DANMARKS OG GRØNLANDS GEOLOGISKE UNDERSØGELSE RAPPORT 2013/30

Sustainable groundwater abstraction

Review report

Hans Jørgen Henriksen & Jens Christian Refsgaard

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

G E U S

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Esbjerg Forsyning
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GEOLOGICAL SURVEY OF DENMARK AND GREENLAND
DANISH MINISTRY OF CLIMATE, ENERGY AND BUILDING

G E U S

Content

Preface

GEUS and six Danish water companies have established collaboration with the long term goals of enhancing the scientific foundation for assessment of sustainable groundwater abstraction and contributing to an improved knowledge based water management practice in this field. The present report is the first output of this collaboration.

The report has been prepared between October 2012 and March 2013 by Hans Jørgen Henriksen and Jens Christian Refsgaard. The work has been financed by the six water companies with a Steering Committee comprising

- Troels Kjærgaard Bjerre, Vandcenter Syd (formand)
- Anne Scherfig Kruse, HO-FOR
- Jørn-Ole Andreasen, Aarhus Vand
- Charlotte Schmidt, TRE-FOR Vand A/S
- Peter Madsen, Esbjerg Forsyning
- Per Grønvald, Aalborg Forsyning

The report was subject to peer review by two international experts

- Dr. Mike Dunbar, Environment Agency, UK. Mike Dunbar is a recognized expert in environmental flow.
- Professor Roland Barthel, University of Gothenburg, Sweden. Roland Barthel is a recognized expert in aquifer safe yield.

The project progress has been discussed at a kick-off meeting and two workshops with participation of the Steering Committee plus the following additional persons (some participated in only one of the workshops)

- Jens Rasmussen, HO-FOR
- Christian Ammitsøe, Vandcenter Syd
- Michael Rosenberg Pedersen, Aarhus Vand
- Ole Silkjær, TRE-FOR
- Eva Hansson, Roskilde Forsyning
- Mads Kjærstrup, Ringkøbing-Skjern Forsyning
- Bo Lindhardt, Nordvand
- Claus Vangsgaard, DANVA
- Henrik Nielsen, Naturstyrelsen
- Martin Skriver, Naturstyrelsen
- Dirk-Ingmar Müller-Wohlfeil, Naturstyrelsen
- Niels Philip Jensen, Kommunernes Landsforening
- Martin Olsen, GEUS
- Anker Lajer Højberg, GEUS

The first workshop on 12 December 2012 focussed on basic concepts and preliminary results. The international peer review was presented by the two reviewers and discussed at the second workshop on 7-8 February 2013, which made conclusions on necessary modifications and improvements of the report that have been implemented in the present final version.

Sammenfatning

Baggrund for rapporten

Vandselskaberne i Danmark har generelt den opfattelse at Vandrammedirektivet og Grundvandsdirektivet udgør en god og fremadrettet lovgivning, som kan bidrage til en helhedsorienteret og bæredygtig vandforvaltning. Vandforvaltningen i Danmark anvender i dag meget simple kriterier til karakterisering af bæredygtig grundvandsindvinding. De er nemme at administrere, men bygger på relativ gammel viden. Samtidig oplevede vandselskaberne at metoder og kriterier i den første vandplanrunde ikke blev anvendt ensartet over hele landet. Vandselskaberne er bekymrede for, at det nuværende faglige grundlag og metodikker for vurdering af bæredygtig vandindvinding ikke er tidssvarende, dvs. at de ikke lever op til den videnskabelige state-of-the-art på området og til den praksis, der benyttes i andre EU lande. Samtidig er vandselskaberne bekymrede for, at en for kategorisk anvendelse af de simple kriterier vil kunne forhindre anvendelse af bedre data og metodikker i ressourcevurderingen.

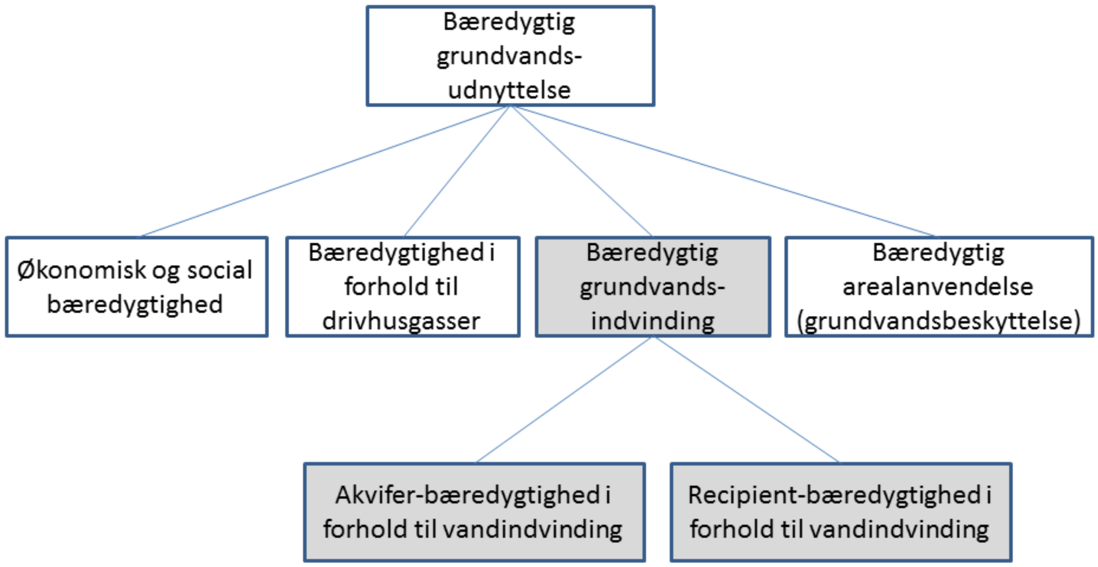
Det blev anledningen til at seks vandselskaber i et partnerskab med GEUS i september 2012 igangsatte et projektforløb (Fase 1), der indeholdt et review af de danske vandplaner, en gennemgang af praksis for implementeringen af vandplaner i andre lande og en indsamling af international viden. På en workshop 12. december 2012 blev grundlæggende koncepter og foreløbige resultater drøftet. En foreløbig udgave af rapporten er blevet reviewet af to anerkendte internationale eksperter Mike Dunbar, Environment Agency, UK og Roland Barthel, Göteborg Universitet, Sverige. Reviewene blev præsenteret og diskuteret på en workshop 7-8 februar 2013.

Nærværende rapport indeholder resultaterne af Fase 1. På de to workshops har der, udover GEUS og styregruppen med repræsenter for de seks vandselskaber, som har finansieret Fase 1, været deltagere fra andre vandselskaber, Naturstyrelsen, Kommunernes Landsforening og DANVA.

Hvad er bæredygtig vandindvinding – definitioner og analysemetoder

Bæredygtig grundvandsudnyttelse er et vidtfavnende begreb. Det inkluderer aspekter såsom (i) økonomisk og social bæredygtighed, (ii) bæredygtighed i forhold til udslip af drivhusgasser, (iii) bæredygtig grundvandsindvinding og (iv) bæredygtig arealanvendelse (grundvandsforurening). Helt overordnet er der i Danmark en samfundsmæssig prioritering, hvor drikkevand prioriteres højest, efterfulgt af vand til natur og vand til erhverv.

Nærværende rapport behandler kun bæredygtig grundvandsindvinding, der inkluderer to nøgle elementer: (a) at undgå uønskede følgevirkninger på grundvandsakviferen af vandindvindingen (akvifer safe yield = *akvifer-bæredygtighed*), og (b) beskyttelse af økosystemers levedygtighed i relation til vandindvinding (environmental flow = *recipient-bæredygtighed*). Det skal præciseres, at Vandrammedirektivets målsætning er god økologisk tilstand, og at god økologisk tilstand omhandler såvel vandkvalitet, fysiske forhold og vandføring. Grundvandsindvinding har derfor størst direkte betydning for påvirkningen af vandføringen (især minimumsvandføringen), men fysiske forhold, som har betydning for fx vanddybde, hastighed og temperatur, samt vandkvalitet kan have lige så stor betydning som vandføring, når det gælder den økologiske tilstand, se Figur I.



Figur I Afgrænsning af bæredygtig grundvandsindvinding (grå bokse) i forhold til bredere aspekter af begrebet bæredygtig grundvandsudnyttelse.

Akvifer-bæredygtighed defineres som: *Den mængde grundvand der kan indvindes uden uacceptable følgevirkninger på grundvandets trykniveau og vandkvalitet, sammenlignet med det upåvirkede magasin.*
Tilsvarende defineres recipient-bæredygtighed som: *Afstrømningskarakteristika (mængde, hyppighed, timing, varighed, fluktuationer og forudsigelighed/variabilitet af hændelser) der er nødvendige for at vedligeholde (eller re-etablere) det naturlige afstrømningsregime, som understøtter specificerede, værdifulde egenskaber ved et økosystem.*

Metoder til vurdering af akvifer-bæredygtighed og recipient-bæredygtighed kan klassificeres i to hovedkategorier:

- *Screeningsmetoder* er relativt simple metoder, som kan anvendes med få, let tilgængelige data. Fordi disse metoder kun i stærkt begrænset omfang er i stand til at udnytte lokale data og viden, er de relativt usikre. De opstilles derfor ofte med indikatorer (tærskelværdier) som afspejler et forsigtighedsprincip og er som sådan velegnede til nationale screeningsformål.
- *Undersøgelsesmetoder* er mere komplekse og kræver flere ressourcer. Disse metoder er stedspecifikke og gør maksimalt brug af lokale data og viden under anvendelse mere raffinerede procesbaserede modelværktøjer. Metoder i denne kategori er derfor i stand til at levere mere pålidelige resultater end screeningsmetoder. Eftersom de administrativt er mere krævende, bør de primært anvendes, hvor simple metoder har påpeget et potentielt bæredygtighedsproblem. Undersøgelsesmetoder til vurdering af akvifer-bæredygtighed kan eksempelvis baseres på stoftransportmodeller, mens habitatmodeller udgør et eksempel på værktøjer, der er egnede undersøgelsesmetoder i forhold til vurdering af recipient-bæredygtighed.

Væsentlige karakteristika for de to metoder anvendt på henholdsvis akvifer-bæredygtighed og recipient-bæredygtighed er præsenteret i Tabel I og II.

Tabel I Karakteristika af screenings- og undersøgelsesmetoder til analyse af akvifer-bæredygtighed i relation til vandindvinding

Formål	Kompleksitetsniveau	Administrativt grundlag	Databehov ved praktisk brug	Vidensgrundlag for udvikling og test
Screening	Simpel	Grundvandsdannelse og ændring som følge af vandindvinding *(Hydrogeologisk)*	Relationer mellem vandindvinding og grundvandsdannelse	Eksisterende databaser. Integrerede grundvands- og overfladevandsmodeller. Kalibrerede relationer mellem ændret grundvandsdannelse og akviferforhold (vandkvalitet og kvantitet)
Detail undersøgelse	Kompleks	Ændret grundvandsspejl i forhold til geologisk lag, havniveau og sårbarhed i forhold til ændret trykniveau Integreret grundvands og overfladevands stoftransport og saltvands ind- og optrængning *(Grundvandstrykniveau og holistiske metoder)*	Historiske afstrømningsdata. Grundvands oppumpning, trykniveau og grundvandskvalitet. Borehulslogging og 3D geologiske og hydrogeologiske modeller. Aldersdatering.	Eksisterende databaser og nye detailundersøgelser af geofysiske/hydrogeologiske data, inkl. kortlægning af grundvandsstand og vandkvalitet. Integrerede grundvands- og overfladevandsmodeller inkl. partikelbanesimulering. Avancerede, detaljerede 3D geologiske og hydrologiske flow, stoftransport og saltvandsmodeller.

Tabel II Karakteristika af screenings- og undersøgelsesmetoder til analyse af recipient-bæredygtighed i relation til vandindvinding

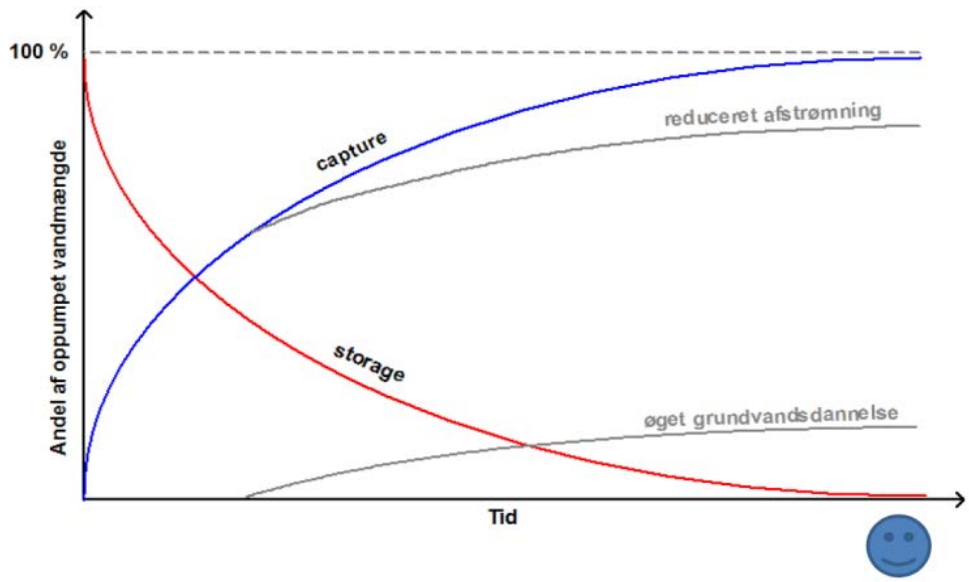
Formål	Kompleksitetsniveau	Administrativt grundlag	Databehov ved praktisk brug	Vidensgrundlag for udvikling og test
Screening	Simpel	Afstrømningsdata og ændringer som følge af vandindvinding. *(Økohydrologisk)*	Historiske afstrømningsmålinger. Vandindvinding og påvirkning af minimumsafstrømningen	Eksisterende hydrologiske og økologiske databaser. Integrerede grundvands- og overfladevandsmodeller. Kalibrerede relationer mellem afstrømning, ændringer i regimet og økologiske indikatorer
Detail undersøgelser	Kompleks	Målspecifikke biota og vandløbshabitater. Hydromorfologi, temperatur og vandkvalitet. Hele økosystemet med alle/de flest individuelle komponenter inkl. grundvand/ådal samt terrestriske interaktioner. *(Hydraulik/habitat, holistisk)*	Historiske afstrømningsmålinger. Vandindvinding og påvirkning af minimumsafstrømningen. Hydrauliske variable for repræsentative vandløbstværsnit. Suitability kurver for målspecifikke arter. Biologiske data om afstrømning og habitat relaterede krav til biota og økologiske komponenter.	Eksisterende databaser og nye økologiske data fra forskellige økosystemer. Integrerede grundvands-overfladevandsmodeller, hydraulisk/habitat modeller og/eller økologiske modeller. Specialviden om hydrologiske, hydrauliske, habitat og økosystem komponenter. Mulighed for at vurdere (kvantificere) påvirkninger på biota der ikke er relateret til afstrømning.

Hvad siger litteraturen

Følgende retningslinjer gælder ifølge litteraturen for brug af *simple screeningsmetoder*:

- Der bør anvendes multiple indikatorer. Herved opnås mere robuste vurderinger af kvantitativ status for akvifer-bæredygtighed og recipient-bæredygtighed
- Akvifer-bæredygtighed og recipient-bæredygtighed må anskues som uadskillelige. Derfor skal der vælges indikatorer, der omfatter begge aspekter.
- Dynamiske, koblede grundvands- overfladevandsmodeller bør anvendes til at understøtte beregninger af indikatorerne.
- Mange indikatorer er skalaafhængige, hvilket medfører at tærskelværdier til skelnen mellem god og usikker tilstand afhænger af den skala, hvorpå de anvendes, det vil sige hhv. størrelse for grundvandsforekomst og/eller oplandsstørrelsen.
- Klimaændringer påvirker grundvandsdannelsen og økosystemer på signifikant vis, hvilket der bør tages højde for.
- Usikkerheder på vurdering af kvantitativ status bør kvantificeres, kommunikeres og kan anvendes ved prioritering af ressourcer til opfølgning. England anvender i dag nogle metoder til karakterisering af konfidens af screenings resultater, som med mindre modifikationer vil kunne overføres til danske forhold.
- Akvifer-bæredygtighed karakteriseres ofte ved hjælp af en indikator hvor grundvandsdannelsen vurderes for en situation uden oppumpning. Det har den medfølgende svaghed, at man derved ikke tager højde for det faktum, at det oppumpede vand delvis stammer fra forøget grundvandsdannelse (kompenseret ved et fald i overfladenær afstrømning til vandløb) og reduceret grundvandsafstrømning til vandløb og dermed er effekter af 'capture'. En indikator baseret på forholdet mellem oppumpning og grundvandsdannelse ved aktuel grundvandsoppumpning vil derfor udgøre et mere fornuftigt grundlag.
- Recipient-bæredygtighed rækker længere end blot minimumsafstrømning og sikring af et statisk afstrømningsregime. Det omhandler det samlede afstrømningsregime, inklusiv minimumsafstrømninger, sæsonmæssige variationer, oversvømmelser og omfanget af fluktuationer i vandføring og vandstand, som er væsentlige for økosystemerne.

I litteraturen har der været en lang debat omkring begreberne 'Water Budget Myth' og 'Capture' (Fig. II). Det første begreb referer til den myte, der har eksisteret i flere årtier blandt fagfolk, at hvis man blot kunne fastlægge størrelsen på grundvandsdannelsen, så kunne man også bestemme den bæredygtige vandindvinding. At det har fået nærmest mytologisk karakter skyldes, at den enøjede fokus på i dette tilfælde akvifer-bæredygtighed, samtidig betød, at man ignorerede recipient-bæredygtighed. Det førte til den vrangforestilling at indvindingen var bæredygtig blot den var mindre end grundvandsdannelsen. Her havde man imidlertid overset hensynet til recipient-bæredygtighed, og at der skulle være noget vand tilbage til økosystemerne (recipient-bæredygtighed), hvis udnyttelsen skulle være bæredygtig. Samtidig overså man grundvandssystemets komplekse sammenhænge, hvor en grundvandsforekomst under naturlige forhold er i en dynamisk ligevægt med balance mellem grundvandsdannelse og grundvandsafstrømning.



Figur II Grundvandsindvinding og betydningen af "capture", dvs. det komplekse sammenspil mellem ændret magasinering, øget grundvandsdannelse og reduceret afstrømning som følge af en introduceret vandindvinding til tiden (t=0) (Figur fra Bjerre, 2012).

Vandindvinding, der igangsættes på et bestemt tidspunkt (t=0 i Fig. II), vil i den første tid blive kompenseret ved ændret magasinering (der udbreder sig en sænkningstragt omkring den nye indvindingsboring). På et vist tidspunkt når sænkningstragten hydrologiske barrierer, fx kildevæld, dræn eller vandløb, sådan at der gradvist sker en påvirkning af den overordnede vandbalance for grundvandsforekomsten, fx i form af øget grundvandsdannelse, reduceret grundvandsafstrømning til vandløb mv. samt evt. ændret underjordisk grundvandsstrømning til nabo grundvandsforekomster. Hvordan en ny ligevægt vil indstille sig afhænger af en række hydrogeologiske og hydrologiske forhold, og det kan tage meget lang tid før en ny ligevægt indstiller sig (år til årtier). Bæredygtig vandindvinding bør derfor vurderes på grundlag af en faglig kvalificeret vurdering af disse komplekse dynamiske forhold. I den sammenhæng er en transient, numerisk grundvands/overfladevandsmodel nødvendige for at kunne kvantificere ændringer i afsænkning og vandbalanceforhold og dermed såvel akvifer-bæredygtighed som recipient-bæredygtighed. Derved kan såvel grundvandsdannelse, tilgængelig grundvandsressource, reduceret afstrømning og ændringer i grundvandsstrømning i forhold til nabo-grundvandsforekomster og/eller havet tages i regning i tid og sted.

Der findes en række eksempler fra litteraturen på metoder til komplekse, detailundersøgelser. Disse studier anvender mangfoldige mere eller mindre raffinerede modelværktøjer, der afhænger af tilgængelighed af lokale data.

For de grundvandsafhængige terrestriske økosystemer (GWDTE) blev der ikke fundet nogen screeningsmetoder i litteraturen. Det vil sige at vidensgrundlaget for sådanne metoder generelt er dårlig. GWDTEs dækker ofte arealer, der er meget mindre og derfor også mere afhængige af lokale forhold, end grundvandsforekomster og vandløbssystemer. Det giver stor usikkerhed på karakteriseringen for GWDTEs ved hjælp af nationale screeningsværktøjer, og betyder at egnede metoder ofte tilhører kategorien komplekse metoder, dvs. baseret på procesbaserede modeller og krav om et betydeligt datagrundlag.

De fire danske indikatorer

Med udgangspunkt i første version af DK-model blev der i 2003 opstillet fire danske indikatorer med henblik på karakterisering af bæredygtig vandindvinding i Danmark. De fire indikatorer var:

- *Indikator 1*: maksimal udnyttelig grundvandsressource udgør 35 % af grundvandsdannelsen (for situation uden oppumpning)
- *Indikator 2*: maksimal udnyttelig grundvandsressource udgør 30 % af grundvandsdannelsen (for aktuel vandindvinding og grundvandsdannelse)
- *Indikator 3:* maksimal reduktion af middelvandføring = 10 %
- *Indikator 4:* maksimal reduktion af minimumsvandføring = (5%, 10%, 15 %, 25% og 50%) afhængig af økologisk (recipient) målsætning for vandløbsstrækningen

De første to indikatorer er relateret til akvifer-bæredygtighed, og indikator 1 tager ikke hensyn til "capture", hvorimod dette er indregnet med indikator 2. De fleste internationale studier benytter indikator 2, med tærskelværdier på mellem 30 % og 100 %. De danske tærskelværdier blev vurderet på baggrund af sammenligning af vandindvinding og akvifer forhold på Sjælland (hovedsagelig forskellige naturligt forekommende vandkvalitetsparametre), uden at de er blevet nærmere testet for andre danske grundvandsmagasin forhold. Indikator 3 og 4 er relateret til recipient-bæredygtighed.

Overordnet set er tærskelværdi niveauer (% satserne i de fire indikatorer) på størrelse med det man kan finde i den internationale litteratur, men litteraturen har en del eksempler på en mere differentieret brug af tærskelværdier fx for forskellige vandløbstyper og forskellige sæsoner. De danske tærskelværdier blev opstillet for oplande i størrelsesordenen fra 300 til 2000 km^2, svarende til de 50 delområder ferskvandets kredsløb opererede med. Test har siden indikeret, at tærskelværdier afhænger af den skala de opstilles, sådan at de kun er gyldige for den skala som de er opstillet for.

Dansk praksis jf. vandplaner

I forbindelse med Vandrammedirektivet og første generation danske vandplaner er overvejende anvendt metoder tilhørende kategorien simple screeningsmetoder. De kan karakteriseres som følger:

- De anvendte kriterier er inspireret af Ferskvandets Kredsløb (Henriksen et al., 2008). Typisk 1-2 af de fire indikatorer er anvendt (Indikator 1 og Indikator 4).
- De fire tests anbefalet af Europa-Kommissionen i CIS (2009) vedr. grundvandsforekomsters kvantitative tilstand er ikke alle anvendt og nogle gange ikke korrekt implementeret (fx hvor grundvandsdannelsen ikke er blevet beregnet for "et helt grundvandsmagasin").
- Der blev anvendt forskellige metoder og med ikke gennemskuelige/forskellige tærskelværdier for Fyn, Sjælland og Jylland (for Jylland endog forskellige metoder for Nord, Vest, Øst og Sydjylland). For eksempel blev det undladt at gennemføre vandbalance testen/uddrage evt. konsekvenser af denne test for Sjælland og for det tidligere Århus amt. Derved er opgø-

relsen til de danske vandplaner ikke sket på baggrund af en national, ensartet landsdækkende screening.

- Opgørelsen var uigennemsigtig som følge af de forskelligartede, og ikke dokumenterede metoder til håndtering af fx grundvandsdannelse, oppumpning, reference situationen i vandløb uden oppumpning og vandløbspåvirkning. Samtidig var fastsættelsen af tærskelværdier, og de lokale variationer heri (bl.a. Formel 7 på Sjælland og højere tilladelige reduktionstal for minimumsvandføring i Østjylland og det tidligere Viborg amt) ikke var grundigt dokumenteret.
- Endelig blev vandbalance testen opstillet for relativt små oplande, der var væsentlig mindre end den skala for hvilke tærskelværdierne oprindeligt var opstillede. Indikator 1 blev anvendt med tærskelværdi på 0,35, selvom den var opstillet for oplande fra 300-2000 km^2, og selvom grundvandsforekomster i gennemsnit er af størrelsesordenen 100 km^2, hvoraf mange i Østjylland, Fyn og for Sjælland er betydeligt mindre, mens andre i bl.a. Vestjylland typisk er noget større end 100 km^2. Indikator 4 blev ligeledes opstillet for meget små oplande af hensyn til en detaljeret opgørelse for bl.a. de øverste vandløbspunkter uden eksplicit at tage hensyn til usikkerheden på den tilgrundliggende DK model, som vides at være er mest pålidelig for simulering af median minimumsvandføringer, Q_{medmin}, for oplande > 30 km^2.

Praksis i andre europæiske lande

En analyse af praksis i andre europæiske lande (England/Wales, Tyskland, Irland og Frankrig) viser, at man i England/Wales, Tyskland og Irland har gennemført vurdering af akvifer-bæredygtighed i overensstemmelse med vandbalance og overflade vands testen, mens man i Frankrig tilsyneladende ser bort fra kravet om vurdering af tilgængelig grundvandsressource, bestemt som differensen mellem grundvandsdannelse og krav til environmental flow til nedstrøms vandløb og terrestriske grundvandsafhængige økosystemer. I alle disse lande tager man i vurderingen udgangspunkt i aktuel grundvandsdannelse ved aktuel vandindvinding, og i visse lande slår man grundvandsforekomster sammen, hvis man har konstateret grundvandsstrømning på tværs af grundvandsforekomster. Dette er ikke tilfældet for Danmark, hvor man dels tager udgangspunkt i grundvandsdannelse uden oppumpning, og dels har undladt at tage horisontal grundvandsstrømning i regning, selvom det anbefales i CIS (2009).

De mest udviklede og modne metoder i forhold til overfladevands testen og recipient-bæredygtighed finder man i England/Wales. Der er en klar skelnen mellem brug af screening i forbindelse med Vandplaner (Vandrammedirektivet) og brugen af metoder i forbindelse med udstedelse af vandindvindingstilladelser (England/Wales). Når det gælder den screening, der anvendes i forbindelse med Vandrammedirektivet / Vandplaner i England/Wales, så benyttes en indikator svarende til den lille minimumsafstrømning der overskrides 95 % af tiden (eller underskrides 5 % af tiden om man vil, Q_{95}) med forskellige tærskelværdier for forskellige relevante årstider og hydrogeologiske/hydrologiske/biologiske typologier. I forbindelse med vandindvindingstilladelser inddrages et udvidet sæt kriterier, hvor der også indgår fx indikatorer for store afstrømninger (der fx overskrides 30 % af tiden/Q_{30}, se bl.a. Appendiks 2). I modsætning hertil er kriterier, der anvendes i Frankrig, Irland og Tyskland til overfladevands testen, mere kvalitative, det vil sige man har endnu ikke fået etableret en oversættelse til kvantitative kriterier, men det forventes i flere af landene at ske i forbindelse med anden generation vandplaner. I Danmark benytter man fortsat Q_{medmin} som statistisk størrelse med tærskelværdier baseret på en vejledning fra Miljøstyrelsen fra 1979, mens man i fx England løbende forsker og udvikler indikatorer og tærskelværdier, med henblik på opstilling af indikatorer for en række forskellige typologier (geologi, opstrøms/nedstrøms, biologiske målsætninger mv.).

Omkring reference situationen er der væsentlige forskelle mellem fx Danmark og Tyskland. I Tyskland tager man udgangspunkt i den nuværende situation, hvorimod man i Danmark, i lighed med fx England/Wales tager udgangspunkt i den upåvirkede situation, altså en situation uden vandindvinding.

Der er store forskelle i størrelsen på grundvandsforekomster og i anvendte tærskelværdier fra land til land, fx har Irland og Danmark relativt små grundvandsforekomster (ca. 100 km^2), Tyskland og England/Wales noget større grundvandsforekomster (ca. 300 km^2), og endelig Frankrig relativt store grundvandsforekomster (ca. 1000 km^2). Samtidig er grundlaget for fx grundvandsdannelsen til grundvandsforekomster beregnet med meget forskellige metoder. Irland og Tyskland anvender input fra såkaldte grundvandsatlas, dvs. nationale kort over grundvandsdannelsen, og det fremgår ikke klart i hvilken omfang disse data fuldt ud er repræsentative for grundvandsdannelsen til de konkrete grundvandsforekomster (disse atlas kort over grundvandsdannelsen kan formentlig bedst sammenlignes med Vandrådets landsdækkende kort over nettonedbøren, bestemt ud fra Q_{medmin} og vandindvinding). Hvor der foreligger data fra hydrologiske modeller indgår de også i vurderingen. Her har Danmark et væsentlig bedre grundlag end de andre undersøgte lande, idet der foreligger en landsdækkende hydrologisk model og for en del områder mere detaljerede hydrologiske modeller fra bl.a. grundvandskortlægningen (se bl.a. Appendiks 1). Der er derfor ikke noget til hinder for at kunne gennemføre fx vandbalance test og overfladevands test samt screening i forholdt til de øvrige kvantitative tests jf. CIS (2009) eller at anvende indikatorer, der inddrager capture. Det er dog ikke ensbetydende med, at der ikke er behov for en videreudvikling af såvel model- som datagrundlag, men der er et klart grundlag for en mere ensartet og konsistent opgørelse indenfor den tidsramme der ligger frem til basisanalysen i forbindelse med næste generation vandplan (se bl.a. Appendiks 1).

Analysen for England/Wales, Frankrig og Irland peger alle på, at man der arbejder med forskellige konfidensniveauer (fx høj og lav konfidens). Det har den fordel, at man bedre kan prioritere indsatsen i retning at opnåelse af god tilstand, og afhængig af konfidens-niveau vælge en forbedret monitering eller egentlige detailundersøgelser med henblik på det videre arbejde i de områder hvor screening ikke har resulteret i god tilstand.

Det gælder for langt de fleste lande at arbejdet med grundvandsafhængige terrestriske økosystemer er en stor udfordring, fordi vidensgrundlaget er meget spinkelt, og fordi der ikke findes særligt egnede screeningsmetoder. Desuden kræver egentlige vurderinger meget detaljerede undersøgelser (næsten på "frimærke skala").

Klimaændringer vil påvirke både vandbalancer, recipienter, grundvandstand, afstrømningsregime, temperatur og havniveau. Det vil påvirke vurderingen af tilgængelig grundvandsressource udtrykt som forskel mellem grundvandsdannelse og environmental flow hensyn. Det håndteres heller ikke i de øvrige lande på nuværende tidspunkt, men EC anbefaler at det inddrages i næste runde.

Med hensyn til læringsmuligheder så er der umiddelbart mest at hente i England, som er klart længere fremme end Danmark mht. standardisering og vidensbaserede løsninger. I en række lande arbejdes der med konfidensniveau som er interessant. I Tyskland har man et relativt omfattende moniteringsnet mht. grundvandsniveau pejlinger, der indgår som et integreret element (som en femte test) i forbindelse med vurdering af kvantitativ status. Samtidig har man i en del lande erfaringer omkring screening i forhold til GWDTEs (fx Tyskland og England) som kan give også dansk inspiration, sidstnævnte gælder formentlig også Holland.

Konklusioner og anbefalinger

Indikatorer for akvifer- og recipient-bæredygtighed er nyttige ved karakterisering af grundvandsforekomsters tilstand. Indikatorerne bør baseres på forsigtighedsprincip, videnskabeligt grundlag og "kalibreres" mod nyeste viden og data. Indikatorer er ikke mål i sig selv – målet kan nogle gange opfyldes selv om indikatoren peger på tvivl herom.

Principper i vandforvaltning

- Multiple indikatorer for akvifer-bæredygtighed og recipient-bæredygtighed bør anvendes til karakterisering af den kvantitative tilstand af grundvandforekomster. Tærskeværdierne for god/usikker tilstand for de enkelte indikatorer bør kalibreres mod data for forskellige hydrogeologiske og økohydrologiske regimer.
- Screeningsværktøjer bør udvikles med indikatorer der afspejler et forsigtighedsprincip. Hvis en vandforekomst opnår god status ved screeningen vil der ikke være behov for yderligere undersøgelser. Vandforekomster, der ikke opnår god tilstand (dvs. usikker tilstand) ved screening vil derimod potentielt have en dårlig tilstand, hvilket skal afklares ved detailundersøgelser med inddragelse af lokale data og yderligere viden. En karakterisering af resultaterne fra screening med usikkerhedsvurderinger (konfidens) kan benyttes aktivt i prioriteringen af ressourcer, fx til detailundersøgelser.
- Screeningsværktøjer bør regelmæssigt opdateres på baggrund af nye data og ny forskningsbaseret viden.

Vidensbehov hen mod 3. runde vandplaner

- Hen mod 3. runde vandplaner er der følgende vidensbehov:
    - Der bør etableres en "bæredygtighedsdatabase" i forhold til akvifer/recipient med data for forskellige grundvandsforekomsters tilstand og deres påvirkning af vandindvinding, så der skabes et grundlag for kalibrering af tærskelværdier for forskellige indikatorer for bæredygtighed.
    - Metodiker til vurdering af bæredygtighed:
        - Der bør udvikles nye metoder til detailundersøgelser af akvifer bæredygtighed (grundvandstand, grundvandskvalitet, stoftransport, mv.)
        - Der bør udvikles nye metoder til detailundersøgelser af recipient bæredygtighed (habitatmodeller, øget inddragelse af økologisk viden, mv.)
        - Der bør udvikles nye metoder til detailundersøgelser af terrestriske økosystemer
    - Indvindingsstrategier, drift og monitering til sikring af bæredygtig vandindvinding
    - Udarbejdelse af vejledninger for anvendelse af detailundersøgelser

Værktøjer til 2. runde vandplaner i Danmark

- Benyt nationale screeningsmetoder med følgende karakteristika:
  - Standardiserede metoder over hele landet.
  - Multiple kriterier inklusiv minimum ét for akvifer-bæredygtighed og ét for recipient-bæredygtighed.
  - Lav en hurtig kalibrering af tærskelværdier mod data fra forskellige hydrogeologi/økologiske situationer.
  - Fastsæt tærskelværdier ud fra forsigtighedsprincip.
  - Karakteriser udfaldet af screening med grad af konfidens/usikkerhed.
  - Udarbejd en vejledning for brug af værktøjer og data til den national screening, så der sikres ensartet og gennemskuelig implementering.
- Udfør detailundersøgelser for vandforekomster, der ikke opnår god tilstand i screening med mulighed for omklassificering (på et gennemskueligt grundlag).

Mulige indikatorer og test til brug i 2. runde vandplaner i Danmark

- Mulige kriterier til national screening (indikatorer skal kalibreres/valideres):
  - Akvifer bæredygtighed: Oppumpning < 30% af grundvandsdannelse
    - Beregnes med oppumpning (”indikator 2”)
    - Beregnes for hele grundvandsforekomster som skal have passende størrelse (indregn horisontal strømning)
    - Evt. differentiering for forskellige hydrogeologiske situationer
  - Recipient bæredygtighed: Reduktion af Q_{medmin} < 10% (eller Q_{95} < 10%)
    - Beregnes for oplande > 30 km^2
    - For oplande < 30 km^2 aggregeres resultater for flere ”headwater” vandløb
    - Evt. differentiering mellem forskellige vandløbstyper, øvre/nedre strækninger mv.
- Andre mulige test (tests skal kalibreres/valideres):
  - Saltvandsindtrængning: Ændring i strømningsretning ved fx kysten
  - Kemisk status: Kan supplere vandbalancetest mht. akvifer bærdygtighed
  - Terrestriske økosystemer: Ændring i grundvandsstand

Rapportens resultater peger på, at det ikke kun er indikatorerne i sig selv, men også måden og det grundlag, de bliver brugt på i vandforvaltningen, der er problemet. Derfor er der behov for kapacitetsopbygning i stat, kommuner og aktørnetværk, sådan at brugen af screenings og detailundersøgelser i vandforvaltningen præciseres jf. state-of-the-art og følger god praksis, og sådan at der opnås bedre forståelse og accept af nye bæredygtigheds indikatorer og tests i vandforvaltningen og på tværs af fagdiscipliner (fx hydrolog-økolog og hydrolog-geolog/geokemiker), i takt med at nye metoder og tests bliver udviklet og taget i brug. Her er der inspiration at hente fra udlandet, fx i England hvor sådanne læringsprocesser har skabt ny udvikling af såvel vidensgrundlag som indikatorer og tests vedr. habitater/recipient-bæredygtighed (se appendiks 2).

En yderligere konsolidering af de indikatorer og metoder der anvendes i Danmark til såvel screening som detailundersøgelse i forhold til akvifer-bæredygtighed og recipient-bæredygtighed kræver som beskrevet ovenfor dels ny viden og dels evaluering af robusthed og usikkerhed på de anvendte indikatorer og tests. En sådan evaluering kan samtidig skærpe overvågningen og gøre den mere målrettet i forhold til valgte indikatorer og tests. Her er der en klar synergi mulighed såfremt den overvågning der foregår i vandforvaltninger og vandselskaber (i forbindelse med drift og detailundersøgelser) koordineres og såfremt det sikres at overvågning fx i forbindelse med detailundersøgelser er standardiseret og tilgængelig. Udviklingen af metoder og tests til detailundersøgelser indeholder en stærk forskningskomponent. Her er der derfor mulighed for at søge ekstern finansiering til sådanne studier. Endelig er der behov for at forbedre såvel datagrundlag som modeller (se appendiks 1).

1. Introduction

1.1 Motivation to commission the present study

The criteria used in the assessments of groundwater quantitative status made in the present first River Basin Management Plans in Denmark were quite simple, but not applied in a uniform manner throughout the country. They resulted in a poor quantitative status for many groundwater bodies indicating non-sustainable groundwater abstraction.

The water companies in Denmark in general find that the Water Framework Directive (WFD), including the Groundwater Directive (GWD), is a good and promising legislation contributing to a more holistic and integrated water resource management. They are, however, concerned that the present criteria build on fairly old knowledge that may be outdated compared to scientific state-of-the-art and to practices in other EU countries. They are furthermore concerned that the simple criteria are implemented differently in different parts of the country and that the criteria, when used too categorically as the final judgement of the groundwater status, do not allow use of better data and knowledge to re-evaluate the status.

On this background the water companies requested GEUS (i) to review the international literature and knowledge base with respect to criteria and methodologies for assessing sustainable groundwater abstraction; (ii) to review management practices from selected European countries and the Danish River Basin Management Plans; and (iii) to prepare recommendations for improvements of knowledge base and tools.

1.2 Content of the present report

Chapter 2 provides a scoping of the content of the present report and presents definitions of aquifer safe yield and environmental flow that are used in the report. Furthermore, a two-level framework for categorising methodologies into different complexity levels used for screening and investigative purposes is presented and basic discussions of the water budget myth and capture are given. Finally, some examples of methodologies are given for each complexity level with respect to aquifer safe yield and environmental flow.

Chapter 3 describes the guidance from the Common Implementation Strategy for WFD/GWD (CIS guidance document 18), with focus on the four tests for evaluating quantitative status of groundwater bodies (saline, surface water, groundwater dependent terrestrial ecosystems and water balance). Furthermore, the key recommendations from the EC implementation report of 2012 including country reports, is summarised. Finally, practices in UK, Ireland, Germany and France are described and evaluated.

Chapter 4 describes the practices and methodologies applied across Denmark with a focus on how the water balance and surface water tests have been applied for Denmark.

Chapter 5 gives a summary of the evaluation of sustainable groundwater abstraction based on six selected themes: aquifer safe yield, environmental flow, reference situation, confidence, groundwater dependent terrestrial ecosystems and climate change impacts. Chapter 6 contains the conclusions and recommendations.

In appendix 1 requirements in hydrological data and models are described.

In appendix 2 a factsheet on the Environmental Flow Indicator from England and Wales (EA) is included.

2. State-of-the-art in international literature

2.1 Scope, terminology and framework

2.1.1 Groundwater sustainability and sustainable groundwater abstraction

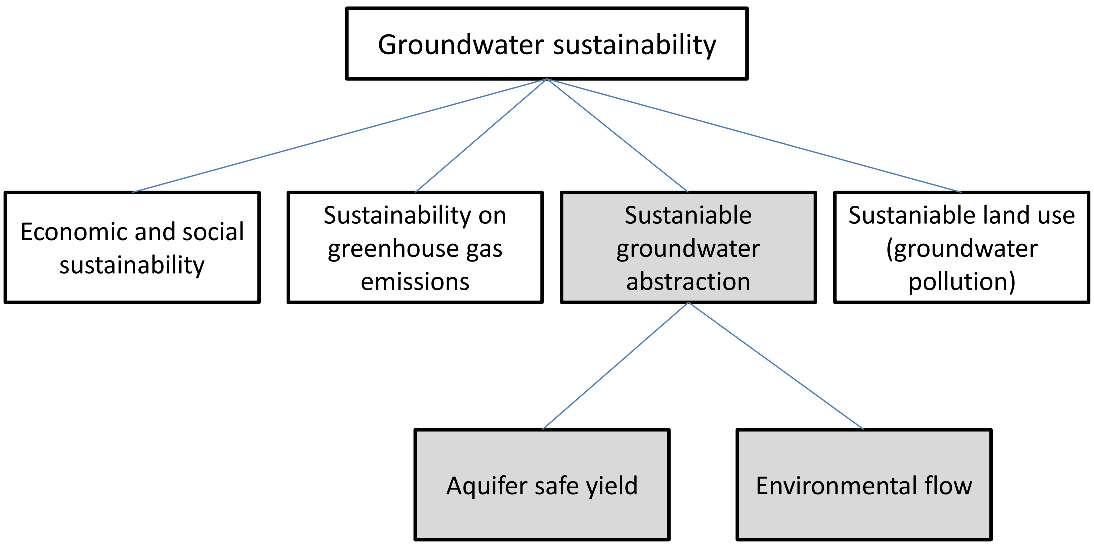


Figure 2.1 Framing of sustainable groundwater abstraction dealt with in the present report (grey boxes) within the context of the broader groundwater sustainability.

Of the sustainability issues shown in Figure (Fig. 2.1), the present report will be confined to sustainable groundwater abstraction, which again has two key elements: (i) avoidance of significant adverse effects to the aquifer due to abstraction (aquifer safe yield); and (ii) protection of ecosystem viability (environmental flow).

It should be noted that the goal of the Water Framework Directive is a good ecological status e.g. for aquatic ecosystems, and that good ecological status is a holistic approach and determined by water quality, water flow and physical conditions of streams. The influence of groundwater abstractions in reducing streamflow, which is the environmental flow focus of this report, is therefore only one of several factors affecting good ecological status.

2.1.2 Definitions of aquifer safe yield and environmental flow requirements

No unified definition for sustainable groundwater abstraction exists (Alley et al., 2002; Alley and Leake, 2004; Bredehoeft, 2002; Sophocleous, 1998/2005; Konikow and Kendy, 2005; Custodio, 2003; Villholth, 2006; Llamas, 2004; Morris et al., 2003; Henriksen et al., 2008).

Some authors see recharge to the aquifer as an important part of any sustainability assessment, arguing that it is evident that sustainability is a function of recharge and that abstraction to recharge ratio, the ***aquifer safe yield****,* cannot be ignored (Devlin and Sophocleous, 2004). Problems with safe yield can be related to various aspects. It can be an abstraction of groundwater, which regionally or locally near the well field lowers the groundwater table to a level, where either saltwater intrusion from coastal areas or saltwater upconing below the well field occurs. It can be adverse water quality effects due to nickel release from matrix to groundwater, in case of lowering of groundwater level and change of redox conditions. In most cases water quality problems e.g. nitrates, pesticides, and arsenic are not directly related to aquifer safe yield, but are more a result of either groundwater pollution from the land use or more specific problems with technical well field design (arsenic). What we therefore consider as part of assessment of aquifer safe yield are the water quality problems that are directly linked to changes in groundwater level or, eventually, increased infiltration from shallow groundwater to deeper groundwater (Henriksen et al., 2008). In some cases with very intensive abstraction, or very vulnerably aquifer types e.g. chalk aquifers (Butler et al., 2012; Soley et al., 2012) or shallow groundwater, pollutions with nitrates and pesticide can be ‗accelerated' to deeper groundwater and as such can be something to incorporate in safe yield assessments.

Other authors focus more on how groundwater abstraction impacts the natural environment that depends on the resource, such as baseflows, riparian vegetation, aquatic ecosystems and wetlands (ASCE, 1998; CIS, 2011) in the following categorized as ***environmental flow requirements***. The argument for the focus on environmental flow is that the exploitable groundwater abstraction will be less than the groundwater recharge (if not there would be a continuous trend with a lowering of the groundwater level), and will depend on potential adverse processes, and there would be a significant reduction in groundwater discharges to wetlands and river systems.

Furthermore, some authors include socio-economics as part of defining sustainable groundwater abstraction, environmental flow and the benefits for ecosystem services providing goods and services to people (Hirji & Davis, 2009; Brown and King, 2003; Dyson et al., 2003; King and Brown, 2009; King and Brown; Walton and McLane 2013). Since the WFD goal setting, including evaluation of trade-offs between different ecosystem services, socio-economic benefits and goods for society is evaluated elsewhere, as part of integrated assessment of measures related to the river basin management plan, ***we will in this report not include socio-economics*** as part of assessment of aquifer safe yield and environmental flow requirements.

In the present report we will use the definitions shown in Box 2.1

BOX 2.1 Definition of sustainable groundwater abstraction

Aquifer safe yield can be defined according to (modified from Henriksen et al. 2008):
The safe yield of a groundwater aquifer is the amount of groundwater which can be pumped from an aquifer without unacceptable negative impacts on groundwater level and water quality, compared to the pre-developmental, virgin situation.

Environmental flow requirements can be defined as (modified from Navarro and Schmidt, 2012; Arthington & and Tharme, 2003):
The environmental flow requirements are the important flow regime characteristics, i.e. the quantity, frequency, timing and duration of flow events, rates of change and predictability/variability, that are required to maintain or restore the natural flow regime in order to maintain specified, valued features of the ecosystem.

2.1.3 Sustainable groundwater abstraction - water myth and capture

Under natural conditions groundwater systems are in a dynamic equilibrium in which long-term average recharge equals long-term average discharge. Pumping groundwater from an aquifer will always cause decrease of groundwater levels. This will induce new recharge and capture. When the pumping rate is larger than the total recharge, groundwater levels will continuously decrease, and groundwater storage will eventually be depleted (Figure 2.2). This indicates mining of groundwater in the aquifer (Aeschbach-Hertig and Gleeson, 2012).

Figure 2.2 illustrates how dynamic equilibrium takes time to develop. At the virgin situation/no abstraction recharge equals groundwater discharge. When the pumping rate is no longer larger than the total recharge, the capture is sufficient to balance the pumping rate so the groundwater system will reach a new equilibrium state. The time to reach the new equilibrium is usually very long; and groundwater level drawdowns could be excessive; depending on the groundwater system characteristics, nature of recharge and discharge, and the pattern and rates of pumping wells. However, the capture may have caused the depletion of stream flows, drying of springs, and loss of riparian ecosystems and wetlands. After a new dynamic equilibrium has established, abstraction equals the change in groundwater discharge.

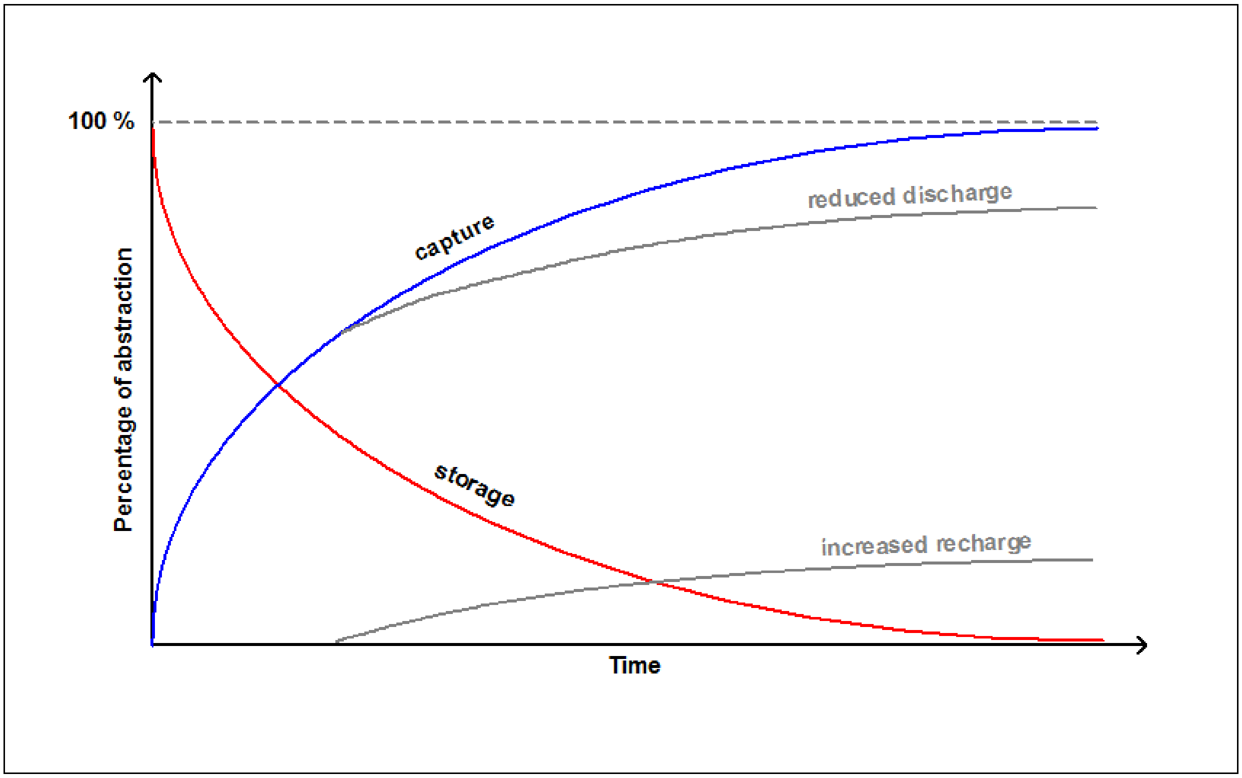


Figure 2.2 What happens when groundwater is abstracted? (Bjerre, 2012). The figure shows the significance of „capture". At time (t=0) a constant groundwater abstraction is initiated. This influences the water balance terms of the aquifer. Groundwater recharge is increased over time. Discharge (river runoff) is decreased. Groundwater storage is decreased. In the initial time period the storage term is balancing the abstracted groundwater, but later on when storage has been changed, capture terms take over the „water balance". At a certain stage groundwater recharge will be increased and discharge decreased.

The principle of water capture is related to the so called ‗water budget myth' (Bredehoeft, 2002; Devlin and Sophocleous, 2004). The water budget myth is the idea that persists within the groundwater community and beyond, that if one can determine the recharge (virgin) to an aquifer system then one can determine the maximum magnitude of a sustainable development. The basis for that is the common sense view, that the pumping must not exceed the recharge, if the development is to be sustainable. This idea is a myth: ―*because it is so ingrained in the community's collective thinking that nothing seems to derail it*" (Bredehoeft, 2002). Bredehoeft argues, that instead of using the water budget (compare abstraction to virgin recharge to aquifer as safe yield criteria), the focus in assessment of safe yield rather should be on how ―*capture"* occurs in an aquifer system which is a dynamic process, where the principal tool for such investigations is the groundwater model. Somehow as reflected by Bredehoeft the groundwater community seems to lose sight of the fundamental principles in understanding groundwater safe yield, first spelled out by Theis in 1940, and getting confused by purely administrative safe yield assessments, not considering the dynamics of groundwater systems, which for many reasons may lead to severe mistakes about the safe yield and quantitative status of groundwater aquifer systems. Devlin and Sophocleous (2005) followed up on the water budget myth and stated that even though the mistaken water budget myth still persists, then on the other hand the assessment of recharge still prevails as an important element. Not as part of estimating sustainable pumping rates, but as part of sustainability assessment, due to the effects recharge is likely to have on water quality, ecology, socio-economic and as requirement for groundwater modelling. The argument by Devlin and Sophocleous (2005) is that sustainability is a goal for the long term welfare of both humans and the environment, and that any modern assessment of groundwater sustainability requires a computer modelling component to assess the behaviour of the aquifer and its sustainable pumping rate. We agree to this. Water balance assessments according to the water balance myth are still prevailing in many European countries and river basins.

Some more recent literature (Bredehoeft and Durbin, 2009; Gleeson et al., 2012; Walton and McLane, 2013) highlights some long-term challenges related to the water budget myth and the capture concept. This literature underlines that time can be a tricky issue as part of assessing aquifer safe yield and environmental flow requirements. This is due to the long term character of processes and problems related to groundwater and to groundwater discharges to surface water (environmental flow). The key here is the fact that groundwater quantity and quality can take many years to fully develop into a new steady state, and decades to form and stabilize into a new equilibrium. This means that assessment of safe yield or environmental flow requirements cannot be based solely on historical monitoring data. Early warning monitoring and hydrological modelling should mutually support each other. As concluded by Bredehoeft and Durbin (2009): *―If a water manager allows more pumping than the pumping can capture, then sooner or later the pumping must be curtailed or a new equilibrium can never be reached and the system will be depleted".*

In the following, it is therefore assumed that any sound assessment of groundwater safe yield and environmental flow requirements must be supported by integrated, dynamic groundwater - surface water models that allow for a proper spatial and transient understanding of the governing processes of groundwater and surface water flow systems.

2.1.4 Methods for assessing aquifer safe yield and environmental flow requirements

The literature reveals that the question of how to translate qualitative policy considerations about sustainability related to aquifer safe yield and environmental flow requirements into quantitative criteria for site specific aquifers and river reaches remains a key challenge for water managers and policy makers (Henriksen et al., 2008). More than 200 different (generic) methods have been developed to derive ‗environmental flows' (Tharme, 2003; Arthington et al. 2006; Navarro and Schmidt, 2012). However a simplistic focus on the number of methods is not necessarily helpful (Acreman and Dunbar, 2004). For example just because a method exists on paper does not tell us how successfully or extensively it has been applied. These different methods take different variables into account.

As a general rule, to maintain a desired level of confidence, more resources and time are required for undertaking quantitative assessments at larger scales. Context is therefore an important issue that should guide the selection of method and structural level. For large scale (e.g. national) screening purposes simpler methods are often used, while more complex methods are used for investigative purposes at smaller scales where results from screening methods indicate sustainability problems.

In Table 2.1 and 2.2 we have summarised a framework for assessment methods for aquifer safe yield and environmental flow requirements.

Table 2.1 Framework for classifying yield assessment methods (screening and investigative)

Purpose	Complexity level	Administrative basis	Data needs for practical use	Knowledge base for development and validation
Screening	Simple	Groundwater recharge and change due to abstraction *(Hydrogeological)*	Abstraction and groundwater recharge relationships.	Existing databases. Integrated groundwater and surface water flow model. Calibrated relationship between alterations in groundwater recharge and aquifer conditions (quality and quantity).
Investigative	Complex	Groundwater level drawdown in relation to geological layers, sea level and vulnerability of aquifer water quality to groundwater level drawdown. Integrated groundwater and surface water / groundwater solute transport and saltwater intrusion models *(Groundwater level and holistic)*	Historical flow records. Groundwater abstraction, groundwater level and groundwater quality relationships. Borehole logging. 3D geological and hydrological models. Age dating.	Existing databases and new investigative geophysical/hydrogeological data, including mapping of groundwater level and water quality. Integrated groundwater and surface water flow model including particle tracking. Advanced, detailed 3D geological and hydrological flow, solute transport and salt water intrusion models.

Table 2.2 Framework for classifying environmental flow requirement methods (simple and complex)

Purpose	Complexity level	Administrative basis	Data needs for practical use	Knowledge base for development and validation
Screening	Simple	River flows and change due to abstraction. *(Ecohydrological)*	Historical flow records. Abstraction and low flow reduction relationships.	Existing hydrological and ecological databases. Integrated groundwater and surface water flow model. Calibrated relationship between flow, hydrological alteration and ecological metrics
Investigative	Complex	Target specific biota and their instream habitat. Hydromorphology, temperature and water quality issues. The whole ecosystem all/most individual components including groundwater/floodplain and terrestrial interactions. *(Hydraulic/habitat, holistic)*	Historical flow records. Abstraction and low flow reduction relationships. Hydraulic variables of representative cross sections. Suitability habitat data for target species. Biological data on flow and habitat-related requirements of all biota and ecological components.	Existing databases and new investigative ecological monitoring data on multiple ecosystems. Integrated groundwater and surface water flow, hydraulic/habitat model and/or ecological models. Specialist expertise on hydrological, hydraulic, habitat and ecosystem components. Ability to tule out (or at least quantify) non-flow impacts on biota.

Screening methods are in general more *simple* to administrate (less resource demanding), with a low resolution of output, less flexible, with relatively low cost for the assessment, but with a high uncertainty regarding obtaining of the good ecological status or quantitative/chemical status of groundwater bodies. As screening methods they are typically designed with an in-built level of precaution. *Investigative methods* are *complex* and administratively more difficult. They are site specific and with more targeted indicators and higher transparency. Furthermore, they allow for a wider range of alternative measures to be evaluated.

2.2 The four Danish indicators

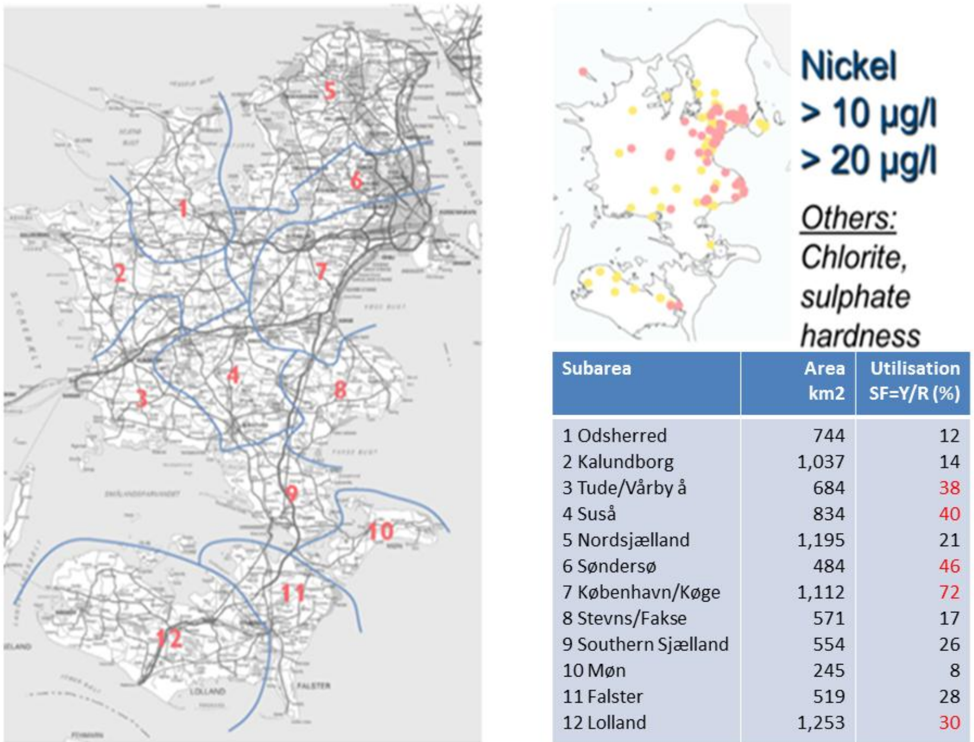
The above concerns on groundwater sustainability were translated into four sustainabiltiy indicators shown in Table 2.3 used for an assessment of the Danish groundwater resources from 2003 (Henriksen and Sonnenborg, 2003; Henriksen et al., 2008).

Table 2.3: Four sustainability indicators for Denmark (Henriksen et al., 2008). Please note that Indicator 2 simply refer to an exploitable fraction of actual recharge and not „a max. increase in recharge", as confusingly miscommunicated in Table 2 in the paper in Journal of Hydrology (Henriksen et al., 2008)

Indicator no.	Indicator	Factor considered
1	Max abstraction = 35 % of natural (pristine) recharge to aquifer	Aquifer sustainability factor (predevelopment)
2	Max abstraction = 30 % of actual groundwater recharge to aquifer	Aquifer sustainability factor (actual yield)
3	Max reduction of annual streamflow = 10 %	Streamflow depletion in relation to mean flow
4	Max reduction of low flows = {5%, 10%, 25%, 50%} depending on ecological objective for river reach	Reduced baseflow

The maximum abstraction of 35% and 30% limits (Indicators 1 and 2) was derived empirically based on an analysis of the actual groundwater quality and abstraction rates for Sjælland, where an observation had been made that areas with intense groundwater abstraction and significant lowering of the groundwater table often extended problems with inorganic trace elements (Figure 2.3; Henriksen et al., 2008).The exploitable groundwater resources were assessed for aquifers at depths of 30-50 m from where the majority of groundwater abstractions takes place. In translation of the abstraction-runoff balancing principle it has been assessed that a 10 % reduction of the average accumulated river runoff from the entire catchment is acceptable (Indicator 3).

The indicator for depletion of low flows (Indicator 4; Figure 2.4) is based on guidelines from the Danish EPA from 1979 (Miljøstyrelsen, 1979).



Subarea	Area km2	Utilisation SF=Y/R (%)
1 Odsherred	744	12
2 Kalundborg	1,037	14
3 Tude/Vårby å	684	38
4 Suså	834	40
5 Nordsjælland	1,195	21
6 Søndersø	484	46
7 København/Køge	1,112	72
8 Stevns/Fakse	571	17
9 Southern Sjælland	554	26
10 Møn	245	8
11 Falster	519	28
12 Lolland	1,253	30

Figure 2.3 The evaluation of indicator 2 (threshold= 30 % of recharge, Henriksen et al., 2008) was based on comparison of exploitation rate ("utilisation" = % abstraction of groundwater recharge to model layer 3 which is the regional aquifer for Sjælland, Møn, Falster and Lolland) and current water quality conditions (like nickel, chlorite, sulphate). It has not been validated for other areas.

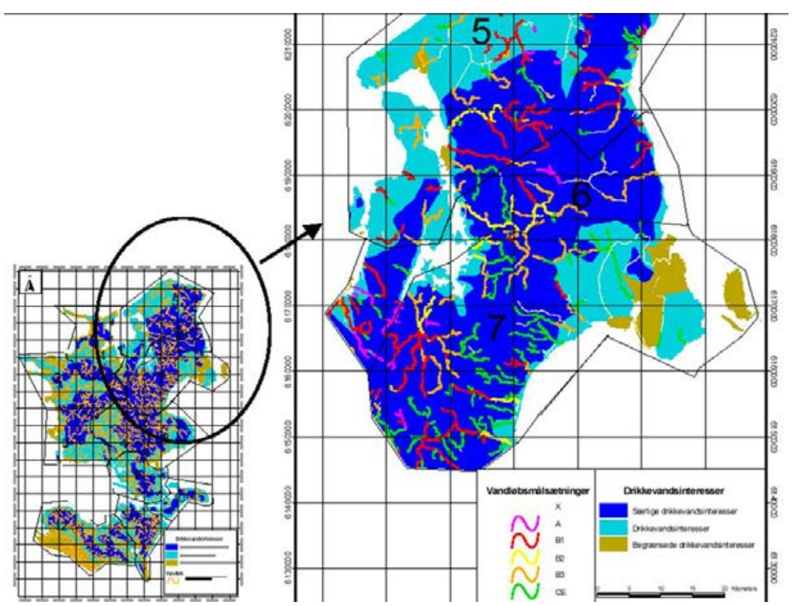


Figure 2.4 Indicator 4 depletion of low flows is acceptable if it is below a 5 % (A), 10 % (B1), 15 % (B2), 25 % (B3) and 50% (C–F) reduction. In the assessment by Henriksen et al. (2008) the incremental (and not accumulated) reduction in baseflow was evaluated by Indicator 4 on the subarea level (e.g. for area 5, 6 and 7). Discharges from waste water treatment plans were not incorporated.

Indicator 4 (Fig 2.4) prescribes that a maximum reduction of low flows depending on the ecological objectives of the river reach, which is categorized in A (Reaches with rare flora and/or fauna classified for high protection and with high research interests), B1 (Salmonid spawning and nursery waters), B2 (Salmonid waters – nursery and living areas for trout), B3 (Cyprinid waters) and C-F (Watercourses solely used for drainage purposes, waters where authorized waste and water discharges cause the quality to be worse, watercourses where the effects of water abstraction render it impossible to maintain fish water objective or watercourses markedly affected by ochre discharge).

Indicator 1 compares actual abstraction to predevelopment recharge (to the groundwater body), and prescribes a max 35 % exploitation of predevelopment (virgin) recharge. In a way the indicator hereby is in line with thinking according to the Water Budget Myth, e.g. the effects of capture are not considered (the range of the indicator is from 0 to above 100 %, since capture and induced groundwater recharge is not included). Contrary to this, indicator 2 (and indicators 3-4) incorporate capture (which means that the range of indicator 2 is from 0 to 100 %, or exploitation factor is between 0 to 1; with a precautionary threshold factor of max. 0.30).

Of the 48 subareas used for the final assessment (Henriksen, 2008; Henriksen and Sonnenborg, 2003) the most constraining of the four were:

- Indicator 1: groundwater abstraction compared to pristine groundwater recharge: 3 subareas
- Indicator 2: groundwater abstraction compared to actual groundwater recharge: 12 subareas
- Indicator 3: reduction in mean flow: 5 subareas
- Indicator 4: reduction in low flow indicator: 28 subareas

This shows that in nearly 70 % of the subareas, indicator 3-4 are the most constraining indicators.

Henriksen et al. (2008) evaluated the approach as appropriate for the WFD, by offering an integration of groundwater and surface water by the use of groundwater and surface models that can be used to analyse the interaction between these two domains, and hereby taking into account the complex and dynamic aspects of the groundwater system and flow components (including capture). The four sustainability criteria focused on avoiding significantly negative impacts of groundwater abstraction on both surface water ecology (criteria 3 and 4) and groundwater quality (criteria 1 and 2) at the national screening level (precautionary). The approach was evaluated well in line with the underlying WFD principles, by providing a transparent, practical and scientifically based methodology for assessment of the sustainable groundwater abstraction. The uncertainty related to the assessment was estimated at +- 10% for the total exploitable resource for Denmark. For the 50 subareas the uncertainty was estimated at +- 40% (Henriksen et al., 2008) when used with or four indicators and the first version of the national hydrological model (DK model) released in 2003. The most significant uncertainties were assumed to be the threshold factors (for indicator 1-4) followed by the uncertainties on the hydrological model.

2.3 General issues

2.3.1 Estimates of the groundwater recharge to aquifers

A main challenge in the assessments is how to estimate the recharge to groundwater aquifers. Different flow processes operate on the shallow hydrological system, which means that only a fraction of the net-precipitation will enter as groundwater recharge to the main aquifer systems (Fig. 2.5).

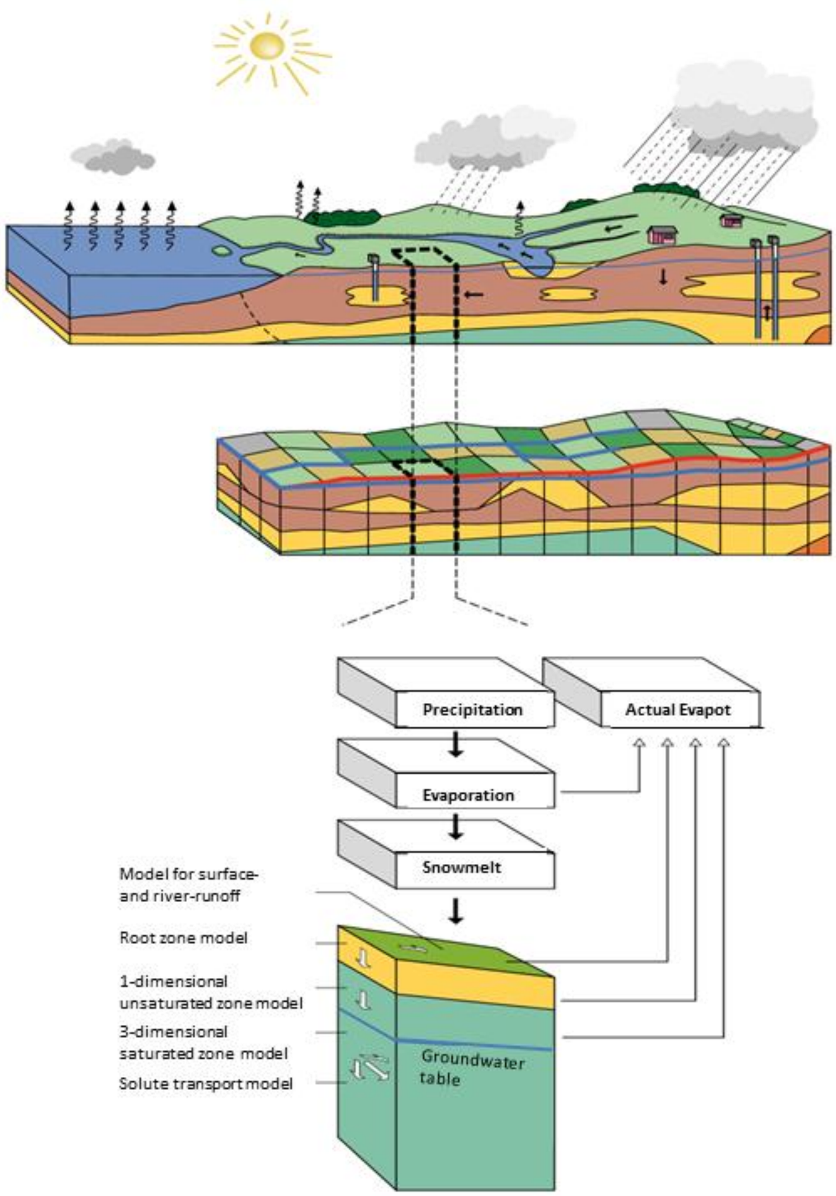


Figure 2.5 Groundwater recharge simulation by use of an integrated groundwater-surface water model (Source: Aarhus county)

As illustrated in Fig. 2.5 input from precipitation may temporarily be stored on vegetation and as snow before eventually evaporating or reaching the ground surface. Here, a small fraction of the water will run off as overland flow to rivers, while the largest fraction will infiltrate into the soil and enter the root zone. From here it is separated into either evapotranspiration or percolation through the unsaturated zone to the phreatic groundwater table. Thus the recharge to the phreatic groundwater table is the precipitation minus the total evapotranspiration.

In the groundwater system, which generally is composed of aquifer and aquitard layers horizontal drainage will occur through tile drains and aquifers. Therefore the groundwater recharge generally reduces with depth implying that groundwater recharge always should be characterised with respect to a particular (depth of) aquifer.

Recharge estimates to aquifers are uncertain for several reasons, and a drawback of ―recharge calculators” is the lack of validation possibility for the recharge estimates. This is a main reason for using integrated hydrological groundwater and surface flow models, which can be calibrated and validated against runoff data and groundwater level observations (Refsgaard et al., 2010). Another issue is the lack of possible evaluation of pristine and actual recharge (without and with pumping). Many of the shallow hydrological flow processes are influenced by the amount of pumping and lowering of the shallow groundwater table (Walton and McLane, 2013). Therefore, infiltration, interflow and rapid runoff may be impacted by different levels of groundwater abstraction, which cannot be considered by simplified recharge calculators. Hydrologists therefore usually use complementary methods for checking the recharge estimates for instance used for ‗steady state or transient groundwater models‘ where actual recharge needs to be entered as input data to the groundwater models. These methods use baseflow index or analyse river runoff, in order to provide a check of the actual recharge.

When moving in the direction from simple to complex aquifer safe yield assessment methods, and also to environmental flow requirements and when including water quality and impacts on terrestrial ecosystems, the challenge (and knowledge gaps) is to link the predicted changes of groundwater levels and flows to the impacts on ecosystems and to groundwater and surface water quality (Zhou 2009; Sophocleous 2007; Henriksen et al., 2008; Walton and McLane, 2013).

In general, groundwater recharge to a specific groundwater aquifer is significantly dependent on both hydrometeorological and hydrogeological conditions. For the entire North Sea Region the CLIWAT project, which focused on climate change impacts, estimated the change in groundwater recharge and dependency on climate as shown in Fig. 2.6 (for a virgin situation without abstraction).

What can be seen from Fig. 2.6 is that for the sandy area, the groundwater recharge to regional (shallow) aquifers amounts to 300 mm/year and is expected to be increased by 10 % by future climate change impacts. For the clayey area, groundwater recharge is only a third (100 mm/year), and will only be increased by 5 % by climate change. Climate change results in rather complex impacts on the hydrological cycle. In some areas groundwater recharge will increase significantly, in cases with a relatively high precipitation compared to evapotranspiration, and significant increase in precipitation especially during the winter, where evapotranspiration is limited. This is the case for sandy areas in the western part of Denmark. For other areas like clayey areas in the western part of Denmark the increase in groundwater recharge is more limited, and evapotranspiration for some climate models and areas eventually can lead to a moderate decrease in the groundwater recharge & level.

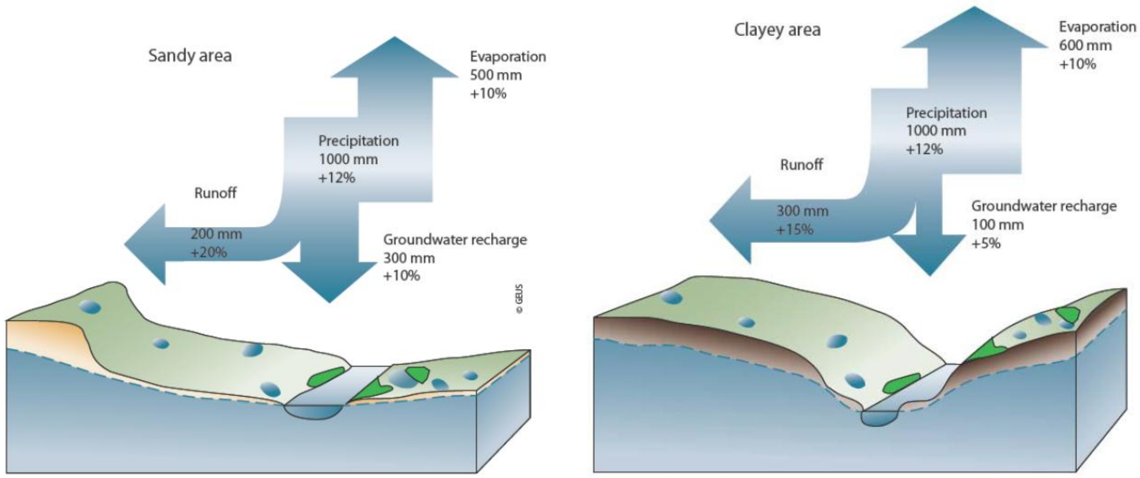
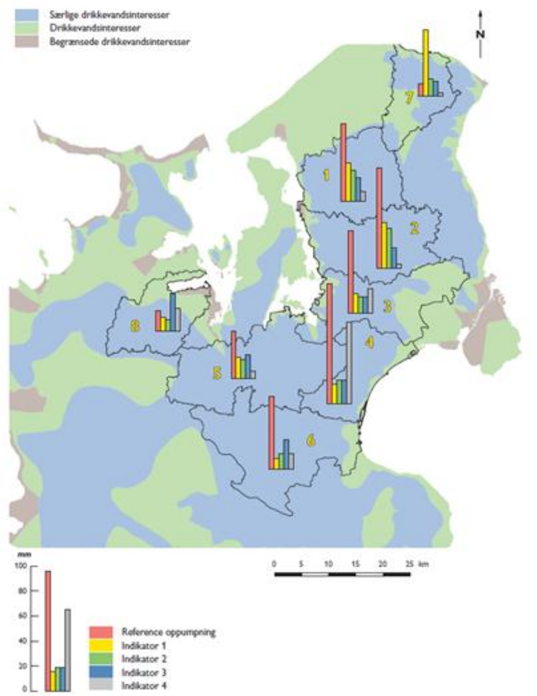


Figure 2.6 Generalized water balance for clay and sandy areas in the North Sea Region and projected changes. Numbers (% change) is due to climate changes for A2 Scenario for a future climate 2071-2100 compared to reference period 1961-1990 (CLIWAT, 2011)

2.3.2 Spatial and temporal scale issues

Simple screening criteria have the weakness that they are scale dependent. This is illustrated by examples for the four Danish screening criteria outlined in Section 2.2. These have been „designed for a specific scale (300–2000 km^2). Use on other scales therefore requires re-assessment of thresholds for the four indicators” (Henriksen et al., 2008). An analysis for KE by Troldborg and Henriksen (2006) illustrated the results of the Indicator 2 if applied for smaller subareas, see Fig. 2.7.

The results indicate a scale dependency, and for subarea 4 utilisation is higher than groundwater recharge which indicate that there is an inflow of groundwater to subarea 4 from neighbour groundwater aquifers, since per definition indicator 2 should always be in the range of [0-1], if the entire groundwater aquifer has been used for the assessment. Lambán et al. (2011) summarised some experiences from applications of groundwater sustainability indicators in different countries concluding that the best scale for applying this type of indicators is the aquifer scale. When used for subares of an aquifer, different (higher) SF criteria‘s eventually can be used (if site specific results can document the criteria), and groundwater inflows across boundaries should be incorporated in the assessment of the total ‚groundwater recharge‘ to the aquifer.



Sustainable abstraction mm/year

Figure 2.7 Test of four indicators for 7 KE subareas to well fields (Troldborg and Henriksen, 2006).

Area	1	2	3	4	5	6	7	8
Recharge (mm/year)	236 km^2	270 km^2	246 km^2	186 km^2	375 km^2	359 km^2	167 km^2	185 km^2
• Pristine	88	102	46	47	48	22	152	31
• Actual	111	128	88	84	70	59	154	37
Abstraction (mm/year)	62	80	66	96	38	58	10	16
Base flow (mm/year)								
• Pristine	61	58	57	87	79	47	52	46
• Actual	33	32	27	34	59	28	45	41
Aquifer safe yield Indicator 2 according to the Danish assessment (mm/year)								
Indicator 2 = Actual abstraction / Actual Recharge to groundwater aquifer (Red values are above SF = 30 %)	**56**	**63**	**75**	**114**	**54**	**98**	6	**43**

Figure 2.8 illustrates the importance of capture, with a quite significant induced recharge due to the intensive groundwater abstraction. For the 8 areas the groundwater recharge to the regional aquifer is, due to pumping of 63 mm/year, increased from 56 mm/year (for pristine/predevelopment situation without pumping) to 87 mm/year. At the same time baseflow is reduced from 61 mm/year (for pristine situation) to 33 mm/year (we will come back to this effect of capture in the section below on environmental flow). In fact, the low flow indicator (Indicator 4 in the Danish assessment) was the most scale dependent of the four indicators.

It is the actual groundwater recharge to aquifers which should be used for assessments of sustainable yield. In Denmark shallow groundwater from the land surface to a depth of 30-50 meter below the surface is significantly contaminated with nitrates, pesticides and other substances. The actual abstractions in most cases are therefore screened below this depth. Therefore, it was not the groundwater recharge below the root zone, but the groundwater recharge to model layer 3 for the eastern islands corresponding to a depth of around 30-50 meter below the surface, which was used (~the regional aquifer). Similarly for Jylland groundwater recharge to model layer 5 was used representing a depth of around 30-50 meter below the surface (reflecting vertical discretization). According to the groundwater monitoring programme the used groundwater recharges had a very limited content of nitrates and pesticides both for Jylland and for eastern islands. A better and more accurate assessment would have required a full mapping of the actual groundwater aquifers (upper top layer, bottom layer and extension according to groundwater boundaries of the aquifers in case of multi layer aquifer systems), and incorporation of groundwater recharge (and eventual inflows from neighbour aquifers).

It is well known that groundwater recharge for a multi-layer aquifer system in general is highest to the uppermost aquifer, and then decreases, so that deep aquifers may have relatively small amounts of groundwater recharge, compared to shallow aquifers. This is illustrated in Fig. 2.8, where the upper regional aquifer for Sjælland (model layer 3) has a groundwater recharge of 62 mm/year, and the lower regional aquifer (model layer 9) only has a groundwater recharge of 36 mm/year.

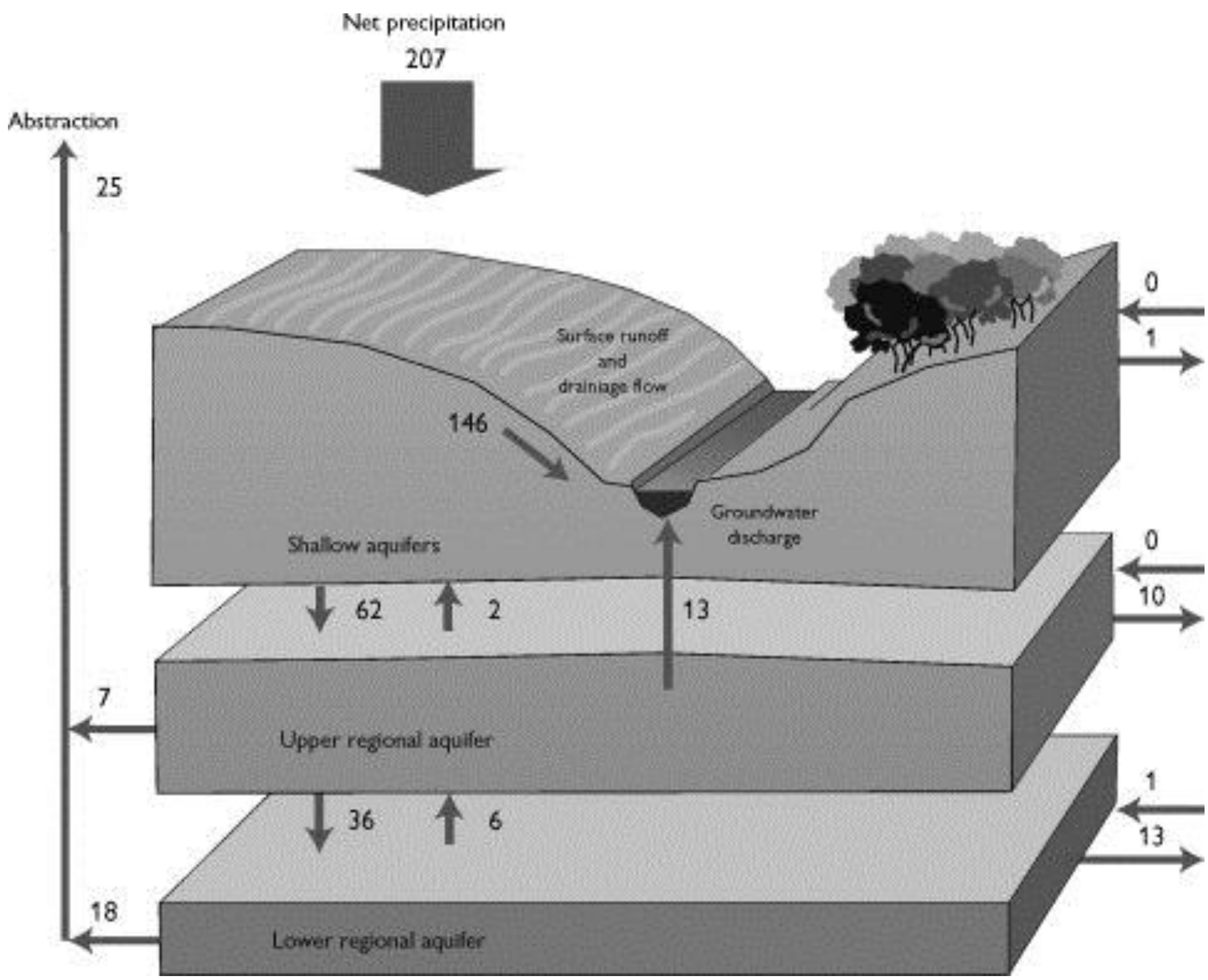


Figure 2.8 Model simulated water balance for Sjælland (Henriksen et al., 2008)

2.4 Examples of simple screening methods from literature

2.4.1 Simple (screening) aquifer safe yield assessment methods

Smith et al. (2010) compare different assessments in the literature about aquifer safe yield by use of the aquifer sustainability factor, SF:

$$SF = Y / R$$

where Y is the extractable yield and R the groundwater recharge to aquifer, see Fig 2.9. The groundwater recharge is here the actual recharge to the groundwater aquifer, and hence SF ranges between 0 and 1. SF corresponds to Indicator 2 in the Danish assessment (Table 2.3). It should be noted, that Fig 2.9 includes also some SF estimates, which are related to environmental flow rather than to aquifer safe yield derived and located between the lines SF = 0.15 and SF = 0.7 and shown with ochre brown triangles on the figure.

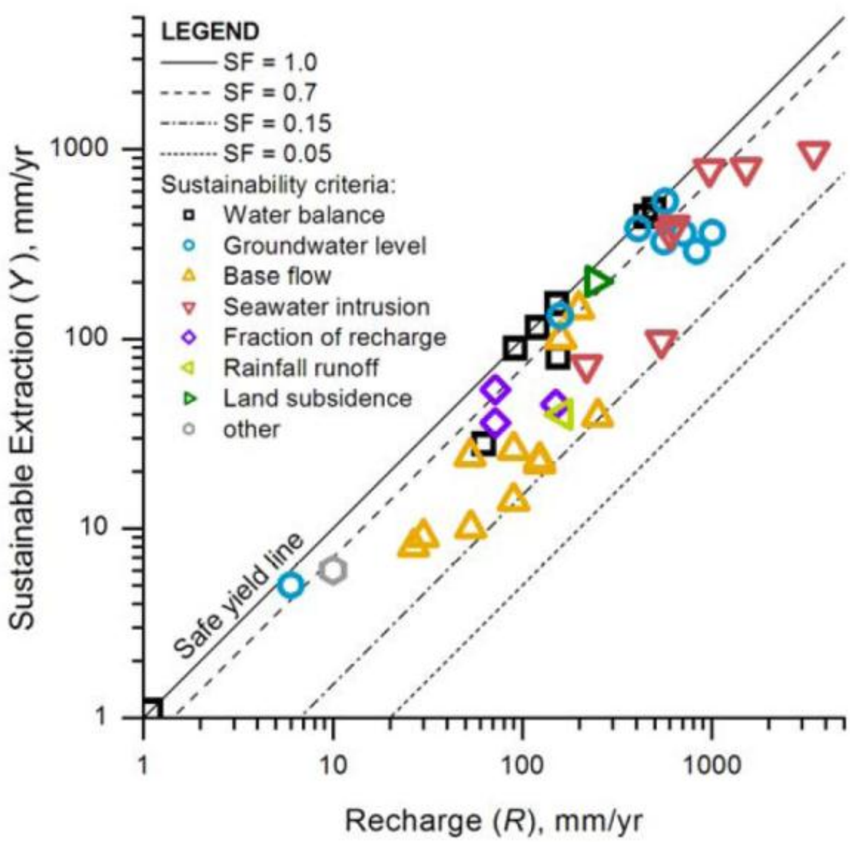


Figure 2.9 Relationship between exploitable abstraction (extraction) and groundwater recharge to aquifers. Sustainable factor (SF = exploitable yield / R = groundwater recharge to aquifer) based on 32 international studies from the literature (Smith et al., 2010).

Smith et al. (2010) note that SF would be derived first from groundwater modelling studies, where recharge to aquifers, R, can be evaluated. SF would then be post-evaluated as a metric for comparing results from different regions. Furthermore, Smith et al. (2010) distinct between safe yield and sustainable yield, where safe yield represents abstraction constrained by the amount of groundwater recharge, and sustainable yield represents abstraction constrained by the feasibly capture of groundwater discharge (which is similar to our thinking regarding the environmental flow requirement).

The literature review in Smith et al. (2010) was based on 32 international studies including one for Denmark (Henriksen et al., 2008). Based on this a range of exploitable recharge factors between 0.15 and 0.85 was estimated. Of the 32 studies, maximum reduction in baseflow was applied for 6 studies, maximum allowable drawdown of groundwater level for 7 studies, water balance stabilization for 7 studies, maximum allowable seawater intrusion for 4 studies, extraction at allowable fraction of the groundwater recharge rate for 2 studies, maximum reduction of rainfall runoff for 1 study and maximum allowable land subsidence for 1 study.

It can be seen from Fig. 2.9 that SF is in the range between 0.15 and 0.7 for most of the review results that evaluate exploitable yield in relation to groundwater level, base flow and seawater intrusion. There is a general tendency that water balance based estimates result in higher aquifer safe yield fractions of recharge, with values near 1 (SF factor in the range between 0.5 and 1). Groundwater level based estimates evaluate SF factors ranging from 0.4 to 0.95. Finally, baseflow studies resulted in the lowest SF factors between 0.15 and 0.8. The studies incorporated a variety of aquifer types and geographic locations, including groundwater systems in Jordan, Australia, Taiwan, USA, South Korea, Denmark, Namibia, China, England, India, Turkey, Israel, Iran and Greece. The authors concludes that a more detailed database derived from a larger number of well-studied groundwater systems would be required to further explore possible general relationships between SF, Y, aquifer attributes and various sustainability criteria. Ideally, Y should be evaluated directly based on quantitative studies whenever possible (Smith et al., 2010).

Summary and evaluation of screening methods for aquifer safe yield assessments

State-of-the-art literature recommends focusing on the effects of *–capture”* of an aquifer system, e.g. the dynamic process by which a change in abstraction is compensated by changes in induced recharge, groundwater discharge to surface water and changed flow across boundaries. The sustainability criteria reflecting this principle is the Sustainable Yield Factor (SF = actual abstraction / actual recharge to aquifer) which is equivalent with Indicator 2 among the four Danish indicators (Table 2.3). Results from studies around the world suggest that the SF indicator has a high uncertainty range [0.2; 1.0] under different conditions. The indicator has only been subject to a coarse test against aquifer conditions on Sjælland and better documentation and validation of the actual threshold values is recommended before using the indicator in a precautionary manner for national screening in Denmark. Tests indicate that threshold values vary with the spatial scale for which they are applied, and hence that they should be applied only to the spatial scale for which they have been derived, typically aquifer systems as a whole.

2.4.2 Complex (investigative) aquifer safe yield assessment methods

When moving from simple methods toward complex methods a shift from flow based indicators towards a combination of water balance based indicators for sub-basins (e.g. single aquifers or subareas within a river basin) and appropriate water level and flow system characterization (e.g. location and distributions of abstractions in relation to rivers, coast etc.) becomes the state-of-the-art (Barthel et al., 2011; Walton and McLane, 2013). However, even though many practical examples are available no unified methodology or guidelines for assessment of aquifer safe yield according to the complex level, to our knowledge, are available in the literature. In the following we will try to briefly explain the knowledge base.

The above difficulties with water balance based evaluation and needs for including climate variability, horizontal flow across boundaries (Uddameri 2005), groundwater recharge in different depths highlight that no assessment of safe yield, whether simple or complex, can be provided without using a dynamic, groundwater - surface water flow model as illustrated by the example of Henriksen et al. (2008). Without such model it will not be possible in a reliable manner to assess the dynamic changes of the water balance and the complex changes in interactions between abstraction, groundwater level, drainage flow, groundwater discharges in space and time that groundwater abstraction will cause (Mansour et al., 2012; Whitman et al., 2012a; Hinsby et al., 2006).

Since complex safe yield methods are –site specific” the conceptual model understanding is a basic tool and starting point. Furthermore, hydrological modelling of groundwater drawdown will be a key component together with water balance evaluation. The latter, not necessarily with a single focus on low flow impacts from pumping at a river reach level, which is generally evaluated as an uncertain business, but also with a broader approach, acknowledging the uncertainty of hydrological models in proper quantification of impact on low flow for various reaches, and instead keeping focus on changes in the overall water balance for subareas, which can be modelled with a proper transient hydrological groundwater and surface water model with a proper discretization.

A couple of examples of a search for complex level methodologies can be found in (Troldborg and Henriksen, 2006; Stisen et al., 2008) for seven Copenhagen well fields. There are many other practical examples from the water supply and groundwater mapping applications, where analyses have been made for well fields in relation to new groundwater abstraction licences, or evaluation of sustainable exploitation at regional or local scale.

A basic premise for complex aquifer safe yield approaches is the site specific focus on protection of groundwater supplies from depletion and that sustainability requires that withdrawals can be maintained indefinitely without creating significant long term declines in regional water levels, or flows, that new equilibrium in groundwater drawdown adequately respects also the protection of ecosystem viability (CCA, 2009). The report by CCA (2009) analysed 11 case studies in USA and Canada and described assessments of groundwater sustainability, amongst others in relation to aquifer safe yield, some of these using comprehensive field investigations combined with hydrological modelling.

Saltwater intrusion is the most imminent threat for groundwater chemical status and to some extent quantitative status for coastal aquifers due to the expected sea level rise. In contrast to many other negative climate change impacts, which vary between different climatic regions, saltwater intrusion is a global problem of great concern, as it reduces the available drinking water resource in coastal regions (Oude Essink et al., 2010). Another threat is the increased contaminant loading to groundwater and dependent or associated ecosystems.

Summary and evaluation of complex investigative methods for aquifer safe yield assessment
Investigative methods that directly incorporate site specific data using comprehensive modelling tools can provide a more knowledge based assessment of changes in groundwater level and quality. Hence, once validated, the investigative methods should overrule the results from screening methods. Unfortunately, there are no professional standards or good practices guidelines for assessment of aquifer safe yield at the complex level, neither in the groundwater handbook (Sonnenborg and Henriksen, 2005), in the Danish best practice guidelines for flow modelling (Refsgaard et al. 2010), nor in the international literature when it comes to analysis of subareas or subcatchments within a groundwater body (WFD only describe how to assess quantitative status for groundwater body basin level). The building blocks for such methods should include site specific conceptual model understanding, groundwater level changes due to groundwater abstractions, location and partial penetration (screening of wells) of groundwater abstraction wells and risk for changes in groundwater quality e.g. due to saline intrusion, sulphate or nickel mobilisation as a result of groundwater drawdown. Finally uncertainty quantification and planning under uncertainty should be acknowledged.

2.4.3 Simple (screening) environmental flow assessment methods

Simple methods related to environmental flow requirements should be applied over large spatial scales, principally for national screening assessment. Hence they need to be expressed as “rules” which indicate acceptable hydrological alteration.

Environmental flow recommendations are designed as acceptable deviations from the natural flow regime and should, depending on the desired level of ecological level of ambition, to a greater or lesser extent reflect this natural flow regime. Although a basic assumption here is, that the full range of natural variability in the hydrological regime is necessary to maintain aquatic ecosystems, the first hydrological methods used in rivers were based on ‚a minimum flow‘ (Gippel, 2001).

Like for aquifer safe yield indicators related to recharge, a similar challenge exists for defining acceptable levels of flow modification for different ecological goals.

Acreman et al. (2005) assembled a panel of ecologists to achieve agreed acceptable limits across the range of UK rivers for key components of biota. The ecologists involved in the study expressed the figures of acceptable flow reductions in terms of points at which they could no longer be certain that good status would be achieved (Dunbar, personal communication), implying that the figures must be seen as precautionary. The project produced a classification system and lookup tables for each river type, specifying the maximum abstraction allowable at different flows (Acreman et al., 2005; Acreman and Ferguson, 2010; Acreman et al., 2008).

To classify river water bodies for fish communities, the assembled expert panel reviewed the typology devised by Cowx et al. (2004) and reduced their original eight fish types to four that have different flow regime requirements

1. High base flow (chalk geology) rivers – groundwater-fed rivers with smoothly varying flow regimes
2. Eurytopic/limnophylic cyprinids, e.g. common bream *Abramis brama* Linnaeus 1758 – low slowflowing water
3. Rheophlic cyprinids, e.g. barbel *Barbus barbus* Linnaeus 1758 – mid-reach, fast flowing water
4. Salmonid (juvenile and trout spawning and nursery areas) – headwater streams

Habitat modelling of Chalk rivers, such as the Itchen (Booker et al., 2004) and Wylye (Dunbar et al., 2000), found different ecological impacts of changes in flow in headwaters and downstream reaches, with headwaters being more sensitive to abstraction, thus requiring different environmental standards, implying that a different percentage of the flow can be abstracted. This may also to be the case for DK, but is not documented.

The Chalk river type was thus sub-divided into headwater and downstream reaches, producing 10 types overall: the eight reach types (A1, A2, B1, B2, C1, C2, D1, D2), with a sub-division of A2 into headwater, A2(hw), and downstream river reaches, A2(ds), plus salmonid spawning and nursery areas (Table 2.4).

Regional studies of macroinvertebrate communities in UK rivers (Extence et al., 1999) suggested that upland rivers are more sensitive to changes in flow than lowland rivers and therefore require more stringent standards of protection. The macroinvertebrate experts concluded that types A2, B1, C2 and D2 were probably more sensitive to abstraction and thus require the highest levels of protection with 10% permissible abstraction. Type A1 rivers are the least sensitive and require the lowest level of protection, with 30% abstraction allowable. For all other types the critical level is 20% (Acreman and Ferguson, 2010).

The fish experts focused particularly on protection of low flows, i.e. very limited or no abstraction when the flow was less than natural flow percentile, Qn_{95}, which is the flow that is equalled or exceeded 95% of the time in the pristine situation with not groundwater abstraction. The working group analyses of threshold flow needs resulted in four broad types with different permitted abstraction levels. The maximum levels of abstraction ranged from 7.5% to 35% of the natural flow depending on river type and flow rate (see Table 2.5).

Worldwide, look-up tables (Acreman and Dunbar, 2004) are the most commonly applied simple method for defining target river flows. It is implicit in these indices that they should be based on statistical properties of the natural flow regime, although this often is not specified clearly. The above UK studies used Qn_{95} as the low flow indicator, while in other cases the median of the observed/historical annual minimum flows (Q_{medmin}) has been used (Henriksen et al., 2008).

Table 2.4 UK River water reach types (Acreman and Fergusson, 2010; Holmes et al., 1998)

Holmes *et al.* type	Holmes *et al.* sub-type	Final WFD48 type
A. Low altitude; low slope; eutrophic; silt/clay-gravel bed; smooth flow	A1 lowest gradients (0.8 ± 0.4 m km^{-1}) and altitudes (36 ± 25 m), predominantly clay	A1 as sub-type
	A2 slightly steeper (1.7 ± 0.8 m km^{-1}), low altitude (55 ± 38 m)	A2 (hw) headwaters as sub-type with catchment area <100 km^2
	Chalk catchments; predominantly gravel beds, base-rich	A2 (ds) downstream as sub-type with catchment area >100 km^2
B. Hard limestone and sandstone, low-medium altitude, low-medium slope; mesotrophic; gravel-boulder (mainly pebble-cobble), mostly smooth flow, small turbulent areas	B1 gradient (4.1 ± 9.9 m km^{-1}), altitude 93 ± 69 m	B1 as sub-type
	Hard sandstone, calcareous shales	
	B2 shallower than B1 (2.7 ± 10.7 m km^{-1}); altitude 71 ± 58 m	B2 as sub-type
C. Non-calcareous shales, hard limestone and sandstone, medium altitude, medium slope, oligo-meso-trophic; pebble, cobble, boulder bed, smooth flow with abundant riffles and rapids	C1 gradient 5.4 ± 6.5 m km^{-1}; altitude 101 ± 84 m; hard limestone; more silt and sand than C2; mesotrophic	C1 as sub-type
	C2 steeper than C1 (7.3 ± 10.8 m km^{-1}); altitude 130 ± 90 m; non-calcareous shales; pebble-bedrock; oligo-mesotrophic	C2 as sub-type
D. Granites and other hard rocks; low and high altitudes; gentle and steep slopes; ultraoligo – oligotrophic; cobble, boulder, bedrock, pebble; smooth with turbulent areas – torrential	D1 medium gradient (11.3 ± 15.6 m km^{-1}); low altitude (93 ± 92 m), oligotrophic, substrate finer than D2 (incl silt & sand); more slow flow areas than D2. Includes acid heaths	D1 as sub-type
	D2 high gradient (25.5 ± 33 m km^{-1}); high altitude (178 ± 131 m); stream orders 1 & 2, bed rock and boulder; ultra-oligotrophic, torrential.	D2 as sub-type
		Salmonid (juvenile salmon and trout spawning and nursery areas) – headwater streams

Table 2.5 Standards for UK river types/sub-types for achieving good ecological status as % allowable reduction of natural flow. Thresholds are for annual flow statistics (Acreman and Ferguson, 2010). Qn_{95} is the natural flow which is equalled or exceeded for 95 % of the time. See Appendix 2 for updated fact sheet with EFI-thresholds for England/Wales from Environmental Agency 2013).

Type or sub type	Season	Flow $> Qn_{60}$	Flow $> Qn_{70}$	Flow $> Qn_{95}$	Flow $< Qn_{95}$
A1	Apr.–Oct.	30	25	20	15
	Nov.–Mar.	35	30	25	20
A2 (ds), B1, B2, C1, D1	Apr.–Oct.	25	20	15	10
	Nov.–Mar.	30	25	20	15
A2 (hw),	Apr.–Oct.	20	15	10	7.5
C2, D2	Nov.–Mar.	25	20	15	10
Salmonid spawning &	June–Sep.	25	20	15	10
nursery areas	Oct.–May	20	15	Flow $> Q_{80}$	Flow $< Q_{80}$
(not chalk rivers)				10	7.5

Bradley et al. (2013), in a study of a selection of streams fed from sandstone aquifers in the English Midlands showed that few ecological impacts were identified when the abstraction was less than 60 % of Q_{75} (median-low flow indicator). The results of these studies suggest that current environmental standards for hydrology, which are considered by expert opinion to support good ecological status (Acreman *et al*. 2008), might be conservative for macro-invertebrates in rivers fed by Permo-Triassic sandstone aquifers.

The advantage of look-up tables, is at the same time the largest disadvantage, because once the general procedure has been developed, application requires relatively few resources, and is as such administratively friendly. However, such rapid approaches tend to be calibrated for a particular region and the transferability to elsewhere is questionable if it is not specifically tested. As based purely on hydrological data, they can be calculated for any region, but they may have very little ecological validity or the ecological data for calibration may be costly and time-consuming to collect. And even then, they do not necessarily take account of site specific conditions (Acreman and Dunbar, 2004). Therefore, they are particularly appropriate for low controversy situations, and tend to be established with worst case assumptions when selecting indicators for UK conditions. To which extent the Danish indicators (Section 2.2) are also precautionary has so far not been analysed.

The key according to environmental flow requirements thinking is that the hydrological flow alterations is one factor determining overall ecological status. Its relative impact will depend on the degree of alteration, the background hydro-meteorological situation and the relative magnitude of other pressures. For Danish rivers this will most likely be either the hydrological alteration at low flow situations (typically in the summer months at low flow or during juvenile fish development for Denmark for trout) or high flow situations (spawning and migration).

Minimum flows, baseflows and the seasonal patterns, flood regime, and rate of hydrological alterations are therefore the key issues (Navarro and Schmidt, 2012) in environmental flow requirements for impacts on streams (Soley et al., 2012). In a Danish report (NERI, 2000) median minimum flow, Q_{medmin}, was positively correlated to DVFI (increased Q_{medmin} better DVFI). The rivers with the highest DVFI corresponded to rivers where Q_{medmin} was relatively high in dry periods. Frequency and duration of low flow conditions was not found correlated to DVFI. Rivers with high organic matter content generally had the poorest DVFI (Clausen et al., 2000). The results in these studies have, however, not been used in the present Danish practice (Chapter 4).

A recent approach in UK is the DRIED-UP, a ―national hydro-ecological model” developed incrementally over past six years (Dunbar, 2013). Mainly funded by EA. Started with 11 monitoring sites in North Anglia, and subsequently tested on 86 upland sites. The ―current model” is based on 146 macro-invertebrate monitoring sites across UK (upland and lowland, spring and autumn samples). The DRIED-UP approach attempt to describe the relationship between LIFE (an ecological metric for macro-invertebrates) and antecedent low (Q95) and high (Q10) flow. The overall messages from the DRIED-UP database/models are that physical habitat modifications influences sensitivity to flow change (depth/velocity habitat is implicit, not explicit; model describes generic response, while still allowing response of individual sites to vary). One application of DRIED-UP is to derive robust site-specific relationships where relatively short series of monitoring data are available.

An extension of DRIED-UP, DRUWID, is a generic framework for creating DRIED-UP type models using lagged flow variables over any number of antecedent years (so far 1-2 previous years summer low flows, including interacting flow sequences e.g. with two subsequent years of low flow resulting in a lower LIFE otherwise expected). Hereby, the tool gives a more realistic quantitative description of how LIFE (macro-invertebrates) responds to intra-seasonal and supra-seasonal drought. The next steps are to develop more regional DRUWID models, to incorporate explicit abstraction impacts and possibly to incorporate hydraulics (Dunbar, 2013).

Summary and evaluation of screening methods related to environmental flow requirements

When defining acceptable levels of flow modification for different ecological goals, it should be noted that river flows has significant seasonal variability, rivers are impacted by pressures other than flow, and river biota vary –naturally” in space and time. Look up tables and classification systems have been established in the literature, e.g. in the UK, for key components of biota (fish, macro-invertebrates and macrophytes) for different flow regimes, headwater/downstream reaches and different hydrogeological settings. Hereby, thresholds for annual and seasonal flow statistics and allowed changes due to abstraction have been derived based on expert judgement, based on precautionary principles. The UK studies used Qn_{95} for different seasons as the low flow indicator allowing reductions between 7.5 and 35%, while Danish studies have used the median of annual minimum flows Q_{medmin}. The only fish type, which according to the UK classification system and Look Up table is considered more restrictive compared to riverine macro-invertebrates (DVFI in Danish terminology), is the –spawning and nursery reaches for Salmon”, and headwater are in general considered more vulnerable than downstream reaches.

2.4.4 Complex (investigative) environmental flow assessment methods

Complex environmental flow methods include understanding of site specific issues. Functional analysis is generally the recommended approach as it combines explicit knowledge of the hydrological and ecological system to provide a site specific solution. Building block methodologies (BBM) developed in South Africa (Tharme and King, 1998; King et al., 2000) explicitly estimate environmental flows (including minimum and high flows) related to adopted environmental flow objectives. Their basic premise is that riverine species are reliant on basic elements (building blocks) of the flow regime, including low flows (that provide a minimum habitat for species and prevent invasive species), medium flows (that sort river sediments, and stimulate fish migration and spawning) and floods (that maintain channel structure and allow movement onto floodplain habitats). A flow regime can thus be constructed by a combination of these building blocks.

The team of experts normally participating in BBM includes physical scientists (hydrologist, hydrogeologist, geomorphologist) and biological scientists (aquatic entomologist, botanist and fish biologist). They follow a series of structured stages, assess available data and model outputs and use their combined professional experience to come to a consensus on the building blocks of the flow regime (King et al., 2000; Acreman and Dunbar, 2004). BBM is used routinely in South Africa and has been applied in Australia (Arthington and Long, 1997; Arthington and Lloyd, 1998), with several functional analysis methods such as Expert Panel Assessment Method (Swales and Harris, 1995), the Scientific Panel Approach (Thoms et al., 1996) and the Benchmarking Methodology (Brizga et al, 2002).

The Flow Events Method (FEM) (Stewardson and Gippel, 2003) is another method highlighting the dynamic nature of rivers, based on a generic method for analysing the frequency of individual hydraulically-relevant flow indices under alternative flow regimes, suited for scenario analysis.

Another set of complex level approaches is based on hydraulic habitat modelling. These approaches use habitat for target species (e.g. trout), and focus on the physical aspects affected directly by changes in the flow regime (Acreman and Dunbar, 2004). Here the wetted perimeter (area of river bed submerged) related to discharge (which will be site or waterbody specific) provides the simplest index of available habitat in rivers and for environmental flow evaluation.

More detailed approaches link data on the physical conditions (water depth and velocities) in rivers at different flows (either measured or estimated from hydrological models) with data on the physical conditions required by key animal or plant species (or their individual developmental stages). Once functional relationships between physical habitat and flow have been defined, they are linked to scenarios of river flow (Waters, 1976; Bovee, 1998; Parasiewicz and Dunbar, 2001; Ginot, 1995; Killingtviet and Harby, 1994; Jowett, 1989; Jorde, 1996; Acreman and Dunbar, 2004; Dunbar, Alfredsen and Harby 2012). The methods have evolved from steady state considerations of flows to time-series analysis considering the entire flow and flow calendars.

Olsen (2010) carried out a PhD study with the aim of developing an integrated hydrology habitat assessment system, which included the capability to describe effects of hydrological changes on ecological conditions in Danish streams designed to work on large scale, making it possible to assess groundwater abstraction impact on ecological conditions on regional and national scale. The approach was a combination of a hydrological model and an instream habitat model approach which included knowledge on habitat requirements for an ecological indicator (brown trout, *Salmo trutta*) based on Habitat Suitability Index (HSI) see Fig. 2.10)

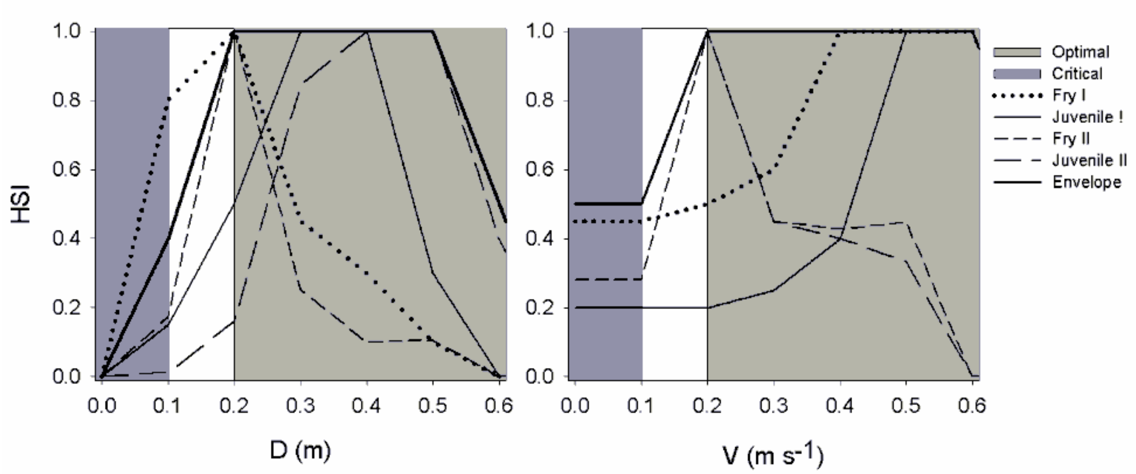


Figure 2.10 Hydrological Suitability Index (HSI) related to depth and velocity of river for different stages of fish (Fry I, Juvenile, Fry II, Juvenile II) (Olsen, 2010)

Conallin (2009) in a PhD thesis studied instream physical habitat suitability in Danish small lowland streams for brown trout (Salmon trutta) as a contribution to complex environmental flow method development in Denmark.

Based on various literature reviews Korsgaard (2006) points out some shortcomings/drawbacks of present environmental flow methodologies (Postel & Richer, 2003; Tharme, 2003; Dyson et al., 2003; Brown & King, 2003):

- Links between flow and ecosystem functions/components are often assumed and not well documented. This uncertainty is frequently used to argue against meeting recommended environmental flows.
- Focus is on minimum flow, although safeguarding of variability is equally important (the Natural Flow Paradigm).
- Focus is on instream/fluvial requirements of riverine systems, while lotic, riparian, floodplain (terrestic), estuarine, and deltaic requirements are often neglected.
- Relatively little attention is given to the requirements of maintaining morphological processes.
- Socio-economic aspects are mostly ignored.
- Validation is difficult, requires long-term monitoring using objectively verifiable indicators.
- None of the methods have been rigorously tested - there is a need for large-scale experiments.
- Habitat simulation and holistic methodologies rely heavily on expert judgements.

Agent Based Modelling can be expanded to biological process modelling and ecosystem modelling to examine trade-offs and predict results of operational changes, impact assessment, and restoration/rehabilitation of aquatic ecosystems. Here advanced modelling approaches can be incorporated, e.g. habitat modelling for quantitatively assessing environmental flows for rivers. The European COST Action 626 ―European Aquatic Modelling Network" reviewed methods and models of assessing the interactions between aquatic flora and fauna and riverine habitats on reach scale and provide transferability to a catchment scale (Harby et al., 2004)). If such methods are to be based on actual data, the collection of the data could well be expensive and time consuming, hence the current reliance on experts (Acreman and Dunbar, 2004).

Site specific evaluations require a lot of data. Complex level environmental flow methods therefore often will require additional monitoring data and coupling of hydrological models and environmental flow models, where hydrological models (and groundwater models) can predict flows throughout a river network, but getting the flow correct is not an end goal, as flow is only an intermediate variable. It is the relationships to hydraulic and habitat variables which matters the most, hence it may be better do relate groundwater models directly to response models.

Summary and evaluation of complex level environmental flow methods

Investigative methods that directly incorporate site specific data using comprehensive modelling tools can provide a more knowledge based assessment of environmental flow requirements. Hence, once validated, the investigative methods should overrule the results from screening methods. The investigative methods and tools should be selected according to the issue and river type. The tool-kit would include look-up tables for scoping and broad-scale analysis, structured expert opinion, functional analysis and physical habitat models. More detailed approaches link data on the physical conditions (water depth and velocities) in rivers at different flows (either measured or estimated from hydrological models) with data on the physical conditions required by key animal or plant species (or their individual developmental stages). Site specific evaluations require a lot of data. Complex level environmental flow methods will therefore often require additional monitoring data and coupling of hydrological models and ecological/habitat models.

2.5 Ground Water Dependent Terrestial Ecosystems (GWDTE's)

There are at least four types of situations, shown in Figure 2.11, where groundwater is essential to a terrestrial ecosystem and where GWDTEs can form. These four categories should be thought of as examples. A subset of these terrestrial ecosystems will be directly dependent on groundwater from groundwater aquifers and therefore considered during characterisation and classification (CIS, 2011):

- Type A: a groundwater source which directly irrigates the ecosystems (spring or seepage)
- Type B: groundwater collecting above impermeable strata (clay or depressions in landscape)
- Type C: high groundwater tables maintaining a seasonally waterlogged condition
- Type D: a seasonally fluctuating groundwater table flooding depressions intermittently

Most European countries monitor GWDTE‘s as part of Natura 2000, but very few countries have specific monitoring of biophysical-chemical parameters in place (monitoring is mostly project based). The same counts for Denmark, where there are project based examples of GWDTE investigations and monitoring, but not systematic monitoring sites for GWDTE‘s in operation. However, monitoring for 5-6 locations has been planned and will be initiated in 2013 (Nilsson et al. in prep.). The idea is to derive knowledge and guidelines for such monitoring based on the systematic monitoring collected at these sites, since the current knowledge is very limited.

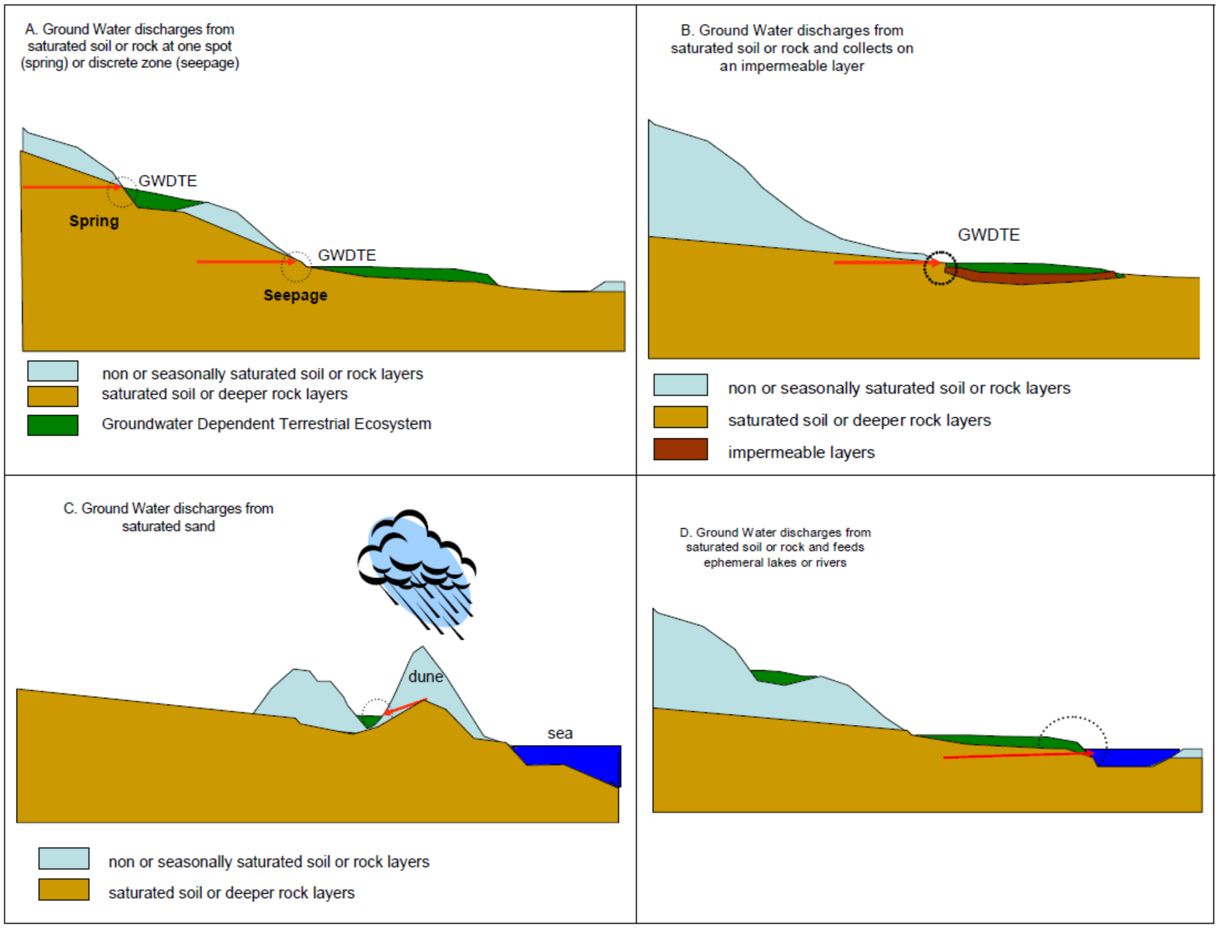


Figure 2.11 Conceptual diagram for Groundwater dependent terrestrial ecosystems - GWDTEs (CIS 2011)

An example of project based monitoring of GWDTEs from Denmark (bl.a. målsatte rigkær) is shown in Figure 2.12, where water level monitoring stations in GWDTEs and in wetlands were planned by Aarhus Vand in an area upstream Spørring in the wider Aarhus area in Jylland, as part of monitoring impacts from a new groundwater abstraction of 1.5 mio m^3/year (even though the project was not finally established the example illustrate a system oriented approach for wetland re-establishment and monitoring as part of establishment of a new groundwater abstraction wellfield).

Locations of water level monitoring stations are in Fig. 2.12 marked with M with the purpose for the northern station to assure that the will not be any degradation of the ecological goals for the fens, even though this area is located within an area with model estimated drawdown in groundwater level from 25 cm to 1 meter (red dashed curve), caused by new well and abstraction of 1.5 mio m^3/year from southwestern well (NØ of Todbjerg, yellow dot). The two monitoring stations to the south has the purpose of evaluating impacts on small wetlands (vandhuller). Bold curve (in red) show areas with drawdown above 1 meter.

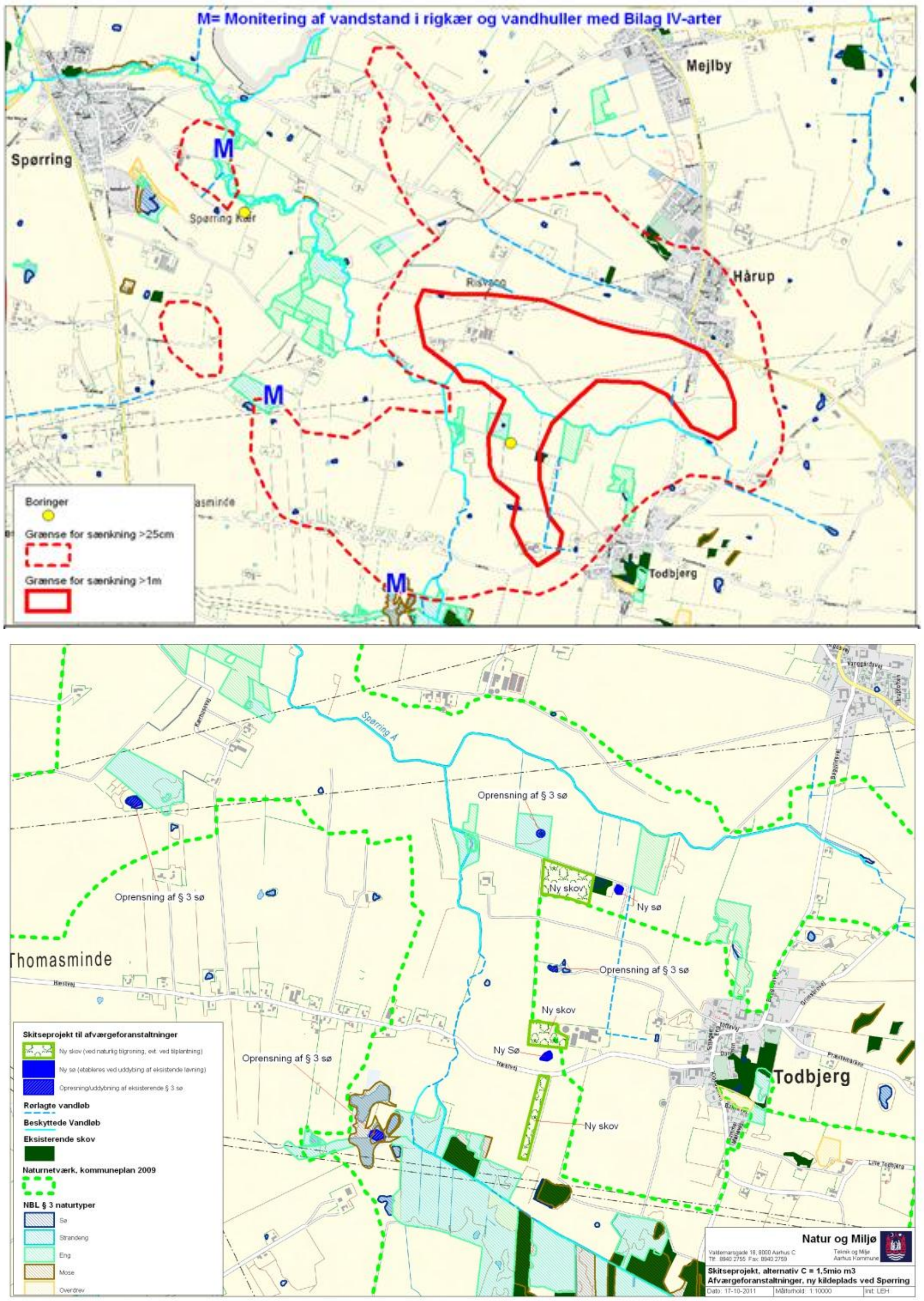


*Figure 2.12 **Upper**: Example of monitoring water levels in a fens (in Danish: rigkær) and in two wetlands. **Lower**: Remediation actions e.g. new forests (three areas west of Todbjerg), new/or cleaned-up lakes (blue areas/dots not shown on upper figure) are other remediations (Source Aarhus Vand).*

3. Practice in other EU countries

3.1 CIS guidance document 18 (Groundwater status and trend assessment)

CIS (2009) indicates that groundwater level should be the principal parameter for assessing good quantitative status. However, whilst the monitoring of water levels is essential to determine impacts and identify long-term trends, it is insufficient on its own and other parameters and information will generally be needed.

3.1.1 Definition of groundwater quantitative status tests

The definition of good quantitative status is set out in WFD Annex V 2.1.2. For a groundwater body (GWB) to be of good chemical and quantitative status each of the criteria covered by the definition of good status must be met (see Fig. 3.1).

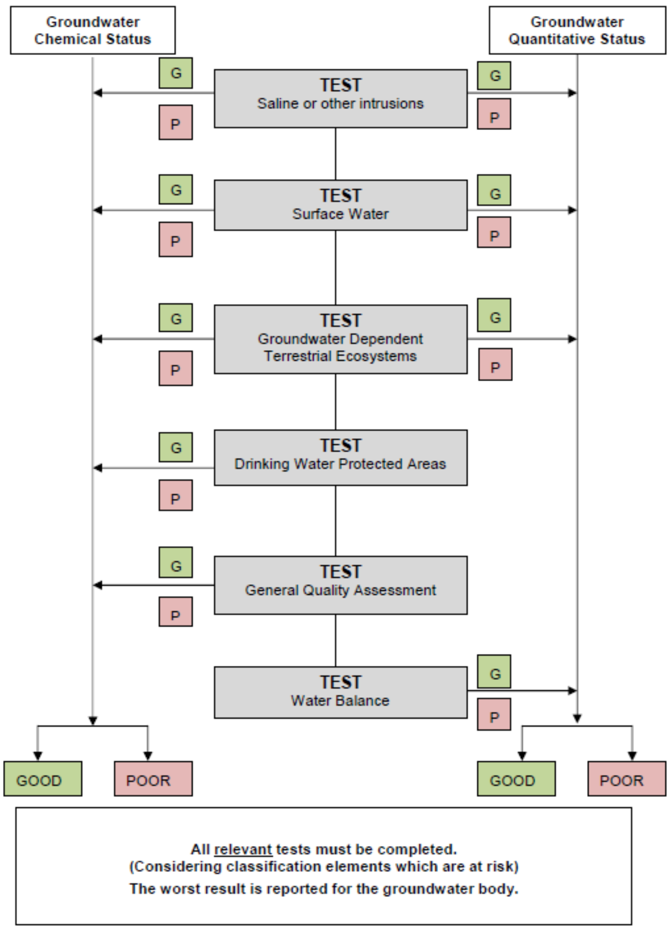


Figure 3.1 Overall procedure of classification tests for assessing groundwater status (CIS, 2009)

Article 4.1(b-ii) states that Member States shall ―…ensure a balance between abstraction and recharge of groundwater with the aim of achieving good groundwater status by 2015”. Each of the tests for groundwater chemical and quantitative status should be carried out independently and the results combined to give an overall assessment of groundwater body chemical and quantitative status. This means that for being evaluated as an aquifer having a good quantitative status, each of the four quantitative tests have to meet the requirements, in order to result in the overall good quantitative status. If any of the tests results in a poor quantitative status, the overall classification of the body will be poor (same yields for chemical status, for the five chemical status tests).

In this report the focus is on the quantitative status of groundwater bodies (aquifers). In the case of quantitative status the following four tests can be briefly summarised (CIS 2009):

a) **Test 1: Saline (or other) intrusion.** For a GWB to be of good status there should be no long-term intrusion of saline (or other poor quality water) resulting from anthropogenically induced sustained water level or head change, reduction in flow or alteration of flow direction due to abstraction.

b) **Test 2: Groundwater Dependent Terrestrial Ecosystems (GWDTE).** For a GWB to be of good status there should be no significant harm to a terrestrial ecosystem that depends on groundwater. The GWDTE tests for both chemical status assessment and quantitative assessment are closely linked. This test requires that the environmental condition required to support and maintain conditions within a GWDTE (e.g. flow or level needed to maintain dependent (plant) communities) are determined. If the conditions are not being met and groundwater level and flow change due to abstraction is determined to be a significant cause, then the GWB is of poor status. In all other cases the GWB will be of good status but potentially at risk (CIS 2011).

c) **Test 3: Surface Water Flow TEST**. For a GWB to be of good status there should be no significant diminution of surface water chemistry or ecology that would lead to a failure of Article 4 surface water objectives relating to surface water bodies. This test includes both river and open water bodies such as lakes to which WFD surface water objectives apply. It requires that environmental flows or water level requirement of surface water bodies (associated with GWBs) needed to support achievement (and maintenance) of good chemical and ecological status is determined. If this flow/level requirement is not being met as a result of a significant impact from groundwater abstraction, then the GWB will be of poor status unless the surface water body remains of good/high ecological status. Under any other circumstances the GWB will be of good status.

d) **Test 4: Water Balance TEST**. For a GWB to be of good status, long-term annual average abstraction from the GWB must not exceed long-term average recharge minus the long-term ecological flow needs (figure 3.2). For the water balance test we must assess annual average abstraction against ‗available groundwater resource‘ in the groundwater body. The available groundwater resource means the long-term annual average rate of overall recharge to the body of groundwater minus the long-term annual rate of flow required to achieve the ecological quality for associated surface waters (specified in Article 4). This is in order to avoid any significant diminution on the ecological status and avoid any significant damage to groundwater dependent terrestrial ecosystems (GWDTE). According to the CIS document, both the surface water and GWDTE environmental flows, and the impacts of groundwater abstraction on low flows must be determined.

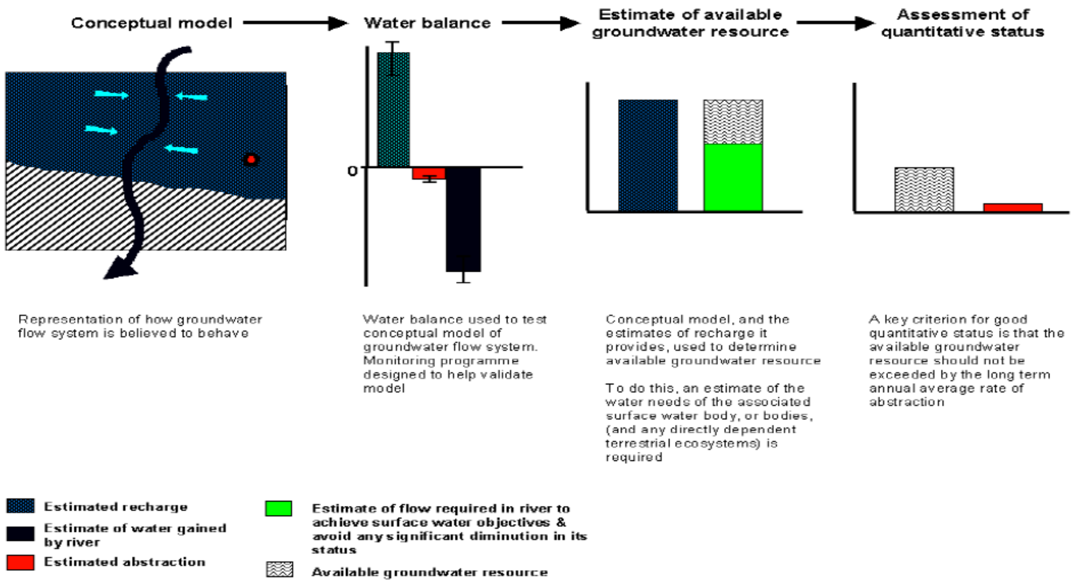


Fig. 3.2 Role of environmental flows in the water balance test (CIS, 2009).

Each relevant test (considering classification elements which are at risk) should be carried out independently and the results combined to give an overall assessment of groundwater body chemical and quantitative status. The first three tests, saline intrusion, GWTDE and surface water flow integrate chemical status with quantitative status, whereas the water balance test is the only one which only considers the quantitative status. Here, especially the significant time scale of groundwater processes related to water quality should be remembered in order to achieve appropriate long term and precautionary assessments of aquifer safe yield and environmental flow requirements that foresee the significant delay in the signal from the groundwater monitoring data.

As seen above, in 3 of the 4 tests for quantitative sustainability it is necessary to determine environmental flows. This gives an idea of the importance of environmental flows, when assessing the quantitative status of water bodies. In the following subsection we will try to unfold more specifically how the different components, e.g. recharge, is estimated according to CIS (2009), and the different requirements for environmental flow in order to assess available groundwater resource. A key criterion (Fig. 3.2) for good quantitative status is that the available groundwater resource should not be exceeded by the long term average annual rate of abstraction.

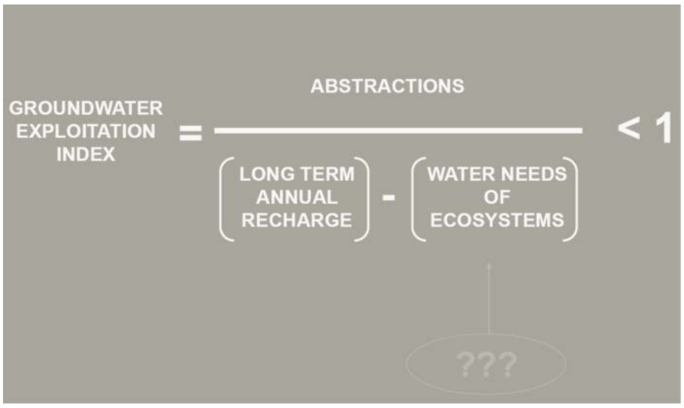


Fig. 3.3 Another expression of groundwater exploitation index according to environmental flow requirements (Sanchez, 2012).

In case the groundwater exploitation index is = 1, then the abstraction corresponds to the ‚available groundwater abstraction‘. Below 1 this indicator signifies that the quantitative status is acceptable, e.g. that abstractions are below available groundwater resources.

3.1.2 Estimate of recharge and environmental flow requirements in order to assess available groundwater resource

Water balance test

The available groundwater resource is an approximate value, based on recharge and the low flow requirements to support the ecology in surface water bodies and terrestrial ecosystems that are dependent on the groundwater body (CIS, 2009). Because the tests described in CIS (2009) are groundwater body wide tests, it may not always be possible to clearly define the local flow needs of rivers and wetlands. Furthermore, the available groundwater resource may not be available for abstraction due to hydrogeological conditions (e.g. transmissivity and storage), that make abstraction difficult economically and practically, or because the distribution of the available resource across the groundwater body varies in relation to sensitive receptors. Therefore, status assessment will need to take this into account and in many cases the poor status boundary will not simply be where abstraction > 100 %. Available resources could in some cases be much lower. In some hydrogeological situations it could be as low as 20 % (CIS, 2009).

It is pointed out that the annual average recharge should be estimated for the whole of the groundwater body including any recharge water deemed to enter the groundwater body from outside (e.g. run off from adjacent impermeable strata). Furthermore, the annual average abstraction rate should include all abstractions from the groundwater body, including any connected confined sections of the aquifer. These abstractions may include evaporation from large open bodies of water, e.g. gravel pits and artificial ground drainage systems. The decision on whether to discount abstracted groundwater that has been locally returned to the aquifer or to a river (during irrigation or at a quarry dewatering operation) should be based on a hydrogeological assessment, taking account of body-wide impacts.

The consideration of long-term abstraction, recharge or water level is to minimise the influence of short-term natural climatic factors and abstraction impacts. Long-term measures allow short-term natural climatic factors and abstraction impacts. For the purposes of the WFD the required length of record will depend on the hydrogeological and environmental conditions associated with the groundwater bodies. It is recommended that as a minimum it should be no less than 6 years (one river basin management cycle).

Both the surface water and groundwater dependent terrestrial ecosystems (GWDTE) ecological flow requirements and the impacts of groundwater abstraction on low flows must be determined. The methods used can depend on the degree to which abstraction pressures affect the groundwater body. This may be by use of either local technical knowledge, simple tools or more sophisticated models.

Where there is flow (lateral or vertical) between adjacent groundwater bodies and other hydrogeological systems this will need to be accounted for when carrying out the water balance test. In some cases these flows may be inflows (recharge) and other cases outflows. Alternatively, groundwater bodies can be grouped to simplify the water balance assessment.

Test surface water flow
For a groundwater body to be of good status there should be no significant diminution of surface water chemistry or ecology that would lead to a failure of Article 4 surface water objectives. The surface water flow test includes both river and open water bodies such as lakes to which WFD surface water objectives apply. Unlike the previous test this test considers whether, at a local scale, the pressures from groundwater abstraction are having a significant effect on individual surface water bodies once all the different pressures on the surface water body(ies) are taken into account. Depending on the delineation of water bodies a groundwater body may contain many different surface water bodies each with their own objectives. This test requires that the flow requirement or water level requirement of surface water bodies (associated with GWBs) needed to support achievement (and maintenance) of good chemical and ecological status is determined. Note that for rivers impacts of groundwater abstraction may be seen as a reduction in flow and in open water bodies a reduction in level (CIS, 2009). If this flow/level requirement is not being met as a result of a significant impact from groundwater abstraction, then the groundwater body will be of poor status unless the surface water body remains of good/high ecological status. Under any other circumstances the groundwater body will be of good status.

CIS (2009) states that it is often not possible to accurately make precise measurements of the reduction in flow/or level caused by groundwater pressures. There is often a time lag between the abstraction pressure occurring and the impacts on the surface water body due to the variability and response of hydrogeological systems. A failure to meet the required environmental flow/level requirements in any surface water body may also be due to either groundwater or surface water abstractions.

Test groundwater dependent terrestrial ecosystems (GWDTE)
For a groundwater body to be of good status there should be no significant damage to a terrestrial ecosystem that depends on groundwater. The GWDTE tests for both chemical status assessment and quantitative assessment are closely linked. The environmental condition required to support and maintain conditions within a GWDTE (e.g. flow or level needed to maintain dependent (plant) communities) has to be determined by this test. If the conditions are not being met and groundwater level and flow change due to abstraction is determined to be a significant cause, then the groundwater body is of poor status. In all other cases it will be in good status but potentially at risk. As part of initial and further characterisation, a screening exercise should have been carried out to identify all GWDTEs that are damaged (or at high risk of damage) as a result of groundwater pressures. This assessment should have been made on the basis of criteria such as ecological indicator communities, likely connection to the groundwater body, proximity to anthropogenic pressures supported by local knowledge and site condition reports. Only sites identified as being currently ‗at risk‘ will need to be considered in the status assessment. The presumption being that GWDTEs ‗not at risk‘ will not lead to a groundwater body being of poor status. For many sites, it will not be possible to quantify supporting conditions required within the GWDTE with a high degree of confidence. This is because sufficiently detailed site-specific information may not be available for all sites. Under these circumstances the groundwater body will be of good status for this test and the results of initial risk screening and any other available evidence should be used to decide if sites are considered ‗at risk‘. These ‗at risk‘ sites should be prioritised for further investigation (CIS, 2009).

Test saline or other intrusion
For a groundwater body to be of good status there should be no long-term intrusion of saline (or other poor quality water) resulting from anthropogenically induced sustained water level or head change, reduction in flow or alteration of flow direction due to abstraction (long-term saline intrusion may also occur even without an alteration in flow direction). Due to the density differences between saline water and freshwater, a reduction in water levels (or head) may on its own lead to saline intrusion. A decrease in hydraulic gradient towards the source of saline water and corresponding decline in groundwater flow all permit saline intrusion to occur before the decrease in water levels is sufficient to produce a change in flow direction. Intrusion is interpreted in this test as intrusion of poor quality water from another water body into a groundwater body (Annex V 2.3.2) rather than movement of a plume of poor quality water within the body. The source of intrusion may be from a water body above, below or alongside the body for which status is being assessed.

This test is combined with the chemical status test for assessing saline intrusion. When making the assessment, consideration should be given to the historical long-term impacts of abstraction particularly in confined aquifers and aquifers with low recharge rates. Historical pumping may have resulted in significantly lowered groundwater levels or piezometric heads (e.g. by hundreds of meters) due to over abstraction but the abstraction has since been reduced to sustainable levels, in terms of a current balance with recharge rates. In these cases, although a water balance may indicate that the available resource is not exceeded continuing intrusion may be taking place and groundwater quality may continue to deteriorate. Where the intrusion is into the body, the saline intrusion test should be applied.

Where anthropogenically altered water levels are leading to geochemical changes within the groundwater body itself and these lead to deterioration of water quality within the body, then where these changes are significant and could potentially lead to an exceedance of a threshold value (or quality standard) or other relevant WFD objective, they should be considered under the chemical status tests. An example of this may be oxidation of groundwater or other geochemical change in a previously confined aquifer caused by over abstraction leading to the mobilisation/release of contaminants. The management of groundwater abstractions to maintain conditions that minimise the potential for status failure due to anthropogenically induced geochemical changes will form part of a Programme of Measures for that groundwater body (CIS, 2009).

3.2 Review reports on WFD implementation prepared by DG ENV

3.2.1 Commission Report

This Commission implementation report is required by WFD article 18 and is based on the Commission's assessment of the RBMPs reported by Member States. It is accompanied by Commission Staff Working Documents (Sub-section 3.2.1) that include a detailed assessment of the RBMPs. It is one of the bases of the Commission Communication on the *'Blueprint to Safeguard Europe's Water Resources' (EC, 2012)*

The third implementation report was published 14.11.2012 to the European Parliament and to the Council. It includes among other things a review of progress in the implementation of the Directive and a survey of the River Basin Management Plans submitted in accordance with Article 15, including suggestions for the improvement of future plans

The Commission Report identified that the main obstacles encountered in each Member State are hydromorphological pressures, pollution and over-abstraction (EC, 2012).

The Commission Report recommends to Member States to:

- Apply ecological flow regimes to ensure that authorities and users know how much water and which flow regime is needed to achieve the objective of good ecological status;
- Improve datasets on water quantity, water availability and demand trend projections to be able to develop coherent and effective sets of measures;
- Integrate climate change considerations into the RBMPs;
- Coordinate the preparation and consultation on the Flood Risk Management Plans with the second RBMPs to ensure coherence.

3.2.2 Commission staff working document and country reports

The commission staff working document about the quantitative status stated that (EC, 2012b): ~~In~~ total 56% of all RBMPs reported that the definition of available groundwater resource‘ was fully or partly applied in accordance with Article 2.27 WFD. Therein, ‚available groundwater resource‘ is defined as the long-term annual average rate of overall recharge of the body of groundwater less the long-term annual rate of flow required to achieve the ecological quality objectives for associated surface waters specified under Article 4, to avoid any significant diminution in the ecological status of such waters and to avoid any significant damage to associated terrestrial ecosystems. For the remaining 44% of RBMPs the respective information was not found or rather unclear”.

„A bit more than half of the RBMPs reported that the balance between recharge and abstraction of groundwater was assessed in order to verify whether the available groundwater resource is exceeded. In the remaining 46 RBMPs (43%) the respective information was not found or rather unclear. The methodologies described very often compare the abstractions with the recharge (considering a safety margin) others conclude from stable groundwater levels to an appropriate balance between recharge and abstraction, while some Member States combine both assessments. Ecological flow needs were frequently mentioned to be considered in the assessments” (EC, 2012b).

„Overall, about 74 % of the groundwater bodies (representing 63% in terms of area) were reported to be both in good chemical and quantitative status in 2009, which is expected to increase to 80% in 2015 (representing 68% in terms of area). About 80% of the groundwater bodies were reported to be in good chemical status in 2009, but nearly 2,000 groundwater bodies were reported to be still in poor chemical status and for 5% of the groundwater bodies the status is still unknown. Poor status is mainly caused due to the exceedance of groundwater quality standards or threshold values affecting nearly 12% of all groundwater bodies in 21 Member States and the main responsible pollutant is nitrate. About 87% of the groundwater bodies were reported to be in good quantitative status in 2009, but nearly 800 groundwater bodies were still reported to fail good quantitative status, mainly 171 due to the exceedance of the available groundwater resource by the long-term annual average rate of abstraction. For 7% of the groundwater bodies the status is still unknown. Although quite high percentage of groundwater bodies are considered to be in good status the methodologies used show significant shortcomings that puts in question the results of the status assessment”, EC (2012b).

EC (2012b): „It is not clear in a lot of cases whether – besides theoretical considerations - associated surface waters and groundwater dependent terrestrial ecosystems were practically included in the groundwater status assessment. Environmental quality standards and threshold values were only reported to be considered for aquatic ecosystems but not for terrestrial ecosystems. Member States reported a considerable lack of knowledge in assessing the needs of terrestrial ecosystems and the interaction between groundwater and these ecosystems”.

„Regarding groundwater quantitative status the methods for calculating groundwater recharge, abstraction and their balance as well as available groundwater resource are different in Member States and in a number of cases those methods are not transparent. It is also not clear whether associated surface waters and groundwater dependent terrestrial ecosystems were included in the assessment in practice. 15 Member States reported having transboundary groundwater bodies, but only 9 of them reported explicitly on the co-ordination of the establishment of their threshold values with all (7 Member States) or at least with some (2 Member States) of the neighbouring countries" (EC, 2012b).

EC (2012b) has the following recommendations about the (miserable transparency and coherence of the) quantitative status:

- Reliability of the status assessment should be improved by extended monitoring and by correctly applying all the required elements of status and trend assessments.
- RBMPs should clearly address all elements specified in the WFD related to both the good chemical and the good quantitative status of groundwater. RBMPs should clearly report the reasons for not considering certain elements.
- Groundwater dependent ecosystems and groundwater associated surface water bodies should always be considered. Member States should take the opportunity of sharing and exchanging experience gathered so far regarding the interconnections between groundwater and the ecosystems and regarding the needs of the ecosystems e.g. in the frame of the Common Implementation Strategy of WFD. Knowledge gaps need to be filled with appropriate studies to inform the RBMP process.
- The definition of ‗available groundwater resource' according to Article 2.27 of the WFD should be fully applied and reported.
- Methodologies to calculate the balance between recharge and abstraction of groundwater should be transparent and better harmonised between Member States. Ecological flow should be considered.
- Information on the status of drinking water protected areas should be included in the RBMPs.

In relation to *drought and water scarcity* the Commission evaluates that (EC, 2012b): ―water quantity can have a strong impact on water quality and therefore on the achievement of good ecological status. Hence quantitative requirements are implicit in the definition of good ecological status and explicitly through the inclusion of flow regime as a supporting hydromorphological element. Good quantitative status is required for groundwater; a balance between abstraction and recharge must be ensured. Furthermore, groundwater levels should not be subject to anthropogenic alterations that might have impacts on surface waters and groundwater dependent ecosystems".

Regarding water scarcity and drought the Commission concludes the following of relevance to water quantity and sustainable exploitation (EC, 2012b):

- Water quantity issues are not sufficiently addressed in the RBMPs, the quantitative datasets are incomplete in many plans, and they are insufficient for pro-active planning. Water demand and availability trend scenarios were not identified in most of the plans.
- The influence of other sectoral policies on the reduction of water scarcity and the mitigation of drought effects is not sufficiently addressed.

About adaptation to climate change EC (2012b) and water quantity the requirements include:

- Assessing direct and indirect (primary and secondary) climate pressures in order to provide information for the pressures analyses.
- Assessing monitoring programmes to ensure early climate impact signal detection.

EC (2012b) gives the following recommendations to how to include climate change in next round:

- Almost all the elements, which are included in the definition of WFD qualitative and quantitative status, are sensitive to climate change. Therefore it is recommended to consider climate change in water management at an early stage. Planning should consider a time period that is longer than the RBMP six-year cycle.
- Use CIS guidance document No. 24 River Basin Management in a Changing Climate as a reference for the activities in the second and third RBMP cycles.
- Member States are requested to demonstrate how climate change is considered in the assessment of pressures and impacts, monitoring programmes and appraisal of measures (climate checking of PoMs) from the second RBMP cycle.

3.2.3 Country report summary about quantitative status

Based on the third implementation report (EC, 2012) a summary of country reports has been drafted (see Table 3.1) explaining the extent of incorporating the four different groundwater quantitative status tests (saline intrusion test, GWDTE test, surface water flow test and water balance test). Information is included on the basis for assessment, the main reason for poor quantitative status, the number of groundwater bodies that fail to pass the tests, and the extent of unknown quantitative status. Finally, a column illustrates the mean size of groundwater bodies for each country (km^2). With green shade, countries which are most promising for comparison with Denmark are shown. This is UK, Germany, Ireland and France, see Table 3.1.

Some countries like Denmark, Ireland, Sweden and Finland have relatively small mean sizes of delineated groundwater bodies (GWB) in the range of 100-150 km^2. Other countries operate with generally larger GWBs, like Germany, UK, Belgium, Slovak Republic, Czech Republic and Norway in the range of 250-400 km^2. Yet other countries like France, Spain, Poland, Romania, Netherland and Slovenia have relatively large GWBs in the range of 800-2000 km^2. Finally, Latvia and Lithuania operate with very large GWBs above 2000 km^2.

Some countries have done all the four quantitative status tests, others only some of them. Most countries have done the water balance test, and in many countries the main reason for not fulfilling the quantitative status tests is reported as a current groundwater abstraction exceeding the available groundwater resource. However, the basis for this assessment and for the analysis of environmental flow requirements and groundwater recharge leading to the available groundwater assessment is rather difficult to evaluate in depth. In the next section we will try to go in depth with the assessment of quantitative status in four selected countries.

Table 3.1 Overview of results from 3rd implementation report (EC, 2012)

Country	Saline Intrusion test	GWDTE test	Surface Water flow test	Water Balance Test	Basis for assessment	Main reason for poor quantitative status	Fail quantitative status DWPAs Fail / total	Unknown quantitative status	Mean size of GWB (country Area/No. Km^2
Austria									
Belgium 13,5 tkm²	?	Mentioned In RBMP	Mentioned in RBMP	YES Described Flemish	7 Assessment Criteria, but No info	Abstr. > Avail.gw.res. (level trend)	14/42 Flemish		~320 (Flemish)
Bulgaria 111 tkm²	YES (Black sea)	NO	Information not available	YES	National approach (abs. versus avai.gw)	?	7/177 DWPAs: 52/109		~627
Cyprus 9,2 tkm²	YES Most aquifers	NO	List of Interactions mentioned	YES, but Avail. Gw. Not clear.	?	Saltwater Intrusion by Overpump.	15/20 DWPAs: 5/13	1~ 5 % of tot	~460
Czech Rep. 79 tkm²	?	YES	?	YES	Long term Abs.<recharge +GWDTE	Abstr.> Avail.gw.res.	33/173 DWPAs: 126/157		~460
Germany 357 tkm²	Considered as far as relevant	Considered as far as relevant	Considered as far as relevant	YES Described In WISE	Long term abs.<recharge + H lev. trend	Abstr. > Avail.gw.res.	38 / 989 DWPAs: 0/870		~360
Denmark 42,9 tkm²		YES, but insufficient data	YES	YES	GIS layers, + DK Novana or WBAL eq.	Abstr. > Avail.gw.res.	136/385 DWPAs: 222/368		~111
Estonia 45,2 tkm²	?	NO	NO	YES	No info	Oil-shale drainage	1/26		~1738
Greece 132 tkm²	NO	NO	NO	NO	?	?	?		?
Spain Catalonia 32,1 tkm²	YES	NO	?	YES	IMPRESS, Abs.<avail.res. + Saline intrui.	Abstr.> Avail.gw.res. +20%Chlorin	6/39 DWPAs: All OK		~823 (Catalonia)
Finland 371 tkm²	YES, Åland	YES	?	YES	Balance abs.<availab. Res.+trend	All OK	0/3804 DWPAs: 0/3804	70~ 2 % of tot	~98
France 550 tkm²	?	YES Lack of Appr.meth.	YES	YES	Available knowledge in different dist.	? New method in 2012	199/574 DWPAs: 65/187		~958
Hungaria 93,0 tkm²	?	YES	?	YES For group of GWPs	Water balance test	Abstr.> Avail.gw.res.	? /185 DWPAs: 15/1739		~1511
Ireland 70,0 tkm²	YES	YES	(NO) Lack info Ecol.flow	YES	Abs.<avail.res. National appr. (risk report)	Abstr.> Avail.gw.res.	4/650 DWPAs 2/731		~108
Italy 300 tkm²	?	NO	?	YES	Abs.<avail.res.	Abstr> Avail.gw.res.	115/733 DWPAs 0/689	232 ~ 32 % of tot	~409
Lithuania 65,0 tkm²	YES	YES	YES (gw modelling)	YES	Statistical analysis + modelling	No problems	0/20	DWPAs 1305 unkno.	~3250
Luzem-Bourg 2,5 tkm²	?	NO No risk/not considered	NO No risk/not considered	YES	Assessment from rainfall, Abs. and piez.	No quantitative risks	0/5	DWPAs 82 unkno.	~500
Latvia 64,6 km²	YES Venta	YES	YES	YES	Not clear how assessment was done	Not considered significant	0/22	No data on DWPAs	~2936
Malta 0,3 tkm²	YES	NO	NO	YES	Simple balance (uncertain)	Abstr.> Avail.gw.res.	4/15 DWPAs 6/7		~43
Nether-Land 41,5 tkm²	YES	YES	YES	YES	Abs.<avai.res.: rech>abs+drai Trend analysis	No trend+ Rech> Abs+ Drainage fl.	0/23 DWPAs 16/16		~1804
Norway 323 tkm²	YES	YES Not clear if Implem.	YES	YES	264 out of 1275 assessed Trend in level!	No assessm. of recharge Or avail.res.	0/1275		~253
Poland 313 tkm²	YES	NO	YES	YES	National appr.	Abstr.> Avail.gw.res.	29/161	DWPAs 145 unkno.	~1944
Portugal 92,1 tkm²	?	?	?	?	NO info on GWB No reporting.	?	?	?	?
Romania 238 tkm²	YES	NO Only qualitative	NO Only qualitative	YES But no avail.res.	Water bal. Trend levels. Abs.<Nat.rech.	No problem	0/142	DWPAs 1423 unkno.	~1676
Sweden 453 tkm²	YES	NO	NO	YES	Only 4 % monitored	98 % in good status?	5/3021 DWPAs 0/856	391 unknown	~150
Slovenia 20,7 tkm²	YES	YES	YES	YES	Abs.< avail.gw.res. Water bal.	No problem	0/21 DWPAs 0/1377		~986
Slovak Re 48,9 tkm²	YES	YES	YES	YES	Abs.< Avail.gw.res.		0/131 DWPAs. 0/170		~373
United Kingdom 245 tkm²	YES	YES	YES	YES	Regional app. Abs.< Avail.gw.res.	?	150/723 DWPAs 56/722		~339

3.3 Practice from selected countries

3.3.1 Practice in England and Wales

Around 20 % of the GWBs in England and Wales are in poor quantitative status. The assessment of groundwater status generally follows a regional approach, with separate methodologies in England and Wales, Scotland and Northern Ireland. The impacts of abstraction have been considered by looking at the balance between long term annual average rate of abstraction compared with the available groundwater resource. Saline or other intrusions, surface waters associated to groundwater and GWDTEs were included in the assessment of quantitative status (EC, 2012).

The groundwater quantitative status assessment for the first round of WFD has been described by UKTAG (2007) and Environmental Agency (EA, 2010) for the four tests. Furthermore, the UK Technical Advisory Group on the Water Framework Directive has developed working drafts for groundwater quantitative classification for the second planning cycle (UKTAG, 2012).

Water balance test
The assessment for first round was carried out in two stages: (1) compare the total abstracted groundwater (ignoring any locally returned water) to the long term average actual recharge total groundwater body resource, and (2) compare impacts of abstraction on low flows (taking into account any locally returned water) to the aggregated naturally available low flow resource which is based on the flow screening standards for all the dependent surface water bodies draining the groundwater body.

Firstly, the long term average actual recharge to the groundwater body is compared with the long term average groundwater abstraction from it. In accordance with the WFD, groundwater bodies, where abstraction exceeds recharge, would be classified as POOR quantitative status (with HIGH confidence). The second stage of the test compared the impacts of groundwater abstraction on low flows to the naturally available low flow resource. The naturally available low flow resource was defined as the natural flow minus environmental flow needs for all of the surface water bodies draining the groundwater body.

If abstraction impacts exceeded the naturally available low flow resource the abstraction pressures exceed limits set by the requirement to protect environmental flows, so the groundwater body is considered to be at POOR quantitative status (LOW confidence). By focusing on the low flow abstraction limits of dependent rivers water bodies, it is a more critical test than the comparison of abstraction to recharge, and results in the status failure of many of the principal aquifer groundwater bodies.

The second part of the groundwater body balance test differs from the separately described ‗Dependent Surface Water Body Status' test in two important aspects:

- it is calculated at the groundwater body scale – aggregating together the natural available low flow resource from all dependent surface water sub-catchments and comparing this with the low flow impacts of all abstractions from the groundwater body; and

- it ignores the impact of surface water abstractions or discharges which may also influence surface water body flows and status.

If both parts of the test are passed, the groundwater body is assigned a GOOD Status result. The level of confidence in the GOOD Status result (HIGH or LOW) reflects the degree to which both parts of the test have been passed when considering the recent actual abstraction scenario. Particular focus was given to groundwater bodies where recent actual groundwater abstraction is a high proportion of recharge (above 50%). Based on a review, the confidence in the POOR Status result was changed from LOW to HIGH in groundwater bodies where there are well documented issues associated with groundwater abstraction - for example, known low flow problems, see Table 3.1.

Review of the results in some of the groundwater bodies with a high abstraction to recharge ratio suggested that the initial result may not be conceptually credible. In some cases the outcome highlights the need to re-consider the delineation of groundwater body boundaries which have been based on simple surface water divides and unreasonably separate a relatively small area off from the remainder of the aquifer. In other cases, the allocation of confined groundwater abstractions to unconfined groundwater bodies appears too simplistic as it ignores leakage from overlying strata. For these reasons, for the first river basin planning cycle, these groundwater bodies have been assigned a POOR Status (LOW Confidence) result (EA 2010).

Table 3.1 Outline summary of the groundwater body balance test in UK (EA 2010)

Status	Confidence	Criteria
Good	High	Groundwater abstraction does not exceed recharge. Groundwater abstraction impacts are less than the aggregated natural low flow resource.
Good	Low	Recent actual groundwater abstraction does not exceed recharge. Fully licensed groundwater abstraction exceeds recharge. Recent actual groundwater abstraction impacts are less than the aggregated natural low flow resource. Fully licensed groundwater abstraction exceeds the aggregated natural low flow resource.
Poor	Low	Recent actual groundwater abstraction does not exceed recharge. Fully licensed groundwater abstraction exceeds recharge. Recent actual groundwater abstraction exceeds the aggregated natural low flow resource. Fully licensed groundwater abstraction exceeds the aggregated natural low flow resource.
Poor	High	Recent actual groundwater abstraction is high or exceeds recharge and Area Staff confirm that there are known issues associated with groundwater abstraction. Recent actual and fully licensed groundwater abstraction exceeds the naturally available low flow resource and Area Staff confirm that there are known issues associated with groundwater abstraction.

The recharge and abstraction rate data underpinning the groundwater body resource balance test in UK were initially nationally derived and have been included in both the 2007 quality review and subsequent Catchment Abstraction Management Strategy (CAMS) process by Environment Agency operational staff (EA, 2012). In some cases improved local estimates of recharge and/or revised groundwater abstraction rate information were provided to add confidence to the assessment results. However, the review process has not provided comprehensive comments for all groundwater bodies.

Estimates of low flow groundwater abstraction impacts used in the second part of the test are very simple. Any non-consumptive proportion of abstracted water locally returned to the catchment is accounted for, but low flow impacts are otherwise generally assumed to be the same as the long term average recent actual abstraction rate. In reality aquifer flow, storage and river interaction mechanisms may often lead to a reduction in low flow impacts when compared to the average abstraction.

The estimates of naturally available low flow resources are based on the Environmental Flow Indicators (EFIs) adopted for surface water flow screening and are therefore subject to the same significant uncertainty regarding the relationship between flow and the ecological status it helps to support. Further refinement of these results should therefore be expected as the assessments are re-visited during the first river basin planning cycle - as part of the CAMS (EA, 2012). For the most recent descriptions of EFIs in relation to the Water Framework Directive and CAMS see Appendix 2.

Groundwater abstraction related deterioration of dependent surface water body status
This test considers the impact on ecological status of surface water bodies. The test seeks to establish whether groundwater abstraction could be resulting in a deterioration of the environmental flow (EFIs) as a regulatory threshold and to allow the screening of abstraction pressures. If there is a failure of surface water body EFIs which is significantly attributable to upstream groundwater abstraction impacts, then the groundwater body upon which both the abstractions and surface water flows depend, should be flagged as being at risk of failing to achieve its groundwater quantitative status objectives. If this is currently true (based on the ‗recent actual‘ abstraction and discharge scenario), the supporting groundwater body is classified at POOR quantitative status. Otherwise the result is GOOD quantitative status.

However, there is uncertainty about the link between the EFI and ecological status, particularly in terms of the impact of abstraction on biological elements such as fish and macroinvertebrates. Therefore, the results of a failure of this test normally results in a POOR Status (LOW Confidence) groundwater body classification. In a small number of groundwater bodies evidence that groundwater abstraction is causing adverse impacts on surface water bodies, HIGH Confidence in the POOR Status classification can be the result.

If a significant potential failure in surface water flow standards related to groundwater abstraction might occur in the future (based on the precautionary ‗full licensed‘ scenario), the groundwater body is assessed as being ‗at risk‘ of failing its ‗no deterioration‘ objective, see

Table 3.2. The test involves calculation at the surface water body scale which is then mapped onto the groundwater bodies on which they partly depend. The aim is to flag-up those river or lake water bodies which are at risk with respect to groundwater abstraction pressures, i.e. where there is a surface water flow ‗deficit‘, ii) upstream groundwater abstraction impacts are ‗a significant part‘ (> 50 % of the average low flow naturally available resource), and iii) groundwater abstraction pressure within the water body sub-catchment is a significant proportion (currently assumed to be 20% of this total upstream groundwater abstraction pressure). The total ‗low flow‘ groundwater abstraction impact within the upstream catchment is estimated as the average of the Q95 and Q70 values taking into account any water locally returned, ignoring non-consumptive abstractions. The naturally available low flow resource is defined as the natural flow minus EFIs (environmental flow indicator). Hydromorphological elements, including flow, help to support surface water body Good Ecological Status rather than defining it. Relationships between flow and ecology remain poorly understood and locally variable, and the EFIs can therefore only be considered to represent a nationally consistent and precautionary screening tool (UKTAG, 2007). In reality aquifer flow, storage and river interaction mechanisms may often lead to a reduction in low flow impacts when compared to the average abstraction.

Table 3.2 Outline summary of the dependent surface water body status test (EA 2010)

Status	Confidence	Criteria
Good	High	EFIs (supporting Good Ecological Status) met at all flows and in all scenarios (RA, FP and FL). EFIs not met but upstream groundwater abstraction impacts are **not** a significant contribution (i.e. >50%) to the failure to achieve flow standards AND groundwater abstraction pressures within the water body sub-catchment are not a significant proportion (>20%) of the total upstream abstraction pressure.
	Low	Failure of some part of the criteria but all three parts not failed.
Poor	Low	There is a surface water flow 'deficit', i.e. the scenario outflows are less than the EFIs at some point on the flow duration curve (usually at low flows) AND Upstream groundwater abstraction impacts are "a significant part" (>50%) of the 'average low flow naturally available resource' or 'allowable abstraction impact' i.e. groundwater abstraction impacts 'are a significant contribution to the failure to achieve flow standards' AND Groundwater abstraction pressures within the water body sub-catchment are a significant proportion (> 20%) of this total upstream groundwater abstraction pressure (note: this may include pressures associated with confined groundwater abstractions which are not located within the sub-catchment itself).
	High	All criteria for this test are failed and Operational staff have strong evidence that groundwater abstraction is causing deterioration in any of the dependent surface water bodies supported by the groundwater body.

Surface water and groundwater body boundaries are often not coincident. If a surface water body maps onto more than one underlying groundwater body, care is needed to ensure that a poor status result is applied to the aquifer from which the groundwater is abstracted, and not to the others. For initial screening purposes it has been necessary to use threshold such that the groundwater body is set to POOR status only if over 20% of its area is drained by POOR status surface water sub-catchments. This avoids the unreasonable spreading of POOR status results from principal aquifers (where most of the abstraction is) to surrounding secondary aquifers. Further refinement of these results should therefore be expected as the assessments are re-visited during the first river basin planning cycle, as part of EA's Catchment Abstraction Management Strategy (CAMS) process.

Groundwater dependent ecosystems
The method used to assess the Water Framework Directive quantitative status of groundwater bodies with respect to Significant Damage to Groundwater Dependent Terrestrial Ecosystems (GWDTEs i.e. wetlands) is carried out in order to evaluate effects from groundwater abstraction on the condition of groundwater dependent ecological features (plant communities) on wetlands. A GWDTE is a wetland ecosystem on the land surface that is directly dependent on a groundwater body and is not part of a surface water body. To assess the impact we need to determine whether, and if so how, groundwater abstraction affects the hydrological conditions on site that support groundwater dependent ecological features. Where significant damage as a result of groundwater abstraction is confirmed the groundwater body will be at POOR status. Otherwise it will be at GOOD status. EA also reports their confidence (HIGH or LOW) in the assessment. This is based on evidence of significant damage to GWDTE, and knowledge from local experts, for example, from previous investigations.

The assessment has two stages: 1) A risk assessment; followed by 2) Site specific analysis. Table 3.3 summarises the methodology.

Table 3.3 Outline summary of the status test for impact of groundwater abstraction on groundwater dependent wetlands (EA, 2010)

Status	Confidence	Criteria
Good	High	No GWDTE in the GWB. All GWDTE sites within the GWB are at no, low or medium risk of significant damage.
	Low	Risk assessment indicates that site(s) within the GWB are at medium risk from quantitative pressures, and there is a clear indication, based on the site condition assessment, that there is a groundwater abstraction pressure on the site. Site(s) within the GWB are at high risk of significant damage but in favourable condition, or unfavourable condition for reasons not apparently related to abstraction pressure acting through the groundwater body.
Poor	Low	Sites at high risk of significant damage and 7 step wetland screening process indicates that abstraction pressures are causing a significant departure from the environmental supporting conditions required to maintain the GWDTE in a favourable state. However, site specific data not adequate to assign HIGH confidence.
	High	Sites at high risk of significant damage and 7 step wetland screening process indicates that abstraction pressures are causing a significant departure from the environmental supporting conditions required to maintain the GWDTE in a favourable state. Site specific data provides HIGH confidence in the result.

The national conservation bodies in England and Wales (Natural England and the Countryside Council for Wales) produced a list of 1368 Sites of Special Scientific Interest (SSSIs) which they consider to be groundwater dependent terrestrial ecosystems (GWDTEs). The risk assessment of these wetlands was based on the source-pathway-receptor model where:

- Source = groundwater abstraction or regional drainage pressures
- Pathway = hydraulic connectivity between the abstraction, groundwater body and wetland
- Receptor = dependency of wetland ecology on groundwater

All wetlands were assessed and given a score, between 0 and 3, related to each of these three components and the individual scores were then added to give a total risk score between 0 (low risk) and 9 (high risk). The risk assessment was carried out in two stages. In the first stage, nationally available GIS data were used to give each site an initial risk assessment score. In the second stage local workshops were held across England and Wales where local expert ecologists and hydrogeologists reviewed these initial risk scores. At this stage additional quantitative pressures, such as artificial drainage, were also identified as potentially affecting sites.

The output from the risk assessment was a list of sites ranked according to their 'total quantitative pressure risk score'. A high score is a relative indication rather than an absolute indication of the risk of significant damage. The ecological condition of all sites was reviewed. Sites at no, low or medium risk of significant damage were screened out (i.e. if these were the only dependent sites, the groundwater body was assumed to be at ‗GOOD status, HIGH confidence'). If, however, there is a clear indication, based on the condition assessment for medium risk sites, of a groundwater abstraction pressure on the site, it was screened out as ‗groundwater body at GOOD status, LOW confidence'.

Sites at high risk of significant damage but in favourable condition, or unfavourable condition for reasons not apparently related to abstraction pressure acting through the groundwater body, will need to be subject to future surveillance but were assigned ‗groundwater body at GOOD status, LOW confidence'. Sites at high risk of significant damage and in unfavourable condition for reasons that suggest an abstraction related pressure could be the cause were subsequently assessed against the 7 step wetland screening process recommended by UKTAG paper 11b(ii). Where abstraction pressures are causing a significant departure from the environmental supporting conditions required to maintain the GWDTE in a favourable state, the groundwater body is at POOR status. The level of confidence (HIGH or LOW) depends on the availability of site specific information.

The final results of this screening relate to wetlands and therefore need to be mapped to the underlying (supporting) groundwater bodies. Where more than one wetland sits on (and is dependent on) a groundwater body, the worst case status result has been taken. Where a groundwater body has no GWDTE associated with it, it has been assigned ‗GOOD status, HIGH confidence'.

The method described in this summary complies with UKTAG guidance, however, there are some issues that need to be taken into account when reviewing the results of this assessment and which may lead to further development and improvement of the approach. These include:

- The impacts of local drainage (including Internal Drainage Board systems) on shallow wetland water levels may be marked at many sites but have mostly been ignored in this assessment which has focussed on groundwater abstraction and regional scale drainage;
- The omission of over 990 sites from the list for consideration may have led to an underestimation of risk in some areas. However, as detailed above, some steps have been taken to consider these sites;
- The risk assessment may characterise whole sites based on localised impacts so that, for large sites, the spatial extent of risk may be over-estimated;
- The characterisation of riverine or multiple site SSSIs may also be poorly targeted in that a risk is applied across the whole site, whereas individual land parcels may not all be at risk.
- Large areas of Wales and some areas of the north-west of England are exempt from groundwater abstraction licensing and, therefore, information is lacking on the location and rates of abstraction. This may have led to an under-estimation of risk in such areas although strata are typically poorly permeable which naturally tends to limit groundwater abstraction pressure;
- It was not possible to assess the risk to over 100 sites in England as it was not possible to identify a groundwater dependency for the ecological features present. As a result these sites scored 0 in the risk assessment;
- A large number of sites in Wales did not have condition assessment data. These were assigned GOOD status;
- Few of the sites in Wales have been surveyed in detail so the botanical communities helping to define groundwater dependency are not known for many.
- In future there is a need to extend the scientific knowledge of groundwater dependency of ecological features so more sites can be scored. Ongoing improvement in the understanding of ecohydrological requirements of plant and animal communities is required. Survey coverage and the availability of condition assessment data for sites in Wales should be extended. Investigations at some sites will also improve confidence in the risk screening and classification methods, and in the ability to determine significant damage from site-based data.

Saline or other intrusion test
The test identifies groundwater bodies where there is intrusion of poor quality water as a result of groundwater abstraction and this intrusion is leading to sustained upward trends in pollutant concentrations or a significant impact on one or more groundwater abstractions.

EA carries out this test to identify where groundwater quality is deteriorating or where impacts on the quality of abstracted water have already occurred as a result of the intrusion of poor quality water into the body. The intrusion must be caused by groundwater abstraction and be sustained. The test therefore focusses on bodies where abstraction pressures may cause intrusion. Impacts associated with mine water rebound are dealt with under the Surface Water and the General Chemical Assessment Test (EA, 2010).

Intrusion can occur when the saline-freshwater interface in coastal regions is drawn inland and upwards by abstraction. Groundwater abstraction can also lead to upward movement (up coning) of poor quality water, the leakage of saline surface waters to an underlying groundwater body or drawing in of poorer quality groundwater from an adjacent aquifer. This test looks at parameters in groundwater that indicate intrusion is occurring, e.g. electrical conductivity, sulphate and chloride.

EA carries out the assessment in two stages: (1) identify monitoring sites that indicate the impact of saline intrusion, e.g. elevated concentrations and upward trends and (2) examine other evidence that indicates that saline intrusion is present, e.g. an impact on a drinking water abstraction. Information from both stages is used to determine status. Where there is evidence of elevated concentrations above natural background and either an upward trend in concentrations/parameter values indicating a worsening condition or there is already an impact on a point of abstraction, then the groundwater body will be at POOR status. Otherwise it will be at GOOD status (EA, 2010).

EA also reports their confidence (HIGH or LOW) in the assessment. This is based on the weight of evidence available for making the status assessment. For example where there is evidence of upward trends in a number of monitoring points and impact has been measured at receptors confidence is HIGH. Where the evidence is less comprehensive, e.g. monitoring is limited, confidence is LOW. Confidence does not indicate how close the groundwater body status is to the GOOD/POOR status boundary.

The steps we take in carrying out the Saline (or other) Intrusion test for a groundwater body are outlined below and summarised in Table 3.4:

a. Construct a conceptual understanding (or model) of the groundwater body to understand the variation in risk across the body and the properties and behaviour of the groundwater system.
b. Review the pressure and risk assessments for all groundwater bodies. If the body is not subject to pressures and is not characterised as at risk the body is at GOOD status. The confidence in this ‗good status' assessment is determined by the amount of evidence we have that confirms that there is no intrusion. Where there are more than five monitoring points and no evidence of elevated concentrations or impacts confidence is HIGH. Where there is less evidence confidence will be LOW.

For those groundwater bodies confirmed as being at risk, EA carry out these steps:

a. Compile groundwater monitoring data body along with any additional information and supporting evidence (see Table 3.4).
b. Screen the results for individual monitoring points to ensure that they are representative for this test and that any elevated concentrations indicate the impacts of intrusion. To do this, compare the measured concentrations of chloride, sulphate, and electrical conductivity to the natural background conditions. This removes any elevated concentrations due to natural conditions.
c. Check monitoring sites with elevated concentrations that indicate intrusion to see if they have increasing concentrations or parameter values (upward trends).
d. Record any other additional evidence that indicates that the extent of saline intrusion in the groundwater body is increasing or that existing abstractions have been impacted by saline intrusion. This may include additional monitoring data that is not on the WFD monitoring network.

e. Where there is an existing, uncontrolled, significant impact on a point of abstraction or an upward trend in one or more monitoring points, the groundwater body will be at POOR status for this test.
f. Confidence in the assessment is assigned on the basis of the availability of monitoring data and any supporting evidence. Where there is extensive monitoring in the groundwater body relevant to this test and/or good documented evidence of impacts or upward trends, confidence will be HIGH. Where monitoring is more limited and there is less evidence confidence will be LOW.

Table 3.4 Outline summary of the saline (or other) intrusion test (EA, 2010)

Status	Confidence	Example criteria
Good	High	**No pressure acting on groundwater** body that could give rise to intrusion. No source for saline or other intrusion. 6 or more monitoring points.
		Based on the conceptual understanding – **Risk** of saline or other intrusion, monitoring points located within area at possible risk from saline intrusion show no evidence for elevated or increasing conductivity and/or chloride and 6 or more monitoring points. Presence of geological barrier that will prevent significant intrusion with supporting data (monitoring) to demonstrate effectiveness as barrier. No detrimental impact on receptors (e.g. groundwater abstractions). No trend. No expansion of the saline or other intrusion and no receptors impacted.
	Low	**No pressure acting on groundwater** body that could give rise to intrusion. No source for saline or other intrusion. <6 monitoring points.
		Based on the conceptual understanding – **Risk** of saline or other intrusion, but less than 5 monitoring points available to confirm assessment (further monitoring may be required) Presence of geological barrier that will prevent significant intrusion with supporting data (monitoring) to demonstrate effectiveness as barrier No detrimental impact on receptors (e.g. groundwater abstractions) No trend No expansion of the saline or other intrusion and no receptors impacted
Poor	Low	Based on the conceptual understanding – **Risk** of saline or other intrusion and monitoring data provide **some** evidence for intrusion. Evidence of rising trend in concentrations or groundwater receptors (e.g. abstractions) currently impacted.
	High	Based on the conceptual understanding – **Risk** of saline or other intrusion and monitoring data show **strong** evidence that the area of intrusion has expanded Upward trend in conductivity and chloride concentrations at points of abstraction and in surrounding monitoring points. Groundwater receptors (e.g. abstractions) currently impacted

Second River Basin Management Planning Cycle

UKTAG (2012) describes detailed procedures for the translation of the definitions of good groundwater quantitative status into an operational classification system, divided on the above four tests using criteria set by WFD. The criteria that define good groundwater quantitative status are fixed within the WFD and cannot be altered. Groundwater status was assessed in 2009 for the 1st River Basin Management Planning cycle. The Groundwater Task Team believes that the production of separate chemical and quantitative assessments (and maps) is more useful than producing ‗overall' status for each groundwater body because the individual outcomes are easier to communicate and use when implementing measures. Even though, the UKTAG 2012 still has a draft status, there is only expected some minor changes in the content of Table 3.1-3.4. However, UKTAG 2012 include an interesting discussion on the use of groundwater level monitoring, which is especially of relevance to complex level assessment methods. See Table 3.5 below. Table 3.5 also shows UKTAG's proposal for incorporating of confidence as a marker for GOOD/HIGH and POOR/LOW confidence in the assessments of chemical & quantitative status.

Table 3.5 Annex 1: Discussion on the use of groundwater level monitoring. Below is shown how the overall status results and confidence can be aggregated (UKTAG, 2012)

UKTAG Paper 11b(ii) on Groundwater Quantitative Classification

Annex 1 : Discussion on the Use of Level Monitoring

A1.1 **Water balance element.** If groundwater levels are falling in a sustained long-term manner, this will confirm that more water is being abstracted than is recharged during the period of the record, thereby indicating poor status from this element. However, long-term, sustained water levels do not necessarily indicate good status, since the water required to maintain this constant level could be drawn from surface water, potentially causing ecological damage.

A1.2 **Surface Water Element.** If there is 100% surface water / groundwater connection, the rivers tend to anchor the groundwater level to the river level so that variation is minimal. In these circumstances groundwater level is not useful in indicating surface water / groundwater interaction. If there is no surface water / groundwater connection, the level in the aquifer can be above, at or below the river level and by itself does not indicate anything about the effects of groundwater on the river.

A1.3 **GWDTE element.** The groundwater level at or around terrestrial ecosystems is fundamental for improving the conceptual model of how a GWDTE functions. It is an essential tool to confirm groundwater connection but there is no single signal from the level monitoring which implies or confirms this. Rather, it is a combination of absolute level measurements, of accounting for variations in the aquifer properties and flow conditions, wetland strata and the open water area. It will almost certainly involve some sort of model developed to confirm the conceptual understanding. This model will include surface water, groundwater or both.

A1.4 **Intrusion Element.** The determination of intrusion is to be based upon quality rather than level measurement.

A1.5 In low permeability aquifers and karst aquifers, monitoring boreholes may not give a true reflection of the piezometric surface and in some areas, the concept of a piezometric surface will have no relevance. In these circumstances, it may be better to use other indicators of quantitative (and qualitative) status such as river flows and spring flows.

A1.6 It is proposed that the best use of level data is to confirm the functioning of the groundwater body and then use the knowledge to inform the determination of status.

Test	Status result	Confidence
No saline or other intrusions	Good	High
Drinking Water Protected Areas (DWPA).	Good	Low
Groundwater Dependent Terrestrial Ecosystem (GWDTE).	Poor	Low
No significant diminution of surface water chemistry and ecology	Poor	High
General Chemical Test	Poor	Low

Overall Chemical Status: Poor Status (High Confidence)

Test	Status result	Confidence
Water Balance Test	Good	High
Surface Water Element	Good	High
Groundwater Dependent Terrestrial Ecosystem (GWDTE).	Good	Low
No saline or other intrusions	Good	High

Overall Quantitative Status: Good Status (Low Confidence)

Here we will only give one example of the most recent UKTAG (2012) proposals for second round, which has some minor changes (draft status). In Table 3.6 the updated version for water balance test is shown.

Table 3.6 Groundwater quantitative classification for 2nd planning cycle (UKTAG, 2012)

Status	Confidence	Criteria
Good	High	Risk characterisation indicates that the groundwater body is not at risk for this test AND Groundwater abstraction impacts are less than the aggregated natural low flow resource.
	Low	Groundwater abstraction impacts exceed the aggregated natural low flow resource but there is no or uncertain evidence of current or predicted groundwater resources depletion (e.g. via modelling) and there is no evidence of existing groundwater resources depletion
Poor	Low	Risk characterisation indicates that the groundwater body is at risk for this test AND Groundwater abstraction impacts exceed the naturally available low flow resource AND There is some evidence that groundwater resources may be depleted at current abstraction volumes (for example using numerical or conceptual models).
	High	Risk characterisation indicates that the groundwater body is at risk for this test AND Groundwater abstraction volumes exceeds recharge volume OR Groundwater abstraction exceeds the naturally available low flow resource and this is corroborated with existing evidence of groundwater resources depletion (falling groundwater levels, disconnection between groundwater and surface water).

Changes are that a risk characterization should now indicate whether the groundwater body is at risk, instead of a comparison of groundwater abstraction and recharge (Good status/High confidence). Instead of comparing actual groundwater abstraction and recharge, fully licensed abstraction and recharge, actual and licensed abstraction and aggregated natural low flow resource, it is stated that „groundwater abstraction impacts exceed the aggregated natural low flow resources‘ but there is no or uncertain evidence of current or predicted groundwater resource depletion (e.g. via modelling) and there is no evidence of existing groundwater resource depletion (Good status/Low confidence).

Risk characterization indicating groundwater body at risk for this test, and groundwater abstraction impacts exceed the naturally available low flow resource and there is some evidence that groundwater resources may be depleted at current abstraction volumes (for example using numerical or conceptual models), instead of the old classification based on comparison of actual/fully licensed groundwater abstractions with groundwater recharge and/or aggregated natural flow resources (Poor status/Low confidence).

Finally, the updated version for poor status/high confidence states that risk characterization must indicate that the groundwater body is at risk for this test and groundwater abstraction volumes must exceed recharge volumes or groundwater abstraction exceeds the naturally available low flow resource and this corroborated with existing evidence of groundwater resources depletion (falling groundwater levels, disconnection between groundwater and surface water), instead of the old formulation that recent actual groundwater abstraction is high or exceeds recharge and area staff confirm that there are known issues associated with groundwater abstraction, and recent actual and fully licensed groundwater abstraction exceeds the naturally available low flow resource and area staff confirm that there are known issues associated with groundwater abstraction.

Key aspects and evaluation of methodology in England and Wales

- England and Wales performed the four tests recommended by CIS. Detailed guidelines were prepared with descriptions on how the tests should be performed. The methodology appears transparent with respect to how it was performed. The test can be categorized as relatively simple screening tools
- Groundwater recharge is evaluated based on the situation with actual abstraction, i.e. including capture aspects, but the methodology for recharge assessment is not fully clear. Aquifers are eventually grouped in case of horizontal flows between aquifers
- Environmental flow criteria were based on Acreman et al. 2008; Acreman and Dunbar 2010
- The confidence of assessment is included. This appears useful.
- Plans have been prepared for improving tests for second WFD round to improve the scientific basis
- Groundwater levels are used as part of validating the result of the overall classification according to the water balance test
- GWDTE‘s evaluations are carried out
- The groundwater bodies of UK have an average size of 339 km^2

3.3.2 Practice in Ireland

Only four GWBs are assessed to be in poor groundwater status in Ireland. The assessment of groundwater status generally follows a national approach *(EPA IE, 2010).* The balance between recharge and abstraction is assessed by comparing the annual average abstraction against available groundwater resource for every groundwater body.

Recharge was estimated for each groundwater body using the Working Group on Groundwater Recharge Map (2008), produced by CDM and Compass Informatics. No account was taken of any potential inflows from the surrounding groundwater bodies, although some groundwater bodies were grouped together for the assessment.

Average annual abstraction was approximated as the sum of all the groundwater abstractions from each groundwater body, including public water supplies, private group schemes, industrial supplies and dewatering of mines and quarries. Private domestic supplies were not accounted for (many of the larger private groundwater abstractions, such as for farms, golf courses and small industries were not included in the assessment, as this information was not available).

Ecological flow requirements were approximately established for average flow conditions, taking into account recommendations in WFD Guidance documents and the SNIFFER Research project WFD 53 Report (2005). In the SNIFFER report a suggestion for initial assessment of available groundwater resources (as a fraction of recharge comparable to SF, but here as a general indicator also for assuring environmental flow requirement). In Table 3.7 suggestions for sustainable Abstraction-Recharge factors are given for different values high and low storage hydrogeological settings. In the WFD 53 Report (2005) alternative governing factors to specific yield, e.g. transmissivity and summer/average flow rations were described.

Table 3.7 Assessment of groundwater abstraction exposure pressures for initial characterisation of available groundwater resources without directly evaluating environmental flow requirements. The figure below the Table shows alternative attempts to relate Abstraction-Recharge thresholds to transmissivity and summer/average flow ration, but the specific yield was chosen in order to keep the assessment simple (Source: WFD Report 43, 2005)

	Exposure Pressure based on average Specific Yield (Sy) of the Groundwater Body	
GWABS/RECH in 2015 (as a % for the groundwater body)	Low Storage (Sy < 5%) (or 'fissured/fractured' flow mechanism)	High Storage (Sy > 5%) (or 'intergranular' flow mechanism)
> 40%[1]	High	High
30 to 40%	High	High
20 to 30%	High	Moderate
10 to 20%	Moderate	Low
2 to 10%	Low	Low
<2%	No	No

Notes:

[1] i.e. GWABS is greater than 40% of the LTA Recharge.

Where a groundwater body was considered to be at risk from over-abstraction and there was evidence of sustained falling water levels in the EPA's National Groundwater Monitoring network, the groundwater body was classified as Poor Status.

The impacts of abstraction on GWDTEs and saline or other intrusions have also been considered. Associated surface waters are considered in the assessment of quantitative status. Final status was assigned based on the abstraction/recharge ratio. There is a lack of information on ecological flow standards, and subsequently the ecological assessment of associated surface waters could not be undertaken and is planned for the second RBMP cycle.

In general there are many similarities with UK methods but also some differences in criteria. The assessment of quantitative status is described in EPA IE (2012). Of the four tests only three have been finalised, as the surface water ecological/quantitative assessment could not be undertaken for Ireland, but it will be undertaken in the second RBMP cycle. The criterion for poor status for the water balance test is defined as either calculated abstraction – or recharge rate above 100 %, the groundwater body automatically defaults to poor status. In the absence of any clear minimum flow requirements for rivers and wetlands in Ireland, an arbitrary figure of 20 % of recharge was left to support the flow in rivers and wetlands. Therefore, abstraction - recharge ratios above 80 %, result in poor status (with high confidence).

A groundwater body is at risk of failing its WFD objectives if the abstraction – recharge rate is above 5 % for groundwater bodies supporting a GWDTE, above 20 % for bedrock groundwater bodies and above 30 % for gravel groundwater bodies (~poor status with low confidence), see Table 3.7. It should be noted that the water balance test is based on an analysis of recharge, ecological flow needs and groundwater abstraction volumes, and a groundwater body test.

Table 3.7 Status category for Ireland based on proportion of recharge used by abstractions (EPA IE, 2010)

Annual Abstraction / Recharge Ratio	Groundwater Body Type	Falling Water Levels	Status & Confidence
>80%	-	-	Poor – High Confidence
30-80%	Gravel	Yes	Poor – Low Confidence
30-80%	Gravel	No	Good – Low Confidence
<30%	Gravel	-	Good – High Confidence
20-80%	Bedrock	Yes	Poor – Low Confidence
20-80%	Bedrock	No	Good – Low Confidence
<20%	Bedrock	-	Good – High Confidence
5-80%	Supporting a GWDTE	Yes	Poor – Low Confidence
5-80%	Supporting a GWDTE	No	Good – Low Confidence
<5%	-	-	Good – High Confidence

Where the Abstraction-Recharge ratio was greater than 80 %, the groundwater body was at Poor Status (with high confidence); otherwise all other Poor Status groundwater bodies had low confidence assigned. Good Status was assigned to all other groundwater bodies, with low confidence assigned to groundwater bodies where the Abstraction-Recharge ration (~SF) was greater than 5 %, 20 % or 30 % for groundwater bodies supporting GWDTE's, bedrock groundwater bodies or gravel groundwater bodies respectively, but there was no supporting water level data. The remaining groundwater bodies were assigned Good Status (with high confidence).

The future developments in Ireland state that the water balance test is a groundwater body wide test, which uses average annual abstraction and recharge values. Therefore estimates of the groundwater abstraction impacts used in the test are basic and the more detailed surface water and GWDTE tests should identify impacts on receptors that are dependent on groundwater. Further consideration of the impacts of groundwater abstraction on low flows will be required in the future, where it is found that groundwater abstractions have, or could have an unsustainable impact on the flow to associated receptors (EPA IE, 2010).

Key aspects and evaluation of the methodology in Ireland

- Ireland performed the three of the four tests recommended by CIS. The methodology appears transparent but not fully consistent with respect to how it was performed and the test can be categorized as relatively simple screening tools. The recharge and available groundwater resources estimates are uncertain, and do not reflect dynamics of capture
- Groundwater recharge was evaluated based on a national map (simplified approach) based on GIS data, where groundwater recharge is estimated based on fraction of net precipitation (taking into account runoff/rejected recharge)
- Some groundwater bodies were grouped (horizontal flow was not accounted for)
- Environmental flow requirements were approximately established for average flow conditions, effects on low flow from actual abstraction not has yet not been assessed
- Groundwater levels play a more dominant role compared to England and Wales in quantitative status assessment
- The confidence of assessment is included. Again, this appears useful
- Plans have been made for improving tests for second WFD round to improve the scientific basis: For instance, thresholds for minimum flow will be established
- GWDTE screening is carried out
- Surface water flow test is not carried out
- Groundwater bodies of Ireland are smaller than for UK and have an average size of 108 km^2 (Denmark has an average size of 111 km^2)

3.3.3 Practice in Germany

In Germany five different tests are included in the assessment of quantitative groundwater status. The tests are the four tests according to CIS Guidance document no. 18, and in addition a trend analysis is provided based on long term groundwater level time series (15-30 years). The trend analysis in Germany plays a major role in the classification of quantitative status, which differs from UK and Denmark and only has some similarity with Ireland.

The assessment of groundwater quantitative status was based on the comparison between long-term average abstractions and long-term average rates of recharge and the analyses of temporal developments of groundwater levels (as far as appropriate time series were available). Indications of impacts of groundwater quantity on the health of associated aquatic and groundwater dependent terrestrial ecosystems and saline intrusion were considered as far as relevant. The assessment methodology is described in the EU WISE database but not in the individual RMBPs (EC, 2012).

According to WISE the following criteria must be met (WFD, Annex V), for a Groundwater body to be of good quantitative status:

1. available groundwater resource is not exceeded by the long term annual average rate of abstraction;
2. no significant diminution of surface water chemistry and/or ecology resulting from anthropogenic water level alteration or change in flow conditions that would lead to failure of relevant Article 4 objectives for any associated surface water bodies;
3. no significant damage to groundwater dependent terrestrial ecosystems resulting from an anthropogenic water level alteration;
4. no saline or other intrusions resulting from anthropogenically induced sustained changes in flow direction
5. long term groundwater level trend

The LAWA guidance document (LAWA, 2003/updated version) state that the most widespread impact on the quantitative status of a groundwater body is from long-term groundwater abstractions. The parameter for testing and assessment is, in all cases, the groundwater level (for confined groundwater this means the groundwater pressure surfaces, for unconfined groundwater the groundwater surface). Over-exploitation of groundwater occurs if, in large parts of a groundwater body, the groundwater levels (or spring discharge) shows a sustained negative tendency which cannot be explained by climatic conditions. Such tendencies generally indicate poor quantitative status, even if the above-mentioned impacts cannot at first be observed. In many cases, these impacts only become apparent after a certain time lag or occur with spatial shifts. With regard to ecological concerns, a local or regional change in groundwater table may also be relevant within the meaning of the Directive. A localised lowering of the groundwater table that has no ecological consequences, e.g. in the vicinity of water extraction facilities, is not a matter for the Directive (LAWA, 2003).

The necessary activities include:

- Wide-scale overexploitation of groundwater body (if the groundwater level hydrograph lines positioned at large distance from abstraction wells show ongoing lowering of groundwater levels of at least ten years, negative trend should be analysed; if a large number of monitoring sites in a groundwater body situated away from immediate impact zone from abstractions, the quantitative status must be defined as ‗poor‘)
- Impairment of hydraulically connected surface water ecosystems or groundwater dependent terrestrial ecosystems (if terrestrial ecosystems are significantly damaged, the status of groundwater body is bad. The decisive parameter for significant damage is the groundwater level. If over a long period values fall consistently short of the lower thresholds, the ecosystems must be considered at risk and the status of the groundwater body shall be identified as poor)

- Influx of salt water or other undesirable constituent substances as a consequence of anthropogenically modified hydraulic heads (in case modifications of influx of salt water or other undesirable substances due to modified groundwater level/hydraulic heads)

A good quantitative status as described by LAWA and other German references (Umweltbundesamt, 2010) means that there is a balance between groundwater abstraction and groundwater recharge. This leads to the assessment of 96 % groundwater bodies in Germany with a good quantitative status, and only 4 % of poor quantitative status of the 989 GWBs. At the time when the RBMPs were established, the Groundwater Directive was not yet implemented into German legislation – as it is now – and the assessment was based on the requirements of the WFD and a guidance document about the technical implementation of the GWD, which was developed by the LAWA (LAWA, 2003) and is in compliance with the WFD and the Groundwater Directive.

Groundwater recharge is typically based on ‗Hydrologischen Atlas Deutchland', e.g. groundwater models are typically not used for assessment of groundwater recharge, and capture dynamics (e.g. induced recharge etc.) therefore not incorporated. A threshold value of 30 % of groundwater recharge has generally been applied. Groundwater abstractions are added based on licensed abstractions. Finally, a trend analysis is added and if relevant salt water intrusion is incorporated in a final evaluation. In some cases it has been decided to base the water test (e.g. for Schleswig-Holstein) for ‗basins' which are larger than the groundwater bodies, in order to avoid problems with horizontal inflows from adjacent groundwater bodies (see Table 3.8).

Table 3.8 Assessment matrix for quantitative status assessment used in Germany (Source: Berthold et al., 2011)

Trendanalysis Groundwater levels	**Abstraction-Recharge fraction (A/R)*)**	**Detailed groundwater recharge Assessment**)**	**Poor status due to saltwater intrusion test**	**Status of groundwater body**
Falling trend	< 0.3	Not required	No	Good status
Falling trend	> 0.3	Positive/OK	No	Good status
Falling trend	> 0.3	Negative	No	Poor status
Strongly falling trend	< 0.3	Positive/OK	No	Poor status
Strongly falling trend	> 0.3	Positive/OK	No	Poor status
Strongly falling trend	> 0.3	Negative	No	Poor status
Not yet available	< 0.3	Positive OK	No	Good status
Not yet available	< 0.3	Negative	No	Poor status
Not yet available	> 0.3	Positive/OK	No	Good status
Not yet available	> 0.3	Negative	No	Poor status
			Yes, significant saltwater intrusion	Poor status

As illustrated by Table 3.8 both an initial screening for the Abstraction-Recharge fraction for the entire groundwater systems (30 % threshold value), and a subsequent (if required) detailed assessment for groundwater body are carried out. The initial A/R factor is calculated based on recharge data from ‗Hydrologischen Atlas Deutschland – HAD' and based on licensed abstraction values (as a worst case assessment of abstraction).

The detailed comparison of groundwater abstraction and available groundwater resource is made for the entire groundwater body. Actual groundwater abstraction is used instead of licensed abstractions in the detailed assessment, and the detailed assessment is according to CIS no. 18 water balance test, taking into account surface water flow requirements (low flow/dry weather conditions) and GWDTE's requirements. Furthermore, exchange flow between adjacent groundwater bodies are incorporated if there are such flows.

The GWDTE test is performed only if a groundwater-dependent country ecosystem is damaged or in risk of harm, which are dependent on a mainly fluctuating groundwater levels due to climate variations. For risk assessment, a test is recommended where amongst others depth to groundwater thresholds are used (<= 3 meter in general and <= 5 m for forest ecosystems). Furthermore, site specific data and knowledge are included in this test, only relevant for terrestrial ecosystems.

Key aspects and evaluation of methodology in German

- Germany therefore performed the four tests recommended by CIS, and used the same methodology in different ‗Ländern'. The methodology appears transparent with respect to how it was performed and the water balance test can be categorized as relatively simple screening tools. In addition to the four CIS tests a fifth test, groundwater level trend analysis, play a key role in the German methodology. An initial and a detailed water balance test are carried out
- Groundwater recharge was evaluated from a national map (simplified approach) based on GIS data at initial stage and detailed stage (not clear if groundwater model results are used in detailed analysis, but horizontal flow is included in detailed test)
- Environmental flow requirements are established for low flow conditions, but it is not fully clear how effects on low flow from actual abstraction has been evaluated as part of detailed water balance test
- Groundwater levels play a guiding role in the quantitative status assessment. This could inspire Danish assessment but require long term systematic monitoring of groundwater levels (> 15-30 years)
- GWDTE screening carried out with thresholds <=3 m (<=5m for forests). This might inspire a Danish methodology. Complex methods (site specific data) are included
- Groundwater bodies of Germany are large with an average size of 360 km^2 (comparable with UK)
- The groundwater recharge assessment is uncertain.

3.3.1 Practice in *France*

The approach to the assessment of the status of groundwater has varied significantly in the different river basin districts (RBD). The different RBD authorities have therefore used the best available knowledge in the different districts. A number of studies have been launched over the past few years all across France in order to develop a sound methodology, and the first results of these studies will be available during 2012 (EC, 2012). Two recent reports describe the quantitative assessment according to the four tests (Blum and Chassiot, 2011; Ministère de l'Ecologie, 2012).

In general the tests are only carried out if the aquifers are at risks (groundwater levels evaluated as in risk of continuous lowering). The conceptual model and the trend of groundwater quality and level are the main foundation for evaluating which tests are carried out (e.g. water balance test, surface water test, groundwater dependent terrestrial ecosystem test and test for saltwater intrusion).

The water balance test is carried out if it is evaluated that there is a long-term risk of the quantitative water balance status of the groundwater body. In this case the groundwater recharge to the aquifer and the groundwater abstraction from the aquifer is calculated as long term average. The ratio between abstraction and recharge is subsequently calculated. If this ratio is above 1.0, then the quantitative status is evaluated as having poor status. If the ratio between abstraction and recharge to the aquifer is below 1.0, the quantitative status is evaluated as good. The water balance test hence use an SF criteria = 1.0.

There are some distinctions for different aquifer types in the water balance test as used in France. It is stated that the water balance test can be difficult to evaluate for karst aquifers which are quite frequent in parts of France, and for multi-layer aquifer systems it is in general recommended to use groundwater models in order to account for the dynamics of capture, and the significant time period before such aquifers reach a new equilibrium after being exposed to changes in abstraction (more than 30 aquifers in France).

The surface water test is carried out in cases where there are downstream surface water bodies considered at ecological or chemical poor status. Piezometric groundwater head maps, hydraulic gradient, hydraulic conductivity of aquifer, river valley land use and groundwater-surface water interaction modelling results are utilized for the evaluation. Based on this, and on the conceptual model, it is evaluated whether or not groundwater abstraction is a cause for good or poor status of surface water bodies.

It is stated that the knowledge base is limited regarding representative spatial relationships/detailed knowledge between groundwater bodies and surface water catchments and groundwater surface water interaction. Also the extent to which groundwater abstraction, and changes in discharges to surface water, impact ecological status, is not well known. There is no translation of qualitative criteria into quantitative indicators in the French methodology as used for first round of WFD implementation.

GWDTEs test in principle follow similar qualitative principles and knowledge base (hydrogeological maps and modelling results). The key is again whether or not there are groundwater dependent GWDTEs, which in France is evaluated based on Natura 2000 sites and all other terrestrial ecosystems (e.g. Ramsar sites, important wet zones etc.). There are two criteria's for evaluating GWDTEs and the anthropogenic impacts from groundwater abstraction (Vernoux et al., 2010):

- Hydraulic interaction between ecosystems and groundwater bodies
- Sensitivity of ecosystems in relation to quantitative and water quality relationships with groundwater bodies

Typology of interactions between groundwater and surface water, irreversibility of changes, dynamics, changes in flow directions, contamination risks from groundwater, vulnerability of GWDTEs in relation to changing groundwater conditions, and monitoring of GWDTEs are some guidance according to the French methodology (Ministère de l'Ecologie, 2012).

The saltwater intrusion test in France is primarily focusing on chemical component of this test and on evaluating Conductivity, Chloride, Sulphate etc. in relation to threshold values for such aquifers.

Like for UK and Ireland the level of confidence is evaluated in France.

Key aspects and evaluation of the methodology in France

- Four tests are made to determine whether a groundwater body is at good quantitative status: (1) the overall water balance; (2) the extent of interactions with dependent surface water bodies; (3) damage to dependent terrestrial ecosystems (wetlands); and (4) the risk of saline intrusion.
- France does not follow recommendations by CIS regarding the water balance test, and instead uses the ratio between groundwater abstraction and recharge such that if the ratio is smaller than 1.0 the status is considered good. This is in line with other Mediterranean countries (e.g. Spain) which use similar principles based on renewable groundwater resource (instead of available groundwater resource).
- The other three tests are carried out based on qualitative criteria, which are not translated into quantitative criteria. The methodology in France and the four test does not appear transparent with respect to how it was performed and the four tests can be categorized as simple screening tools, with significant differences from basin to basin.
- Groundwater recharge was evaluated with a simplified approach based on available data (not clear how groundwater model results are used in detailed analysis e.g. for multi-layer aquifers).
- Environmental flow requirements have not been established for low flow conditions, and it is not transparent how effects on low flow from actual abstraction have been evaluated as part of surface water test.
- Groundwater bodies of France are large with an average size of 958 km^2 (which is three times larger than UK and Germany and nearly ten times larger than Denmark).
- The groundwater recharge assessment is uncertain.

4. Review of Danish River Basin Management Plans ("Vandplaner")

4.1 Detailed description of quantitative status assessment for Denmark

The quantitative status assessment for Denmark has been carried out by the Danish Nature Agency (NST). The assessments were made in somewhat different ways for different areas. Basically Jylland, Fyn and Sjælland used different approaches especially for evaluating the test related to impact on surface water systems.Jylland used a mix of methods for different subareas, Fyn used results from the first version of the National model (DK model; Henriksen et al., 2008) but with no new simulations, and Sjælland used an updated version of this model (NOVANA DK model; calibrated for low flows).

The test for water balance (aquifer safe yield) has been done for most groundwater bodies in Denmark using the criteria Abstraction <= 0.35 * Recharge (-> Good Status). However, even though the approach was carried out in most areas, one exemption is the old ‗Aarhus Amt', where this criterion was not applied. Furthermore, since the DK model version 2 (NOVANA model) introduced a layer concept which respected the top layer of the various groundwater bodies, whereas the DK model version 1 that was applied for Jylland used a voxel approach. Therefore, the recharge had to be interpolated from different model layers for the water balance test. This introduced some additional uncertainties.

In the following we will describe the overall methodology with the example of Fyn (Odense river basin), and from there introduce differences for Sjælland and Jylland.

4.1.1 Fyn

The water balance test was calculated for all regional groundwater aquifers, but not for the shallow aquifer which was considered as too uncertain due to a merge of shallow and deeper aquifers. All abstraction wells have been included. Screen levels have been related to unique groundwater bodies. The licensed abstraction (1991-2000) is typically 25 % higher than the actual abstraction. All abstractions above 25,000 m^3 per year are represented in the model.

The most recent low flow observations were used (1976-1991). For this period 1800 discharge (Q) measurements have been collected for 600 stations in rivers. The more detailed calculation of low flows was based on the ‗synkronmåling approach', whereby the median of annual minimum flow (Q_{medmin}) is calculated from campaign measurements, and using time series at gauging stations for corrections from actual flow to Q_{medmin}. Apart from the Q_{medmin} dataset from 1995, which mainly reflects the hydrometeorological and abstraction conditions of the 1980s, there is no systematic observation based assessment of minimum flows for Fyn.

The evaluation of groundwater abstraction impacts on low flow was based on DK model Fyn (Henriksen et al., 2008) with seven model layers, where the uppermost model layer merges three geological layers. This means that the simulation of the shallow aquifer system is more uncertain than the regional and deep aquifer simulations. In order to enable an assessment on a river reach basis Fyn was subdivided into 459 subareas (mean area ~ 2 km^2).

The guiding criteria for evaluating High/Good ecological status for river reaches impacted by groundwater abstraction are:

- Groundwater abstraction should not give reductions in Q_{medmin} above 5 % and 10-25 % of the virgin (natural without abstraction) Q_{medmin}, for river reaches with High and Good ecological status goals. The more specific evaluation of acceptable reduction is based on river type and vulnerability of the river.
- For areas impacted by water supply for drinking water it is possible to define specific reduction criteria for rivers with High or Good ecological status goal, which accept a higher %-reduction that defined above. If a concrete knowledge is available that documents good hydromorphological and physical-chemical conditions, then higher %-reductions than defined above can be accepted.

Waste water treatment plant discharges were not included in natural Q_{medmin}. For the assessment for Fyn Q_{medmin} reduction due to abstraction are based on the DK model Fyn.

The model generally overestimates the low flow compared to observed Q_{medmin}. Therefore it was decided to use the observed Q_{medmin} instead of the modelled, and to assume that the model can provide a good estimate of the change in Q_{medmin} caused by the abstraction. In order to estimate the reference conditions the model calculated difference due to abstraction at low flow is added to the observed Q_{medmin}. The calculation of needs for reduction in groundwater abstraction is subsequently evaluated by accumulating the calculated reference situation Q_{medmin}, environmental flow requirements (minimum requirements in relation to max reduction in abstraction at different stations), whereby the need for reductions in abstractions were assessed.

The calculations resulted in a need for reduction of abstraction by an amount of 8 million. m^3/year for Fyn, in order to meet the environmental flow demands at 341 river reaches included in the analysis. This is approximately the same order of magnitude as the water balance test described below. The conclusive map for Fyn highlights 202 areas, where environmental flow requirements could not be meet by actual abstraction (~17 % of the river reaches).

The water balance test is based on the assumption that the available groundwater resource ~0.35 * Recharge. A virgin situation without abstraction is used, and the DK model is used to calculate the groundwater recharge to groundwater bodies, based on ―downward" vertical flow to the model grids comprising the layer corresponding to the uppermost layer of the various aquifers (regional and deep aquifer; based 1 km^2 grid cells in DK model 2003). This downward groundwater recharge is accumulated for the entire area of the groundwater body and multiplied with the factor (SF = 0.35). If the long term groundwater abstraction exceeds the available groundwater resource, the groundwater body in the water balance test is marked as ‗Poor‘.

For the Odense Fjord catchment area ‗Poor quantitative status‘ was the result for four out of 12 regional groundwater bodies. NST evaluates that a reason for this could be the uncertainty associated with the assessment (the geological model has been updated, and horizontal flow between neighbouring groundwater bodies was not included in the calculation of ―recharge”; this is not in agreement with CIS guidance doc. No 18). This means that the assessment of the remaining resource is also only indicative for new groundwater abstraction licences. For two of the groundwater bodies the evaluation showed that the available groundwater resource was only ~10 % of the current groundwater abstraction. For the other two current groundwater abstraction was 1.5 - 2 times the available groundwater resource.

NST evaluates that several sources of uncertainty should be considered. First, the version of DK model Fyn has a grid of 1 km^2. Next, the geological model has not been updated. The model was not calibrated for all the observed Q_{medmin} data, only data from Q gauging stations with time series has been included in the calibration (11 Q stations). The water balance test does not incorporate horizontal flow components, which especially near river valleys and for sub-aquifers to groundwater aquifers may give misleading results.

4.1.2 Sjælland

The approach is similar with the one for Fyn but with some differences especially for the evaluation of low flow reduction due to groundwater abstraction. A concrete assessment of a higher reduction-% based on site specific evidence, was applied for Sjælland. This assumption was used for many river points (based on the so called ―formula 7 approach”). However, for Lolland, Falster and Møn the approach was like Fyn, only using the a priori reduction goals of 5% and 10-25 % reduction of Q_{medmin}.

The approach for estimating natural Q_{medmin} (without waste water) was in principle the same as used for Fyn. However, the estimate of how to assure good status was different for Sjælland compared to Fyn. We will not go into details about the method used for Sjælland, but for each of the 763 observations of Q_{medmin} and for additional 177 pseudo-stations for ungauged catchments demand values for maximum acceptable reduction of low flow in relation to reference situation Q_{medmin} was assessed based on the following information:

- The delineated river reaches, where there was a risk of not achieving good ecological status
- Guiding values for maximum Q_{medmin} reduction based on Regionplan 2005 and actual biological state assessed based on Danish Stream Fauna Index (DVFI), Miljøstyrelsen (1998)
- Maximum Q_{medmin} reduction values calculated from an empirical model linking good ecological status to median min Q (formula 7)
- The actual Q_{medmin} reduction due to abstraction

- The topographic catchment area

The formula 7 approach is based on 15 stations on Sjælland, where there is evidence for a good ecological status. For these 15 stations, a relationship between topographic catchment area and the environmental flow (in l/s) has been calibrated.

Formula 7 was used for 558 of 940 stations (area > 10 km^2). It was decided also to use this formula for bottom slope < 0,05 %, which previously had been assumed problematic. Formula 7:

$$\text{Med min Q requirement (l/skm}^2\text{)} = 0{,}1676 \times (\text{km}^2 \text{ oplandsareal}^{\,0{,}2201}) - 0{,}0128 \qquad \text{(Formula 7)}$$

GEUS did a review of the approach used by NST for Sjælland (Refsgaard, 2010). For use in the river basin management plan it was recommended to describe and present the uncertainties related to the assessment method, in order to distinct between decisions which are robust in relation to the uncertainties, and decisions which are vulnerable. Furthermore, it was recommended to improve the hydrological model in order to reduce uncertainties. It was further recommended, to evaluate the environmental flow requirements, which define how much water will be available for groundwater exploitation. It was recommended to use methods at the complex level (habitat models) acknowledging that this would require new knowledge about relationships between flow, water quality and ecological conditions, so this would be a longer term developmental effort.

4.1.1 Jylland

For Jylland a different approach was applied compared to Fyn and Sjælland. The water balance test for Jylland was more difficult regarding the recharge estimate. There was not an available DK Jylland model with hydrostratigraphical based layers related to aquifers. Therefore, the estimate of recharge was based on interpolations between different layers in the voxel based DK model Jylland, where vertical flow components were extracted and summarised for the different shallow, regional and deep aquifers. This added an extra source of uncertainty to the assessment, compared to how it was made for Fyn, where model layers were more consistently related to the top of the groundwater bodies.

The vertical identification of the top layer and the distinction between regional and deep groundwater bodies had to be approximative. For Ringkøbing Fjord it was done in one way relating it to top of Bastrup formation. For Limjorden another distinction had to be made, assuming that the distinction between shallow and regional was located 25 meter below the surface, and between regional and deep in 75 meter below surface etc.

Due to the high number of irrigation wells in western, southern and northern Jylland the groundwater abstraction was based on permissions/licensed abstraction rates instead of actual abstraction.

For former Aarhus county a different approach was applied, where it was decided to skip the water balance test, because it was evaluated as being too uncertain (in Aarhus the sizes of groundwater bodies were rather small which automatically increase the uncertainty of the recharge estimates). Instead of using groundwater recharge from DK model Jylland, Aarhus attempted to stick to the old method, where water balance (groundwater recharge) was estimated based on Q_{medmin} and added to the groundwater abstraction. However, it was evaluated that this did not provide any meaningful result in terms of quantitative status for the water balance test. Another complication was the selection of rather big areas for the calculated water balances (Randers Fjord and Århus Bay catchments).

For the test for surface water impacts from abstraction there were also rather big differences in methodology. For former Ringkøbing, Viborg and Ribe counties analytical methods, such as Jenkins or similar, were applied for evaluating impacts from irrigation wells and other wells on low flow in rivers. It was realised that due to the structural governance reform many previously applied methods (e.g. Stang's method) was lost or forgotten by the administrative system. It was not possible to find the material for digital calculations so a new approach had to be invented. By support from a consultant (Orbicon) a GIS model was developed which subdivided Jylland into subcatchments based on Q_{medmin} stations. However, Århus did not participate in this joint effort by the other units for establishing a shared methodology. Subcatchments had areas of 5-25 km^2, with a few interpolated ‗stations'. The data were based on Q_{medmin} from synchronous low flow measurement campaigns from the 1990s. Orbicon correlated the estimates and updated Q_{medmin} corresponding to the 1971-2000 period.

For each subcatchment abstractions were accumulated divided on irrigation and water supply abstractions. The aggregation only considered abstractions to a certain depth (shallow and regional), as the deep aquifers were assumed not to result in reduced Q_{medmin}.

Next an ‗impact factor' was evaluated for Ribe county based on a regression analysis, because the best available knowledge regarding abstraction amounts and impacts on low flow existed for this area. Return flow was also assumed for irrigation wells. No common method could be agreed upon between the different regions for impacts from water supply abstraction. Aarhus used the equation: Low flow reduction (%) = summarized abstractions / (Q_{medmin} + summarized abstraction), but for rather big subcatchment areas (unrealistic dataset).

For the western part of Jylland, where a few surface water abstractions exist in the downstream part of Skjern river, Storå and Karup å, a reduction factor of 4 was applied, due to three months of irrigation / seasonal impact on low flow.

A slightly different approach was also applied for the reference situation natural Q_{medmin} (the historical reference situation without any abstraction). For some areas the observed Q_{medmin} was used and to this the evaluated low flow reduction was added (Limfjorden), for other areas a model was used (Ringkøbing and Nissum Fiord catchments) for evaluation of natural Q_{medmin} (without abstractions). For Århus calculated low flow impact was added to observed Q_{medmin}
The evaluation of good ecological status was also different for different parts of Jylland. For most areas the 10-25 % reduction goals was applied. For two areas, Viborg and Århus, goals were for some river stations selected based on site specific information about a good

ecological quality (> 25 % reduction), despite the impact on low flow for such reaches. For Viborg analytical methods were used for the analysis of low flow impacts, for Århus numerical from groundwater mapping models was used.

From the above it is clear that there could not be established a general consensus among the NST units regarding the methodology for water balance test and surface water impacts from abstractions for Jylland. It was decided to use different methods for the first round of the WFD planning cycle, and in the next round to decide on more unified methods.

Another issue is the differences in scale for subareas and groundwater bodies. For Aarhus initially 800 groundwater bodies had been suggested, however these were reduced to 100 after the final gap analysis (basisanalysen). For Fyn a similar reduction in the number of groundwater bodies had been made, however a bit less drastic. Despite this attempt to homogenise the sizes still a huge difference prevails between the sizes for the western part and eastern part of Jylland.

The test for groundwater dependent terrestrial ecosystems was evaluated as impossible due to lack of knowledge about groundwater surface water interactions. This test was not applied for Jylland or for any other areas in Denmark. The test for saltwater intrusion was made for rather large groundwater bodies in western Jylland (1000 km^2). The focus here is on changes in flow direction due to abstractions.

4.1.2 Evaluation of Danish implementation

From the description above the Danish implementation for Fyn, Sjælland and Jylland can be characterized as shown in Table 4.1.

Table 4.1 Danish implementation for Fyn, Sjælland and Jylland.

	Recharge and abstraction	Median min Q	Water balance criteria	Surface water flow criteria	Impact on low flow (accumulat.)
Fyn	Recharge based on DK model (Henriksen et al., 2008) Abs:1995-2005	1976-1991 dataset (1995 dataset for 600 stations)	Abstraction < 0,35 * virgin, natural recharge *(horizontal flows not included)*	High ecological status: max 5% Good ecological status: max 10-25 %	Observed median min + modelled low flow reduction = natural med.min
Sjælland	Recharge based on DK Novana model Abs: same as Fyn	763 observations of low flow and 177 pseudo stations	Abstraction < 0,35 * Natural recharge	Same as Fyn, but Formula 7 and higher reduc. % for 558 of 940 stations	Same as Fyn
Jylland	Recarge interpol.from DK model or based on waterbalance estimates Abs: Permissions	Updated median min Q for 1971-2000 (Orbicon) For west,south and North. Aarhus different	Abstraction < 0,35 * Natural recharge for large part, but not appied for Aarhus	Same as Fyn, but Viborg and Aarhus, higher reductions allowed	Impact factor based on mixed methods, models, or estimated (analytical)

The following findings can be emphasised:

- There is a lack of similarity and consistency in the application of methodology across Denmark (Fyn, Sjælland and Jylland), with different recharge assessment methodologies and different model results used. The same yield for the use of median minimum discharge, Q_{medmin}, where different time periods and datasets were applied. For Jylland there was no consensus between NW-S and E Jylland methodology in use of Q_{medmin}.
- The water balance test was evaluated with reference to groundwater recharge without pumping. The poor quantitative status was communicated differently for different parts of the country. For Fyn and parts of Jylland “flagging” the poor status, while the results of water balance test for East Jylland (e.g. Aarhus area) and Sjælland was almost ignored in the RBMPs (only surface water test was applied). Since the groundwater recharge was assessed with different methodology for different areas, results were less comparable from catchment to catchment.
- The criteria for impact on low flow were diverse and lacked consistency. For Fyn higher %-reduction was not considered for any reaches. For Sjælland a major part of the stations were allowed a higher reduction % based on “Formula 7”. For parts of Jylland (within former Viborg and Aarhus counties) higher reduction % were adopted. Furthermore, the reference situation and impact on low flow from pumping was estimated with ad hoc methods (water balance, model based and analytical methods). Even though the DK model for small catchments has significant uncertainty (especially for stations representing catchment areas < 30 km^2), this was not addressed in the design of ‘stations‘ used for the assessment.
- The water balance test was not applied in compliance with CIS methods. Available groundwater resource was not quantified with respect to surface water and ground water dependent terrestrial ecosystems. And instead of using the actual groundwater recharge, the pristine recharge (situation without groundwater abstraction) was used with a threshold value of 0.35, and without including horizontal flows, even though many groundwater bodies have significant horizontal flows across boundaries (this means that capture is not incorporated in the assessment). If a fixed ration are used e.g. 0.30 for indicator 2 (and 0.35 for indicator 1) these should be better documented for different hydrogeological conditions (indicators have not been validated for Fyn and Jylland, and scale dependency have not be properly addressed in the first round uses). A database with experience values should be established for different hydrogeologies and scales so that precautionary established criteria can be derived.
- The surface water test in principle is in compliance with CIS methods, but the uncertainty in the assessment was not considered, and transparency was not assured. Actual Q_{medmin}, pristine Q_{medmin}, assessment of reduction in low flow due to groundwater abstraction and relevance of the indicator is difficult to compare across different basins (Fyn, Sjælland and Jylland) and across different catchments within basins (Sjælland and Jylland).
- There is a need for using indicator 2 (actual abstraction/recharge relationship) instead of/or in combination with indicator 1, and of using the same methodology across Denmark in the next round for recharge and water balance calculations. Either groundwater bodies should be merged so that there are no significant horisonal inflows, or horizontal flows should be included in the assessment by use of DK model for all catchments. Furthermore, the design of vertical discretization should

be in full compliance with top and bottom of groundwater aquifers for which the assessment are made. The indicators should be applied for the all catchments.

- The surface water test should be applied in a more consistent way by use of the DK model for all catchments. River setups should be improved, but the assessment should be designed with stations honouring the uncertainties in low flow/and low flow reduction simulation due to pumping. Furthermore, methodology for irrigated areas should be clarified.
- At the moment there is no monitoring data included in the evaluation of quantitative status and no application of level of confidence. This should be included. Monitoring data should be related to aquifer safe yield and environmental flow relevant aspects (complex methods) e.g. groundwater levels, water quality in relation to selected water quality indicators, and environmental flow indicators better describing relevant physical, macroinvertebrate and fish conditions.

5. Summary of evaluation of sustainable groundwater abstraction

Aquifer safe yield. The WFD does not prescribe any fixed relationships between groundwater abstraction and recharge. Instead WFD suggests a water balance TEST where the environmental flow requirements are directly evaluated (in relation to surface water and GWDTE requirements for flow), and the saltwater intrusion test. But since such evaluations are difficult, not only for surface water flow systems but also for GWDTEs, simplified screening criteria are needed which are precautionary and incorporate the water balance for the aquifer (groundwater recharge and horizontal inflow in cases where such are significant). The actual situation is key, e.g. actual groundwater abstraction compared with actual groundwater recharge/horizontal inflow, or actual lowering of groundwater level compared to constraints in hydrogeology (geological layers, screenings of wells, sea level, hydraulic conductivity and porosity, etc.). But due to capture and the significant time lags (years to decades) it is required also to incorporate the groundwater level and/or its trend as indicators (trends in water quality are covered by chemical tests). This can only be satisfactorily assessed when using integrated hydrological models. In Germany, UK and Ireland indicators are applied which comply with Groundwater Directive/WFD guidance document no 18, and which also integrate groundwater levels in such assessments. This is not the case for Denmark, where aquifer safe yield is evaluated based on the Danish indicator 1, which refers to ‗pristine' groundwater abstraction (a situation without pumping), and where groundwater levels trends, and the dynamics of capture are not integrated into the methodology. There is a need for establishing a better knowledge base for actual abstraction-recharge rations for screening purposes (precautionary based), for various hydrogeological setting typologies, scales and exploitation configurations (deep/shallow well fields etc.).

Environmental flow. The surface water TEST represents the other side of the sustainability coin. The literature analysis shows that UK has the most developed and mature methodology regarding the surface water TEST (see Appendix 2) in comparison to the other four evaluated countries. This methodology has a clear distinction between the need for classifying the status according to environmental flow indicators in relation to WFD status assessment and in relation to management of groundwater abstractions and licenses (CAMS). The WFD part of the test mainly link to Q_{95} (a statistical low flow value which is exceeded 95 % of the time), but with distinction between different seasons in the evaluation, whereas CAMS relate to four different flow criteria also including e.g. high flow (Q_{30}). Criteria applied in other countries like the France, Germany and Ireland are mainly qualitative and without any translation of qualitative criteria into quantitative criteria, or yet without quantitative assessments of this flow criteria. Denmark has a long tradition for using the median minimum discharge, Q_{medmin} for water management and it is still used for WFD: However, the Q_{medmin} datasets are generally old and needs updating (mostly collected before 1990). Furthermore the Danish approach is not, like in the UK, searching for a new foundation and knowledge base from habitat models and expert knowledge. In Denmark the knowledge base is from an old document by Danish EPA (Miljøstyrelsen, 1979) aimed as a design basis for waste water treatment plants to maintain good water quality in rivers receiving the effluents from treatment plants. Maybe the biggest difference between Denmark and UK is that there has not been any expert evaluation of the criteria and thresholds used in Denmark, with proper linking to more recent biological monitoring data (macrophytes, fish, morphology/physical conditions and relationships to groundwater abstraction), whereas in England and Wales there has been an ongoing development and data collection of methodology and knowledge base during the last 1-2 decades, which so far has resulted in a set of precautionary thresholds applicable for UK rivers for environmental flow evaluation in relation to WFD and CAMS. There is no doubt that Denmark could learn from UK, not only regarding monitoring and habitat modelling, but also on the policy and water management field in relation to environmental flow assessment, and especially transparency, consistency, relation of indicators to various hydrogeological and biological conditions.

Reference situation. The ultimate aim of the WFD is that the European surface waters achieve ‗good chemical and ecological status' and the European ground waters achieve ‗good chemical and quantitative status'. This requires meeting environmental objectives. The designation of water bodies does not occur on the basis of the same criteria. While Denmark looks at the best status that a water body can achieve, other countries like Germany looks at the initial status irrespective of the status that a water body can achieve. This difference in reference conditions (Kessen et al., 2010) for quantitative status combined with in general larger groundwater bodies in Germany compared to Denmark, has a huge influence on the result of the quantitative status assessment (where Denmark is many times worse than Germany). When compared to France the difference in size of groundwater bodies are even bigger, and here the threshold (SF=1.0) corresponds to the old Water Budget Myth, whereas Denmark and Germany apply an SF threshold ~ SF=0.30.

Confidence. The assessment methodology from UK for the water balance test includes an interesting way of introducing uncertainty in the status assessment (High/Low confidence) in order to operate with four different classes. Similar examples can be found from other European countries (France and Ireland). The rationale behind this is to provide an indicator for prioritizing action (where four classes are better than two). Confidence in poor status will be reported as ―high", and ―low", depending on the test. ―Low" confidence will usually mean that further investigation should be carried out as a priority to improve confidence and measures taken. It is stressed that the assessment of confidence in status should not be used as the only driver for instigating measures. Good status groundwater bodies may require higher priority attention, if they are predicted to fail either the trend objective in the long term or some other measure of the risk of future deterioration in status. Confidence in good status will be reported either ―high" or ―low"; being defined as follows: ―High" confidence will usually mean that the only requirement is to assess potential deterioration using surveillance monitoring. ―Low" confidence is associated with a more limited evidence base. Further monitoring will be required to improve the level of confidence. Similar levels of confidence without doubt also could be relevant for Denmark.

Groundwater dependent terrestrial ecosystems (GWDTEs) are main challenges regarding best practices. Of the four quantitative test criteria, the GWDTEs in many countries (including Denmark) have not yet been incorporated in the river basin management plans. Better illustrations of how environmental flow concept can be applied for the assessment of

GWDTEs is required (Grath and Hall, 2012). The discussions on these challenging issues in working groups in EC and elsewhere are still ongoing, and next steps include a summary report for a workshop on groundwater and climate change in the spring 2013 and a WG C Groundwater meeting in Ireland 16/17 April 2013 (Grath and Hall, 2012). According to CIS (2011) key groundwater body dependencies in relation to GWDTEs are to assure discharge of groundwater via springs and seepages, maintenance of an upward hydraulic gradient from the groundwater body to GWDTE, maintenance of an upward flow of groundwater to GWDTEs and water saturation of the soil/soil moisture. Almost by nature, GWDTEs are site specific, and thus complex methods are required. Furthermore, the scales of GWDTEs in many cases are challenging, since they require detailed groundwater and surface water models. Others evaluate that for a thorough handling of water quantity and water quality issues of GWDTEs, complex methods will be required that include water quantity as well as water quality issues in order to develop indicators for application at the complex level. In the Appendix 1 on Requirements in data and modeling, GWDTEs are not addressed specifically, since the requirements in data and modeling still need more clarification regarding best practices.

Climate change impacts. WFD qualitative and quantitative status tests are sensitive to climate change impacts (EC, 2012b). Predicting climate-change effects on groundwater is challenging and uncertainties are present in all steps of the process, from greenhouse gas emission scenarios to global climate models and the downscaling methods applied to adapt their projections to the scale of aquifers, and finally to hydrological models and the effects of climate change on vegetation and recharge dynamics (Aeschbach-Hertig and Gleeson, 2012; van Roosmalen et al. 2009). Climate change will impact the water balance, groundwater levels, flow regime (minimum, mean and maximum flow), temperature and sea level and therefore impact aquifer safe yield and environmental flow requirement assessments. Salt water intrusion evaluations will require monitoring and complex methods. Water balance test will require a reassessment of available groundwater resources, due to changes in groundwater recharge, abstraction (irrigation) and environmental flow requirements, which eventually indirectly also may require adjustment of SF values. Accumulated impacts from abstraction on low flow in rivers could also be affected. Finally, climate change will have complex impacts on GWDTEs which need assessments, monitoring, etc. Some of the impacts will first be significant in 50-100 years, but planning should consider a time period that is longer than the six-year cycle.

6. Conclusions and recommendations

6.1 Conclusions

6.1.1 Sustainable groundwater abstraction – scope of report

Sustainable groundwater development can be understood in a very broad context including aspects such as (i) economic and social sustainability; (ii) sustainability of greenhouse gas emission; (iii) sustainable groundwater abstraction; and (iv) sustainable land use management preventing groundwater pollution. The present report only deals with sustainable groundwater abstraction that includes two key elements: (i) avoidance of significant adverse effects to the aquifer due to abstraction (*aquifer safe yield*); and protection of ecosystem viability (*environmental flow*).

6.1.2 Scientific state-of-the-art

Framework for classifying methodologies
A variety of definitions and methodologies have been reported in the international literature. Today it is commonly accepted that the requirements for a groundwater abstraction to be sustainable imply that no unacceptable impacts occur to either the aquifer itself or to the associated aquatic ecosystems receiving baseflow from the aquifer. We have adopted the following definitions:

- *Aquifer safe yield* is the amount of groundwater which can be pumped from an aquifer without unacceptable negative impacts on groundwater level and water quality, compared to the pre-developmental, virgin situation.
- *Environmental flow* is flow regime characteristics, i.e. the quantity, frequency, timing and duration of flow events, rates of change and predictability/variability, required in order to maintain specified, valued features of the ecosystem.

The methodologies for assessing aquifer safe yield and environmental flow can be classifies in two categories:

- *Screening methods*, which are relatively *simple* and have low requirements to local data. Due to their low knowledge base they are characterized by a high level of uncertainty. If they are designed with built-in precautionarity, they are well suited for national screening purposes.
- *Investigative methods*, which are more *complex* and requires more resources to implement. They are site specific with maximum use of local data and knowledge and use more sophisticated process based modelling tools, hence resulting in less uncertainty compared to the screening methods. They are administratively more difficult, why they are typically used in situations, where the screening methods suggest that there may be sustainability problems. Investigative methods for aquifer safe yield can for instance be based on solute transport groundwater models, while habitat models are examples of tools that can be suitable as investigative methods for environmental flows.

Simple, screening methods

When using screening methods the experience reported in the literature suggests the following guiding principles for designing sustainability criteria:

- Multiple criteria should be used, because they provide more robust characterization of the status of aquifers and ecosystems.
- The two aspects of aquifer safe yield and environmental flow must be seen as inseparable, implying that the selected set of criteria should include both aspects and that dynamic, coupled groundwater/surface water models should be applied to support the analyses.
- Many criteria are scale dependent, implying that the numerical threshold value for good/poor status change if the criteria are used for different sizes of groundwater bodies and catchment areas. Therefore the criteria should generally not be used for scales above/below those for which they have been calibrated.
- Climate change may significantly affect the groundwater recharge and ecosystems and should hence be accounted for.
- Uncertainties in the assessments should be characterized and properly communicated. The methodologies used in the UK regarding confidence in assessments appear suitable and operational at a screening level.
- Aquifer safe yield is often characterised by an indicator being an exploitable fraction of the recharge in a virgin situation (without groundwater abstraction). This has the weakness that it does not account for the fact that the abstracted water comes from increased recharge, i.e. less surface near flows to streams, and reduced aquifer flow to streams and increased (capture). Therefore, an indicator based on the ratio between the abstraction and the actual recharge calculated with abstraction, is a more sound indicator.
- Environmental flow goes much beyond a minimum and static flow regime of rivers. Thus the entire flow regime, including minimum flows, seasonal patterns, flood regime and rate of hydrological alterations are important.

Complex, investigative methods

Many cases of sustainability studies using complex investigative methods have been reported in literature. They use a variety of more or less sophisticated modelling tools dependent on the availability of local data and the character of the problem to be addressed, reflecting that investigative methods should be designed case dependent.

Groundwater dependent terrestrial ecosystems (GWDTE)

No screening methods for sustainability of GWDTEs were found in the scientific literature reflecting that the knowledge base for GWDTEs generally are poorer than for aquifers and river systems. The EC CIS Guidance document 18 prescribes that GWDTE tests should be performed, but does not explain how to do it. Most countries have not implemented GWDTE tests in the first WFD round. GWDTEs often cover areas that are much smaller and hence more dependent on local conditions than groundwater bodies and rivers systems. This makes it very uncertain to characterise the status of GWDTEs with national screening tools, why appropriate methodologies are often characterised as being more complex process based models based on substantial local data.

6.1.3 The four Danish indicators (Henriksen et al., 2008)

Based on the first version of the National Water Resource Model (DK-model) four Danish indicators were developed and used to characterise the sustainability of the groundwater abstraction in Denmark (Henriksen and Sonnenborg, 2003; Henriksen et al., 2008). The four indicators are:

- *Indicator 1*: Maximum groundwater abstraction = 35% of natural recharge to aquifer (calculated without abstraction)
- *Indicator 2*: Maximum groundwater abstraction = 30% of actual groundwater recharge to aquifer (calculated with actual abstraction)
- *Indicator 3*: Maximum reduction of annual streamflow = 10%
- *Indicator 4*: Maximum reduction of low flows = (5%, 10%, 15%, 25%, 50%) depending on the ecological objective of the river reach.

The first two indicators are related to aquifer safe yield. Indicator 1 does not account for the significance of capture, while this is included in Indicator 2. Most international studies use an indicator like Indicator 2 accounting for the important capture aspects, but with thresholds varying between 30% and 100%. The threshold values for the Danish indicators (35% and 30%) were established from comparison of water abstraction and aquifer conditions on Sjælland, but they have never been tested for other areas of Denmark.

Indicators 3 and 4 reflect environmental flow aspects. The overall threshold values are in the same order of magnitude as typically found in the international literature, which however, includes examples of threshold values that are more differentiated spatially (between river types) and temporally (different values for different seasons).

The Danish criteria were developed for spatial scales ranging between 300 and 2,000 km^2, corresponding to 50 subareas in Denmark. Tests have subsequently shown that the actual threshold values vary with the spatial scale for which they are applied.

6.1.4 Practice in Denmark during the first WFD planning cycle

The Danish methodologies used during preparation of River Basin Action Plans during the first WFD planning cycle were simple, screening tools that can be characterised by:

- The criteria were inspired by the four indicators from Henriksen et al. (2008). However typically one, or in a few cases two out of the four indicators were used.
- The four tests recommended by CIS were not all implemented. In some cases they were wrongly implemented (e.g. where groundwater recharge was not based on the entire groundwater aquifer).
- Different methodologies and different threshold criteria were applied in different parts of the country with a lack of transparency in the use. Hence there was no national screening based on a standardised methodology.
- There was a high degree of confusion and mixed ways of calculating groundwater recharge and groundwater abstraction to aquifers.

- The manners in which groundwater bodies in some areas, due to local data and knowledge, were accepted as having good status in spite of a negative output from the screening tests, were not transparent.

6.1.5 Practice in other countries

Review of implementation practices in England, Ireland, Germany and France resulted in the following key findings:

- There are considerable differences in implementation practices among the countries. The recommended CIS tests are used to different degrees in different countries
- All countries are working towards enhancing the knowledge base for the second round of WFD implementation.
- Some countries are, from a scientific point of view, ahead of Denmark in one or more aspect, and one country (England) is generally the country with the most advanced knowledge based practices.
- Some countries, including Denmark, used practices that in some aspects were not scientifically sound.
- EC demands a larger degree of transparency in calculations of groundwater availability and environmental flow requirements in connection with the CIS water balance test.
- Some countries associate a confidence to the results of the various tests of GOOD/POOR status. This may be useful in prioritizing resources for further detailed studies.

6.2 Recommendations – water management

6.2.1 Principles in water management

- Multiple criteria for aquifer safe yield and environmental flows should be used as the basis for characterizing the quantitative status of groundwater bodies. The criteria for good and uncertain status should be scientifically based, clearly described and generally accepted. They should be calibrated and validated against data for different hydrogeological and eco-hydrological regimes.
- Criteria belonging to the group of simple screening methods (Table 2.1 and Table 2.2) that are easy to use and require little data should be developed with a build-in precautionarity. The groundwater bodies that successfully pass these screening tests should be classified as having a good quantitative status. Groundwater bodies which fail should be classified as having uncertain status. For those, the status must be revaluated by use of complex investigative methods requiring more efforts in terms of local data and more sophisticated methods. A characterization of the results of the screening with an evaluation of level of confidence should be used for guiding prioritisation of resources for investigative methods.
- The criteria should be regularly updated, e.g. in connection with each WFD cycle, as new data, knowledge and tools become available.

6.2.2 Knowledge needs toward third WFD planning cycles

Toward third WFD planning cycle the following knowledge gaps have been identified:

- A sustainability database should be established. This database should comprise data on the status and stresses for groundwater and surface water bodies in order to obtain a database that can be used for calibrating and testing various indicators and threshold values for future national screening.
- Methodologies for investigative level evaluation with general applicability in Denmark should be developed:
    - Development of new investigative methods for evaluation of aquifer safe yield (groundwater level, groundwater quality, solute transport etc.)
    - Development of new investigative methods for evaluation of environmental flow (habitat models, incorporation of environmental flow knowledge etc.)
    - Development of new investigative methods for evalutation of terrestrial groundwater dependent ecosystems (GWDTEs)
- Groundwater abstraction strategies, operational issues and monitoring in order to assure sustainable groundwater abstraction.
- Guidance documents for investigative methods should be developed.

6.2.3 Tools for the second WFD planning cycle in Denmark

- Use national screening methods based on standardised methodologies:
    - Standardised methodologies to be used across Denmark
    - Multiple criteria should include at least one indicator for aquifer safe yield and one indicator for environmental flow.
    - Provide a quick calibration of thresholds based on data for different hydrogeological and ecological typologies
    - Assess thresholds based on precautionarity
    - Characterize the result of the screening also with level of confidence (uncertainty)
    - Provide guidelines for use of tools, methods and indicators (thresholds) for the national screening with a possible re-classification (assure full transparency)
- Conduct investigative methods for the groundwater bodies having uncertain status, i.e. those who did not result in good ecological status according to the nationwide screening

6.2.4 Possible indicators and tests for the second WFD planning cycle in Denmark

Below is given some estimates of possible indicators and threshold values for use in the second WFD planning cycle. It must be emphasized that the threshold values should be calibrated/validated against actual data.

- Thresholds for screening of good/uncertain status (indicators should be calibrated and validated):
    - Aquifer safe yield: Abstraction less than 30 % of groundwater recharge (SF = 0.3)

        - Calculated with actual abstraction (-indicator 2 according to the four Danish indicators)
        - Applied for entire groundwater body having a sufficient size (include horizontal groundwater flow from adjacent groundwater bodies)
        - Possibly, make a differentiation of thresholds for different hydrogeological settings
    - Environmental flow: Max. reduction of Q_{medmin} due to abstraction less than 10 % (or Q_{95} < 10 %)
        - Calculated for catchments larger than 30 km^2
        - For catchments less than 30 km^2 results should be aggregated from several headwater reaches/tributaries
        - Possibly, differentiate between various reach types, upper/lower reaches
    - Other possible tests:
        - Salinity test: Change in flow directions e.g. near the coast
        - Chemical status: Can supplement water balance test in relation to aquifer safe yield evaluation
        - Groundwater dependent terrestrial ecosystems (GWDTE test): Change in groundwater table

The review underlines, that not only the use of indicators lack validity and transparency, also the way they are used and the foundation for their use as part of groundwater governance is a problem. Therefore, capacity development is needed targeting policy makers and groundwater managers in state, municipalities and actor networks, in order to strengthen the proper use of tests and indicators used at the screening and investigative level according to state-of-the-art good practices, and in order to enable a better understanding and acceptance of new indicators and tests used in water management and across different scientific disciplines (e.g. hydrology-ecology and hydrology-geology/geochemistry), as new methods and tests are developed and applied as part of WFD implementation. Here, there is inspiration available from e.g. England and Wales where such new learning processes have created an ongoing development of knowledge base and indicators/tests related to habitats and environmental flow requirements (see Appendix 2).

A further consolidation of indicators/methods used in Denmark for screening and investigative level assessments of aquifer safe yield and environmental flow require as described above new knowledge and a further evaluation of robustness and uncertainty related to the used indicators/tests. Such an evaluation at the same time can strengthen/focus the monitoring and further target it towards selected indicators/tests in use. Here there is a clear possible synergy if the ongoing monitoring conducted by water companies (operational management of wellfields and water supply) is better coordinated with monitoring related to WFD by the state, which require that this monitoring is fully standardised and available. Development of new investigative methods has a strong research component. Therefore, extern financing for such studies is possible (e.g. strategic research programmes). Finally, there is a need for improved data sets and models (see Appendix 1).

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Appendix 1 Requirements in hydrological data and models

Ap. 1.1 Scope of appendix - topics covered

Many factors affect the performance of a hydrological model. The most important ones include:

- The quality and quantity of the available data for the particular study (Section Ap. 1.1).
- The conceptual understanding including the knowledge of the governing processes (Sec Ap. 1.2).
- The suitability of the model code selected for the particular study (Section Ap. 1.2).
- The competence of the modeller and the modelling practice adopted (Section Ap. 1.2).
- The type of problem to be addressed such as simulation of streamflow and water balance or simulation of the effects of groundwater abstraction on streamflow (Section Ap. 1.3).
- The scales used for calibration, prediction and modelling (Section Ap. 1.4).

The discussion of these issues in the following sections does not intend to be general and all inclusive but is rather brief focussing on aspects of particular relevance for the topics of the present report. This leads to conclusions regarding how to improve model performance when predicting streamflow depletion due to groundwater abstraction (Section Ap. 1.5)

Ap. 1.2 Data quality and quality in Denmark

The national monitoring network in Denmark has been dramatically reduced during the past decade. Between 2005 and 2010/2011 the number of recording stations have been reduced for river discharge from 415 to 244, for precipitation from 432 to 228 and for groundwater heads from 800 to 400 (the number of wells is only for Sjælland), i.e. a reduction by about 50% (Refsgaard, 2012). This will inevitably have consequences in terms of a reduced reliability of the models that are calibrated against less data.

He et al. (submitted) reported a study on the relationship between the station network and the accuracy of a model calibrated against different densities of precipitation station network. They analysed the impact of the number of precipitation gauges for the Ringkøbing Fjord catchment and found that the model performance for river discharge simulation measured in terms of the Nash-Sutcliffe coefficient averaged over 10 discharge stations decreased from 0.76 to 0.57 when reducing the number of precipitation gauges from 87 to 40. The reduction in performance was generally larger for smaller catchments.

While the study of He et al. (submitted) focussed on the effects of precipitation gauge network, similar analyses could relatively easy be performed for river discharge stations and groundwater head observation wells. However, no systematic analysis of the relationship between the density of the station network and the performance of hydrological models has been performed in Denmark. Therefore, we do not have sufficient knowledge to optimally design the monitoring network, even if we had information about the desired level of accuracy. The effects of changing the station network will be different for different model variables such as discharge, groundwater heads and stream-aquifer interaction as well as for different spatial scales.

Based on the analyses of He et al. (submitted) and our general modelling experience, we expect the following general relations between changes in the existing monitoring network and model performance:

- Considering the broad range of sophisticated modelling software (model codes) that has been developed during the past two decades, the fundamental barrier for improved model performance is the limitation of available data. This applies both to system data such as geology and to time series data such as climate, discharge, groundwater heads and concentrations.
- The reduction in discharge stations during the past decade has mainly been designed with the objective to minimise the adverse impact on the assessment of nutrient load to the marine ecosystems, implying that the most downstream stations covering large catchments have been preserved, while many small stations at upstream tributaries have been closed. While this is a rational priority for assessments of nutrient loads to coastal areas, it is not optimal in connection

with modelling of streamflow depletion due to groundwater abstractions, where it is often critical to have discharge data for small tributaries. Thus, different study objectives will result in different monitoring network designs.

- The deterioration/improvement of model performance with decreasing/increasing data coverage is more pronounced at smaller scale than at larger scale.

Ap. 1.3 Conceptual understanding, model code and modelling practice

Adoption of a good modelling practice based on state-of-the-art methodologies is of paramount importance for a successful modelling study. Recommendations in this respect can be found in Refsgaard et al. (2010).

A good conceptual understanding of the hydrogeological conditions is generally recognised to be the maybe most important condition for achieving a good groundwater model (Sonnenborg and Henriksen, 2005; Refsgaard et al., 2010). Another important precondition of a modelling study to become successful is the selection of a suitable model code. With the availability of several well proven and multi-purpose modelling software systems comprising a variety of options for process descriptions, e.g. MODFLOW and MIKE SHE, this is most often not a problem. Guidance on selection of model code and how to configure it for particular study purposes can be found in Sonnenborg and Henriksen (2005) and Refsgaard et al. (2010).

Ap. 1.4 Differences in accuracy for simulations of streamflow and streamflow depletion due to groundwater abstraction

The capability of hydrological models to simulate water balance in Denmark has been subject to numerous studies during the past 30 years. Recently, Refsgaard et al. (2011) provided recommendations on how to use the existing hydrometeorological data, e.g. precipitation data dynamically corrected to account for gauge undercatch, to best ensure sound water balance simulations. An overview of the capability of the national water resources model (DK-model) to simulate water balances for Denmark on this basis is provided in Stisen et al. (2012). One of the results from this study is shown in Fig. Ap.1. It should be noted that the DK model uses distributed national data on geology, soils, land use, etc. but uses uniform hydraulic parameter values such as hydraulic conductivities, time constants for drain flow and leakage coeffic I ents for stream beds, within each of the six model domains.

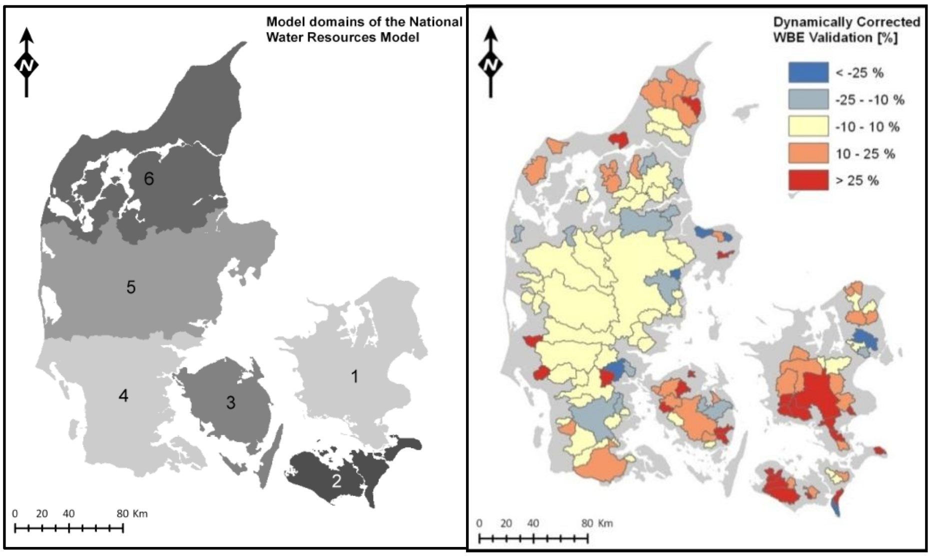

Fig Ap. 1 The six domains of the Danish national water resources model, DK-model (figure to the left). Performance of the DK-model (figure to the right) for simulation of water balance in a validation period. The numbers are % deviations between observed and simulated discharges averaged over the four years validation period (Stisen et al., 2012).

Due to calibration hydrological models can usually simulate annual runoff (Fig. Ap.1) and discharge hydrographs outside calibration periods under similar conditions (model validation) with good accuracies that are only slightly less than the performance during the calibration periods. When models are used for prediction of changed conditions beyond their calibration base the accuracy is generally deteriorating with less performance the further the extrapolation from the calibration conditions is (Refsgaard et al., 2012). Model prediction of changes in streamflow caused by changes in groundwater abstraction is usually such a situation, because there are no data for the situation after the groundwater abstraction (if data exist there would be no reason to use a model for prediction). Model predictions of changed conditions are therefore less accurate than the accuracy of model validation for un-changed conditions. How much the prediction accuracy is reduced depends on the local conditions and the site specific model, and no general conclusions can be made on this.

Table Ap.1 shows an example of model predictions for reductions in minimum flows using historical data from three catchments, where discharge measurements existed for periods before the start of major groundwater abstractions. The results in Table Ap.1 show that the model gives a very good prediction for Havelse Å, while there are deviations by factors of 2 and 4 for the two other rivers. At a first glance this is not very impressive and may raise concern with respect to directly using the results for water management decisions. When evaluating this performance the following factors should, however, be taken into account:

- The model is intended as a large scale national model. This is reflected in a rather coarse grid scale (500 m) and a calibration strategy, where the calibration parameters are identical for the entire Sjælland, which, as illustrated in Fig. Ap.1, causes significant internal spatial biases that disappear when aggregating over the whole model domain.
- The model is not targeted particularly towards simulation of low flows.

While it may be questioned whether the model used in ALECTIA (2010) is sufficiently accurate as a basis for controversial water management decisions and while a better performing model would definitely have been desirable, the alternative of not using a model at all does not appear very attractive, as this would lead to pure guesswork.

Table Ap.1 Comparison of model predictions and observations for changes in low flows (median of annual minimum flows) from a period without to a period with groundwater abstraction (ALECTIA, 2010)

River	Køge Å, Lellinge ($126\ km^2$)	Græse Å, V. Hørup, Lindebjerg ($25\ km^2$)	Havelse Å, Strø Bro ($102\ km^2$)
Simulated reduction in low flow (l/s)	11	11	24
Observed reduction in low flows (l/s)	48	20	28

In another study Seifert et al. (2012) analysed the performance of six hydrological models (200 m grid) that were identical except for differences in geological conceptualisations for predictions of amongst others changes in streamflow in the Langvad Å catchment near Lejre resulting from changes in groundwater abstraction. They found a large variation between the six models, where some of them were relatively good in predicting changes in low flows while the performance of others were relatively poor.

Modelling scales

Model performance depends on several types of spatial scales:

- *Model grid size*. As low flows are very dependent on baseflow, often from deeper aquifers, it is very important to be able to simulate the stream-aquifer interaction as good as possible. The flows from aquifers to streams are calculated as a function of the differences between the groundwater heads and the water table in the river. It is therefore important to have the correct elevations of the rivers and the river valleys in the model. As many river valleys are narrow, coarse model grids like 500m will many places not be able to sufficiently resolve the topography in the river valleys. This is a limitation for the model performance.

- *Catchment size*. Performance is generally better for simulations of larger catchments, because random errors at smaller scales, e.g. caused by uncertainty on local rainfall or local geology, will cancel out when aggregating to larger catchments. This is illustrated in Fig. Ap.2 based on DK-model simulations of discharge.
- *Scale of calibration*. Performance can generally be improved by adding more free parameters to the calibration. If the DK-model had not used the same hydraulic parameter values throughout a model domain, but different parameters in different catchments, the performance could be improved. Fig. Ap.1 for example shows that there are different biases in the water balances for different catchments on Sjælland. These biases could be removed or at least reduced significantly, if different parameter values had been used for different catchments. The drawback of using too many local parameters is the risk of overparameterisation, whereby the calibration ends up in pure curve fitting leading to a reduced predictive capability, when a model subsequently is used under changed conditions outside the calibration period.

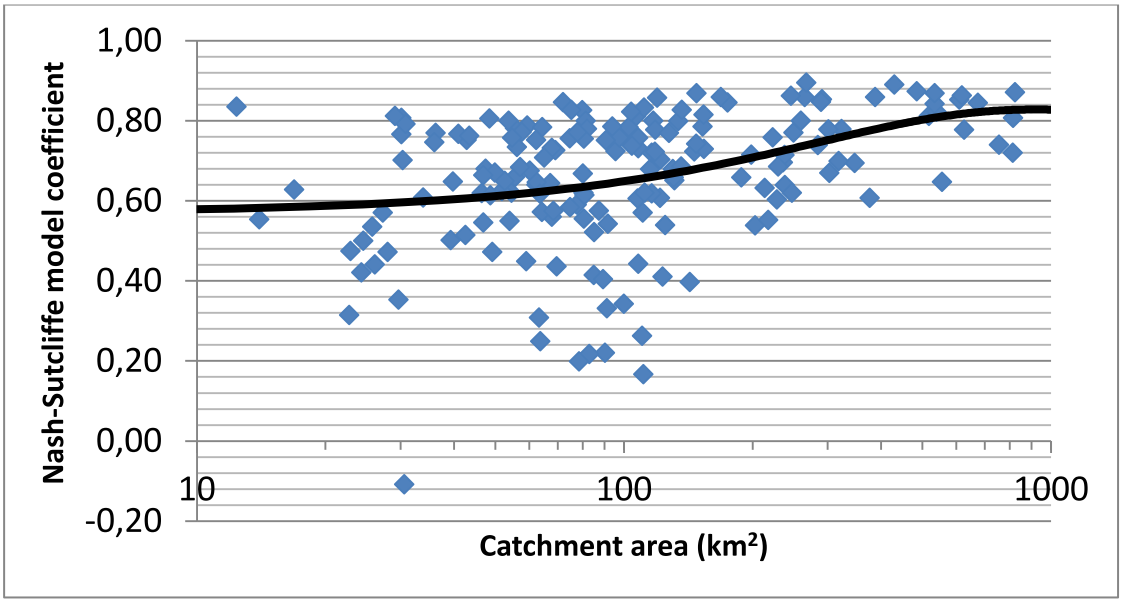

Fig. Ap. 2 Performance of the DK-model for simulation of discharge at 182 stations during the period 1990-2010 as a function of catchment size. Each diamond represent a discharge station and the full line is a power function fitted to show the trend. Data from Højberg et al. (2012).

How to improve model performance

When establishing a hydrological model with the purpose of predicting the impacts of groundwater abstraction on streamflow and groundwater heads in riverine areas the following aspects should be considered:

- The model should be a coupled groundwater-surface water model that is able to simulate dynamic conditions.
- The model should have a spatial discretization that is sufficiently fine to resolve the river valley topography for the particular site. This implies grid sizes in the order of 50 – 250 m depending on topography, location of abstraction wells and catchment size.
- Good modeling practices such as those described in Refsgaard et al. (2010) should be followed.
- The model should be calibrated specifically for the catchment in question.
- The calibration and validation strategy should best possible make use of dynamic aspects of the data, ie. including data and objective functions on fluctuations of groundwater heads and data from periods with different groundwater abstraction or test pumpings, if such data exist. To enable assessment of the expected model prediction accuracy validation tests should be performed that to the extent possible include some of the dynamic aspects and aspects of changed pumping conditions.

- The option of collecting additional field data to support the model calibration and validation should be considered. Data of particular importance that may be obtained in short campaigns include: i) synchronous measurements of discharge at many locations during a low flow situation; ii) synchronous measurements of groundwater heads in existing wells, preferably from the same period as the synchronous discharge measurements; iii) continuous recording of streamflow and groundwater heads in wells located close to the river.

Appendix 2 Environmental flow indicator (EA, UK)

The following fact sheet from January 2013 (the fact sheet is regularly updated: http://www.environment-agency.gov.uk/business/topics/water/119927.aspx) sets out how the Environmental Flow Indicator (EFI) was developed, how it is used and what assumptions can be drawn from its application. EFI describes an acceptable percentage deviation from the natural flow represented using a flow duration curve. This percentage deviation is different at different flows, and also depends on the ecological sensitivity of the river to changes in flow.

The background for EFI are flow standards for the Water Framework Directive developed by UK Technical Advisory Group (Acreman et al., 2005) and UK TAG (2008) which have been adapted to set the EFI. The EFI is set through expert opinion and at a level to support good ecological status.

Here a short introduction to the fact sheet based on UK TAG (2008) is provided. For promoting the sustainable use of water the effect of flows and water levels on ecological communities is vital. Identified parameter used important to the ecological status e.g. biological elements like fish, macrophytes and macro-invertebrates, is per cent change from natural flow conditions. The abbreviation, QN95, stands for the natural flow that is exceeded for 95 per cent of the time. QN95 is the 95 percentile for normally 10 years of flow records e.g. a statistical low flow value selected for the evaluation in relation to WFD.

For river reaches of High Status UK TAG (2008) prescribes that for flows less than QN95 the allowed per cent change from the natural flow is up to (<=) 5 %. For flows greater than QN95 the allowed per cent change is up to 10 %. However, the fact sheet only focuses on Good Status.

For Good Status the standard proposed were derived from UK TAG (2008) expressed as the maximum permitted amount of change from the natural flow (the abbreviations QN60 and QN70 for example refer to the natural flow exceeded for 60 or 70 per cent of the time, see Table 2.4 and 2.5 in Chapter 2 of the present report). The standards thus reduce the degree of change allowed during the spring and summer, and have been designed to protect macrophytes in spring and early summer, and macro-invertebrates and fish in the later summer and early autumn. A summary of the outputs from UK TAG (2008) is given in Table 1 of the fact sheet.

Table 3 of the fact sheet describe the use of EFI in Water Framework Directive, where EFI is used in the hydrological classification to identify where reduced river flows may be causing or contributing to a failure of good ecological status (the compliance assessment). Compliance has been assessed at low flows (Q95) using recent actual scenario, and shows where specific scenario flows are below the EFI (in case not compliance indication of how much the non-compliant Band 1, 2 or 3 is provided as part of the classification). The Abstraction Sensitivity Band (ASB 1-3) refer to sensitivity of different river types (Table 1 of fact sheet for flow < Q95; referring back to Table 2.4 and 2.5 of the present report).

Environmental Flow Indicator

What it is and what it does

January 2013

The Environmental Flow Indicator (EFI) plays a crucial role in the management of Water Resources in England and Wales. This factsheet sets out how the EFI was developed, how it is used and what assumptions can be drawn from its application.

- **EFIs are used to indicate where abstraction pressure may start to cause an undesirable effect on river habitats and species. They don't indicate where the environment is damaged from abstraction.**
- **Compliance or non-compliance with the EFI helps to indicate where flow may or may not support Good Ecological Status.**
- **The EFI is not a target or objective for resolving unsustainable abstractions. It is an indicator of where water may need to be recovered. The decision to recover water in water bodies that are non-compliant with the EFIs should only occur when supported by additional evidence to provide ecological justification.**
- **In Catchment Abstraction Management Strategies (CAMS) EFIs help to indicate where water may be available for future abstraction without causing unacceptable risk to the environment.**

What is the EFI?

The Environmental Flow Indicator (EFI) is a percentage deviation from the natural river flow represented using a flow duration curve. This percentage deviation is different at different flows. It is also dependant on the ecological sensitivity of the river to changes in flow.

The EFI is calculated within the Resource Assessment and Management (RAM) framework. This assessment gives an indication of where and when water is available for new abstractions. Where the assessment fails a more detailed assessment is required to understand if current abstractions and use of full licensed quantities are threatening the long term health of the river ecology.

Development

Flow standards for the Water Framework Directive (WFD) developed by the UK Technical Advisory Group (TAG) (Acreman et al, 2005 and UK TAG, 2008) have been adapted to set the EFI. The EFI is set through expert opinion and at a level to support good ecological status. The adaptation was necessary for the Environment Agency to use it within the existing abstraction regulatory regime.

UK TAG (2008) identified the percentage deviation from natural flow (that supports GES) for differing river 'types' and at different flows: low flows (Q95) and flows above Q95. A summary of the outputs from this report is given in Table 1. This was translated for use within the Resource Assessment Methodology to be used in the Environment Agency's Water Resources work (EA and Entec, 2008, Hall, 2008).

River type	**Flow > Q95**		**Flow < Q95**	
Months	*Mar - Jun*	*Jul - Feb*	*Mar - Jun*	*Jul - Feb*
Predominantly clay. South East England, East Anglia and Cheshire plain	25%	30%	15%	20%
Chalk catchments; predominantly gravel beds; base-rich	15%	20%	10%	15%
Hard limestone and sandstone; low-medium altitude; some oligotrophic hard rock	20%	25%	15%	20%
Non-calcareous shales; pebble bedrock; Oligomeso-trophic; Stream order 1 and 2 bed rock and boulder; ultra-oligo trophic torrential	15%	20%	10%	15%
Months	*Oct - Apr*	*May - Sep*	*Oct - Apr*	*May - Sep*
Salmon spawning & nursery (not chalk rivers)	15%	20%	10%	15%

Table 1: Flow standards for UK river types for supporting good ecological status given as the percentage allowable abstraction of natural flow (UKTAG, 2008).

Use in Catchment Abstraction Management Strategies

The Catchment Abstraction Management Strategy (CAMS) process has 3 main stages to it:

- Water resource availability assessed using our Resource Assessment Methodology (RAM),
- The licensing strategy,
- ‘Measures’ appraisal process – that is identifying and delivering things we want to change

Resource availability is expressed as a surplus or deficit of water resources in relation to the EFI. This is calculated by taking the natural flow of a river, adding back in discharges and taking away existing abstractions. This results in a scenario showing both a recent actual and fully licensed river flow. The difference between the fully licensed scenario flow and EFI gives us the amount of water which is available for abstraction and when it is available.

The Environment Agency abstraction regime uses fixed ‘hands-off flows’. These give a more effective use of water from the environment by enabling abstraction to cease at set flows, but also enable abstraction from periods of time when more water is available. The EFI is defined for four conditions, ranging from naturally low (Q95) to naturally higher (Q30) flows. To help manage abstraction at higher flows and protect flow variation greater percentages of flow is allowed to be abstracted. Table 2 shows the percentages of flow to be abstracted at three different sensitivities to abstraction (abstraction sensitivity bands) at different flows.

	high flow	→	→	low flow
Abstraction Sensitivity Band	**Q30**	**Q50**	**Q70**	**Q95**
ASB3. high sensitivity	24%	20%	15%	10%
ASB2. moderate sensitivity	26%	24%	20%	15%
ASB1. low sensitivity	30%	26%	24%	20%

Table 2: Percentage allowable abstraction from natural flows at different abstraction sensitivity bands.

Details of all the abstraction licences are recorded in CAMS ledgers (volumes and location and discharges). The ledgers are updated every time a new licence is issued, changed or revoked and are updated to inform future licensing decisions.

The EFI is a fundamental component of how we set out clearly what water is available where and when for potential abstractors. This is detailed in licensing strategies that are developed for each CAMS catchment and are available on the Environment Agency's internet site. The strategies set out the hands off flow and other conditions that will be applied to licence applications. They also include any local constraints that potential abstractors will need to be aware of such as higher levels of environmental protection for designated conservation sites, or where local information has shown that different amounts of water are available in the catchment.

Use in Water Framework Directive

The EFI is used in the hydrological classification for WFD to identify the water bodies where reduced river flows may be causing or contributing to a failure of good ecological status. This is called the compliance assessment. Compliance has been assessed at low flows (Q95) using recent actual scenario.

The compliance assessment shows where specific scenario flows are below the EFI, and indicates by how much. This is used to identify areas where flows may not be supporting good ecological status and target further investigation of what measures are needed to achieve good ecological status.

The degree of non-compliance has been split into three compliance bands, each band indicating the certainty that flow conditions do not support good ecological status. The compliance bands help to prioritise action where the abstraction pressure, and therefore the risk of not supporting good ecological status are greatest. The percentage below natural flow for each compliance band is shown in Table 3.

	Flow adequate to support GES	**Flow not adequate to support GES: Low to Moderate Confidence (uncertain)**		**Not adequate to support GES: High Confidence (quite certain)**
Abstraction Sensitivity Band	***Compliant with EFI***	***Non-compliant Band 1*** (up to 25% below the EFI at Q95)	***Non-compliant Band 2*** (25-50% below the EFI at Q95)	***Non-compliant Band 3*** (up to 50% below the EFI at Q95)
ASB3. high sensitivity	<10%	<35%	<60%	>60%
ASB2. moderate sensitivity	<15%	<40%	<65%	>65%
ASB1. low sensitivity	<20%	<45%	<70%	>70%

Table 3: The percentage difference from natural flows for each compliance band and how this relates to supporting good ecological status (GES). Percentages given are the range below natural flow for the relevant abstraction sensitivity band.

Glossary

Abstraction Sensitivity Bands (ASB)	There are three abstraction sensitivity bands assigned to each water body in England and Wales: ASB1 – low sensitivity; ASB2 – moderate sensitivity and ASB3 – high sensitivity. Each of the ASB has a different EFI associated with it allowing less abstraction in high sensitive sites and more in sites with lower sensitivity. Each of these sensitivity bands was developed from assessment of 3 components: Physical typology – using the river 'types' used in Acreman et al (2005). Macroinvertebrate typology – using expected Lotic index for Flow Evaluation (LIFE) scores Fish typology – using identification of a fish 'guild' expected under particular physical parameters. Scores and confidence ratings from each component are combined to give the overall ASB for the waterbody.
Good Ecological Status	Good Ecological Status (GES) defines a water body as only being a little way from being in its totally natural state. It is the main objective of the WFD to return all water bodies to this near natural condition, although it does recognise that this will not always be possible. GES covers a variety of elements that give an indication of the health of a water body and its ability to support life. Hydrology is a supporting element for good ecological status - but in some situations, flow may be the limiting element for biology and for achieving good ecological status.
Natural Flows	The river flow that would have occurred without any human influences. This is calculated by starting with a gauged flow/recent actual flow and adding back in the abstractions and taking out the discharges. It can also be calculated from other surface water or groundwater models.
Scenario Flow	The scenario flow that is generated by denaturalising the natural flow taking into account abstractions and discharges operating at their recent actual rate (**recent actual scenario**) or abstractions operating at their full licensed limit and discharges operating at their recent actual rate (**fully licensed scenario**).
Waterbody	A manageable unit of surface water, being the whole (or part) of a stream, river or canal, lake or reservoir, transitional water (estuary) or stretch of coastal water. A 'body of groundwater' is a distinct volume of underground water within an aquifer.
WR GIS	The WR GIS uses ArcView. The abstraction, discharge, natural flows and complex impacts information from the CAMS ledgers is uploaded onto this central system. The WRGIS uses this information to calculate the current resource availability for each water body at four flow percentiles.