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Large scale screening of seawater intrusion risk in Europe

Methodological development and pilot application along the Spanish Mediterranean coast

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Summary

Seawater intrusion caused by overabstraction has become an important problem in coastal areas of the Mediterranean. Intrusion processes highly depend on complex local conditions and studies generally focus on specific problems of individual aquifers. The European Environmental Agency has carried out a survey of observed intrusion cases to provide an overview of the problem over Europe. For policy making and water management at EU level, information on the main driving forces, emerging risk areas and possible future trends are necessary and require more detailed analysis and integration of models and GIS approaches. However, seawater intrusion problems have hardly been addressed at large scale due to the strong impact of local conditions and the problems related to data collection for large geographical areas.

To fill this gap and to explore the potential use of readily available data sources, we developed a simple screening methodology for large scale assessment of seawater intrusion risk along the Mediterranean coast of the EU based on a two tiered assessment procedure. Tier 1 is a simple risk assessment based on the balance of groundwater recharge and water abstractions for coastal areas. A positive net groundwater recharge results in transition of the saltwater-freshwater interface to a new equilibrium state. No equilibrium exists with negative net recharge (over-abstraction) and seawater is drawn into the aquifer compensating freshwater losses. Tier 2 provides a quantitative characterization of seawater intrusion for standardized aquifers considering generalized local geological conditions: A simple analytical intrusion model calculates freshwater loss and seawater progression for specified combinations of aquifer properties, aquifer dimensions and boundary fluxes (recharge and abstractions). An unstressed quasi-natural state (recharge only) is compared with a stressed state adding the current level of abstractions.

The estimation of groundwater recharge and the spatial disaggregation of national water abstraction data are still tentative and future improvement of the procedures is necessary.

A pilot application was carried out for the coast of SE-Spain. As geological data did not support an assessment of individual ‘true’ coastal aquifers, a schematic approach was chosen: A calculation of abstractions and recharge was carried out for a set of hypothetical aquifer domains of different size and repeated in regular intervals along the coastline. The fraction of aquifer domains falling into the transient or over-abstraction class defines the local risk of transient intrusion or intrusion due to over-abstraction. The seawater intrusion model (Tier 2) was applied to each aquifer domain using local geology and boundary fluxes with standardized aquifers.

Freshwater loss and progression of the saltwater-freshwater interface illustrate the potential severity of potential intrusion processes.

The analysis does not predict real occurrences of seawater intrusion and can not be used for quantitative assessment of real-world intrusion problems, as the approach is not applied and validated to real aquifers. Due to the lack of suitable hydrogeological data this situation is unlikely to change in future.

The approach supports screening of intrusion risk over large geographical areas based on local relation of abstraction and recharge. The methodology is principally promising, even though input data used for the pilot studies are still based on tentative approaches and need to be replaced by more detailed analysis of statistical information and modelling. Water demand and groundwater recharge as basic input data can be linked to model applications, allowing assessment of future intrusion problems in the context of scenario analysis (for example assessing changes of intrusion risk based on climate change, land use changes and changes in water demands).

1 Introduction

Coastal aquifers in the Mediterranean are put under pressure due to the concentration of the Mediterranean population in coastal areas, due to intensive tourism especially during the summer period and due to the large extent of irrigated agriculture. Groundwater abstractions for agricultural purposes and securing drinking water supply have caused seawater intrusion in many coastal aquifers of the Mediterranean area (EEA 1999, Mediterranean Groundwater Working Group 2007). In coastal aquifers, increased abstractions and overabstractions result in subsequent intrusion of seawater.

Ensuring long-term sustainable rates of extraction from water resources and promoting sustainable water use and prevention of intrusion are explicit policy targets of the EU, being set out in the 6th EAP (6 EAP 5.6 (p.45-46) [1]) and the Water Framework Directive (WFD Article 1 [2]). Application of the WFD shall ensure a balance between abstraction and recharge of groundwater and a chemical composition of the groundwater body not exhibiting the effects of saline intrusion, with the aim of achieving good groundwater status by 2015 (WFD Article 1 [2]). With respect to seawater intrusion, the WFD requires that for achieving good quantitative status of a groundwater body "alterations to flow direction resulting from level changes may occur temporarily, or continuously in a spatially limited area, but such reversals do not cause saltwater or other intrusion, and do not indicate a sustained and clearly identified anthropogenically induced trend in flow direction likely to result in such intrusions". National, regional and local authorities are required to introduce measures to improve the efficiency of water use and to encourage changes in agricultural practices necessary to protect water resources (and quality) (WFD Article 9 indent 2 [2]).

Studies on seawater intrusion generally focus on individual aquifers systems at local or regional scale. Intensive monitoring and complex and data demanding modelling tools are applied to analyse and to solve intrusion problems. Such studies are necessary to develop mitigation strategies at local level, but they can not provide a general (large scale) overview of the problem to support policy making at international level. A large scale assessment of intrusion risk including analysis of main pressures and possible impacts can support policy making at EU level and facilitate communication with authorities and stakeholders at different regional levels. Only little research is being undertaken at large scale due to the strong dependency of intrusion processes on the local settings and approaches are very limited. Rajan et al. (2006) presented a global assessment of freshwater losses in coastal aquifers due to sea level rise in response to climate change. A general assessment of sea-level rise impacts in sea-water intrusion was presented by Werner and Simmons (2009). Such studies remain very general and focus on sea-level rise. More specific large-scale assessments, taking into account potential effects of water use (or over-use) in coastal regions and giving better insight into local conditions of driving forces as well as potential impacts are still missing.

EEA carried out a survey of observed seawater intrusion cases and released the latest update of a map of observed seawater intrusion in 2005 (Figure 1). This survey provides a good overview on the current distribution of seawater intrusion problems and is likely to capture the high-risk zones. However, it can not detect emerging risks nor can it be used to make projections into the future. Considering the increasing need to estimate environmental impacts of environmental change, policies and socioeconomic developments also at large scale, it would be desirable to provide a link between large scale assessment of seawater intrusion and scenarios of driving factors.

The work presented in this report aims at developing an approach for continental-scale assessment of sea-water intrusion risk focussing on potential impacts of water abstractions. We describe a methodology developed for large scale assessment of seawater intrusion risk at continental scale using generally available data and simple modelling approaches. The core issue of the proposed methodology is the local balance of recharge and water abstractions as a risk-indicator. A pilot application along the Spanish Mediterranean coast demonstrates the potential capabilities of the methodology and explores limitations of data availability and challenges in data processing.

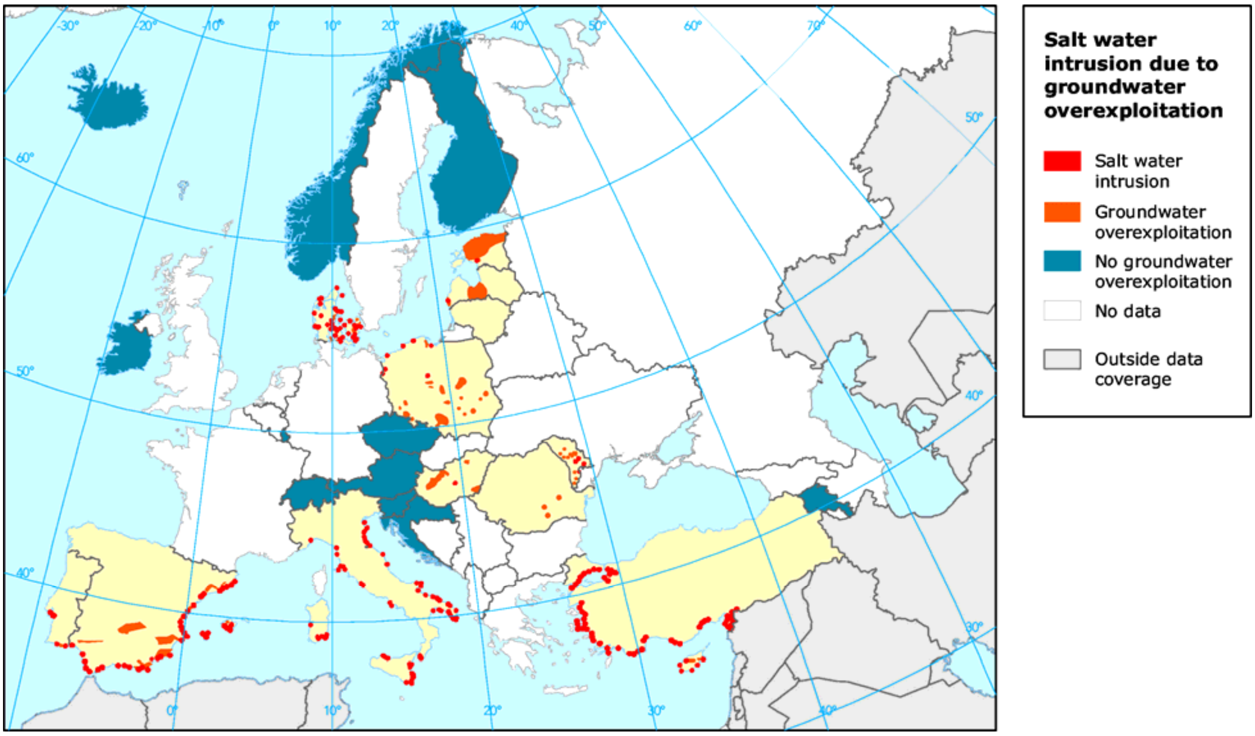

Figure 1: Observed cases of seawater intrusion (EEA European Environment Agency, 2005)

2 Theoretical background

Coastal aquifers vary in size from local scale aquifers that are a few square kilometres or less in areal extent to multi-layer, regional-scale aquifer systems that are tens of thousands of square kilometres in areal extent. Although such multilayer, regional aquifer systems may be discontinuous locally, hydrological they act as a single system on regional scale (Sun and Johnston, 1994). The most important water bearing formations are unconsolidated sands and gravels; semi-consolidated sands; and, among the consolidated rocks, carbonates (lime stones), sandstones and fractured crystalline rocks (e.g. granites).

The general pattern of fresh ground-water flow in coastal aquifers is from inland recharge areas towards coastal discharge areas. Fresh groundwater comes in contact with saline ground water at the seaward margins of coastal aquifers. The seaward limit of freshwater in a particular aquifer is in a state of dynamic equilibrium. This dynamic equilibrium stage is controlled by the amount of freshwater flowing through the aquifer, the thickness and hydraulic properties of the aquifer and adjacent confining units, and the relative densities of saltwater and freshwater, among other variables. The freshwater and saltwater zones within a coastal aquifer are separated by a transition zone within which there is mixing between freshwater and saltwater due to dispersion and molecular diffusion. The width of the transition zone can range from some hundred metres up to several kilometres.

For simplification purposes, the freshwater saltwater transition zone can be assumed as a sharp boundary (freshwater-saltwater interface). The depth of the interface in an unconfined aquifer can be approximated by the so-called Ghyben-Herzberg relation.

$$z = \frac{\rho_f}{\rho_s - \rho_f} h \quad \approx 40h \qquad [1]$$

where z = thickness of freshwater zone below sea-level [L], h = thickness of freshwater zone above sea-level [L], ρ_s = density of saltwater [M/L^3], ρ_f = density of freshwater [M/L^3].

This equation relates the elevation of a water table to the depth of the interface between the freshwater and saltwater zones of the aquifer. Roughly speaking, each metre of fresh groundwater elevation above sea-level would correspond to a freshwater column of 40m of below the sea-level.

Groundwater withdrawals and other human activities disturb the natural balance between freshwater and saltwater in coastal aquifers. Basically, all activities that lower groundwater levels and reduce fresh groundwater flow to coastal waters ultimately cause seawater to intrude coastal aquifers. Apart from groundwater abstractions, also the construction of drainage canals and reduced recharge (e.g. in urbanized areas) affect the seawater-freshwater balance. Also the placement of water abstraction sites (and management of pumping) and the distribution of other measures affecting groundwater tables can have considerable effect on the saltwater freshwater interface and intrusion processes. Consideration of these factors is therefore essential to develop local management strategies.

As long as there is a net flow towards the sea, a reduction in flow due to water abstractions will result in a relocation of the seawater-freshwater interface towards a new dynamic equilibrium position. The time needed for transition from one dynamic equilibrium stage to another is considerable, in the order of tens of years at least or even centuries (Bear et al. 1999). It depends on the degree of alteration of the groundwater table and/or recharge fluxes and the areal extent of these changes as well as on hydrogeological properties of the aquifer system. Given the long time frame to restore an equilibrium position of the interface, saltwater intrusion is most frequently in a transient state. In case of abstractions exceeding freshwater recharge (overabstractions), seawater is drawn actively into the aquifer by an inward hydraulic gradient replacing abstracted freshwater. This process will continue until either all freshwater is lost or a net seaward flow is restored.

The seawater-freshwater balance is also affected by natural processes superposed to human interventions. Such processes include:

- Climatic changes affecting groundwater recharge and sea-level
- Eustatic and tectonic movements resulting in sea-level fluctuations
- Land subsidence due to compaction processes

Depending on local geologic conditions and history, it is therefore possible that a transition zone is not in equilibrium with actual conditions. Barlow (2003) presents the effect of Pleistocene sea level fluctuations on the transition zone in the Northern Atlantic coastal plain aquifer system. Fossil groundwater and brines may be sources of salinization, when mobilized by human pumping, as observed for example in the Floridan aquifer system (Barlow 2003).

Saltwater can contaminate freshwater aquifers through several pathways: lateral intrusion from the ocean (also referred to as encroachment), upward intrusion from deeper zones and downward intrusion from coastal waters. A specific type of vertical intrusion is saltwater upconing, which refers to the upward movement of saltwater in response to pumping at a well or well field.

Aquifers can roughly be classified in porous aquifers, fractured rock aquifers and karstic aquifers, which differ considerable in hydrologic behaviour. Porous aquifers in consolidated and unconsolidated sediments store and conduct water in primary pores. Fractured rock aquifers (crystalline and metamorphic rocks) store groundwater in secondary pores made up of cracks and fissures. Flow direction depends on the orientation of the secondary pore system.

Compared with unconsolidated aquifers and consolidated crystalline rocks, karstic aquifers have specific features causing a different hydrodynamic behaviour, also affecting saltwater intrusion processes. Karstic aquifers are dual porosity systems. The primary porosity is small and has low permeability. Water flowing through fractures gradually and irregularly dissolves the carbonate material creating fractures, fissures, caverns and channels, which make up the secondary porosity. In the secondary pore system, flow is often non-Darcian. Discharge is often concentrated in a few springs, which also includes submarine springs. Karstic aquifers respond rapidly to natural recharge, human activities (groundwater abstraction) but also to the effect of tides, storm surges and pressure variations, also and especially in terms of saltwater intrusion. Convective flow of fluids dominates the mixing of saline and fresh water rather than hydrodynamic dispersion. Therefore the fresh-saline relation only partly follows the Ghyben-Herzberg relation in karstic regions. A description of groundwater flows and saline intrusion processes in karstic areas with mathematical formulas is complex or even impossible, unless very rough assumptions are permitted (Bear et al. 1999).

Due to the complex nature of saltwater intrusion processes and the dependency on local and regional hydrogeological conditions, there are little to no studies available focussing on large scale assessment (in terms of sub-continental to global scale) of saltwater intrusion processes. Full understanding of the specific local and regional conditions is required to prevent or reduce saltwater intrusion or to develop aquifer management strategies. On global scale, Ranjan et al. (2006) investigated freshwater losses in coastal aquifers due to sea-level rise in response to climate change. Though modelling approach and parameterization of coastal aquifer systems was subject to considerable simplifications, the approach enabled regional differences of aquifer response to be analysed.

Modelling approaches to calculate the position of the seawater-freshwater interface and to simulate seawater intrusion processes include analytical and numerical approaches.

Analytical models (e.g. Glover, 1964, Fetter 1972, Strack 1976) provide solutions for simplified aquifer geometries and boundary conditions, where only few input data are required or available. Analytical solutions normally do not directly solve “real-world” problems (Cheng and Ouazar, 1999). They often serve as instructional tools and as tools for first-cut engineering analysis, providing first insights into specific problems when only few input data are available.

Various numerical models exist to simulate one- to three-dimensional, density dependent and transient groundwater flow and transport. Well known models are for example SEAWAT 2000 (Guo and Langevin 2002, Langevin et al. 2003), SUTRA (Voss and Provost 2002) and FEFLOW (Diersch 2005). Numerical approaches have the potential to simulate intrusion processes under transient conditions and for complex geologic conditions. They are therefore especially suited to analyse intrusion processes of specific aquifer systems. Limitations are put at large scale due to excessive data demand and numerical considerations. "A sophisticated model without the support of reliable input data does not provide more accurate result (Cheng and Ouazar, 1999)."

Lumped aquifer models calculate water and solute balance of aquifer systems (e.g. Milnes and Renard, 2004). As for analytical approaches, aquifers have to be transformed into simplified (lumped) representations. Temporal evolution of water storage and solute concentrations can be taken into account.

Intrusion problems require development of specific management strategies. Possible measures could be:

- Demand management (modernization, consumption behavior, restrictions)
- Injection barriers to restore seaward hydraulic gradients at the coastline and protect pumped regions of the aquifer further inland
- Placement of pumping wells (distance from the coast, aquifer layer and depth)
- Management of pumping wells (volume, intervals)
- Alternative water sources
- Artificial groundwater recharge
- Adaptation of land use
- Deconstruction of drainage systems (restore higher groundwater table)

Concerning large scale assessments, the following key points can be summarized:

- Any change of net groundwater recharge will result in a transition of the seawater-freshwater interface towards a new equilibrium position. In case of overabstraction, seawater is drawn actively into the aquifer replacing freshwater.

- Many factors affect intrusion processes, but the balance of recharge and abstractions can be seen as the main driving force. These main forces are also accessible for large scale analysis, while local factors can not adequately be accounted for.
- Human induced intrusion may be superposed by natural processes. The intended analysis will confine to intrusion caused by human activities.
- For modelling purposes, there is a trade-off between model complexity, data demand, uncertainty resulting from availability of input data and simplifications according to the scale and spatial resolution of the analysis.
- Seawater intrusion is a dynamic process. However, for risk and impact assessment, changes have to be analysed with respect to an initial or reference state. As the initial states of coastal aquifers are practically unknown at large scale, a hypothetical reference state needs to be defined.

3 Materials and Methods

3.1 General outline of the risk assessment

Available data don't support characterisation of single aquifers and the related water fluxes. Assessing seawater intrusion based on 'real' aquifers is therefore difficult. The large scale perspective of this study further requires a schematic approach capable of treating a large area. We abandon the concept of single aquifers and assess a theoretical seawater intrusion risk based on hypothetical aquifers instead. Boundary conditions can be derived for these hypothetical aquifers from available data.

The assessment was carried out for specified **assessment points** distributed regularly along the coastline. For each assessment point a set of virtual **aquifer domains** of different spatial extent was defined. The further processing was based on a two-tiered approach.

Tier 1 is an assessment of seawater intrusion risk based on the balance of recharge and abstractions as main driver of seawater intrusion processes. The recharge balance is the only factor which is accessible for large scale assessment, while numerous local effects such as the placement and management of abstraction wells, drainage activities, geological conditions and other can not be account for. At each assessment point, net recharge and water balance terms are calculated for each virtual aquifer domain. A risk indicator was based on the number of aquifer domains falling into a specific risk class.

Tier 2 is a semi-quantitative assessment of intrusion characteristics based on a simple seawater intrusion model. The analytical seawater intrusion model estimates freshwater loss, groundwater level decrease and progression of saltwater wedge. Boundary conditions (net recharge) were taken from Tier 1 and geological properties were assigned according to dominant parent material of each aquifer domain. A set of pre-defined standard **aquifer settings** (confined-unconfined, deep-shallow) was assigned to each assessment point and each aquifer domain.

While Tier 1 allows for general assessment of seawater intrusion risk, Tier 2 supports more specific assessment and quantitative comparison for standardized aquifer configurations and local geologic conditions. A schematic outline of the 2-tiered assessment approach is presented in Figure 2.

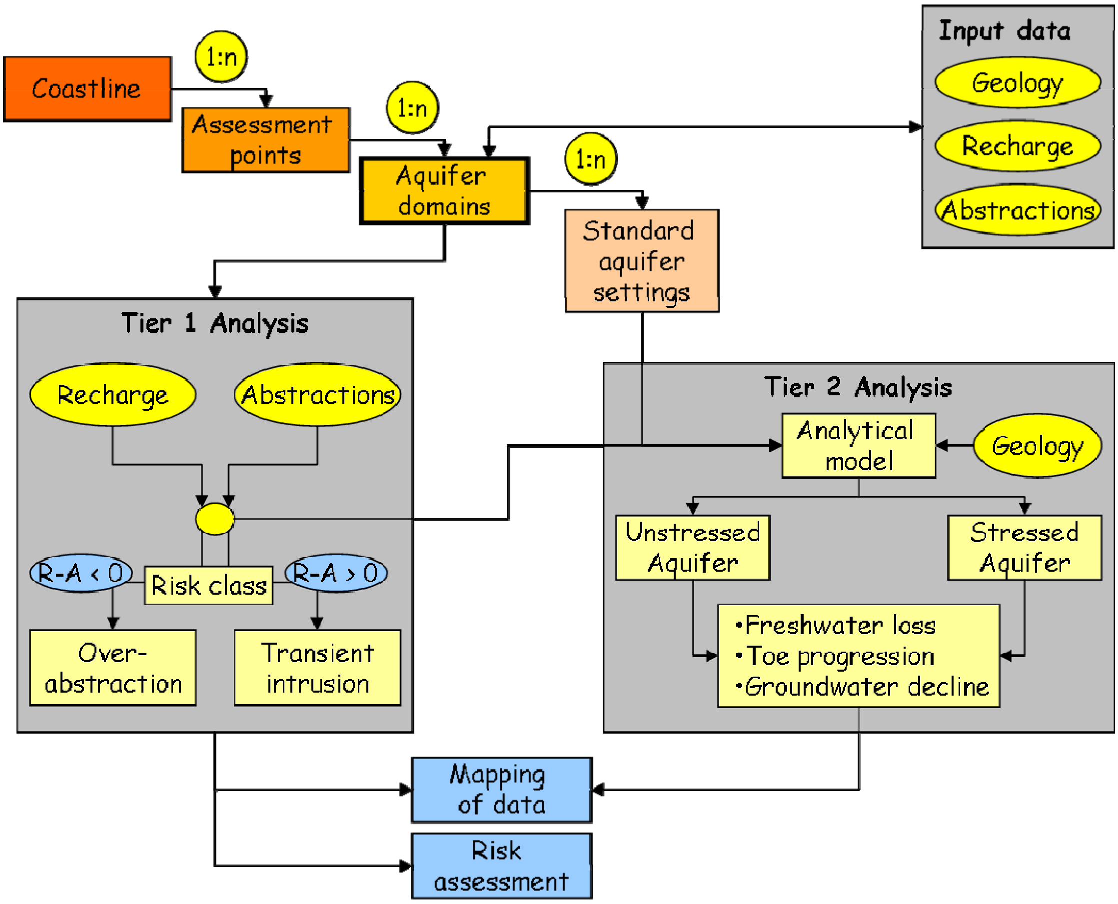

Figure 2: General outline of the 2-tiered assessment approach

3.2 Tier 1 – Recharge balance assessment

3.2.1 Defining assessment points and aquifer domains

Assessment points were distributed in more or less regular distance along the coastline based on the following procedure:

We created a centreline keeping a constant distance of 2500m from the coastline.

All points of a 2500m raster falling within a certain distance of the centreline (tolerance of 880m) were selected as assessment points.

At each point, we create a set of virtual **aquifer domains**. The aquifer domain is a circle around the assessment point. The aquifer domains increase in size, applying a radius of 2.5 km, 5 km, 10km, 20 km and 40 km. The effective area of the aquifer domain depends on the distribution of land and sea within the area covered by the domain. Ideally, with increasing radius the effective area transforms from a circle to a half circle. Basic input data, such as groundwater recharge, total water abstractions, agricultural water abstractions and substrate were then defined for each aquifer domain as the average values over the domain (most frequent values in case of substrate).

Both, recharge and water abstractions depend on the area of the aquifer domain taken into consideration. Especially in the case of water abstractions, we can only infer the locations of water use but not of abstractions. Abstracted water may locally be distributed within a certain distance, for example to settlements or agricultural areas. Therefore increasing the aquifer domain allows the effect of increasing aquifer size and/or water uses within a certain environment to be considered.

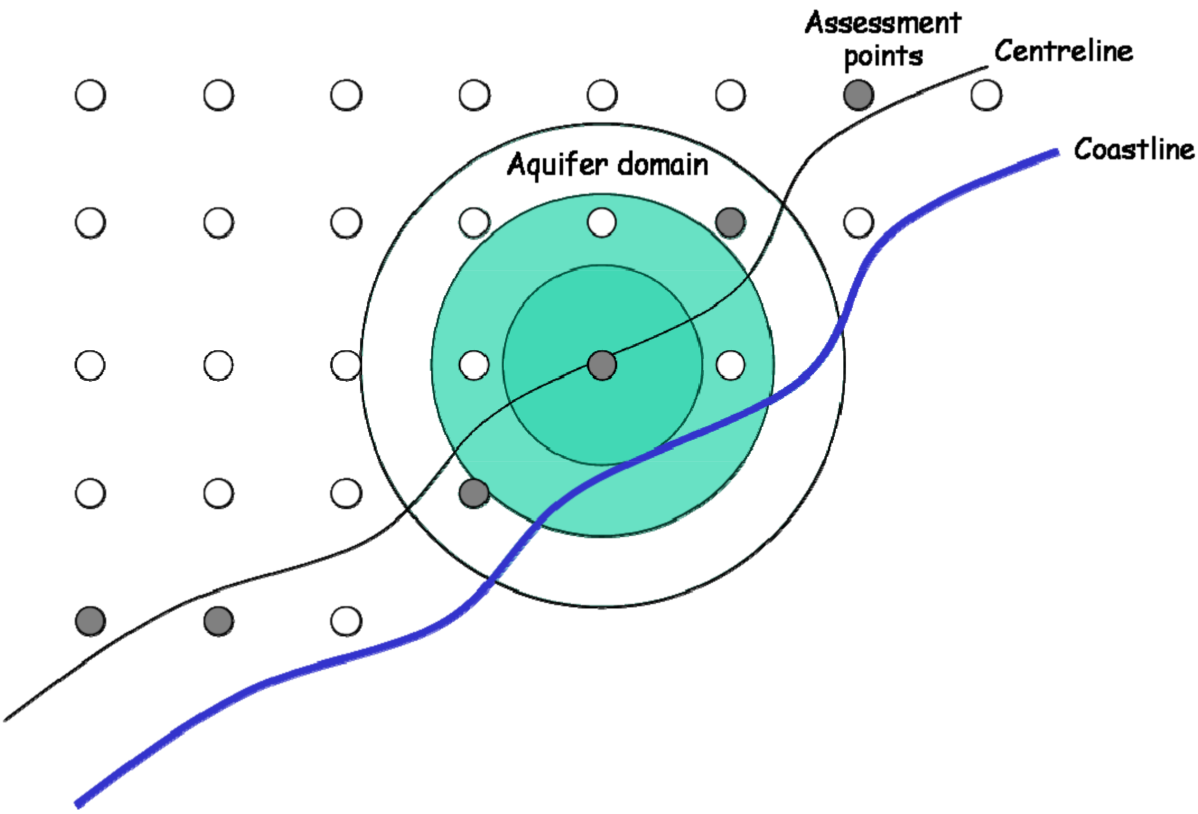

Figure 3: Delineation of assessment points and aquifer domains

3.2.2 Groundwater recharge

Natural groundwater recharge is affected by soil cover, land use, topographic factors controlling surface runoff generation and hydrogeological properties of the underground, depth to groundwater, and climate characteristics. An appropriate estimation of groundwater recharge needs to be based on a general hydrological modelling approach, taking into consideration infiltration, evaporation and the generation of fast surface and subsurface runoff components. For the first testing of the simulation procedure, a general large scale hydrologic modelling approach was not yet available. Therefore groundwater recharge was provisionally estimated based on the annual climatic water surplus.

Climatic data were taken from the MARS-Database, which provides climatic data in a 50x50km station raster for Europe. Based on the MARS climatic data, we calculated monthly water balances and determined the climatic water surplus and water deficit according to the following procedure:

The monthly water balance was defined as

$$WBal_M = \sum_d \left(P_d - ETP_d\right) \qquad [2]$$

where WBal = water balance of month m (mm), P=precipitation (mm), ETP= potential evapotranspiration (mm), M=month index, d=day index.

The annual water surplus was calculated as the sum of positive water balances

$$WSurplus_A = \sum_M WBal_M, \quad M \middle| WBal_M > 0 \qquad [3]$$

where WSurplus = annual water surplus (mm), A = year index.

Water surplus separates into groundwater recharge and surface runoff. Groundwater recharge was estimated according to the following formula:

$$GWR_A = WSurplus_A \cdot \left[1 - \frac{WSurplus_A}{MaxWS}\right]^{0.5} \cdot \left[1 - \frac{Slope}{100}\right]^2 \qquad [4]$$

where GWR = groundwater recharge (mm), Slope = Slope (%), MaxWS = Maximum water surplus (=2000 mm).

The above formula constitutes a decrease of groundwater recharge with increasing slope and limits groundwater recharge also under conditions of excessive recharge in flat areas. Given a range of climatic water surplus from 0 to 1800mm over Europe, groundwater recharge ranges from 0 to approximately 550mm.

Other factors such as land use, soil and geology were not taken into account. Corresponding to the data on water abstractions, the groundwater recharge was estimated for the year 2000.

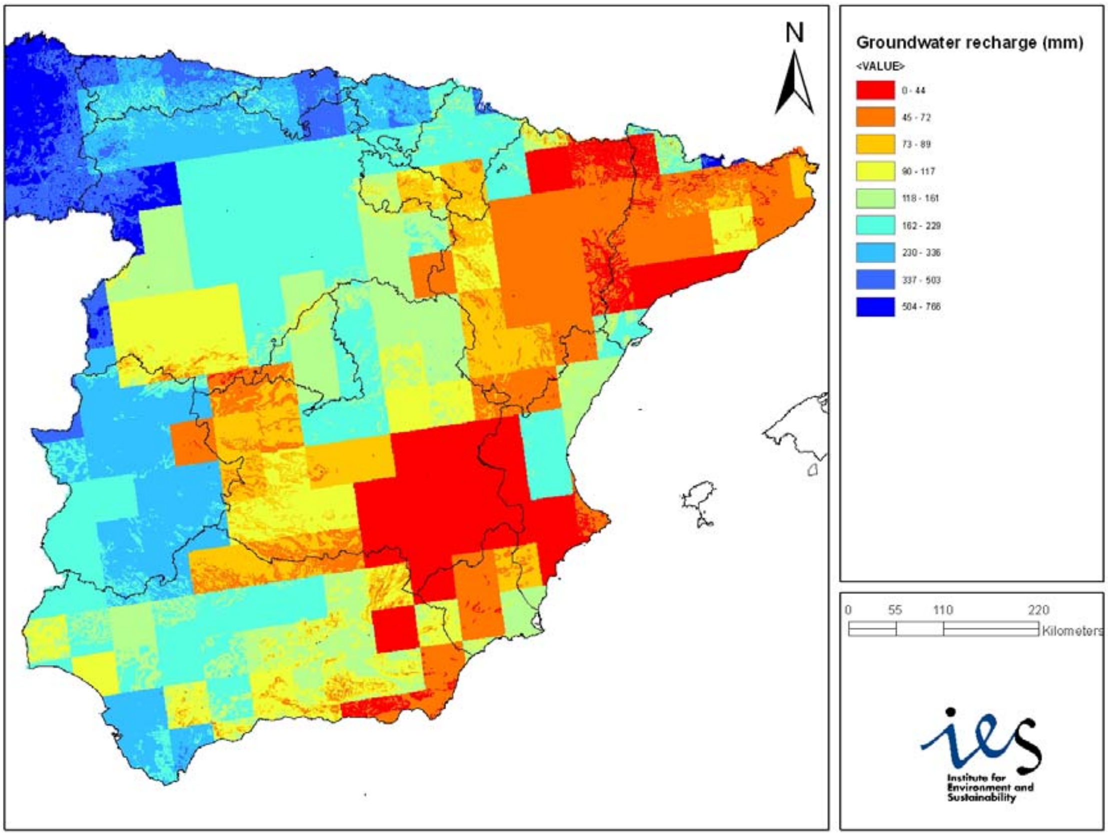

Figure 4: Preliminary groundwater recharge based on MARS climatic database (50x50km) (Example: Spain)

3.2.3 Water abstractions

Water abstractions are used in agriculture (e.g. for irrigation purposes), public and domestic water supply, industrial supply, energy cooling and additional demands for tourism water consumption. Generally, ground- and surface water sources can be used to meet water demands. The water abstractions are the most important information to assess seawater intrusion risk apart from groundwater recharge. For the seawater intrusion assessment, we needed to compile datasets of total water abstractions and for individual water uses with sufficiently high spatial resolution.

Readily available data sources are usually national or regional aggregates. For example, the EUROSTAT data collection currently includes only water abstraction data on national level (Joint OECD/ESTAT questionnaire).

For specific countries and specific water uses it is possible to find data of higher resolution. For example, French data on irrigation abstractions are available at NUTS3 level.

However, even a NUTS3 level is still to coarse to estimate water abstractions in the coastal zone. It was also beyond the capacities of this study to collect and process data from numerous scattered sources. We therefore decided to use the ESTAT data on national water abstractions and disaggregate them to an appropriate level.

The disaggregation procedure followed a two step procedure (Figure 5). First, water abstraction data were disaggregated based on the spatial distribution of suitable proxy measures:

- Public and domestic abstractions - by population to communal level (NUTS5)
- Industrial abstractions - by gross domestic product to NUTS3
- Abstractions for electricity cooling - by electricity production (thermal and nuclear plants) to NUTS2
- Agricultural abstractions by irrigated area to NUTS3

Second, the so derived abstraction data were further disaggregated to a local level:

- Industrial abstractions were disaggregated according to industrial areas as defined in CLC2000 (100m-raster).
- Agricultural abstractions were disaggregated based on irrigated areas as defined in the European Irrigation Map (Wriedt et al. 2008).
- Abstractions for electricity cooling were not further processed. We assumed that cooling water abstractions affect surface water bodies only and are therefore not relevant for the scope of this study.
- Public and domestic abstractions were not further disaggregated and a uniform distribution within each commune was assumed. The communal level already provides a considerable spatial resolution. Even though water use may concentrate on urban and rural settlements, this assumption is not valid for water abstractions. Water distribution within a commune and between communes may take place. Corresponding to the other abstraction types, public abstractions were also converted to a 100m-raster.

Finally, we obtained a raster data set of 100m resolution for each abstraction type. The final raster data sets then serve as a base to summarize water abstractions for arbitrarily defined spatial units, for example coastal areas or catchments or aquifer domains. It has to be pointed out that the final raster data sets reflect the locations of water use rather than of water abstractions. They do not localize water abstractions.

Water may be abstracted from surface water bodies as well as from groundwater bodies. The use of specific water sources depends on numerous factors, including physiogeographic settings, historic evolution, socioeconomic factors and water management. However, from national or even regional data it is not possible to reasonably estimate local water abstractions from specific sources. Even assuming that in coastal plains with good aquifers water may be predominantly taken from local groundwater sources, large distance transport from mountainous areas can not be excluded, especially, when large cities or intensive industry or agriculture require considerable amounts of water. For this analysis, groundwater abstractions in coastal areas were defined as the sum of agricultural abstractions, industrial abstractions and public and domestic abstractions. As pointed out before, cooling water is assumed to be taken from surface waters only. The adopted definition of groundwater abstractions provides an estimation of the maximum water abstractions that can be taken from local aquifers. The real abstractions are probably smaller taking into account local surface water sources or large distance transport.

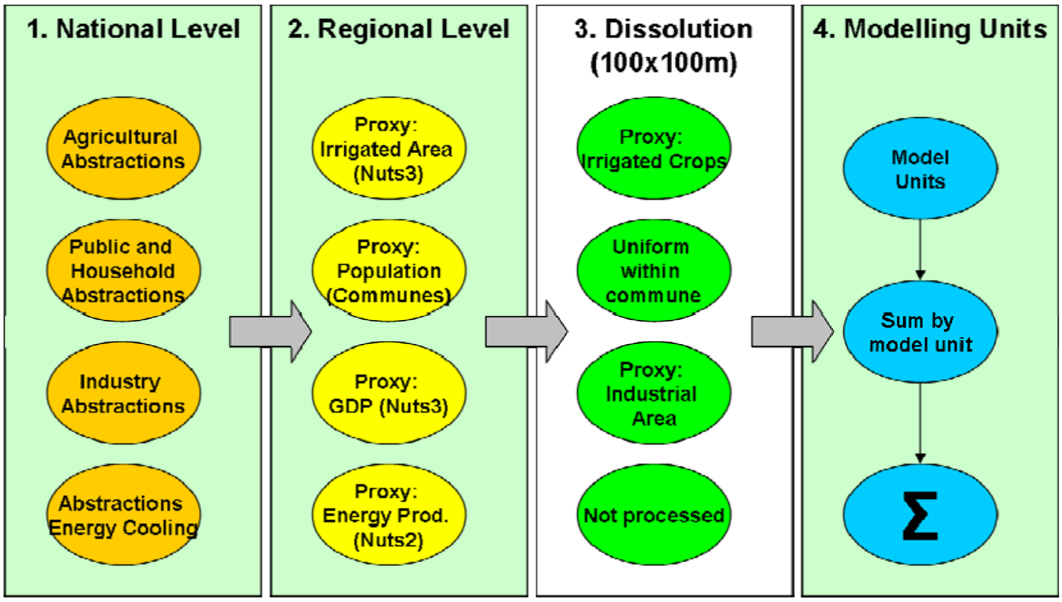

Figure 5: Scheme for disaggregation of water abstraction data

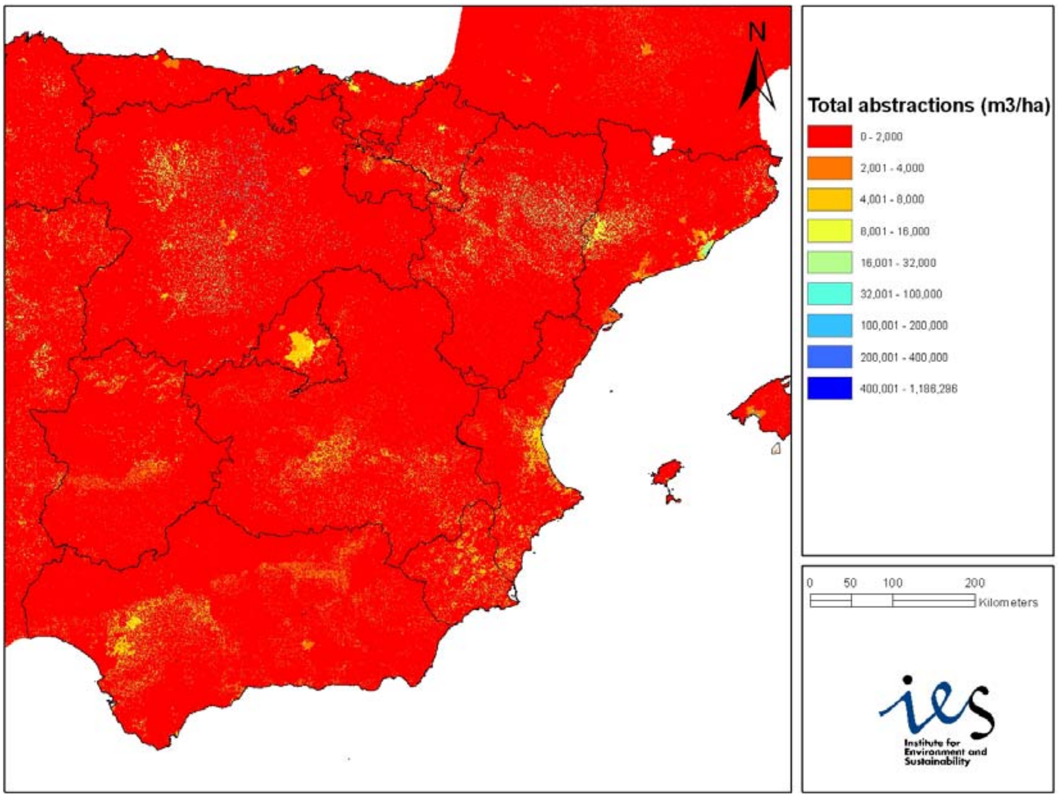

Figure 6: Disaggregated total water abstractions in m3 per ha (Example: Spain)

3.2.4 Evaluation of Tier 1 results

At each assessment point, we obtain a set of results (recharge and abstractions for each aquifer domain). The Tier 1 approach provides the basic information to characterise seawater intrusion risk. Based on the recharge-abstraction balance, we define two general risk classes:

Risk class 1 - Transient intrusion: If the net recharge is positive, groundwater recharge exceeds water abstractions and a net flow is directed seaward. The seawater-freshwater transition zone has an equilibrium position. Seawater intrusion takes place during transition from a previous position to an equilibrium position representing the current state of pressures.

Risk class 2 – Intrusion caused by overabstraction: If net recharge is negative, the aquifer is overexploited. Seawater actively intrudes the aquifer compensating water abstractions. There is no theoretical equilibrium position and intrusion is permanent (at least until a positive net recharge is restored increasing recharge or decreasing abstractions).

At each assessment point, we determined the number of aquifer domains falling into the transient intrusion or overabstraction class. The fraction of aquifers falling into a specific class determines the specific risk of transient intrusion or overabstraction. This risk was classified as: High risk (>0.66), Medium risk (0.33-0.66) and Low risk (<0.33).

Further, we determined the average ratio of water abstractions and groundwater recharge at each assessment point. This abstraction-recharge ratio serves to characterise the potential pressure on water resources. For the transient risk class, the ratio is smaller than one and we classified as follows: moderate pressure (ratio < 0.5) and high pressure (ratio > 0.5). We confined to two pressure classes, considering possible data uncertainties. However, a refined classification better suited to separate classes of very low pressures with negligible freshwater loss and seawater intrusion is possible.

For the overabstraction risk class, the ratio is greater than one and we classified pressure as overabstraction (ratio <5) and extreme overabstraction (ratio > 5). Considering the high uncertainties associated with input data, we refrained from a more detailed classification of pressures in the pilot study. There is also no class without risk, except where abstractions are zero (ratio = 0).

As water abstractions have been compiled per water use sector, the pressure exerted on groundwater by individual sectors can be assessed as well. In this study, we asses the role of agricultural water abstractions only, but other sectors may be assessed in a similar way.

At each location and for each aquifer domain, we calculated two indicators of agricultural pressure on coastal groundwater:

- Share of agricultural water abstractions in total water abstractions, characterizing importance of agriculture as water user.
- 'Agricultural pressure' as ratio of agricultural water abstractions and groundwater recharge, characterizing the individual pressure of agricultural abstraction in the abstraction-recharge balance. Agricultural water abstractions exceeding groundwater recharge (ratio > 1) indicate that agricultural water abstractions alone are sufficient to cause over-abstraction and seawater intrusion processes.

Where high share of agricultural abstractions coincides with high agricultural pressure, agriculture can be seen as one of the main drivers of seawater intrusion at a specific location.

3.3 Tier 2 – model based characterisation of intrusion processes

Quantitative characterization of intrusion processes, combining geological properties, aquifer dimensions and boundary fluxes was implemented in Tier 2 based on an analytical modelling approach (Strack solution, Strack (1976) as described in Bear et al. 1999).

At large scale, available data sources provide only generalized information and can not adequately represent 'true' local conditions. A schematic and simplified aquifer representation is required to facilitate data processing and modelling over a large geographical area. Therefore results of Tier 2 provide indicative information rather than estimates of real-world processes. The analytical approach was considered to be appropriate in this context, as the additional capabilities of complex models are outweighed by data availability, uncertainty and generalisation.

The analytical model calculates the steady-state position of the freshwater-saltwater interface and the freshwater volume for a simplified 2-dimensional aquifer, combining net recharge, inflow and geologic properties of the aquifer. Information on intrusion processes were derived comparing two corresponding steady state simulations with different conditions of abstraction and recharge. Specifically, an initial or 'unstressed' state with unmodified natural groundwater recharge was compared to a 'stressed state', where human water abstractions were considered additionally. Results as freshwater loss, displacement of saltwater wedge and characteristic time allow an indicative characterization of intrusion processes.

3.3.1 Analytical solution

It is generally assumed, that an equilibrium interface potentially exists and no other processes such as sea-level rise or vertical movements of the coastal sediments create permanent disequilibrium. Further it is assumed that recharge and abstractions are evenly distributed within the model domain. The impact of different placement of abstraction wells is not accounted for.

The Strack-Solution (Bear et al. 1999) is defined for confined and unconfined aquifers. The aquifer is separated into two zones, a freshwater zone (zone 1) and a zone where freshwater exists on top of a saltwater wedge (zone 2). The interface of zone 1 and zone 2 is the toe of the saltwater wedge. Potential functions are defined for confined and unconfined aquifers, depending on density difference, depth and thickness of the aquifer and the freshwater head. Also, an independent equation for the potential function is presented, depending on lateral and vertical inflows and hydraulic conductivity. The potential function can be used to solve freshwater head and to calculate depth of the saltwater interface and thickness of the freshwater lens. The model variables are explained in Table 1. The model representations of an unconfined and a confined aquifer are given in Figure 7.

Table 1: Variables of the analytical solution

Variable	Description	Unit
b	thickness of freshwater layer	[L]
d	depth of lower confining layer below sea level (aquifer bottom)	[L]
ξ	depth of saltwater interface below sea level	[L]
B	Aquifer thickness (confined aquifers)	[L]
L	Length of vertical cross section (perpendicular to coastline)	[L]
x	Position along cross section	[L]
h_f	freshwater head	[L]
Φ	potential	[L]
Φ_{Toe}	potential at toe location	[L]
x_{Toe}	Toe location (location of interface between zone 1 and zone 2)	[L]
w	Recharge	$[L^3/ L^2/ T]$
q	Inflow	$[L^3/ T]$
K	Hydraulic conductivity of aquifer	[L/T]
ρs	Density of salt water (1.025)	$[M/L^3]$
ρf	Density of freshwater (1.000)	$[M/L^3]$
FVol	Freshwater volume	$[L^3]$

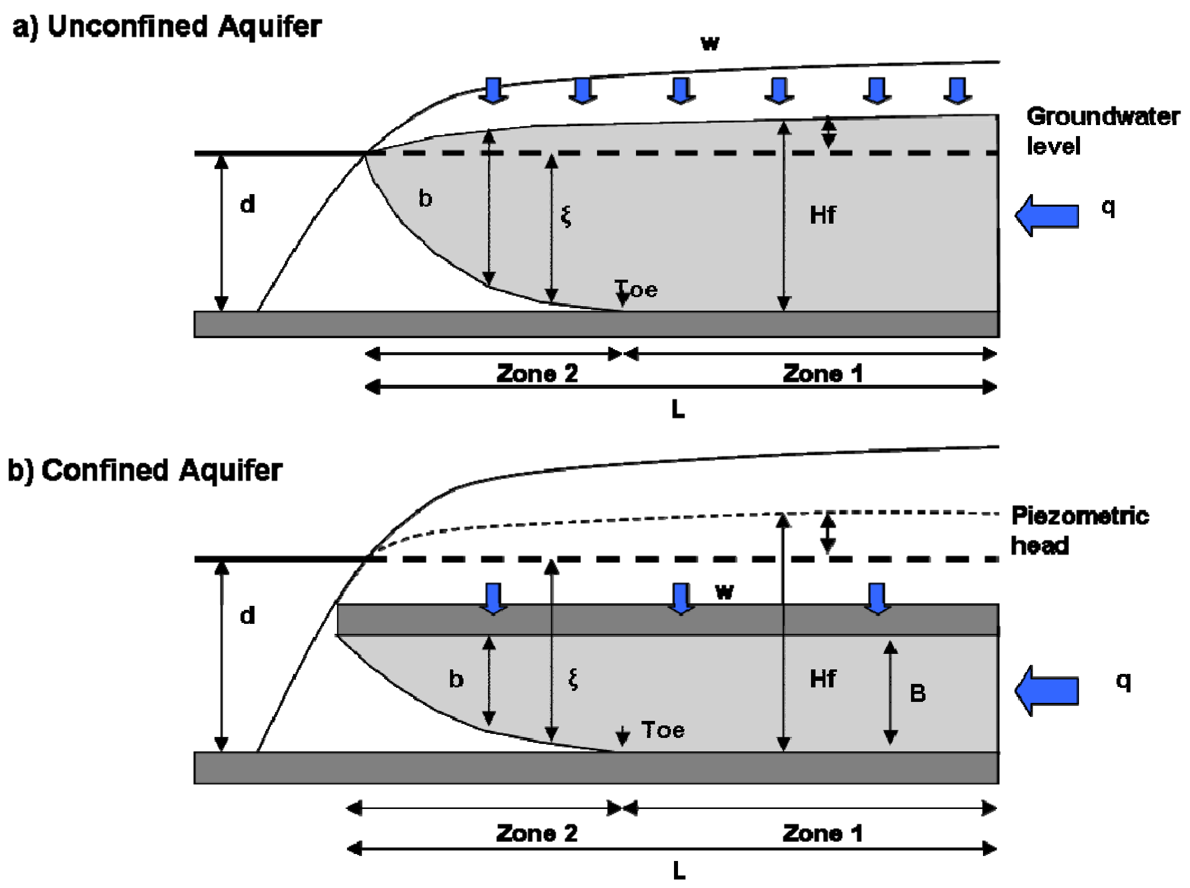

Figure 7: Model representation of an unconfined (a) and a confined (b) aquifer system

The position of the saltwater-freshwater interface and the thickness of the freshwater lens are calculated as follows:

Δs is the density ratio of fresh and salt water, given by

$$\Delta s = \frac{\rho_s - \rho_f}{\rho_f} \qquad [5]$$

The Ghyben-Herzberg relation is defined as:

$$h_f - d = \Delta s \xi \qquad [6]$$

A potential function is defined for each zone and is expressed for unconfined aquifers:

$$\Phi = \frac{1}{2}\left[h_f^2 - (1+\Delta s)d^2\right] \quad Zone\,1 \qquad [7]$$

$$\Phi = \frac{(1+\Delta s)}{2\Delta s}\left(h_f - d\right)^2 \quad Zone\ 2 \qquad [8]$$

and for confined aquifers:

$$\Phi = Bh_f + \frac{\Delta s B^2}{2} - (1-\Delta s)Bd \quad Zone\,1 \qquad [9]$$

$$\Phi = \frac{1}{2\Delta s}\left[h_f + \Delta s B - (1+\Delta s)d\right]^2 \quad Zone\ 2 \qquad [10]$$

The potential at the toe of the saltwater wedge, i.e. the interface of zone 1 and zone 2, is continuous across the interface and takes for unconfined aquifers:

$$\Phi_{Toe} = \frac{\Delta s(1+\Delta s)}{2} d^2 \qquad [11]$$

and for confined aquifers:

$$\Phi_{Toe} = \frac{\Delta s}{2} B^2 \qquad [12]$$

A solution is given for the potential function in a vertical cross section with constant surface recharge w and inflow q across the inland boundary of the cross section:

$$\Phi = -\frac{w}{2K}x^2 + \left(\frac{q}{K} + \frac{wL}{K}\right)x \qquad [13]$$

Substituting the potential value at the toe location, the toe location x_{Toe} can be solved

$$x_{Toe} = \frac{q}{w} + L - \sqrt{\left(\frac{q}{w} + L\right)^2 - 2\frac{\Phi_{Toe}K}{w}} \qquad [14]$$

The potential functions (Equations 7 and 8/ 9 and 10) can be rearranged to solve for freshwater potential h_f at any location x, substituting the potential $\Phi(x)$ with the values obtained from equations 13. The resulting equations for confined and unconfined aquifers in zone 1 and 2 are:

Unconfined Aquifer, Zone 1

$$h_f(x) = \pm\sqrt{2\Phi(x) + (1+\Delta s)d^2} \qquad [15]$$

Unconfined Aquifer, Zone 2

$$h_f(x) = d \pm \sqrt{\Phi(x)\frac{2\Delta s}{1+\Delta s}} \qquad [16]$$

Confined Aquifer, Zone 1

$$h_f(x) = \frac{\Phi}{B} - \frac{\Delta s B}{2} + (1+\Delta s)d \qquad [17]$$

Confined Aquifer, Zone 2

$$\begin{aligned} & h_f(x) = -\frac{p}{2} \pm \sqrt{\left(\frac{p}{2}\right)^2 - q} \\ & \text{where} \quad p = 2\Delta s B - 2d(1+\Delta s) \\ & \text{and} \quad q = \Delta s^2 B^2 - 2d\Delta s B(1+\Delta s) + d^2(1+\Delta s)^2 - 2\Delta s\Phi(x) \end{aligned} \qquad [18]$$

Given the freshwater potential $h_f(x)$, the thickness of the freshwater lens b at each location x is given by the following set of equations:

$$\begin{aligned} b &= h_f && \text{unconfined} && \text{zone1} \\ &= h_f - d + \xi && \text{unconfined} && \text{zone2} \\ &= B && \text{confined} && \text{zone1} \\ &= \xi - d + B && \text{confined} && \text{zone2} \end{aligned} \qquad [19]$$

Assuming unit width of the aquifer, the so derived thickness of salt- and freshwater can be transformed into salt and freshwater volumes, integrating thickness of the freshwater lens over the extent of the model domain:

$$FVol = \int_{i=0}^{L} b \, dx \approx \sum_{i=1}^{nx} b_i \Delta x \qquad [20]$$

3.3.2 Comparison of paired simulations

As long as an aquifer is not overexploited, seawater intrusion can be seen as transition from one equilibrium state to another, caused by changes of recharge or water abstractions.

The model provides a steady state solution, information on intrusion processes therefore have to be derived comparing paired simulations representing different states. For example, a potential intrusion process may be defined as transition from an 'unstressed' state to a 'stressed' state. For this study, the 'unstressed' state was defined setting water abstractions equal to zero (recharge only) while in 'stressed' state full water abstractions are applied.

For corresponding pairs of stressed and unstressed simulations the following parameters were calculated to characterize intrusion processes:

- Change of net groundwater recharge
- Change of lateral inflow
- Change of total water flow (recharge and inflow)
- Decrease of average groundwater level
- Freshwater loss
- Progression of saltwater wedge (toe location)

Absolute and relative changes were calculated for each variable using the formula:

$$\Delta X = X_1 - X_2$$
$$r_X = \frac{X_1 - X_2}{X_1} \qquad [21]$$

where ΔX = absolute change of X, X = state variable, rX = relative change of X, Index {1, 2} = indicating state 1 ('unstressed') and state 2 ('stressed').

The freshwater loss and the change of water flow can be used to calculate the characteristic time T_c. T_c is a measure of the time scale needed for the transition from one stage to the other (ranging from years to centuries). It gives an indication on how fast the aquifer reacts to altered levels of net recharge. The 'characteristic time' must not be mistaken as 'response time', which is the time actually needed for the transition. This information requires detailed numerical modelling on a case study basis and can not be captured by the simple concept of 'characteristic time'. T_c can be estimated according to:

$$T_C = \frac{\Delta V_f}{\Delta q_a} = \frac{V_f^1 - V_f^2}{(q_2 + w_2) - (q_1 + w_1)} \qquad [22]$$

where ΔVf = freshwater loss [L^3] and Δq_a = change of water flow [L^3].

Given the characteristic time T_c, the total changes between two states can also be expressed as ‘characteristic’ annual changes, which may be more comprehensible (for example ‘characteristic’ annual loss of freshwater volume or ‘characteristic’ annual progression of the saltwater wedge. As for characteristic time T_c, these values are indicative only.

3.3.3 Aquifer settings and hydrogeological properties

For each aquifer domain, we defined a model transect having an extent equal to the domain radius (i.e. ranging from 2.5 – 40 km). Each domain was further associated with a geological substrate (and corresponding properties) and with a set of different **aquifer settings** (compare also Figure 2).

The spatial distribution of geological substrates was derived from the European Soil Database 2.0 (ESDB), which provides information on dominant and secondary parent material of the soil units. Parent materials include igneous and metamorphic rocks and sedimentary rocks in a hierarchical classification of three levels. Not all materials are permeable aquifers but may be more or less impermeable. With given boundary fluxes (net recharge) high potential gradients are required to maintain water flux, resulting in unrealistically high groundwater levels and extreme response to changes in boundary fluxes. In practice, such substrates can not be exploited for groundwater and should be excluded from the analysis. However, even in areas where the main substrates are impermeable, permeable substrates may be of local importance and form relevant aquifers. Applying properties of the dominant substrates may therefore not be appropriate, as relevant local aquifer systems may be missed. Additionally, the available data give only incomplete information on aquifers, as geological maps at this scale display a generalized distribution of geological units at the surface without giving usable information on the vertical structure of permeable and impermeable materials. Therefore parent materials designated in the map may not correspond to the materials forming the main aquifers.

To overcome these problems (at least partly), we associated locations and parent materials according to the following rules: First, the most frequent dominant parent material within an aquifer domain was assigned. If the dominant parent material was impermeable, the secondary parent material was assigned. If also the secondary parent material was impermeable, the aquifer domain was considered impermeable and excluded from the simulation.

Further large-scale data sets considered for this study include the Hydrogeological Map of Europe (IHME 1500, BGR 2007) and the USGS Map showing Geology of Europe (USGS-GME, Pawlewicz, M.J., Steinshouer, D.W. and Gautier, D.L. 2003). The USGS-GME includes only stratigraphic information and can not be used to derive distribution of substrates. The IHME is currently available in printed form only and the scanned map can be viewed online. Due to the analogue format, the data can not be made available for this study. A digitized version will possibly become available within FP7 (Peter Winter, BGR, oral communications). Visual comparison, however, showed a reasonable consistency with the parent materials of the ESDB.

For modelling purposes, basic information on hydrogeological properties of substrates needs to be linked to the parent materials.

Substrate properties were derived from literature sources. Morris & Johnson (1967) provided an analysis of basic hydrogeological properties of typical igneous and metamorphic rocks and sediments. Hydrogeological properties included hydraulic conductivity, porosity, effective porosity, specific yield. The data provided by Morris & Johnson (1967) were based on laboratory analysis. They do not and can not reflect the natural variability of aquifer properties ranging in orders of magnitude. They also do not reflect differences between the analysed rock matrix and macro-scale properties originating from cracks, fissures and other openings present in the rock material. This is especially true for fractured rock aquifers and karstic aquifers, where the formation of a secondary pore system depends on the association with other rock types, genetic, diagenetic and tectonic history, the age of the substrate and other factors. In summary, the substrate properties do not reflect the broad range of possible values within on and the same material. Therefore they are indicative only and do not necessarily reflect true local conditions.

Substrates for which information was available only partly correspond to parent materials taken from ESDB. Where correspondence was not obvious, substrates were associated or new substrates were defined based on expert judgement. The substrates were additionally classified into aquifer materials (having a hydraulic conductivity > 0.1 m/d) and impermeable substrates.

There are numerous possible aquifer configurations, ranging from confined and unconfined aquifers to shallow and deep aquifer systems. As the true vertical structure and extent of the coastal aquifers was not known, we defined a set of pre-defined **aquifer settings** representing different aquifer systems (Table 2 and Figure 9). These aquifer settings were adapted to local conditions, applying recharge, abstractions and geological characteristics determined for each aquifer domain.

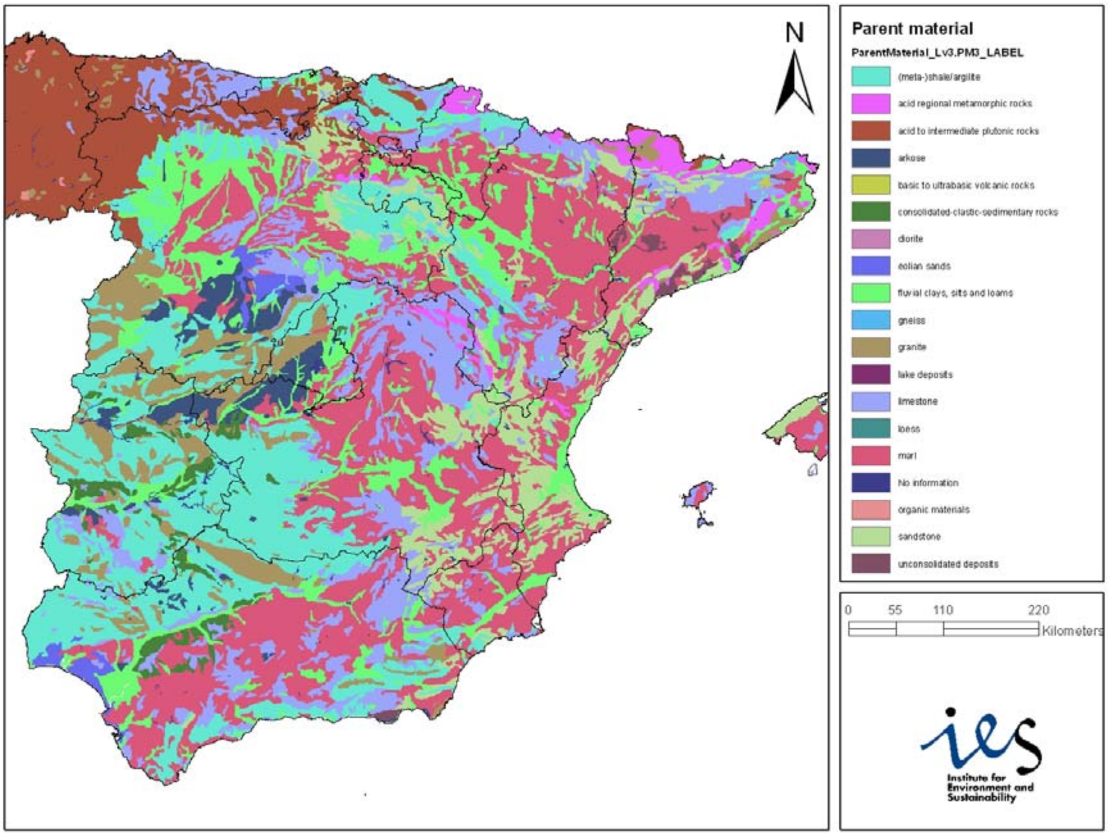

Figure 8: Dominant parent materials derived from ESDB 2.0 (Example: Spain)

Table 2: Characteristics of aquifer settings applied in Tier 2.

Code	Description	Depth of lower confining unit [m]	Aquifer thickness [m]	Aquifer type unconfined = 0, confined = 1
SHAL_UCF	Shallow unconfined	50	-	0
DEEP_UCF	Deep unconfined	150	-	0
TOP_CF	Shallow confined	55	50	1
BOT_CF	Deep confined	155	50	1
DEEP_CF	Deep confined	155	150	1

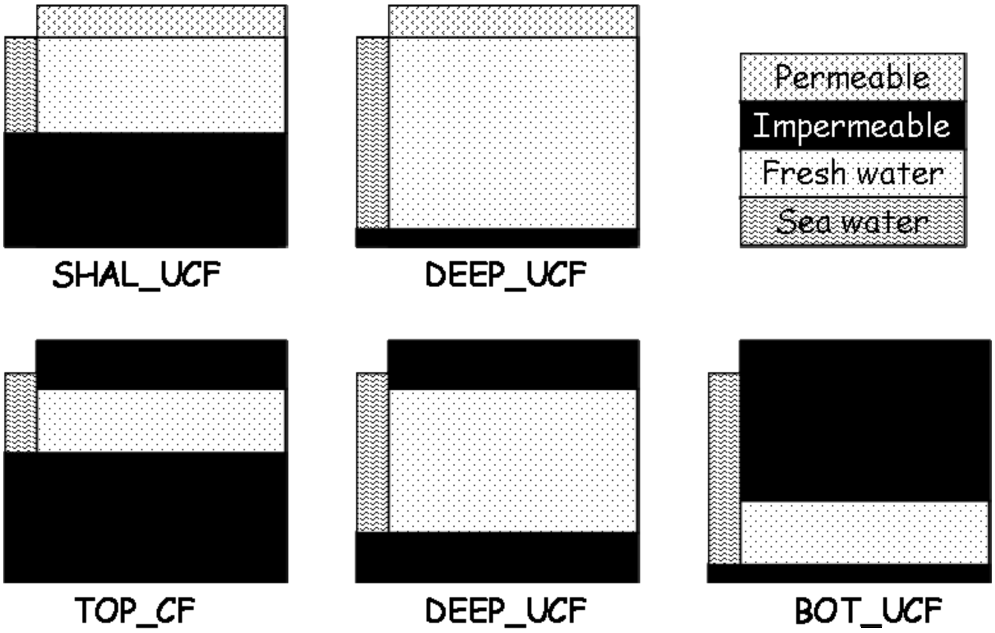

Figure 9: Schematic illustration of pre-defined aquifer settings applied in Tier 2

3.4 Implementation notes

Input data on geology, recharge and abstractions were provided as raster data sets. The assessment points and aquifer domains were generated via a series of GIS operations implemented as a Python script. Another Python script extracts input data for each aquifer domain from the input raster data sets (Geology, Recharge and Abstractions). The results are stored as tables in a MS-Access database. Pre-defined database queries calculate the recharge balance and classify intrusion risk for each assessment point and aquifer domain for Tier 1 evaluation. The analytical model was implemented using the Visual Basic programming language. It comprises two independent core modules: the analytical solution of the saltwater-freshwater interface and the comparison of paired simulations. The analytical solution module reads input data directly from a database table storing input data for multiple simulations. A database query generates the simulation input table from the various input data tables. Results are returned as a database table for further evaluation. This feature enables processing of an arbitrary number of simulations, for example to simulate individual aquifers along a coastline or to carry out sensitivity and uncertainty analysis. We processed simulations for ‘unstressed’ and ‘stressed’ state separately generating separate input data tables and result tables. The comparison module calculates differences between paired simulations. Two result tables with paired simulations have to be provided as input, a single output table is generated and written to the database.

4 Results of the pilot study

4.1 Results of Tier 1 – risk characterisation

The results of the risk classification are displayed in Figure 10 for transient intrusion risk and in Figure 11 for risk of intrusion by over-abstraction. The maps display very well the distribution of intrusion risk along the coastline. The distribution patterns of both maps complement each other, as a high risk of over-abstraction is associated with low risk of transient intrusion and vice versa. The coastal segments between Tortosa and the French border, between Valenzia and Cartagena and at the southern coast of Andalucia have a high risk of intrusion due to overabstraction of groundwater.

The share of agricultural water abstractions in total water abstractions is displayed in Figure 12. In many parts of the Spanish coast, agricultural abstractions account for more than 50% of water abstractions. Only in few segments (e.g. around Almeria, Barcelona), agriculture is of minor importance.

The 'Agricultural pressure' as ratio of agricultural water abstractions and groundwater recharge is shown in Figure 13. Especially in the areas between Valencia and Cartagena and around Tortosa agricultural abstractions alone account for sufficient water abstraction to put aquifers at risk of overabstraction or even extreme overabstraction.

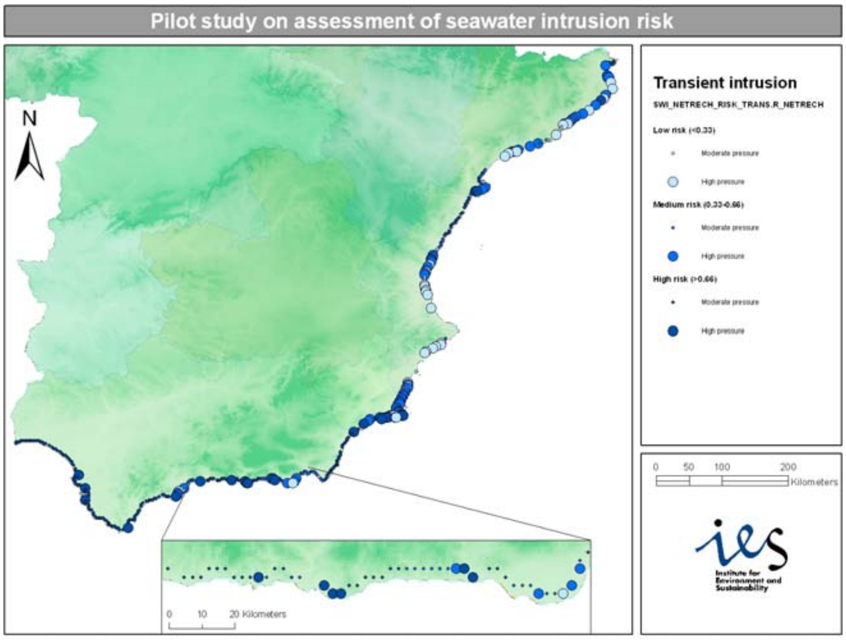

Figure 10: Risk of transient intrusion. Risk classes are based on the number of aquifer domains at each location falling into the transient risk class. Separation into moderate and high pressure is based on the abstraction-recharge ratio (> 0.5).

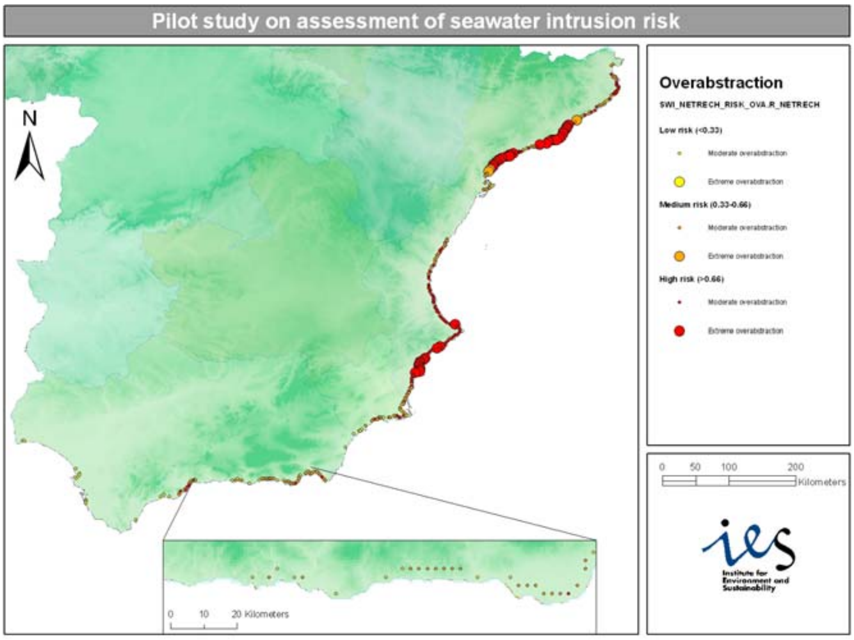

Figure 11: Risk of intrusion caused by over-abstraction of aquifers. Risk classes are based on the number of aquifer domains at each location falling into the transient risk class. Separation into moderate and extreme over-abstraction is based on the abstraction-recharge ratio (> 5).

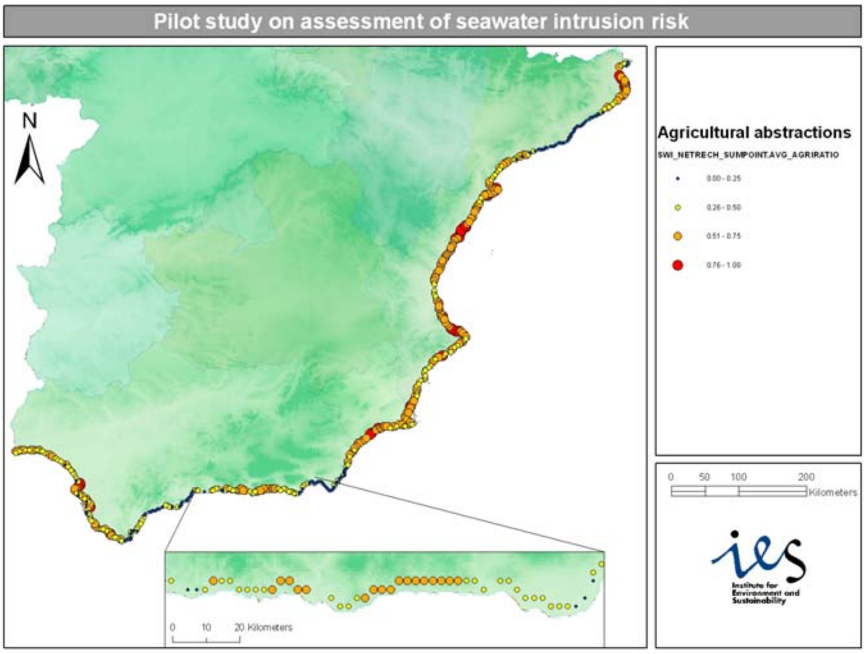

Figure 12: Share of agricultural abstractions in total water abstractions as average of all aquifer domains at each assessment point.

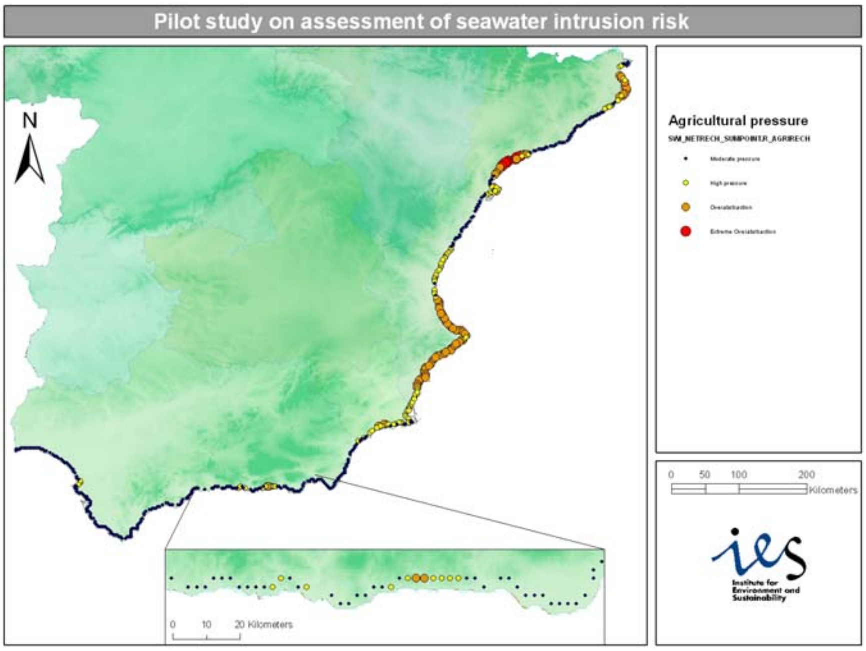

Figure 13: Agricultural pressure on coastal aquifers based on the ratio of agricultural abstractions to groundwater recharge (as average of all aquifer domains at each assessment point): moderate pressure < 0.5, high pressure 0.5-1, overabstraction 1-5, extreme overabstraction > 5.

4.2 Results of Tier 2 – quantitative characterisation

The results of Tier 2 can not be aggregated at assessment point level to built summarized indicators. Results have to be analysed individually for defined aquifer domains and aquifer settings.

Due to the uncertainties associated with the underlying data and the standardized approach not respecting in detail local conditions, the results are indicative only. Nevertheless, they allow ranking and relative comparison of different locations.

The results presented here refer to a deep unconfined aquifer with an aquifer extent of 5000 m (Scenario DEEP_UCF). The results of Tier 2 are differences with respect to a hypothetical 'unstressed' state. The true initial state is unknown.

The patterns of freshwater loss (relative: Figure 14, absolute: Figure 15) follow the patterns of intrusion risk and associated pressure, as the freshwater loss depends directly on the ratio of abstractions to recharge. A relative freshwater loss of 1 occurs for simulations falling into the overabstraction class, while relative freshwater losses < 1 correspond to simulations within the transient intrusion class.

Further results displayed are average decline of groundwater level (Figure 16), characteristic time (Figure 17) and progression of toe of saltwater wedge (Figure 18). The characteristic time ranges from several years up to some centuries, according to the (generalized) hydraulic conductivities.

While the Tier 1 approach allows risk assessment at any point along the coastline independent of aquifer characteristics, Tier 2 results depend on the association with and parameterization of geological substrates. Various coastal segments are assigned impermeable substrates (neither dominant nor secondary material are permeable substrates) and therefore no Tier 2 results are available at these locations (e.g. Ebro-Delta/Tortosa, coast north of Valencia, coast east of Malaga).

Generally, it becomes clear, that possible impacts cover the entire range from negligible water losses until complete loss of the groundwater resources and the processes may be slow or fast.

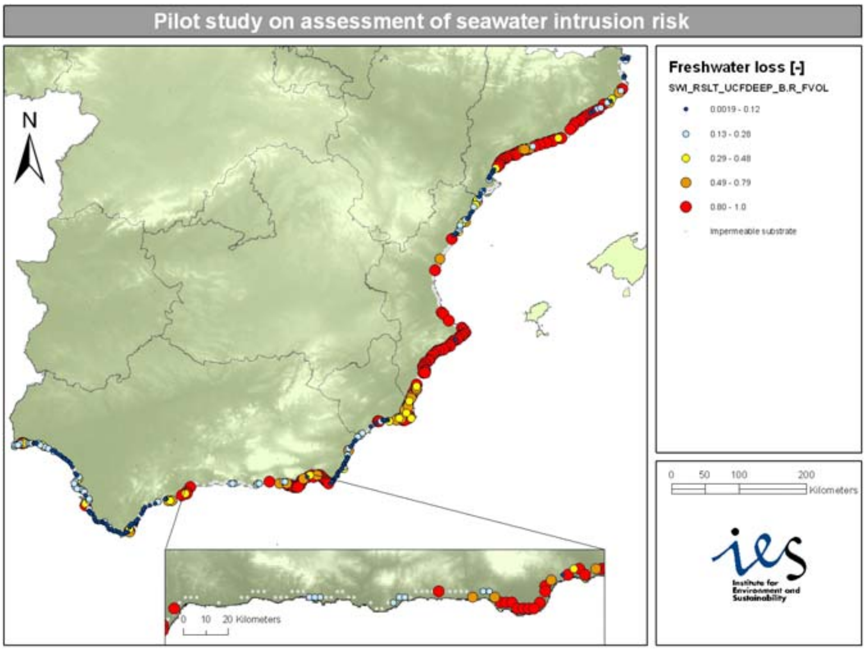

Figure 14: Relative freshwater loss during transition from unstressed to stressed state. Results are displayed for simulations falling into the transient intrusion risk class only.

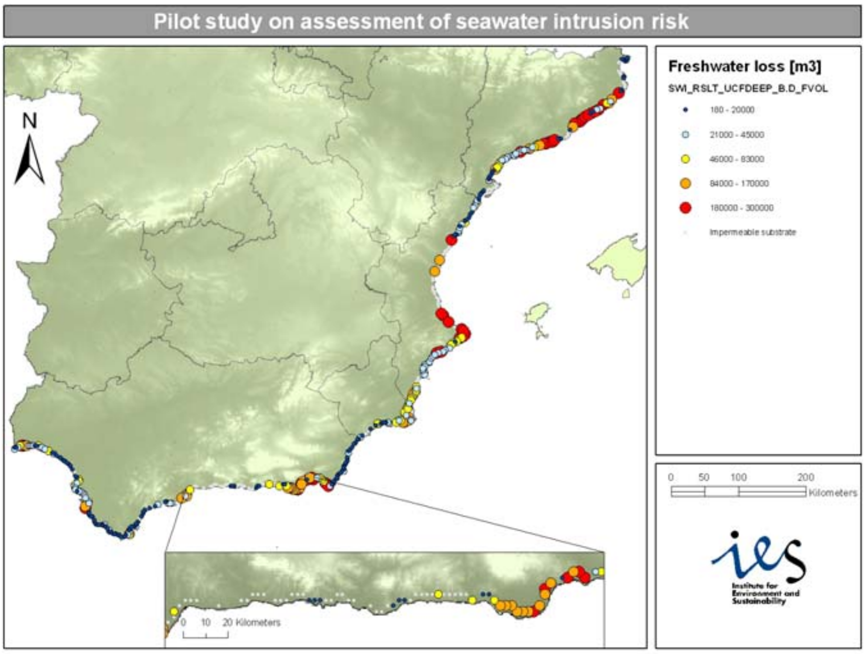

Figure 15: Absolute freshwater loss during transition from unstressed to stressed state. Results are displayed for simulations falling into the transient intrusion risk class only.

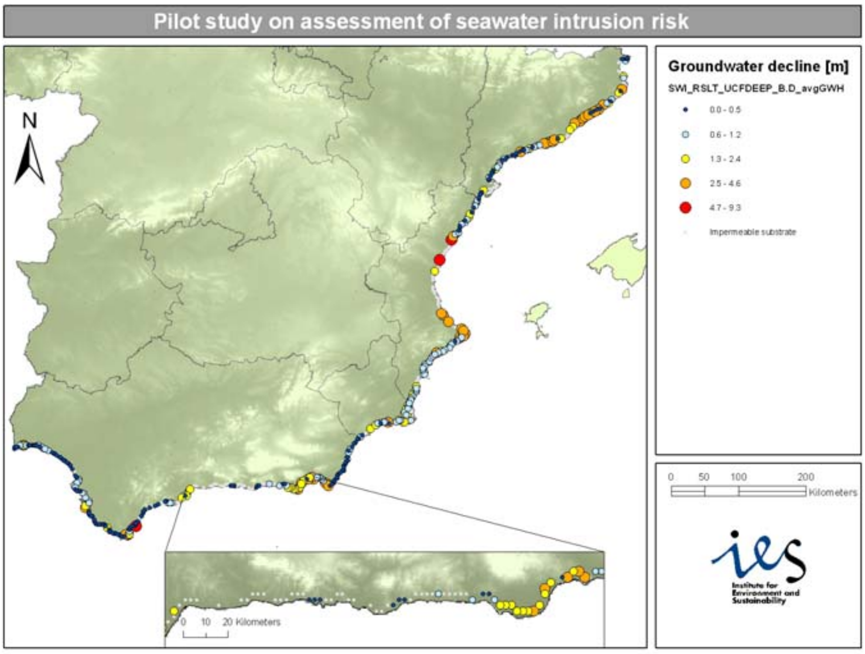

Figure 16: Decline of groundwater level during transition from unstressed to stressed state. Results are displayed for simulations falling into the transient intrusion risk class only.

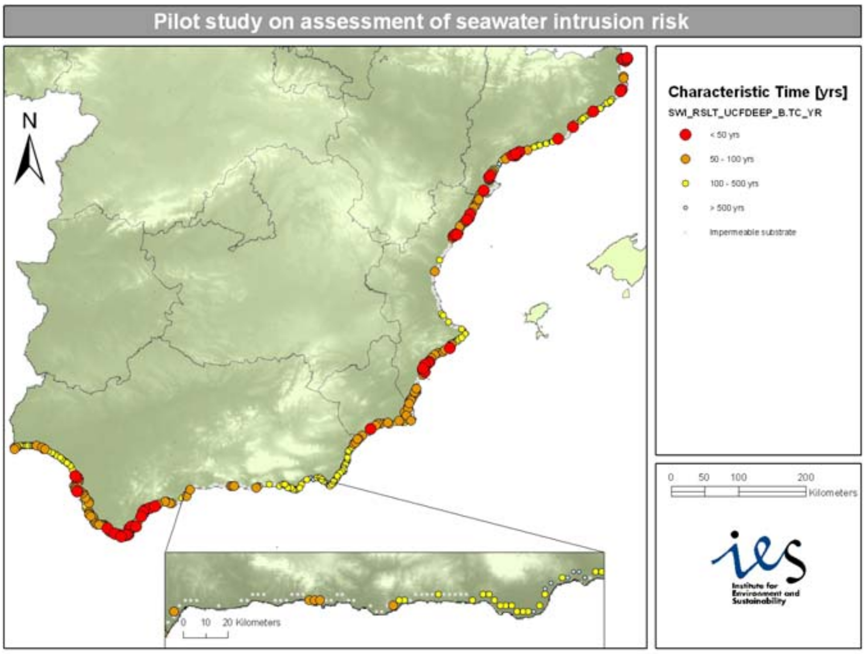

Figure 17: Characteristic time (years), representing time scale for transition of interface from unstressed to stressed state. Results are displayed only for simulations falling into the transient intrusion risk class.

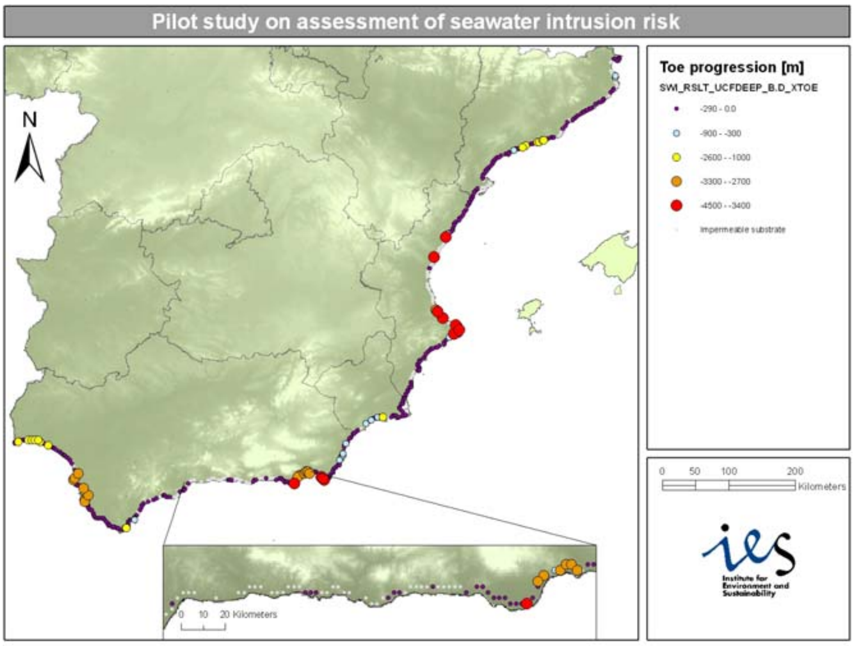

Figure 18: Progression of saltwater-freshwater interface from unstressed to stressed state (given as displacement of toe of saltwater wedge) (for simulations in transient intrusion risk class only).

5 Discussion and conclusion

The assessment approach developed in this study was designed as a tool for large scale analysis of potential hot-spots of seawater intrusion, exploring the potential use of currently readily available data for this type of analysis. In future, the approach shall be applied to draw a general picture of seawater intrusion risk in Europe. The risk assessment of Tier 1 is based on the local ratio of abstractions and recharge. This approach allows a simple and easy-to-use general assessment of intrusion risk, independent of typically poorly known real-world aquifer conditions. In Tier 2, simulation results provide quantitative information for standardized aquifers respecting generalized local geological properties. We demonstrated the potential information to be derived in a pilot application.

For a general risk assessment, Tier 1 is the most important part. It does not require any hydrogeological information or modelling and can directly be linked to information on natural water balance and human water requirements. The Tier 1 approach is sufficiently generic and can also be used to assess water quantity pressure on inland aquifers, given the availability of suitable input data. Accuracy of the screening results is directly related to input data. Required input data for Tier 1 are water abstraction and groundwater recharge.

Disaggregation of water abstraction data starts at national level and the disaggregation procedure introduces some uncertainty. Strictly speaking, the spatial disaggregation of water abstractions determines where the water is used, but not where it is really abstracted. Water may be transported to water users over long distance, which has not yet been considered. Further, there is currently no way to reasonably separate water abstractions into ground-and surface water abstractions. Even if statistics are given, the administrate units are too large (national or regional level data) and factors influencing the source appointment are too many, to reasonable disaggregate in space. Only water abstractions for cooling purposes are exclusively drawn from surface water. We attributed all remaining abstractions to groundwater draws a worst-case scenario. True water abstractions from groundwater may be overestimated and a certain yet unknown fraction may be taken from surface water or alternative sources instead. A higher resolution of initial data would be desirable to reduce the disaggregation uncertainty and improvements in the disaggregation procedure and source appointment are necessary. Aquifer recharge may be limited by climatic water surplus but true recharge is affected by numerous factors, such as topography, land cover, unsaturated zone and protection of aquifer, which have not been taken into account. As the focus of this work was on the methodological development, recharge data are partly provisional and have to be replaced by a more detailed water balance assessment. In the current assessment, optimistic recharge estimation is combined with a worst case estimation of abstractions.

However, within the overabstraction risk class, the ratio of abstractions to recharge indicates considerable discrepancies, frequently obtaining ratios exceeding 5 by far. Even a considerable change of recharge or abstractions may therefore not alter the general picture (i.e. spatial distribution of intrusion risk). Only within the transient intrusion class we would expect considerable shifts in pressure and also switching from transient risk class to overabstraction risk class and vice versa.

The high local discrepancies of abstractions and groundwater recharge also result from the problem of allocating water abstractions in space and separating ground- and surface water abstractions and further research is required to solve this problem. A more appropriate approach should consider optimizing the spatial distribution of ground- and surface water abstractions with the availability of ground- and surface water resources at least at a river basin level.

Given the need to improve the assessment of recharge and water abstractions, we have started a more detailed analysis of water demands in Europe and the development of a European water balance model for groundwater recharge estimation. The results of this study could not yet be included in this assessment and have to be left for a future update, that may also include a European assessment of intrusion risk.

In extension of Tier 1, Tier 2 provides more specific information on the potential intrusion problems. It helps to improve understanding of the intrusion processes and facilitates risk assessment and regional comparison including the (generalized) geologic properties of the substrate. This is quite important, as a high abstraction/recharge imbalance assessed in Tier 1 may result in short or long transition times (requiring urgent or long-term actions), depending on the substrate conductivity. Ideally, a large scale intrusion risk assessment like this should be based on individual aquifers to be as close to reality as possible, targeting the assessment more specifically on the actual local settings and conditions. It is, however, currently not possible to get detailed information on the shape, size, depth and geological properties of coastal aquifers in Europe. Data sources covering large geographical areas such as European and national geological maps are too general and do not provide information meeting the specific needs of groundwater modelling. Even local data are difficult to obtain and are not generally available, as data collection requires intensive hydrogeological investigations. Also collection of local aquifer data is not centralized at European level and it is practically not feasible to collect specific data at the level of the responsible authorities. Therefore a risk assessment based on real-world aquifers will remain limited to local or regional scale, but is currently not feasible for continental scale overviews. The use of schematic aquifers nevertheless illustrates the consequences of intrusion problems for different aquifer types and it is therefore possible to analyse the results specifically for the aquifer setting best representing a local situation. As true local situations can differ considerably from the simplified large scale representations, the information obtained remains indicative. The spatial distribution of observed intrusion cases as shown by EEA (Figure 1) supports a first plausibility check.

The model applied in Tier 2 can be used for a schematic representation of any coastal aquifer, given some information on aquifer extent and depth, hydrogeological properties, recharge and abstractions. The analytical model applied in Tier 2 is a simple but valid representation of hydrodynamics of saltwater-freshwater aquifers. It can therefore also be applied for a screening of real-world aquifers. A similar analytical approach was applied by Werner and Simmons (2009) for a general assessment of sea-level rise impacts on sea-water intrusion.

A potential source for aquifer specific information could be data collection within the WFD, as it requires the identification of groundwater bodies. Collection of this information and the related aquifer and water recharge and abstraction data would help to gather information for a European assessment. However, the WFD Directive does not require delineation of groundwater bodies based on the hydrogeological situation. Other features, like chemical composition and geological boundaries can be taken into account, neglecting the hydrological functioning of the aquifer. Therefore it is questionable if WFD groundwater bodies can be used as aquifer representations.

Using the analytical model, the intrusion process can only be assessed as change between two equilibrium states. A time variant numerical model, even simple, can help to overcome this limitation. However, a numerical model will not increase the accuracy of the results, unless accompanied by a big leap forward in data availability.

The ongoing development of European water balance and water demand assessment will result in the availability of water recharge and abstraction data based on either a regular grid over Europe or on a catchment basis. We also expect considerable improvement regarding the procedures to estimate and disaggregate the information. Including this information in Tier 1, assessment points along the coastline could alternatively be replaced by cells and catchments, directly overlaying the data layers. Tier 2 could then be applied to catchments and coastal cells, rather than to aquifer domains.

Most studies dealing with seawater intrusion problems focus on local problems or problems of limited regional extent. Often complex models or intensive data collection are used to analyse processes and to develop specific management strategies. While development of measures to combat seawater intrusion always requires consideration of local factors, it is difficult to draw a general picture of the problem from individual cases. To our knowledge, there is only one study focusing on large scale assessment of seawater intrusion, investigating the impact of sea-level rise on freshwater resources at a global scale (Rajan et al. 2006). In this context, our work closes a gap between coarse global approaches and detailed local studies, contributes to a large scale overview including a generalised analysis of pressures and geological conditions. Naturally, such large scale approaches have to deal with simplifications and details of high local relevance can not be included. The assessment approach extends the EEA survey of observed cases, highlighting potential or emerging risks where no intrusion is yet observed or helps to understand the driving forces behind the intrusion problem, as various information of the pressure exerted by different water use sectors and the potential severity of intrusion processes can be derived. The pressure analysis can guide policy actions at large scale in addition to local mitigation strategies. The analysis does neither predict real occurrences of seawater intrusion nor can be used for quantitative assessment of real-world intrusion problems. The approach is relatively simple and robust, supporting screening of intrusion risk over large geographical areas. The main input data can be derived from future projections and models, thereby allowing application of the approach for scenario analysis to analyse effects of climate change, land use change, and changes in water demand on seawater intrusion and associated freshwater loss. The approaches to derive reasonable input data for the assessment require further research and methodological development. The tentative results used so far have methodological limitations and more sophisticated approaches are required. Given the potential of the approach developed, future work shall improve the underlying information and apply the approach to assess the risk of seawater intrusion across Europe.

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Abstract

Seawater intrusion caused by overabstraction has become an important problem in coastal areas of the Mediterranean. Intrusion processes highly depend on complex local conditions and studies generally focus on specific problems of individual aquifers. Seawater intrusion problems have hardly been addressed at large scale due to the strong impact of local conditions and the problems related to data collection for large geographical areas. To fill this gap and to explore the potential use of readily available data sources, we developed a simple screening methodology for large scale assessment of seawater intrusion risk along the Mediterranean coast of the EU based on a two tiered assessment procedure. Tier 1 is a simple risk assessment based on the balance of groundwater recharge and water abstractions for coastal areas. A positive net groundwater recharge results in transition of the saltwater-freshwater interface to a new equilibrium state. No equilibrium exists with negative net recharge (over-abstraction) and seawater is drawn into the aquifer compensating freshwater losses. Tier 2 provides a quantitative characterization of seawater intrusion for standardized aquifers considering generalized local geological conditions: A simple analytical intrusion model calculates freshwater loss and seawater progression for specified combinations of aquifer properties, aquifer dimensions and boundary fluxes (recharge and abstractions). An unstressed quasi-natural state (recharge only) is compared with a stressed state adding the current level of abstractions. The estimation of groundwater recharge and the spatial disaggregation of national water abstraction data are still tentative and future improvement of the procedures is necessary. A pilot application was carried out for the coast of SE-Spain. As geological data did not support an assessment of individual ‘true’ coastal aquifers, we defined standardized aquifer domains of different size and applied the seawater intrusion model (Tier 2) to each aquifer domain using local geology and boundary fluxes. Freshwater loss and progression of the saltwater-freshwater interface illustrate the potential severity of potential intrusion processes. The approach supports screening of intrusion risk over large geographical areas based on local relation of abstraction and recharge. The methodology is principally promising, even though input data used for the pilot studies are still based on tentative approaches and need to be replaced by more detailed analysis of statistical information and modelling. Water demand and groundwater recharge as basic input data can be linked to model applications, allowing assessment of future intrusion problems in the context of scenario analysis (for example assessing changes of intrusion risk based on climate change, land use changes and changes in water demands).

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