected parameters Gamma, Conductivity, Resistivity and Caliper. By pressing *View Full-screen and Edit*, a new window opens where it is possible to adjust the scale of the parameters in order to get a better view of the data.
Temporal data is of specific importance to this project because it is possible to display temporal data as groundwater chemistry and water levels as graphs or tables. When data are plotted on graphs it is furthermore possible to include different parameters from the same or different boreholes on the same graph. More help is available on-line at the eacoweb website: https://eacoweb.cowi.com/water4coasts/.



Figure 12. Graphing capabilities in EacoWeb, where data sets from different parameters and even different boreholes can be investigated online on the same graph.

7. Discussion and perspectives

One primary obstacle encountered in the monitoring of chloride and nitrate with the ion selective electrodes was fouling of the electrodes when monitoring in the surface waters. Analysing the data, it is not possible to see how long the sensors remained responsive in the surface waters. However, it seems apparent that they were responsive for less than two weeks. This is not a satisfactory length of time if the objective is to monitor concentrations over longer periods. It is also apparent that fouling affected all three electrodes – the nitrate, chloride and the reference electrode. The fact that fouling affected all three electrodes in less than two weeks was disappointing. The prevention of fouling of the sensing surfaces will need to be improved upon in order to establish a more effective monitoring system.

On the other hand, the sensors when applied in the monitoring well seem to be more robust and not affected by fouling. By measuring down in the monitoring well, the system is isolated from biological activity, and thus the sensing surfaces on the sensors were not degraded at nearly the same rate as in the surface waters. Thus the potential for long-term monitoring down in wells is very promising. However, when measuring for changes in monitoring wells, the sensors should be located as close to the screen as possible to be sure that the water being monitored is representative of the aquifer. There should also be assured that water is flowing through the well, and not just stagnant. If possible, it would be ideal to apply the sensors in a well that is being pumped from – this will provide the most representative monitoring. However, if pumping from the wells, it will need to be certain that the wires and sensor array do not get caught up in the pumps themselves.

Because of the simple silver surface of the chloride electrode and the glass membrane of the reference electrode, it was possible to recondition both electrodes after fouling allowing them to be reused. This, of course is important with regards to overall life-span of the electrodes, making it more economical in the long run. However, the membrane on the nitrate electrode was more susceptible to degradation and needed to be replaced. If the membranes of the electrodes need to be replaced often, this can become costly, both in terms of time and money, and make the sensors less attractive to use.

When the data logger was able to send the signal to the server, the system worked very well. The website established for the system was able to display the monitored concentrations in real-time. A simple formula using Nernst Law was able to convert the signal (in mV) being sent from the data logger to actual concentrations, and displayed directly on the website. Thus by simply accessing the website, you can have a complete overview of the monitoring as it is happening. Using the mouse to scroll in and out allowed for a quick and easy adjustment of the time-scale one is looking at the data from, allowing to see data from the entire monitoring period to the last day in just seconds. In addition, the downloading of data from the website to a text file was very simple, being just a right click with the mouse. This allowed for the data, as displayed on the website, to be downloaded over the wanted time frame and quickly loaded into programs such as Excel, where it can then be processed even further.

Traditionally, when data loggers are used in the field, they have to be connected via cable to a computer in order to retrieve the data. This means having to physically go to the field to download the data at certain intervals. Therefore, even though data is being continuously collected, it cannot be seen until it is downloaded. In addition, if the system stops working, there is no way of knowing when it stops until the fields site visit occurs. Having the data transferred automati-

cally from the data logger to a server via GPRS has the distinct advantage of not only being able to access the data being collected in real-time, but seeing straight away if the system starts to have problems. This advantage was illustrated very nicely in this study – it was known within less than 24 hours every time the data logger lost power or stopped operating properly. This allowed for the site to be visited and the logger to be fixed. Unfortunately in this case, the data logger proved to be very unstable and much of the monitoring time was lost anyway in the numerous attempts to fix the logger. In a fully operating system the real-time uploaded data can be used in the automatic control of pump yields etc. at water works in a fully automated SCADA (Supervisory Control And Data Acquisition) system. The project group will in collaboration with new Danish SMEs continue to develop such tools in an up-coming EU project on subsurface water solutions developed to control salt water intrusion in coastal aquifers.

8. Recommendations and conclusions

8.1 Sensor Development

More work still needs to be done in order to use the sensors in surface waters. The test showed that both nitrate and chloride sensors had a limited lifespan in the surface waters that were tested in this study – likely no more than two-weeks. Further studies need to be conducted looking at preventing fouling of the sensing surfaces and reference electrodes under different conditions when monitoring in surface waters.

The study shows that there is a good potential use for the sensors directly in groundwater monitoring wells. Even after two months in the wells, the chloride sensors remained sensitive to changes in chloride concentrations. Further development in the applications will be to test how long the cable between the sensors and data logger can be, so that it would be possible to monitor even deeper in the well. The sensors should also be tested in wells where there is a known change in concentrations to determine whether or not the sensors can pick up the concentration changes. Further tests in e.g. active water supply wells with monitoring of the specific electrical conductivity and frequent chloride analyses are recommended.

When applying the sensors directly in the field, the data logger which measured the potential differences from the sensors, proved to be more sensitive to the changing weather conditions than other applications such as monitoring of temperature, pressure and humidity, which the manufacturers of the data logger conduct successfully. Therefore further testing must be done in order to get a more stable data logging and data transmission capabilities for the more sensitive potential measurements from the ion selective electrodes. It should be noted that a data logger from only one manufacturer was tested, and that there may be other more stable options available.

The project group recommend to improve the collaboration between public and private partners within water research and innovation for continued research on improved sensor, data logging and transfer technologies to be used for the development of efficient supervisory control and data acquisition (SCADA) systems, as well as online and interactive data visualisation, for optimized assessment and control of salt water intrusion and sustainable management of coastal freshwater resources.

8.2 Enhanced use of data integration strategies and EacoWeb data presentation

International projects where data are used in between many organizations will benefit greatly from a centralized approach to structuring of their source data in order to ensure a single version of the data and one entry point for the data. EacoWeb possess great possibilities for presenting the source data in different ways and much functionality have not been explored as a part of this project. Mobile Sample Manager (MSM), a field access tool for the SiteFX database is one tool that was considered to be used but not applied due to the basic challenges with the loggers. MSM include field upload of data as well as access to the source from the field. Powerful visualization capabilities for geological models as well as any other gridded data could also further enhance the user experience and interpretation capabilities when using EacoWeb if such data were added to the project. Finally, EacoWeb is also used to store large document

archives of georeferenced data such as reports, photos, drawings ect. that all can be geocoded and displayed on the map as well as downloaded for subsequent use at the will of the user. We recommend that the work on such tools is continued and the tools continuously improved for efficient and sustainable management of the quantity and quality of the global water resources (Scharling et al., 2014).

9. Acknowledgement

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