JRC Scientific and Technical Reports

Towards a General Water Balance Assessment of Europe

Gunter Wriedt, Fayçal Bouraoui

EUR 23966 EN - 2009

EUROPEAN COMMISSION

Institute for Environment and Sustainability

The mission of the Institute for Environment and Sustainability is to provide scientific-technical support to the European Union's Policies for the protection and sustainable development of the European and global environment.

European Commission
Joint Research Centre
Institute for Environment and Sustainability

Contact information
Address: Fayçal Bouraoui, JRC, TP 460, Via E. Fermi 2749, 21027 Ispra, Italy
E-mail: faycal.bouraoui@jrc.ec.europa.eu
Tel.: +39 0332 78 5713
Fax: +39 0332 78 5601

http://ies.jrc.ec.europa.eu/
http://www.jrc.ec.europa.eu/

JRC 53258

EUR 23966 EN
ISBN 978-92-79-12979-7
ISSN 1018-5593
DOI 10.2788/26925

Luxembourg: Office for Official Publications of the European Communities

Printed in Italy

Table of Contents

Summary

Large proportions of water supply in European countries rely on groundwater resources, and many aquifers in water scarce regions are overexploited. Water management relies on reasonable information on water availability as well as on water demands by different sectors. Information on water availability and water needs are crucial to identify hot spots of quantitative pressures on water resources. This report focuses on estimating natural water availability across Europe. Simple water balance models were applied for an assessment of available water and potentials and limitations of their application are shown. Special emphasis is given to the role of groundwater in the water cycle and we explore ways to derive groundwater balance terms for large scale assessments. We further develop indicators of water quantity pressure relating water availability to water use and losses at different spatial scales. A short overview on the functioning of groundwater systems is given, highlights properties, processes and problems relevant for groundwater quantity and quality assessment. Some concepts to address groundwater issues at large scale are derived. The methodological part combines a general water balance assessment at large scale with more specific approaches to characterize groundwater systems and to quantify groundwater balance terms at large scale. Two different water balance modelling approaches are applied estimating the amount of water available for direct and subsurface runoff. The modelling approaches are compared to observed values and to each other. The available water is compared to water abstractions developing to indicators for human pressures on water resources. Focusing more specifically on groundwater systems, different methods to calculate baseflow and groundwater recharge are applied and compared and a prototype groundwater recharge map of Europe is presented. The report concludes with a synthesizing discussion of methods and results and an outlook on possible future studies. The individual studies have not yet been integrated into a common framework. Rather, they show various restrictions that require further research on various specific issues relevant for water management at European scale. The approaches laid out in this report and related reports provide a starting point for further development of screening approaches to be integrated in a common water resources assessment framework.

1 Introduction

1.1 Background and objectives

The 6th Environment Action Programme (EAP) (1600/2002/EC) and the Water Framework Directive (WFD, 2000/60/EC) set out the main policy objectives in relation to water use and water stress at EU level, aiming at ensuring a sustainable use of water resources. Specific environmental objectives related to groundwater resources include implementation of measures to limit input of pollutants into groundwater and to prevent deterioration of groundwater; protection, enhancement and restoration of groundwater and ensuring a balance between abstraction and recharge of groundwater; implementing measures to reverse significant and sustained upward trends in pollutant concentrations resulting from human activities (MGWWG 2007).

Large proportions of water supply in European countries rely on groundwater resources, and many aquifers in water scarce regions are overexploited. Such situations are frequently (but not only) observed in Mediterranean countries, sharing typical features with respect to water use and water availability. The arid and semi-arid regions of the Mediterranean combine winter rainfall and summer droughts. Only a small amount of water flows into rivers or percolates into aquifers. The limited natural availability of water resources and the high climatic variability are confronted by high water demand of the Mediterranean population, which has doubled in the second half of the last century (MGWWG 2007) and is expected to further increase due to ongoing population growth and urbanization. The important role of irrigated agriculture in Southern Europe, accounting for more than 60 % of total water abstractions, also contributes considerably to the overuse of water resources.

The imbalance of demands and available water resources cause water scarcity, causing conflicts between water users and damaging water resources by overexploitation and related processes (e.g. salinisation). Management measures can address both, water demands as well as water supplies. Climatic conditions and physical geographic settings determine the availability of ground- and surface water resources and the potential to supply water for human needs. Water demands are defined by the need to maintain drinking water supply, domestic water, industry, food production. The abstracted volume is largely determined by social conditions, behavioural factors, economy, technological development and the institutional framework. Considerable savings can be achieved by careful demand management (Ecologic, 2007) to reduce water demands or pressures on water resources (Figueres et al., 2003).

"(…) A different approach to groundwater management is now developing based on the recognition that users need to have incentives and effective management instruments and mechanisms to preserve the resource. (…) Ideally, an aquifer would be managed so that it would not be polluted and it's yields would be perpetually sustained (Kemper 2003)", the final goal being explicit reduction of total groundwater abstractions to a sustainable level. To meet societies' water demands, ground- and surface water resources can be exploited. Changes of water demands therefore do not necessarily translate to groundwater resources (or surface water respectively) and groundwater use has to be seen in the context of general water use and availability.

Water management relies on reasonable information on water availability as well as on water demands by different sectors. Information on water availability and water needs are crucial to identify hot spots of quantitative pressures on water resources. Climate change is expected to intensify problems of water scarcity, especially in the Mediterranean region (IPCC 2007, Goubanova and Li 2006, Rodriguez Diaz et al. 2007) and approaches to evaluate the current situation and future scenarios of water scarcity are therefore needed to support policy measures and planning. Given that water management takes place at different administrative and political levels, ranging from local to international scale, also background information must be prepared at appropriate scales and resolution.

The research behind this report contributes to the assessment of water availability and water demands at European scale and to the development of indicators for quantitative pressure on water resources. This report focuses on estimating natural water availability across Europe. Simple water balance models were applied for an assessment of available water and potentials and limitations of their application are shown. Special emphasis is given to the role of groundwater in the water cycle and we explore ways to derive groundwater balance terms for large scale assessments. We further develop indicators of water quantity pressure relating water availability to water use and losses at different spatial scales. In a related report (Wriedt et al. 2009b) we developed spatially distributed background information on the actual level of water abstractions, consumption and returns. This was achieved by disaggregating water abstraction data to locations of (likely) consumption, using national statistics and suitable proxy data for disaggregation. Originally, the research covered by this and other reports was aimed at a selective assessment of water quantity pressures on groundwater resources in Europe only. Considering the complex relations between groundwater systems with the general hydrological cycle and human water consumption, and the lack of suitable data for European scale assessments a broader approach was adopted, starting from the general water balance and a water demand assessment and adding groundwater specific aspects as far as possible.

The report is organised as follows: First, a brief but comprehensive overview on the functioning of groundwater systems is given, highlighting properties, processes and problems relevant for groundwater quantity and quality assessment. Some concepts to address groundwater issues at large scale are derived. The methodological part combines a general water balance assessment at large scale with more specific approaches to characterize groundwater systems and to quantify groundwater balance terms at large scale. Two different water balance modelling approaches are applied estimating the amount of water available for direct and subsurface runoff. The modelling approaches are compared to observed values and to each other. The available water is compared to water abstractions developing to indicators for human pressures on water resources. Focusing more specifically on groundwater systems, different methods to calculate baseflow and groundwater recharge are applied and compared and a prototype groundwater recharge map of Europe is presented. The report concludes with a synthesizing discussion of methods and results and an outlook on possible future studies.

1.2 Short review on the functioning of groundwater systems

This chapter gives a brief but comprehensive overview on main characteristics of groundwater bodies and important groundwater flow and solute transport processes, as to be found in standard text books (e.g. Hiscock, 2005). Further we highlight main threats and pressures on groundwater resources. Given the complexity of groundwater systems, this overview can not be complete and exhaustive. Nevertheless, it serves i) to illustrate different aspects of groundwater systems in terms of water balance and transport of solutes and water, ii) to demonstrate the complex interactions of groundwater related processes and iii) to develop an understanding on needs, possibilities and limitations of large scale groundwater related assessments.

1.2.1 How does a groundwater system work?

An aquifer is a geological unit made of permeable rock or loose material that can store and conduct groundwater. Groundwater is held within the pores of the aquifer material. The size of aquifers ranges from a few hectares to thousands of square kilometers.

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 (the soil matrix). 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.

Karstic aquifers are dual porosity systems and form in calcareous rocks. 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 convective. Discharge is often concentrated in a few springs, including 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.

Unconfined aquifers are only partially filled by groundwater. A free groundwater surface (or groundwater table) exists. In a confined aquifer, the groundwater surface is restricted by an impermeable layer overlaying the aquifer and preventing the development of a free surface.

The term aquifer relates to the geological unit containing groundwater. This does not mean that all groundwater within an aquifer forms a homogeneous groundwater body. For example, within a homogeneous aquifer, different groundwater catchments may develop, with groundwater flowing towards discharge boundaries (typically streams) and a groundwater divide between to discharge boundaries. Having relatively little interactions between these catchments, pollution problems in one may not affect the other, although located in the same aquifer. Further, a groundwater catchment may include several aquifers (also made of different substrates), which are hydraulically connected and therefore behave effectively as a single aquifer. Aquifers closely interact with other compartments like vegetation, soils, unsaturated zone and surface water systems as lakes, rivers and the sea. They are connected by exchange fluxes of water and solutes. An aquifer or an aquifer system can have local and regional flow systems. In case of considerable seasonal or interannual dymanics of the groundwater table, changing groundwater levels may alter the connection to surface waters and subsequently alter the flow patterns. The geological history of an area plays an important role in the transport of groundwater and pollutants, as the arrangement of permeable and impermeable substrate can be very complex and defines possible flow directions, mixing of groundwater, areas of recharge and discharge areas. Impermeable layers may separate groundwater bodies or protect aquifers from percolation of pollutants.

The groundwater storage is refilled by recharge and depleted by discharges and (human) abstractions. Recharge results from soil leaching or infiltration from surface waters. Recharge from soil leaching is closely related to (and controlled by) the soil cover, climate conditions and land use. The groundwater surface water interaction can go in two directions. On the one hand, groundwater can discharge (exfiltrate) into the surface water system. This is a typical situation in humid regions. On the other hand, groundwater can receive water by infiltration from streams and rivers. Especially in arid and semi-arid regions, this infiltration of water from rivers and streams into the groundwater can be a significant component of the total recharge. The recharge area of a particular aquifer is not identical with the spatial extent of an aquifer. Only where the aquifer is exposed to the surface and not covered by impermeable layers, water can actually percolate down to the aquifer. For confined aquifers, the recharge area can therefore be a small fraction of the entire aquifer area. Recharge is also limited by the transport capacity of an aquifer. If the recharge flow is higher than the outflow from a particular aquifer, storage increases, resulting in rising groundwater levels and gradually increasing discharge until groundwater eventually reaches the surface, creating wells or wetlands and impeding further recharge of water.

Discharge of groundwater occurs at springs and wells, at surface waters such as streams, rivers and lakes, and in groundwater dependent wetlands. The discharge areas are small compared with the aquifer area and usually bound to the spring or direct surrounding of a stream.

The flow of groundwater is driven by pressure gradients in the aquifer (hydraulic gradient), typically measured as slope of groundwater levels or groundwater potentials. The permeability or hydraulic conductivity defines the water flux through a cross sectional area resulting from a given hydraulic gradient. The rate of groundwater flow is very slow compared to the flow of water on the surface and is usually in the range of several centimeters per year to several meters per year. The flow of groundwater (and dissolved substances) is a four-dimensional process (in space and time). Flow directions have a considerable vertical component depending on the distance to/from the watershed or the stream. Close to the watershed, the flow direction may have a significant downward component, while groundwater is rising close to or below a stream. Each location within an aquifer is related to a certain recharge area (connected via the flow path) and a groundwater age (defined by the time needed to transport water and solutes from the connected surface area to this point).

Travel times can be as short as months and as long as centuries, depending on the travel distance and the transport velocity within the substrate. Groundwater discharging into a stream is typically a mix of older and younger components, representing the travel time distribution of the aquifer. Changes in water inputs can still have direct effects on the outflow, as water flows result from pressures, which are conveyed immediately. A change of solute inputs may however not have observable effects in groundwater discharge or even in the groundwater itself (depending on position of the observation well) within reasonable observation periods, as the solute needs to be physically transported through the aquifer to reach the connected surface waters or observation wells.

Depending on the mineralogical composition of the substrate and the chemical characteristics of the groundwater, various biogeochemical processes take place in groundwater, typically mediated by specific microorganisms. At this point, we would like to highlight some important pathways relevant for various groundwater related pressures:

Organic matter and sulphides are important reactive compounds. As long as the groundwater contains oxygen, oxygen is used for the aerobic microbial transformation of organic matter (mineralization) and sulphides. Once oxygen is depleted, substances like nitrate (denitrification), nitrogen dioxide (denitrification, step 2), iron and manganese oxides and sulphate (desulfurication) will be used (in sequential order) for anaerobic microbial degradation of organic matter and sulphides.

Not all organic matter and sulphide minerals are chemically active. Organic matter may be leached from the soil, but high groundwater tables are required to receive substantial loads of reactive organic matter. Sulphide containing minerals are of sedimentary origin or formed in place (only over geologically time scales). The pool of reactive substances is therefore practically limited and not renewable.

These reaction pathways have considerable relevance for the pollution of groundwater bodies, as i) also pesticides and BTX are organic substances, which are partly biodegradable (consuming oxygen) and ii) nitrate leaching into the groundwater may undergo denitrification irreversibly consuming the pools of reactive substances in the aquifer. Changes in water balance, water level, land use etc. can potentially alter the chemical processes in the aquifer.

The unsaturated zone is the zone between the soil surface and the groundwater surface. A certain time is required for water leaching from the soil to reach the groundwater surface, depending on the thickness of this zone. Solutes in percolating water may undergo significant biogeochemical transformations (for example degradation of organic matter).

Not all aspects and processes are as relevant in all aquifers at the same time. In fact, aquifers and groundwater systems show highly individual characteristics, which are also highly variable in space. Given the three-dimensional nature of groundwater flow and the slow transport velocities (resulting in considerable travel times), groundwater quality observations must take into consideration the connected surface area and the groundwater age for an appropriate assessment of the status of groundwater bodies, trends and future developments. Wells located in greater depths or old groundwater can not indicate pollution problems in shallow and younger groundwater. Deterioration of quality indicators in individual wells may indicate transition of a polluted groundwater parcel, but does not necessarily show response to reduced pollutant inputs. To better understand these processes, modelling has become a valuable tool for interpretation of observed data within a consistent framework.

1.2.2 Threats to groundwater systems

The main threats to groundwater systems can be summarized as alterations of the ground water balance and groundwater pollution. Alterations of the water balance are mainly associated with groundwater abstractions. Changing the abstraction level can result in overabstraction, where abstraction permanently exceeds the recharge, or development towards a new dynamic equilibrium of the groundwater system, having effects on discharge, storage and also recharge. Also changes of land cover due to sealing or de- or afforestation can have impacts on the water balance, mainly by alteration of groundwater recharge.

The main drivers of water abstractions from groundwater are domestic uses, industrial uses, and agricultural water use (mainly for irrigation purposes). Abstractions for cooling purposes are almost entirely taken from (and returned to) surface waters and therefore do not affect groundwater resources. Water demand for tourism has a considerable effect on local water abstractions in many regions, especially coastal regions in Southern Europe, putting ground water resources at risk.

Water abstractions by pumping wells alter the flow direction in the vicinity of the wells or the well field, forming local well catchments with a water flow directed towards the well. The position of a well within the aquifer has considerable effect on groundwater quality and the trends to be expected, as the position relates the well to flow path and travel time. Intersecting different geological layers may cause mixture of previously isolated groundwater bodies. These issues are not necessarily a threat on groundwater, but can become relevant in case of pollution problems.

Changes of groundwater recharge due to human intervention are mainly related to land use changes and to soil cultivation practices. Groundwater recharge is typically higher below agricultural land than below permanent vegetation such as forests and grassland, due to the lower crop density, shallower root systems and periods where the soil is bare and unprotected and evapotranspiration and plant interception are reduced. Sealing of land surfaces by extension of city areas, industrial areas, construction of streets etc. renders the related areas impermeable preventing groundwater recharge and increasing direct runoff, if no special measures for re-infiltration are taken. Sealing and forestation have a decreasing effect on groundwater recharge, while deforestation and conversion of grassland into arable land has an increasing effect. Also soil compaction and loss of soil aggregation due to inadequate management of soils have the potential to severely decrease infiltration potential, increasing surface runoff on cost of groundwater recharge.

There are also human activities increasing groundwater recharge. Conveyance losses in channels and the water distribution network (either wastewater or drinking water) may be responsible for substantial additional recharge. Conveyance losses occurring between water abstraction and delivery to the final recipient can account for more than 50% of water abstractions (indicative figure based on conveyance efficiencies in irrigation systems and EUROSTAT statistics on water returns and total water use). In the case of urban water distribution networks, they can be related to substantial input of microorganisms, organic matter and pollutants (waste water leakage). Also intentional re-infiltration of water occurs, for example as a part of waste water treatment or as measures for groundwater protection (for example creating barriers against seawater intrusion).

It was often perceived that once knowing the recharge and defining an environmental flow (i.e. a discharge required to maintain functioning of connected surface water bodies), the volume of water available for abstraction and human consumption would be defined. This issue is heavily discussed as the “water budget myth” (Bredehoeft, 2002). Indeed, pumping may result not only in decreased discharge, but also in increased recharge; therefore sustainable pumping rates can even be higher than the pristine recharge rates. A sustainable pumping rate is defined by the combined effect of increase in recharge and decrease in discharge, known as well-capture. Changing the balance of recharge and discharge increasing abstractions, the groundwater system adjusts to a higher pumping level by decrease of groundwater levels and storage. Therefore sustainable pumping rates can have negative or undesired environmental impacts. Instead, the term groundwater sustainability has a much broader meaning than just sustaining pumping rates, including not only long-term security of water supply but also avoidance of related environmental impacts such as decrease of groundwater levels, discharge reduction, sea-water intrusion, ground-subsidence etc. In this context, knowledge of groundwater recharge, discharge and abstractions is necessary to evaluate the sustainability of actual or prospected abstractions and to indicate potential risks.

Pollution problems result from pollutant leaching from diffuse sources or from point sources. Groundwater pollution poses a great risk to public health since the majority of the fresh water supply occurs as groundwater. Many of the groundwater pollutants are colourless, odourless, and tasteless. Degradation of groundwater supplies also occurs as a result of poor waste-disposal practices or poor land management.

Diffuse pollution is mainly related to agricultural activities. Fertilizer and organic chemicals percolate through the soil and reach the groundwater level with the recharge flux. Nitrate and pesticides are typical diffuse pollutants related to agricultural activities. Landfills, septic tanks, leaky underground gas tanks act as point sources from which pollutants leak into groundwater. Pollutants infiltrate to the groundwater either as dissolved components (with the recharge flux), as gas or they are liquids themselves, either immiscible or miscible with water.

In case of seawater intrusion and groundwater salinisation, the pollution problem is closely related to the alteration of the groundwater balance. Coastal groundwaters have an open boundary towards the sea. The position of the transition zone (interface) from sweet groundwater to salty seawater is defined by the groundwater discharge into the sea, depth of the aquifer and the density difference between saltwater and freshwater. Under stable conditions, freshwater is underlain by a saltwater wedge, with an outflow of freshwater at the top of the aquifer and the intrusion of saltwater below balanced by the discharge flow and pressure gradients in the system. An increase of abstractions (irrigation, tourism, population growth, etc.) or decrease in recharge (climate change, land use change) lower discharge fluxes and groundwater levels. Drainage measures may directly lower groundwater levels in the aquifer. This will result in an alteration of the seawater-freshwater balance, causing seawater to intrude into the aquifer (Bear et al., 1999). The intrusion process can be directed towards a new equilibrium (if net recharge is positive) or be a permanent process replacing freshwater with saltwater until abstraction is stopped, when a negative net recharge reverses the flow direction from the sea into the aquifer. A methodology for large scale assessment of seawater intrusion risk was proposed by Wriedt and Bouraoui (2009c).

Capillary rise of water from the groundwater to the soil is a typical mechanism of soil salinisation. The evaporating water leaves the dissolved salts at the soil surface, deteriorating soil quality. A rise of groundwater levels can potentially dissolve salts stored in the unsaturated zone (dryland salinization) and result in increased pollution of upper groundwater and increased salt loads to surface waters. Circulating water within a closed aquifer-irrigation system can result in groundwater salinization, as salts extracted with irrigation water are returned to the aquifer during irrigation, while part of the irrigation water is lost by evaporation thus concentrating the salts in the groundwater (Milnes and Renard, 2004).

Groundwater systems closely interact with other ecosystem compartments, such as soils, rivers, lakes, wetlands, swamps and bogs. Declines of groundwater levels may change rivers from gaining to loosing streams, decrease river discharge, and dry out groundwater dependent ecosystems. Even wetlands geographically separated from groundwater systems can be affected, as groundwater related changes of the river discharge will affect such wetlands.

1.2.3 Modelling tools

In groundwater research, different types of models are applied. Numerical groundwater models include models for groundwater flow, groundwater transport and geochemical models. The model domain is typically subdivided into defined spatial units and time steps (discretization). The flow and transport equations are defined for each unit and solved simultaneously applying specialized numerical algorithms. Numerical models are the most flexible class of models and can be applied to a broad variety of groundwater related problems in multiple dimensions and under steady or transient boundary conditions. Numerical constraints on the size of spatial units and time steps and the required calculation time limit either the spatial extent of the problem that can be analysed or the spatial detail in which the model can be applied. The application to real world problems is quite data demanding and labour intensive. Available input data on aquifer characteristics and water fluxes are typically scarce and model results are subject to considerable uncertainty.

Geochemical models facilitate analysis of complex chemical processes and reaction systems in subsurface environments. Typical applications include modelling the distribution and fate of pollutants, changes of groundwater quality by natural processes or human intervention, remediation measures and so on. While initially this class of models was constrained to simple transport scenarios (batch experiments, one-dimensional, steady state flow), there has been extensive development to integrate geochemical modelling approaches with numerical transport models. This allows analysing pollution problems under transient conditions and in more complex (multi-dimensional) flow systems.

Analytical models solve flow and transport processes by closed mathematical equations. They can be solved only for relatively simple and well constrained situations, as complex problems can not be solved analytically. They are typically applied for illustrative purposes, for screening purposes, or as benchmark solutions to test numerical models.

Lumped or black box models simplify aquifers into single spatial units, to which the water and solute balance equations can be applied (using numerical or analytical solutions). A further differentiation within the aquifer is neglected. They can be based on effective properties and require less input data than spatially differentiated numerical models. They can be applied to problems where an aquifer is considered as a whole, but are not applicable when process differentiation within an aquifer is required.

Integrated hydrological models also consider groundwater flow and sometimes also solute balance for specified substances. Typically, the groundwater storage is simplified by lumped models for a shallow and deep aquifer. The designation of aquifers is however not based on their true extent but bound to the definition of spatial units in the hydrological models (catchments, sub-catchments, hydrological response units). Again, they can not be applied to all groundwater related problems, especially solute transport and chemical reactions have to be neglected or extremely simplified. Couplings to numerical groundwater models exist for some hydrological models, but they are difficult to implement. Groundwater recharge as a main input for groundwater models often has to be derived from related water balance modelling. Groundwater discharge to rivers and riverbed infiltration also require surface runoff and river systems to be taken into account. On the other hand, hydrological models must implement groundwater systems to properly account for the effects of groundwater recharge and discharge on the catchment runoff characteristics.

1.2.4 Requirements for large scale assessments at European scale

In section 2.1 we listed various data sources supporting hydrologic modelling at continental scale. Such information included topography, river and catchment networks, climatic data, soil data, runoff time series and even dominant surface geology.

To focus more specifically on groundwater resources, an ideal solution to assess the state of groundwater resources would be to define a lumped groundwater balance over a specified region (ideally a groundwater system defined as a closed functional unit). While this can be done at local or regional scale, severe limitations have to be tackled when it comes to model based assessment of groundwater systems at continental scale.

- Groundwater balance components recharge, abstractions and discharge must be determined modelling soil water balance and rainfall runoff relations. They can not be determined independently and are also interacting with each other. Abstractions must not be neglected in many areas. This requires an integrated approach quantifying the entire water cycle in a consistent way.

- Consistent geological data of European extent are not available that would allow deriving information on extent, depth, and volume of aquifer (or aquifer systems) and representative hydrogeological properties. It is therefore not possible to define functional groundwater modelling units a priori with reasonable settings. The WFD requires identification of water bodies serving as fundamental units for assessment and management. The WFD allows groundwater bodies to be distinguished according to chemical, geologic or other suitable criteria. There is, however, no requirement to distinguish groundwater bodies as functional hydrological units considering the groundwater shed, flow system and interaction with other groundwater bodies. This renders the designation of groundwater bodies in WFD useless for model based assessments, as the boundaries may be practically open and neither water balance assessments nor rough estimations of transport characteristics such as travel time can be based on these units.
- Infiltrating rivers can potentially contribute substantially to groundwater recharge. Groundwater-surface water interactions can not be represented in European scale assessment, as the relevant process scale can not be represented at the modelling unit scale.
- Hydrograph data for validation of rainfall runoff models and baseflow estimations can only be applied when the catchment remains small. In large river basins the hydrograph response may be affected by the impact of reservoirs and lakes, the internal organisation of the basin with interacting catchments and groundwater systems and non-homogeneous climatic, topographic and edaphic conditions, resulting in an apparent behaviour not representing runoff related processes. Remote sensing approaches may be useful to directly calculate evapotranspiration and estimating groundwater recharge from the local soil water balance.
- The extent of the geographical region of interest puts limitations on a detailed collection of relevant local data and their integration into databases and modelling tools.

For water quality aspects additional information is necessary. An indication of travel times and reactive capacity of groundwater can give valuable information on the potential response and vulnerability of aquifers to pollution risks. This requires information on extent, volume and structure of groundwater systems to evaluate flow path lengths and turnover times. A geochemical classification of aquifers can help to assess natural attenuation potential.

2 Studies on water balance and groundwater recharge

2.1 Methodological approach

A general water balance assessment shall provide information on the amount of water available for surface and groundwater runoff (hydrological excess water). It should further allow distinguishing runoff components such as direct runoff, interflow and groundwater runoff, providing an interface to a more specific analysis of the groundwater balance.

As indicated in section 1.2, the groundwater balance is defined by recharge, abstractions and discharge, the interaction of all these components defining storage and groundwater levels.

In contrast to other parts of the water cycle, such as precipitation and total runoff, groundwater recharge is difficult to quantify. It is not only affected by meteorological input and soil properties, but also by aquifer properties, and groundwater storage itself (groundwater levels, gradients and surface water interactions). The scale of interest has considerable impact on conceptualising flow components and separating groundwater discharge and non-groundwater subsurface runoff (interflow). Also human interventions in form of groundwater abstractions or artificial recharge can feed back into recharge processes.

For large scale assessments, water balance models in combination with suitable spatial data are the only option to estimate groundwater recharge. A validation of such simulations is hardly possible at

European scale. Experimental approaches are only applied locally and on a case-by-case basis, consideration in a general European concept is therefore not suitable. Potentially remote sensing approaches to calculate water balance components can provide a way out of this dilemma.

Groundwater discharge and baseflow indices can be quantified from hydrograph analysis (Eckhardt, 2008). Such approaches allow separation of fast and slow runoff components. Slow runoff components are generally interpreted as baseflow or groundwater discharge, which is conventionally used as an estimator of groundwater recharge. It is important to note that different methods result can produce considerably different results (Eckhardt, 2008). It must be assumed, however, that abstractions are negligible compared with total runoff. Consequently, hydrograph data from rivers highly affected by human interventions are not suitable for estimating groundwater discharge.

As pointed out by Wriedt (2009d), a spatially distributed estimation of groundwater abstractions at large scale can not be based on available data and a suitable estimation procedure still needs to be developed, requiring iterative allocation of water abstractions in space and to ground- and surface water sources to meet water requirements.

At the current state of work, we present different analyses that were carried out to evaluate the potential of available data for groundwater specific assessments. These analyses include the following:

- We implemented three hydrograph separation methods presented by Ruthledge, (1998), Smakhtin (2001) and Bogena et al. (2005) and tested them with the available monthly and daily runoff time series.
- Further we calculated baseflow indices, defining the ratio of groundwater runoff and total runoff. The hydrograph separation approach of Bogena et al. (2005) allows direct calculation of baseflow indices and recharge estimates.
- An a priori estimation of the baseflow index from physio-geographic catchment characteristics and heuristic assumptions was based on the approaches presented by Bogena et al. (2005), de Witt (1999), Meinardi et al. (1994) and Döll et al. (2002).
- Based on the available physio-geographic data, we also calculated a characteristic groundwater travel time, combining travel distance and substrate conductivity at catchment level and providing information indicating groundwater response to pollution inputs.
- The generic water balance model and the heuristic baseflow index were combined to calculate groundwater recharge potential in EU27 and Switzerland at catchment level.

2.2 Available data sources

2.2.1 Climatic data

Climatic data were taken from the MARS climatic database (Micale and Genovese 2004) developed by MARS/AGRIFISH Unit of the Joint Research Centre. The database was created by interpolating observations of meteorological stations across Europe to a 50 km square grid. Meteorological data included are daily maximum and minimum temperature, vapor pressure, wind speed, rainfall, potential evapotranspiration, global radiation (calculated) and snow depth. The database used contained daily data for the time period 1990 – 2003.

2.2.2 Land use information and elevation model

CORINE Land cover 2000 (ETC, 2005) provides a high resolution data set of land use over Europe at a resolution of 1 ha. The minimum mapping unit is 25 ha and the map scale is 1:100000. CORINE Land Cover maps the spatial distribution of various land use categories, including irrigated and non-irrigated arable land, rice and various permanent crop classes. Permanently irrigated land and rice fields were distinguished based on detection of irrigation infrastructure (water supply and drainage canals).

The elevation model is based on SRTM data and covers Europe in a resolution of 1ha. This elevation model was used as a basis to calculate slopes and to delineate catchments and river basins.

2.2.3 Soil properties

The soil data required were derived from the European Soil Database 2.0 (ESDB) developed by the European Soil Bureau Network (ESBN). The European Soil Database is the only comprehensive source of soil data in Europe harmonised according to a standard international classification (FAO, Jones et al. 2004). The soil mapping units of the geographical database are derived at a scale of 1:1.000.000 and can be linked to Soil Typological Units containing soil parameter data. The geographical database also reports soil properties for dominant and secondary soil types, for example texture and available water capacity and information on parent materials.
Water holding properties of soils were estimated from soil texture using the pedotransfer functions defined by Rawls and Brakensiek (1985):

$$WP = 0.026 + 0.005 \cdot CL + 0.0158 \cdot OC \quad [1]$$

and

$$FC = 0.2546 + 0.002 \cdot SA + 0.0036 \cdot CL + 0.0299 \cdot OC \quad [2]$$

where WP = Wilting point (cm/cm), FC = field capacity (cm/cm), SA = sand content (%), OC = organic matter (%), CL = clay content (%).

2.2.4 Geology and aquifers

The European Soil Database 2.0 (ESDB) provides information on dominant and secondary parent material of the soil mapping 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. This does not exclude, however, existence of local aquifers which are not resolved in the data set.
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. The ESBN has also compiled a European map of groundwater resources. The map includes only few member states and the classification does not go beyond a rough indication of aquifer types. The information included does not go beyond the ESDB 2.0 and therefore the map was not further considered.
While the ESDB can give information on the distribution of geological substrates at the surface, it does not give information on the spatial extent of groundwater bodies or the vertical structure of aquifer systems (also none of the other datasets discussed above can provide this information). Though national agencies collect more detailed hydrogeological information, including borehole data and structural data, there have not been attempts so far to combine this information to develop a European scale information system on hydrogeological units that aggregates the country specific information in a way applicable for large scale modeling purposes. This is, however, not surprising having said that aquifer management requires analysis of individual aquifers and groundwater bodies. It would be desirable, however, to develop a classification of groundwater systems that can be mapped and backed up with ‘representative’ aquifers.

For modelling purposes, the hydrogeological properties of the substrates need to be characterised. 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 and 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. Further, substrate properties are highly variable in nature and data obtained at one location may not be valid for the same substrate type in another location. In summary, the substrate properties do not reflect the broad range of possible values within on and the same material. They are indicative only but may not reflect true local conditions.

For the purposes of this study, we combined the parent materials defined in ESDB with the substrate properties by Morris and Johnson (1967). Substrates for which information was available only partly correspond to parent materials defined in the ESDB. Where the 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.

2.2.5 Spatial units for statistical data

Statistical information is usually based on administrative regions. For European statistics, a system of territorial units was developed at different hierarchical levels (Nomenclature of Territorial Units for Statistics NUTS). We applied a customized layer of NUTS-regions (NUTSFATE) that was developed previously (Bouraoui and Aloe, 2007; Mulligan et al., 2006) combining NUTS regions from level 2 ('provinces') and level 3 ('districts').

2.2.6 Hydrography and River discharge data

A European catchment database (HydroEurope database) was developed at the JRC's IES-RWER[1] based on the SRTM digital elevation map and compatible with the ArcHydro data model, allowing storing different hydrological information in a consistent way for various types of analysis. The catchment database comprises a catchment layer and a river basin layer. River basins were defined by all catchments draining towards a single sea-outlet. Catchments were delineated requiring a minimum size of 100 km^2 based on topological features.

The shape and extent of *groundwater catchments* depends also on the geological set up of a region (in three dimensions) and may therefore differ considerably from surface water catchments. However, large scale geological information is not suitable to make assumptions on the extent of aquifers and groundwater bodies. We therefore use the surface water catchments as basic hydrological unit for general water balance and groundwater balance considerations.

An intersection layer of the catchment layer and the NUTS regions was created to transfer data between NUTS regions and catchments. Another intersection layer was created to transfer data between catchment and a layer of 10x10km grid cells used in databases and modelling applications of the FATE working group at IES-RWER.

Discharge data were collected from various sources, such as national water boards, the global runoff data centre (GRDC) and research institutions. The collected data include daily, monthly or annual discharge data for varying time periods (the late 60's until 2005, differing by station). River discharge

[1] Joint Research Centre (JRC), Institute of Environment and Sustainability (IES), Rural, Water and Ecosystems Resources Unit (RWER)

data are related to gauging stations. The gauging stations were assigned to the corresponding catchments defined in the HydroEurope database.
For modelling purposes, the original catchment and basin structure was enhanced by an additional structure defining basins and sub-basins by monitoring stations rather than sea-outlets. The most downstream monitoring station defines a basin, consisting of all catchments draining towards this station. A basin may consist of several nested sub-basins, where each sub-basin is defined by all catchments draining towards a monitoring station, excluding catchments belonging to another sub-basin defined by an upstream monitoring station (Figure 1).

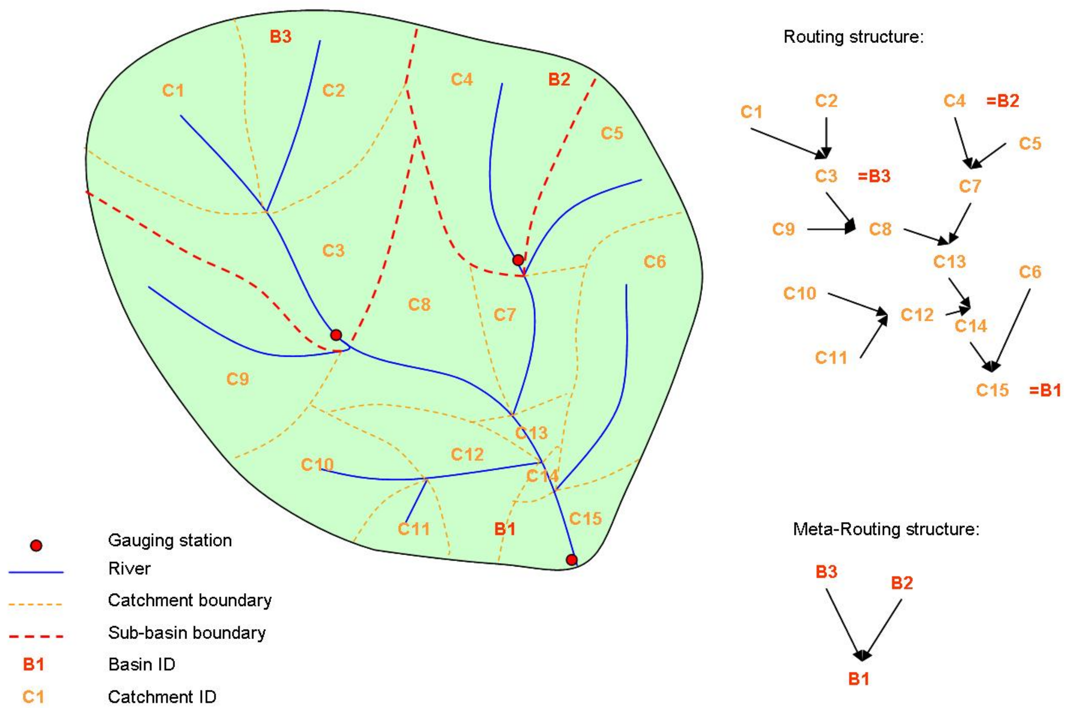

Figure 1: Conceptual basin and catchment structure

2.3 Water balance modelling using a generic monthly water balance model

2.3.1 Generic water balance modelling approach

A generic monthly rainfall-runoff model to calculate hydrological excess water (water available for surface and subsurface runoff) was applied for each catchment.
The rainfall series was corrected for effects of snow accumulation and snow melt using a simple degree-day approach (Schulla and Jasper, 2007). The original approach was developed for daily data. It was here applied at monthly time scale to account for longer winter periods with snow storage.

Step 1 $$f_T = \frac{T_{RS} - T_{Trans} - T}{2 \cdot T_{Trans}}, \quad 0 \le f_T \le 1 \qquad [3]$$

Step 2 $$SNOW(t) = SNOW(t-1) + f_T \cdot P(t) \qquad [4]$$

Step 3 $$MELT(t) = c_{0m} \cdot \left(T - T_{0,m}\right) \cdot \frac{\Delta t}{24}$$ [5]

Step 4 $$SNOW(t) = SNOW(t) - MELT(t)$$ [6]

Step 5 $$P^*(t) = (1 - f_T) \cdot P(t) + MELT(t)$$ [7]

where SNOW = snow storage (mm), P = precipitation (mm), f_T = snow fraction of precipitation, T_{RS} = temperature at which 50% of precipitation falls as snow (ºC), T_{Trans} = transition range from snow to rain (ºC), T = Air temperature (ºC), c_{0m} = melt factor, T_{0m} = melt temperature, P* = effective precipitation (mm)

The soil water balance was based on the modelling approach suggested by Pistocchi et al. (2008) using simple phenomenological relations describing direct runoff (RO), actual evapotranspiration (ETA) and infiltration (=deep percolation) F* at a monthly time scale.

Step 1 $$RO(t) = (1-\gamma) \cdot \phi \cdot P^*(t) + \gamma \cdot \frac{\left(P^*(t) - 0.2 \cdot S\right)^2}{P^*(t) + 0.8 \cdot S}$$ [8]

Step 2 $$ETA(t) = \frac{P^*(t)}{\left(\alpha + \left(\frac{P^*(t)}{ETP(t)}\right)^{\beta}\right)^{\frac{1}{\beta}}}$$ [9]

Step 3 $$F^*(t) = P^*(t) - ETA(t) - RO(t)$$ [10]

Step 4 $$HXS(t) = RO(t) + \left(F^*(t) \middle| F^*(t) \geq 0\right)$$ [11]

Step 5 $$Q_{SF}(t) = RO(t)$$ [12]

Step 6 $$Q_{SSF}(t) = \left(F^*(t) \middle| F^*(t) \geq 0\right)$$ [13]

where RO = direct runoff (mm), P(*) = (effective) Precipitation (mm), S = Storage potential (coarse soils: 400 mm, other: 250 mm), γ= retention coefficient (=0.75), ϕ = runoff coefficient (coarse soils: 0.47, other: 0.59), ETA = actual evapotranspiration (mm), α = 1, β=1.5, F* = net infiltration (mm), HXS = hydrological excess water (mm), Q_{SF} = direct runoff (mm), Q_{SSF} = fast and slow subsurface runoff (mm).

The calculation of direct runoff and infiltration does not take into account effects of relief, slope or geology on the runoff process. The calculated runoff components are mainly determined by climatic factors and soil infiltration capacity on a horizontal plane. Reinfiltration, interflow and other runoff processes relevant at catchment level and for groundwater recharge are not considered in this approach. As actual evapotranspiration was calculated independently, the total hydrological excess water is not affected by these issues.
The model and the routing routines were implemented as VB.NET applications. An external routing module allows accumulation of runoff and other water balance terms from upstream catchments to downstream catchments and to further aggregate results to river basins and monitoring stations.
Input data were derived from the sources described above, i.e. climatic data from the MARS database, soil data from the European Soil Bureau Database and land use information from CORINE land cover. The model was applied to all catchments and routing was performed for the natural catchment network structure (to river basins) and for the extended structure according to the monitoring stations. The model was applied without specific calibration, using the default values given by Pistocchi et al. (2008).

2.3.2 Results of the generic water balance simulation

The generic water balance simulation provides an estimate of hydrological excess water (Figure 2) and further allows distinguishing a direct runoff (Figure 3) and a subsurface runoff component (Figure 4). The ratio of precipitation to potential evapotranspiration ratio (Figure 5) shows patterns similar to the hydrological excess water, indicating that the model is consistent with the main input data.
The simulation period 1990-2003 covers a period of 14 years only. Hydrological excess and baseflow vary considerably as indicated by the minimum and maximum annual values in Figure 7.
The separation of direct and subsurface runoff components is based on soil storage potential and precipitation only. Topographic factors do not come into play. The association of the calculated flow components with surface and subsurface runoff is therefore tentative. A further separation of the subsurface runoff component into fast and slow subsurface runoff can be based on the heuristic baseflow index, adding the specific physiogeographic information of a catchment to the identification of flow components (see 2.11).
A direct comparison of average annual hydrological excess water and observed runoff shows no obvious direct relation (Figure 8). This does not, however, invalidate the model per se. The model at the current stage lacks various features that are necessary to represent runoff dynamics at river basin level limiting the capacity for direct comparison of observed and simulated results:

- The model was applied with default parameters that were not calibrated individually to catchments or basins. Topographical and geological features are not considered in the runoff generation process that might have been captured by individual calibration.
- The simulated runoff components are immediately released as runoff and the retarded runoff of groundwater and the effects of lakes and reservoirs are not included. The visual comparison of simulated and observed hydrographs revealed that a frequent failure is the incapability of the model to represent the correct timing of hydrograph peaks.
- Water abstractions are not considered in the model, which can have impact on discharge at least in southern European river basins, where abstractions can be a substantial part of runoff.

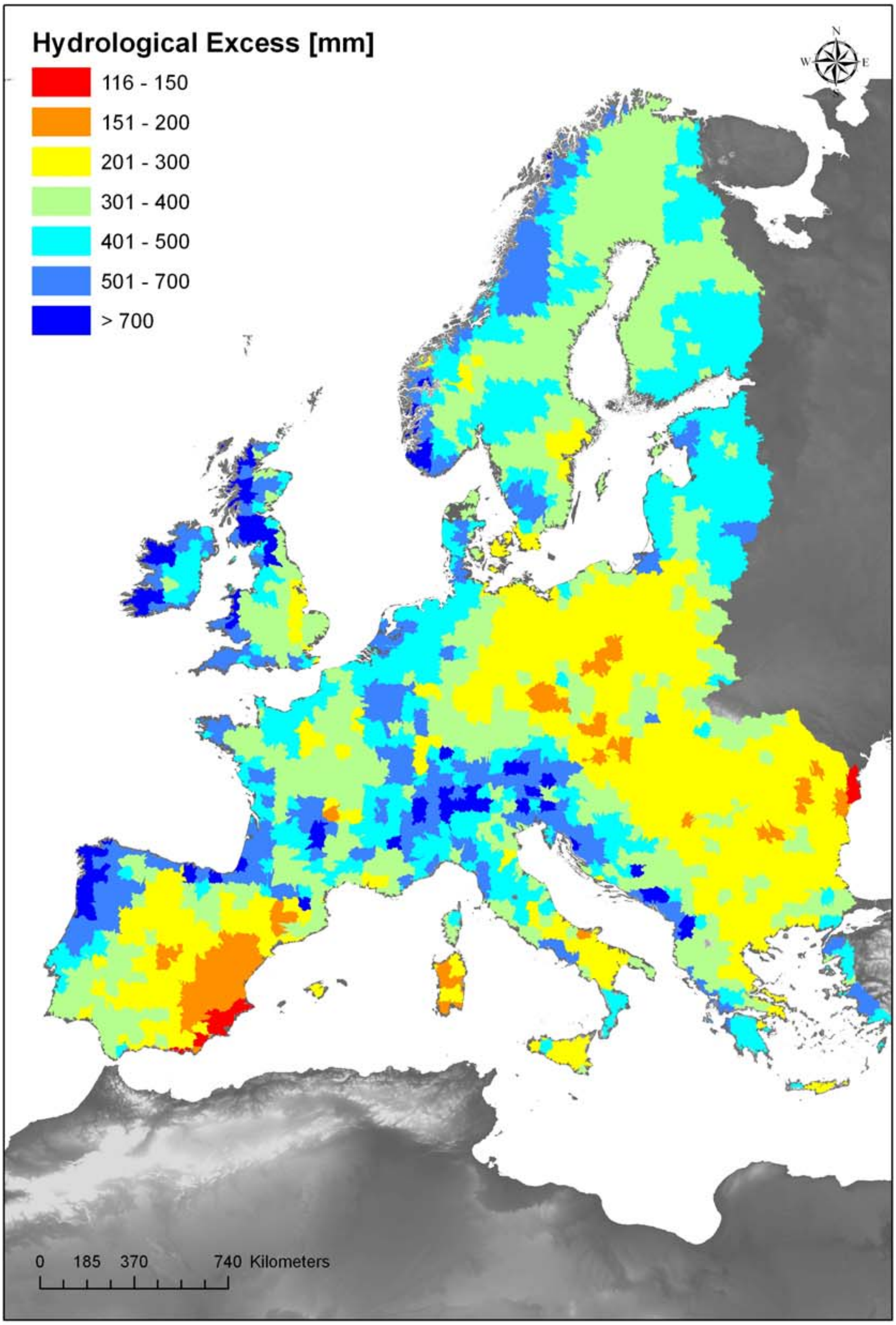

Figure 2: Calculated hydrological excess water [mm] available for surface and subsurface runoff

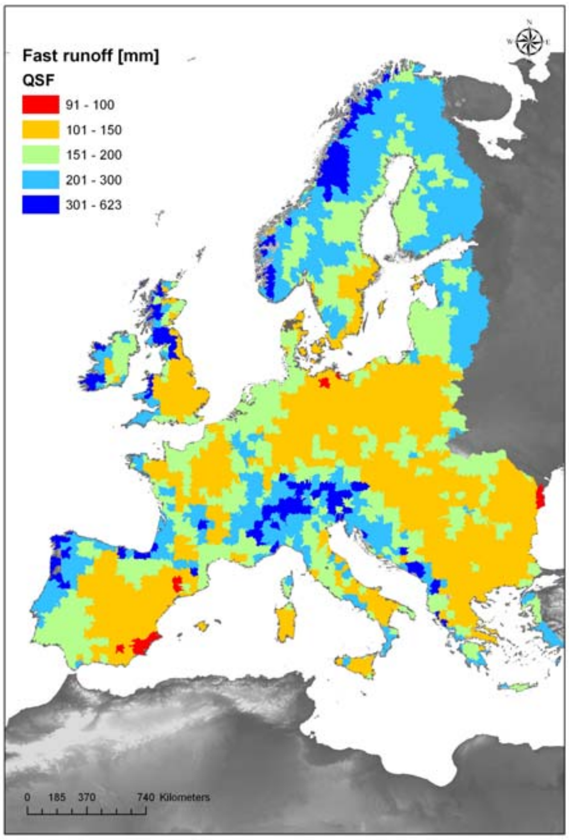

Figure 3: Calculated direct runoff (partly including fast subsurface runoff) [mm]

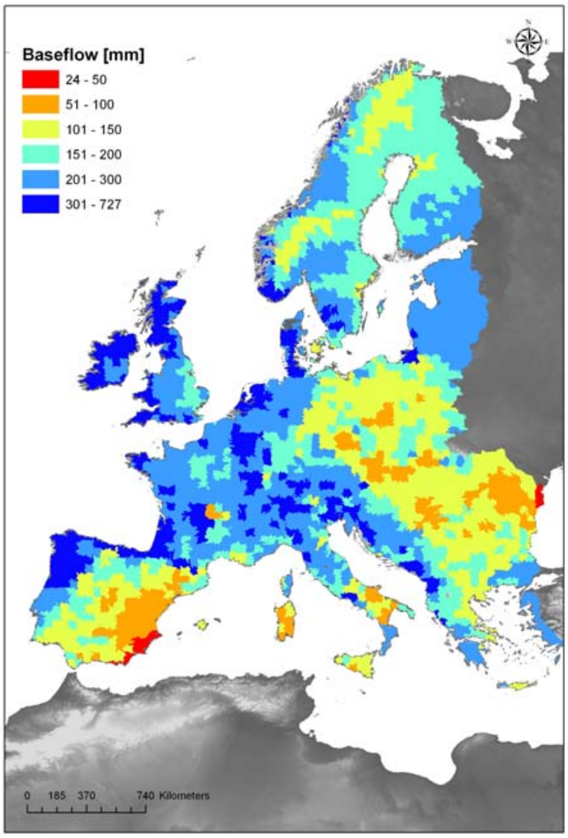

Figure 4: Calculated subsurface runoff (fast and slow subsurface runoff) [mm]

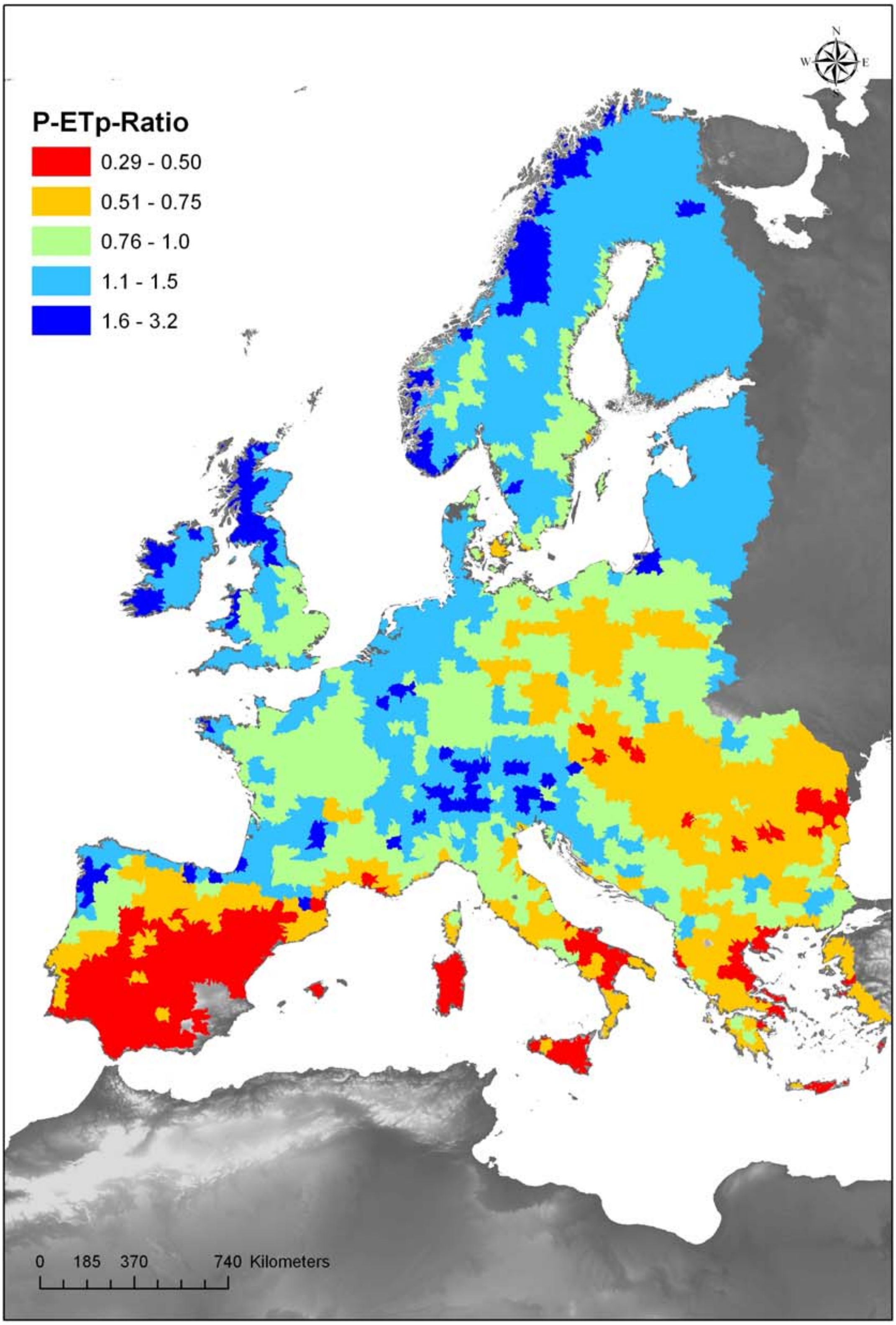

Figure 5: Ratio of precipitation and potential evapotranspiration [-]

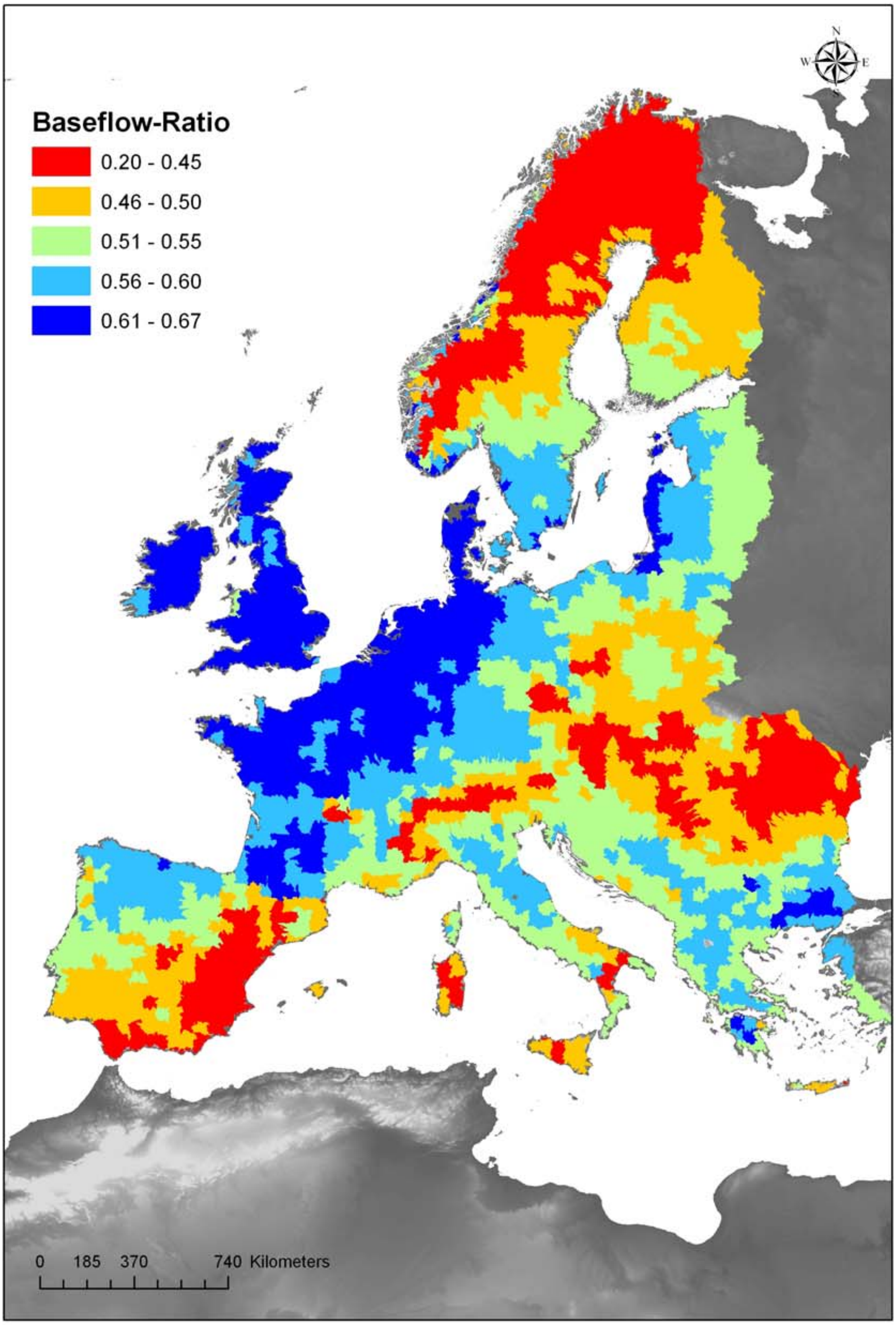

Figure 6: Calculated ratio of infiltration to total runoff (hydrological excess)

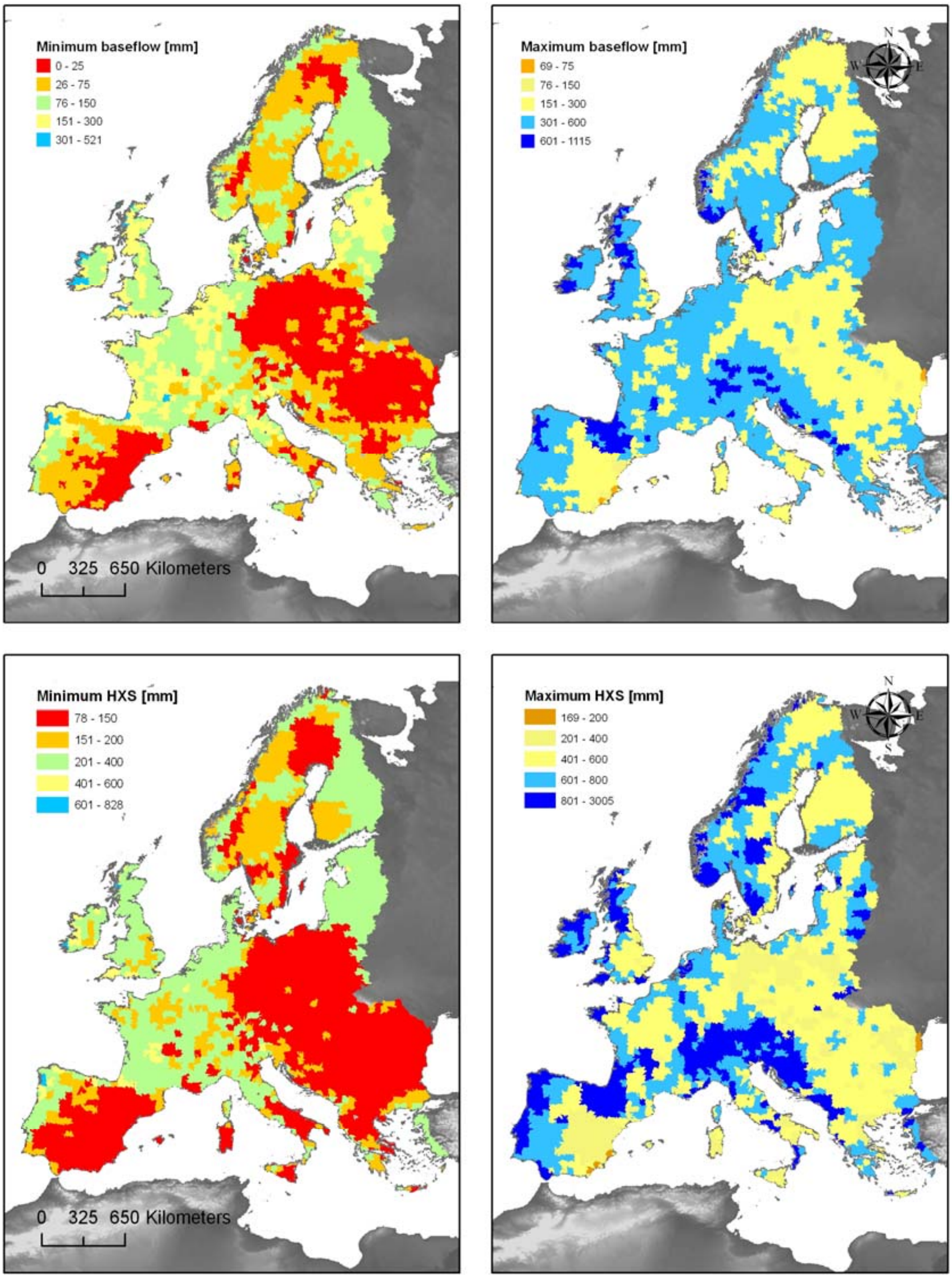

Figure 7: Minimum (left) and maximum (right) of annual infiltration (top) and hydrological excess water (bottom) in [mm]

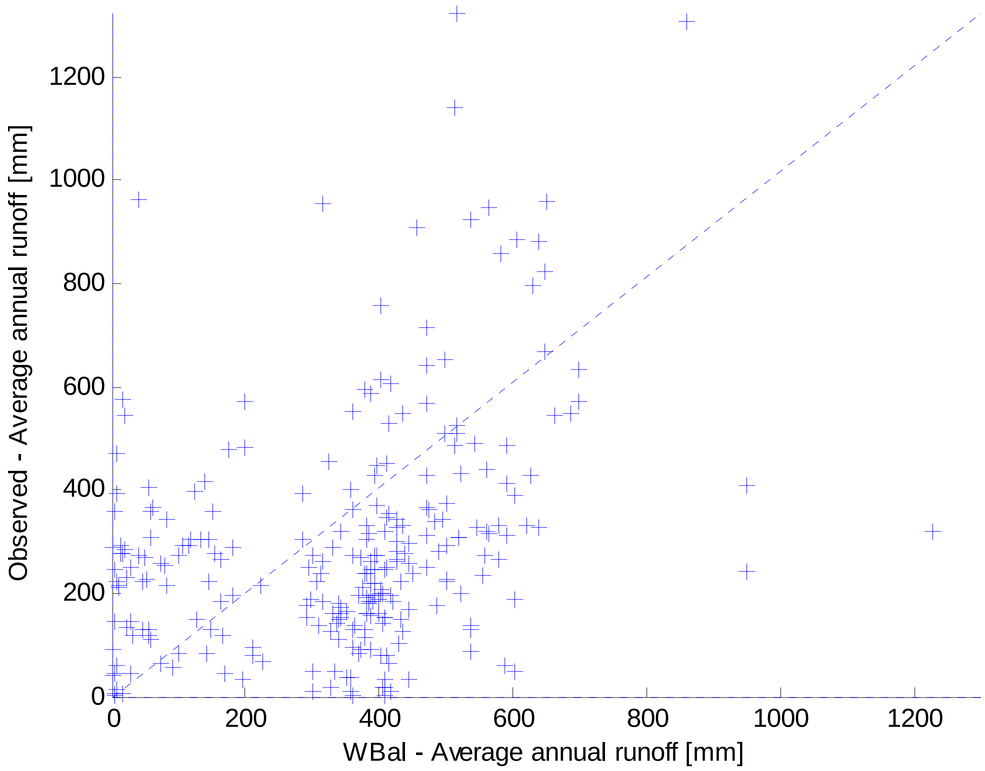

Figure 8: Comparison of average annual runoff calculated with the generic water balance model (WBal) and observed values. The dashed line indicates a 1:1 relation.

2.4 GR2M rainfall runoff simulation

2.4.1 GR2M rainfall runoff model

GR2M is a monthly rainfall-runoff model. It was developed at CEMAGREF, France by Mouelhi et al. (2006). Developed at a monthly time step, GR2M is relatively simple and has only two model parameters and two initial conditions. The model parameters X1 defines the maximum size of the production storage and X2 defines the water exchange to other sinks not related to catchment runoff (exchange with other basins, abstractions). Initial conditions are the initial production store storage and the initial routing store storage. Figure 1 presents the conceptual model scheme of the GR2M model, including the basic model variables and parameters. The corresponding model equations are given in Table 1. For more specific information on the GR2M model, we refer to the available literature.

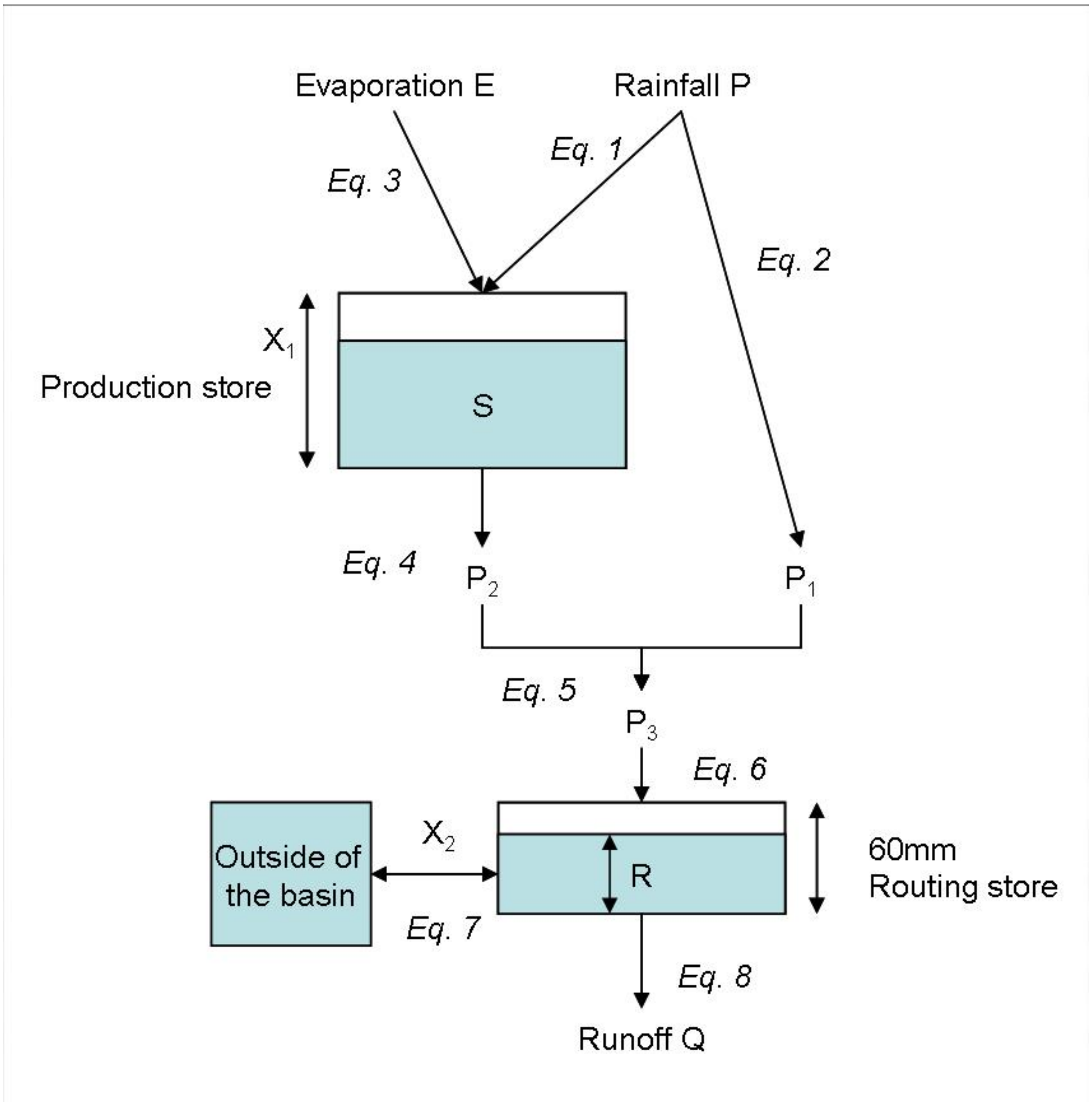

Figure 9: GR2M model scheme (Mouelhi et al., 2006, modified)

Table 1: GR2M model equations (Mouelhi et al., 2006)

Nr. Figure	Equation	
(1)	$S_1 = \dfrac{S + X_1\varphi}{1+\varphi\cdot\dfrac{S}{X_1}} \quad where \quad \varphi = \tanh\dfrac{P}{X_1}$	[14]
(2)	$P_1 = P + S - S_1$	[15]
(3)	$S_2 = \dfrac{S_1\cdot(1-\psi)}{1+\psi\cdot\left(1-\dfrac{S_1}{X_1}\right)} \quad where \quad \psi = \tanh\dfrac{E}{X_1}$	[16]
(4)	$S = \dfrac{S_2}{\left[1+\left(\dfrac{S_2}{X_1}\right)^3\right]^{1/3}} \qquad P_2 = S_2 - S$	[17]
(5)	$P_3 = P_1 + P_2$	[18]

(6)	$R_1 = R + P_3$	[19]
(7)	$R_2 = X_2 \cdot R_1$	[20]
(8)	$Q = \dfrac{R_2^2}{R_2 + 60} \qquad R = R_2 - Q$	[21]

The original model is deployed as FORTRAN code. We developed a version programmed in VB.NET and added various functionality for model evaluation, routing model results in a basin structure and accessing databases to automatically generate model input files. An additional software package GR2MBasins was developed to run GR2M in the context of river basins consisting of inter-connected catchments. GR2MBasins applies GR2M to individual catchments and accumulates the discharge flux downstream along the catchment network. Results are given for each sub-basin and river basin.
The GR2M model was applied to nested sub-basins as defined by the available gauging stations. Within each sub-basin, the model parameters for each catchment were considered to be identical. This was necessary because monitoring stations are the only reference for model calibration. Further differentiation within the basin was not possible because model parameters are not linked to measurable catchment properties. Climatic data were taken from the MARS database. Specific soil and land use information were not required. The catchment structure was derived from the HydroEurope database.
The model was calibrated using the Gauss-Marquardt-Levenberg Algorithm implemented in the PEST software package (Doherty, 2002). To avoid errors due to poor model performance in upstream sub-basins, observed runoff was used as input from upstream sub-basins instead. We calibrated model parameters (X1, X2) and initial conditions (Si and Ri) for each sub-basin. All available data were used for calibration and no separate calibration-validation periods were distinguished. We refer to the calibration run as GR2M-A. Having the calibrated parameters, the model was re-run routing simulated fluxes instead of observed fluxes (GR2M-B).
To evaluate model performance we calculated Nash-Sutcliffe index (NSC), Index of Agreement (IOA) and Index of Volumetric Fit (IVF) for each monitoring station.

2.4.2 GR2M simulation results

The GR2M simulation included 492 catchments for which daily or monthly runoff data were available. The geographical focus was on Germany, France, Spain and Portugal.
Figure 10 displays the scatter of observed average annual runoff and the simulated runoff with observation routing (GR2M-A). GR2M represents the observed values relatively well, with a tendency to overestimate the observed values. The calibrated model parameter X1 allows taking into account soil storage and topographical features. Gr2m also implicitly accounts for abstractions by calibrating X2. However, X2 determines a fixed fraction of water loss, seasonal patterns of abstractions, for example resulting from irrigation during the growing period, can therefore not be considered.
As main GR2M results we present the simulated runoff (Figure 11), the Nash-Suttcliffe efficiency with and without routing observed fluxes (Figure 12 and Figure 13) and the model parameters X1 (production storage, Figure 15) and X2 (exchange coefficient, Figure 16). In some basins, especially in Spain and Southern France, Nash-Sutcliffe efficiencies indicate insufficient model performance, while in Central European basins the model performs better. Omitting the observation routing, a drop in model efficiency can be observed in basins that receive considerable amounts of water from upstream basins, as poor simulation results are then propagated downstream (Figure 13).

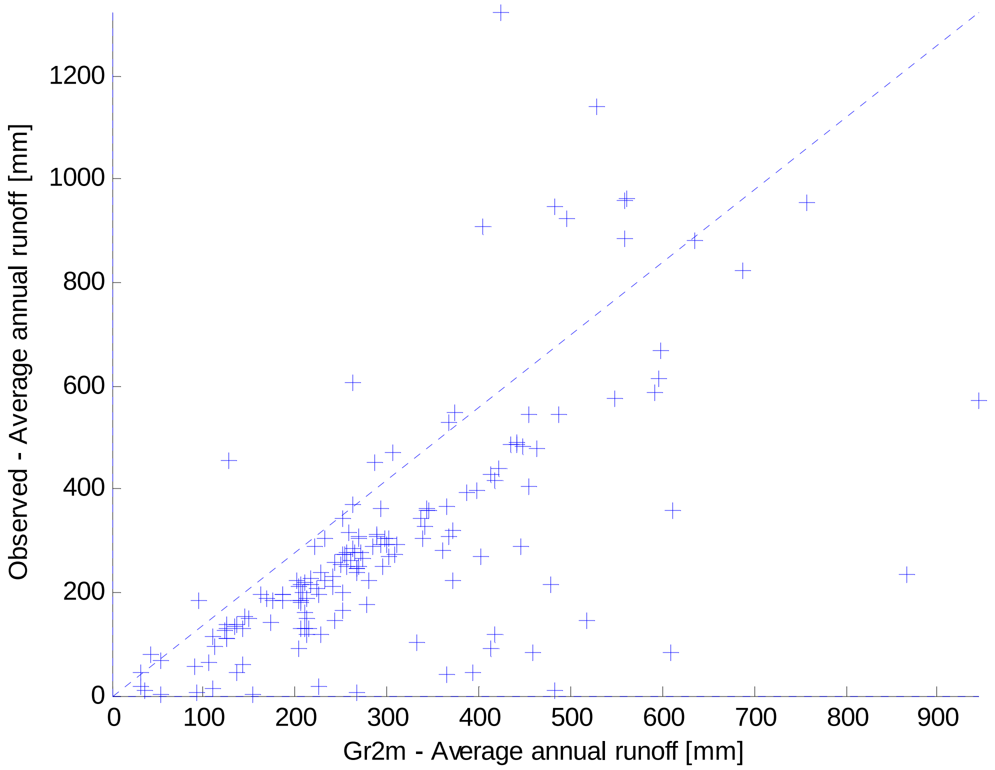

Figure 10: Comparison of average annual runoff calculated with theGr2m model and observed values. The dashed line indicates a 1:1 relation.

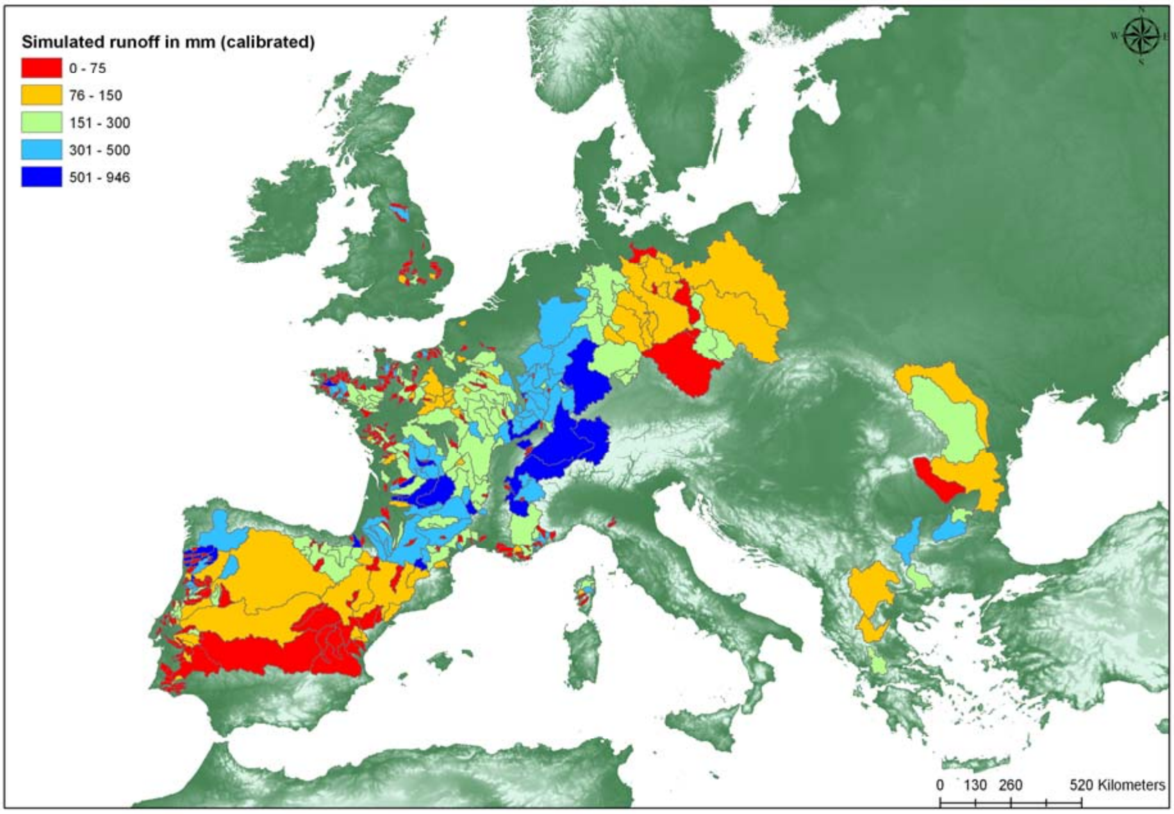

Figure 11: GR2M - Simulated runoff in mm

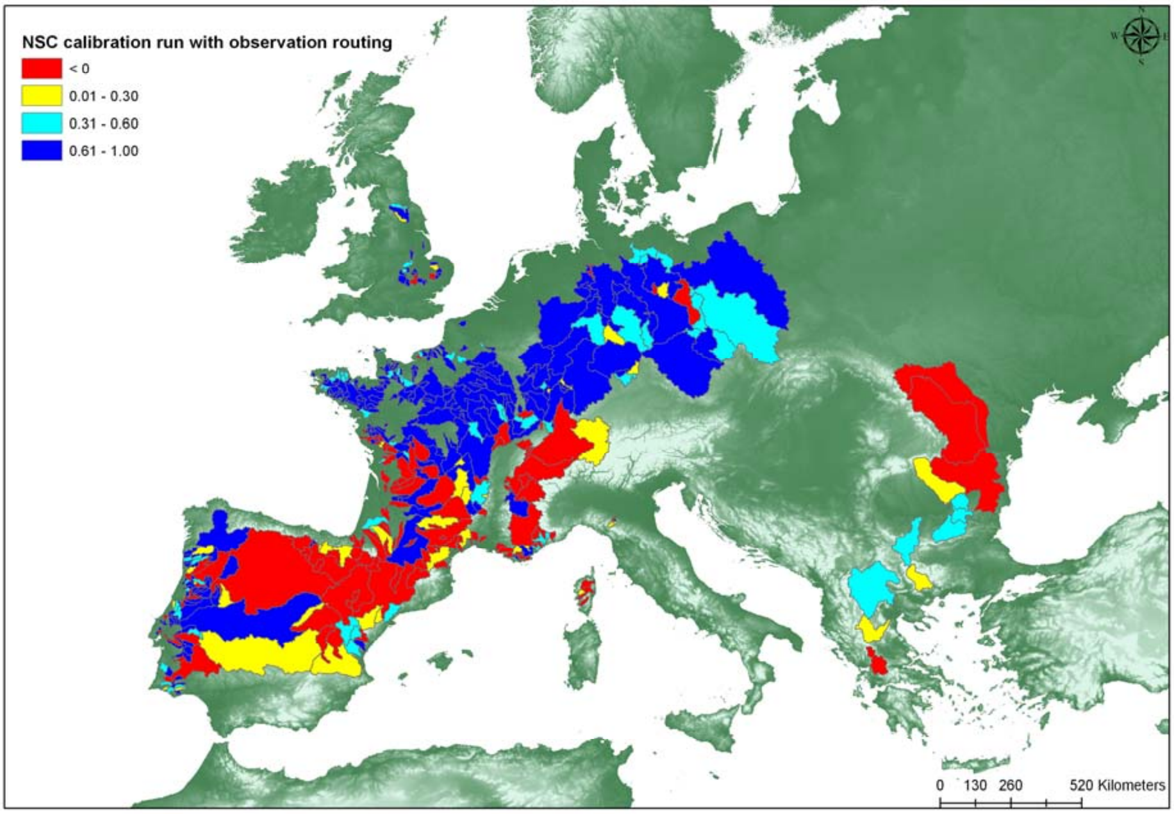

Figure 12: GR2M A - Nash Sutcliffe index obtained during calibration with observation routing

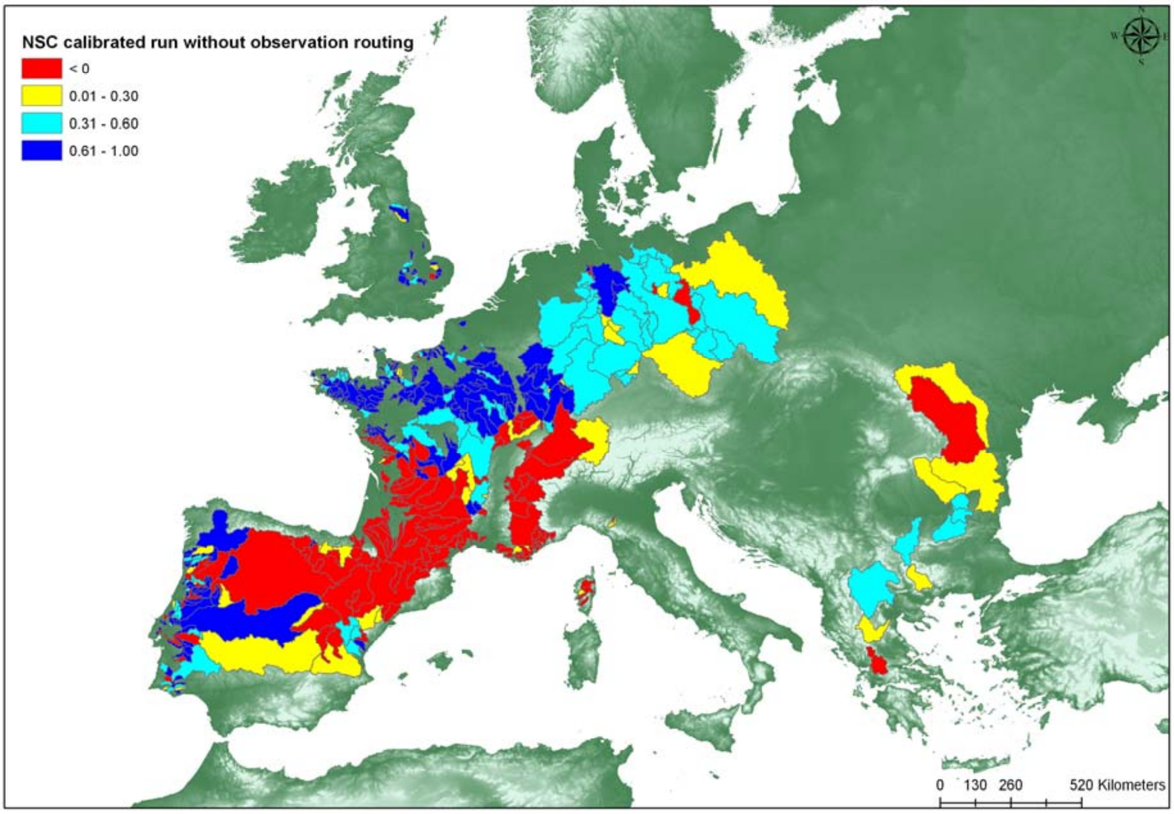

Figure 13: GR2M B - Nash Sutcliffe index after calibration without observation routing

A visual comparison of simulated and observed hydrographs and scatter plots of simulated and observed values gave further information on model performance that can not be derived from performance indices per se (Figure 14).

The basins were classified into six categories. Categories 1-3 refer to simulations with a good visual fit, a reasonable fit and a poor, but still visible relation between observed and simulated values. Category 4 contains basins that showed a reasonable fit for a part of the simulation period and a divergence of runoff behaviour during another part, indicating a possible change of runoff regime or inconsistent climatologic input. Category 5 contains basins showing much less variation of simulated runoff than observed runoff. This is considered a calibration problem, where the best fit is achieved when the simulated series approaches the mean value of the observed series. Another category 6 contains all simulations which showed no visible relation between observed and simulated values or where unusable for insufficient length of observation time series. Please note that also catchments with a high Nash-Sutcliffe index were eventually classified as 'unusable', while in some other cases also catchments with low Nash-Sutcliffe index behaved reasonable in the visual comparison. Problems related to the correct timing of runoff peaks were observed only for a few sub-basins.

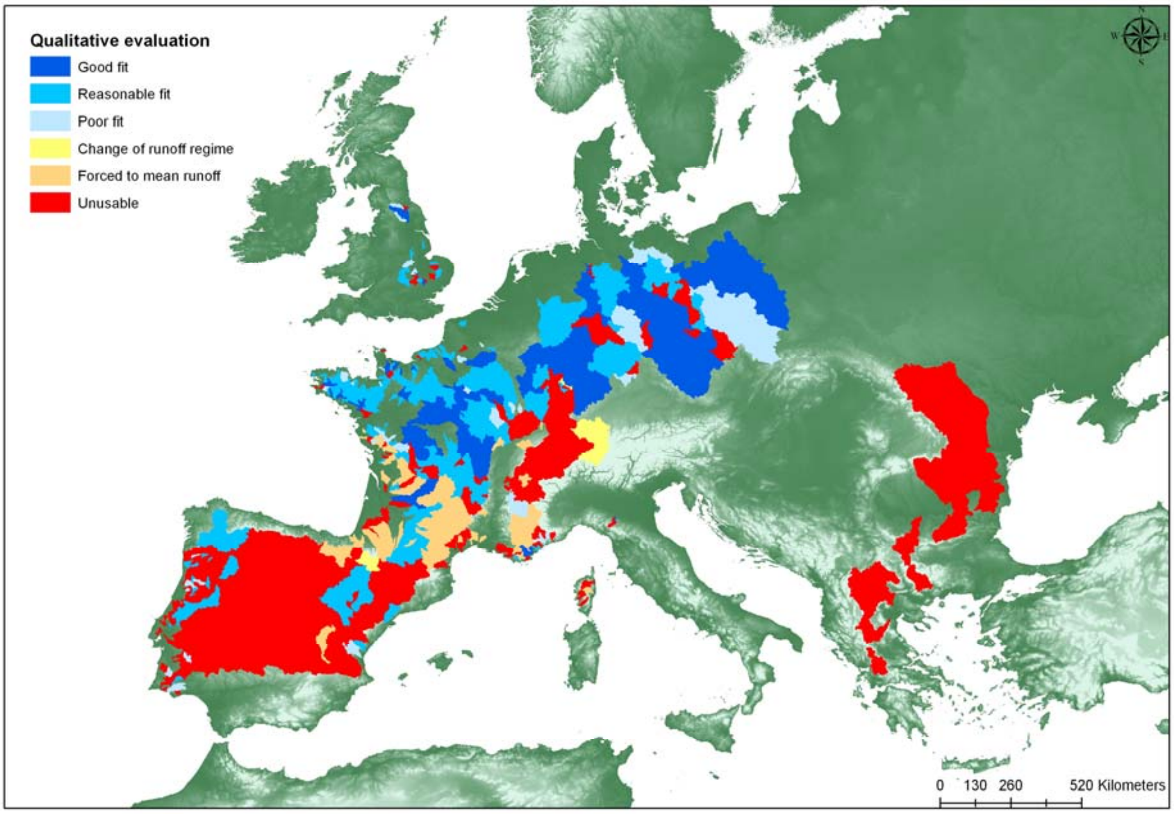

Figure 14: GR2M - Results of qualitative evaluation of model performance by visual comparison of simulated and observed hydrographs

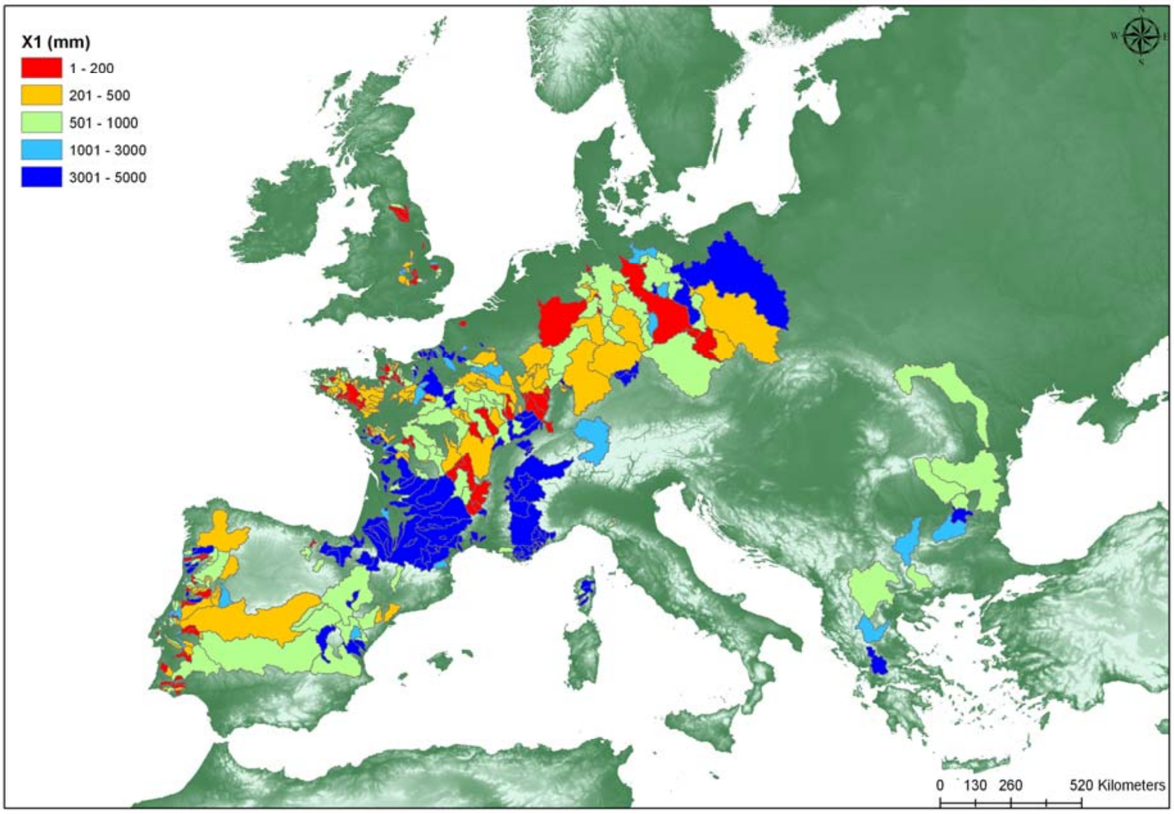

Figure 15: GR2M - Model parameter X1 (Productive storage) in [mm]

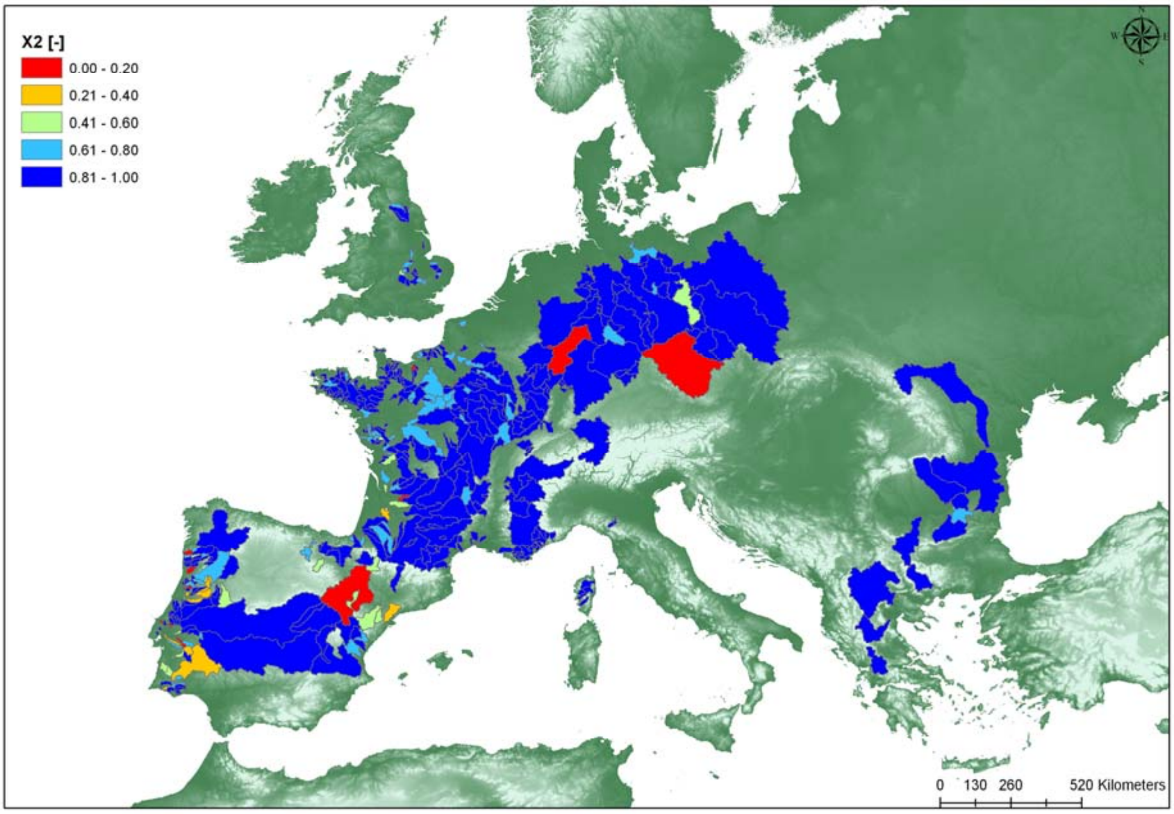

Figure 16: GR2M - Model parameter X2 (exchan ge coefficient) as fraction

Model parameters had no obvious relations to basin characteristics, such as size, number of internal catchments, drainage area, relief and anthropogenic water abstractions and losses. For a general water balance assessment the GR2M model can only be applied if model parameters can reasonably be transferred to basins without observation data.
The regionalisation approach was based on the characterisation of the monitored basins by principal components and investigating the relations of calibrated parameters with the extracted principal components.
We characterized the monitoring basins using a principal component analysis (PCA) using basin area (excluding connected upstream basins), number of catchments, total drainage area, available water capacity, heuristic baseflow index, groundwater travel time, mean slope, water abstractions and water losses as variables. As water capacity, baseflow index and slope were originally given on a per catchment basis, the data were aggregated to monitoring basin level calculating the area-weighted mean. The dataset was logarithmized (Figure 17) and standardized before the PCA.
The first three principal components explained 77% of total variance (Figure 18). The components can be associated with catchment size and water abstraction pressure, hydrological controls, and a mixed component reflecting size, abstractions and hydrological controls (Table 2).
The calibrated model parameters and the goodness of fit (expressed as Nash-Sutcliffe index) showed no obvious relation to the first three principal components (Figure 20). We therefore conclude that the given criteria can be used to characterize the catchments, but the principal components are not sufficiently reflecting the runoff generation process to support regionalisation of model parameters to ungauged catchments.

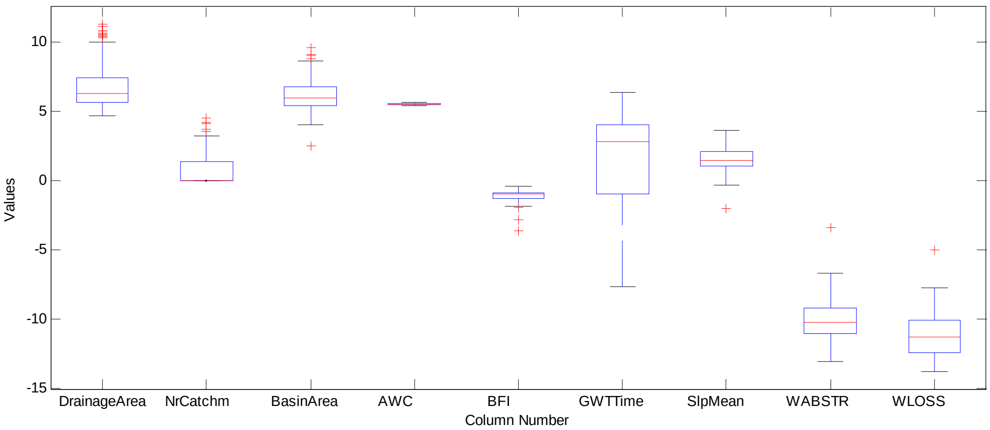

Figure 17: Boxplot of logaithmized variables

Principal component	Explained variance (%)
1	43.12
2	22.04
3	12.40
4	7.85
5	7.22
6	3.24
7	2.23
8	1.41
9	0.48

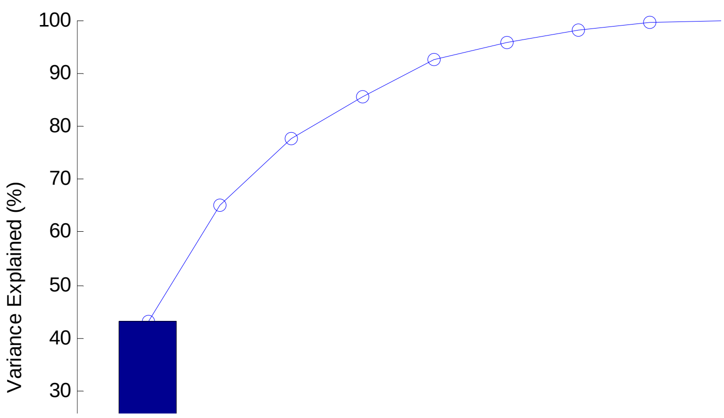

Figure 18: Variance explained by principal components

Table 2: Results of the principal component analysis - Loadings of the first three principal components

Variable	Loadings Component 1	Loadings Component 2	Loadings Component 3
Drainage Area	-0.3887	0.0254	-0.3488
Number Catchments	-0.4549	0.0342	-0.1778
Basin Area	-0.4734	0.0850	-0.0613
Soil water capacity	0.0639	0.3137	0.6443
Baseflow index	-0.0123	-0.6045	0.2835
Groundwater travel time	-0.2150	-0.3714	-0.1779
Mean slope	0.0083	0.6185	-0.1877
Water abstractions	0.4397	-0.0750	-0.3515
Water loss	0.4172	-0.0331	-0.3963

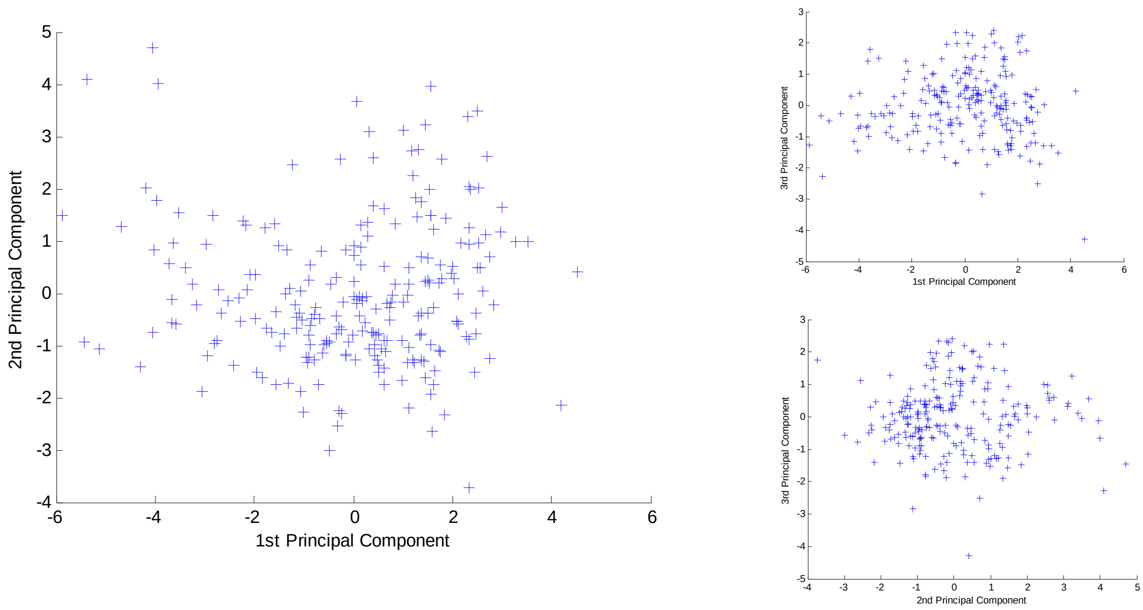

Figure 19: Representation of catchments in the PCA space

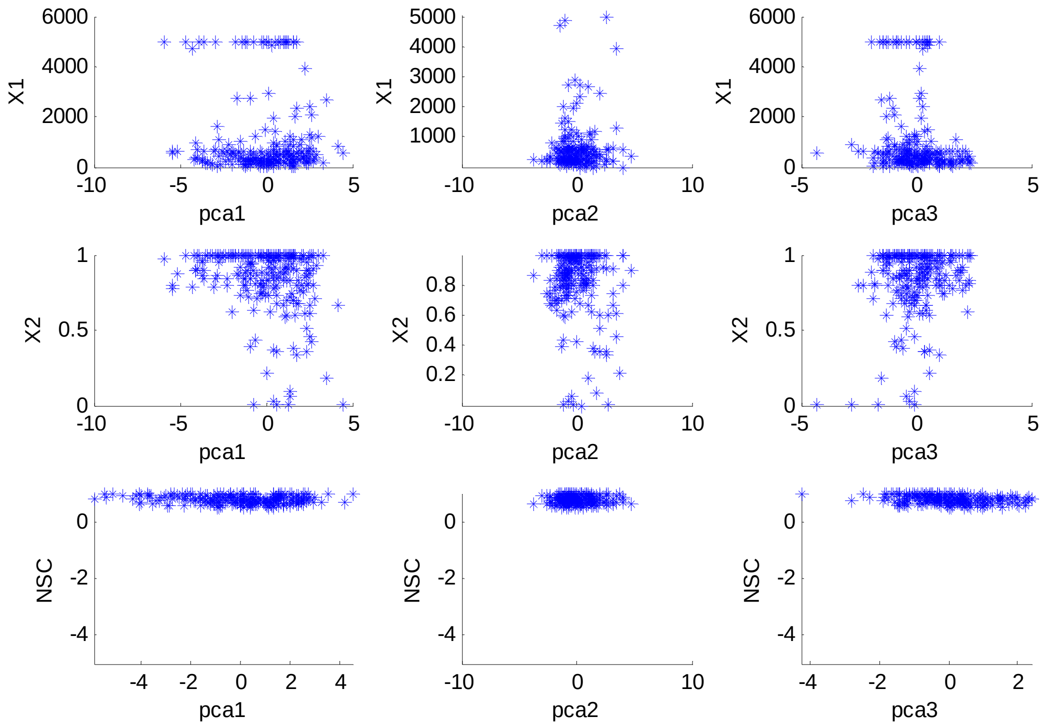

Figure 20: Relation of model parameters and model fit to the first three principal components

The direct comparison between GR2M results and the generic water balance model (as average annual runoff, Figure 21) again shows no clear relations, consistent with the relations between the water balance model and the observed data.

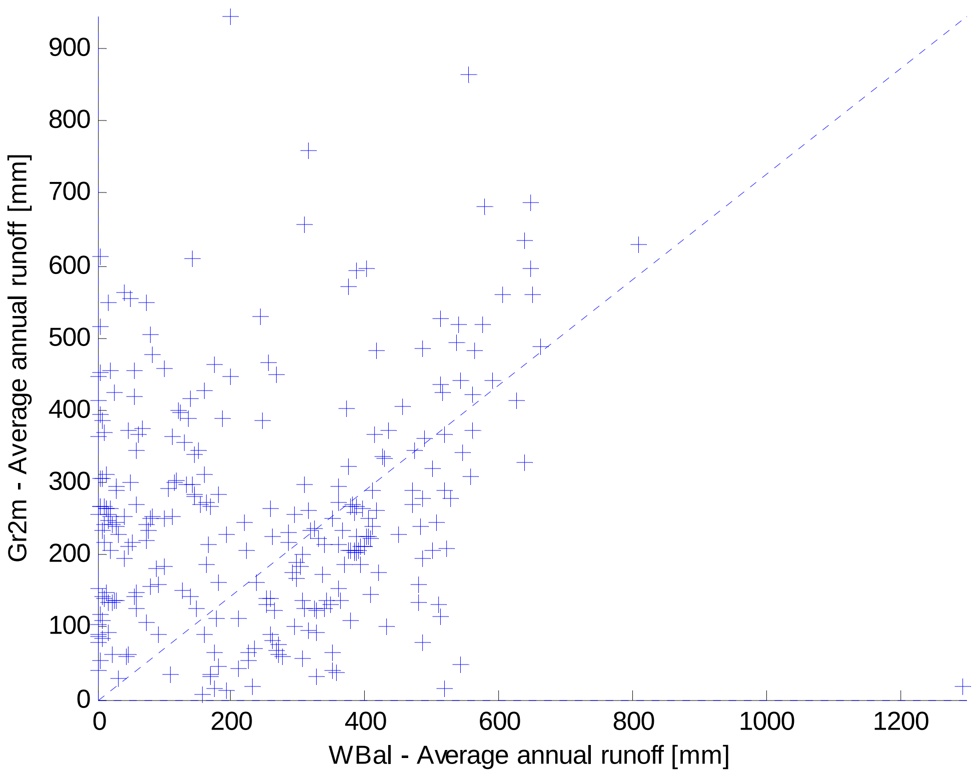

Figure 21: Comparison of average annual runoff calculated with the generic water balance model (WBal) and the Gr2m model. The dashed line indicates a 1:1 relation.

There are various reasons, why the GR2M model performs well in some basins and does not reasonably apply to other basins.

- Artificial structures are not considered; especially reservoirs can change runoff regimes within a river basin, making it impossible to obtain a reasonable fit of observed and simulated runoff.
- Water losses as a result of human abstractions can make up a significant amount of or even exceed natural water supply. Especially in Mediterranean basins, abstractions and losses alter runoff quantity and runoff regime. The model parameter X2 implicitly allows accounting for water losses as long as they do not show a considerable seasonal variation (which is the case in regions with irrigated agriculture).

It can be concluded that the GR2M model can be applied reasonably well after calibration, however there are limits regarding the regionalisation of model parameters and the capability of the model to adapt to anthropogenic impacts on the runoff regime. In the end, each basin has to be analysed individually and a automated procedure as followed in this context is not a promising approach as long as the anthropogenic impacts can not be better represented in the model. Routing runoff from upstream basins to downstream basins causes propagation of uncertainties; calibration should therefore include routing of observed runoff as it was made in this study.

2.5 Assessment of water abstractions, losses and returns

In a related report we presented a pan-European analysis of water abstractions and water losses. For a detailed description of the methodology and data sets we refer to Wriedt et al. (2009, in preparation). Main outcomes were datasets on total water abstractions (excluding cooling water abstractions), water losses (consumptive losses, evapotranspiration losses) and water returns. Water abstraction data for public water supply, from households and for industrial uses were disaggregated in space to a

10x10km grid covering EU27 and Switzerland. The disaggregation procedure associates water abstractions with the locations of water consumption based on proxy indicators such as population distribution, economic indicators and land use. Agricultural abstractions were based on the irrigation assessment published by Wriedt et al. (2008, 2009a, 2009b). Total water abstractions and losses were transferred to catchments and river basins as sum over all grid cells overlapping a catchment weighted by overlapping area.

Yet unsolved were the problems of allocating water abstractions in space to the possible locations of abstraction and to the abstraction source (ground water and surface water). The allocation of water abstractions to water sources plays a key role in determining pressures specifically for ground- and surface waters. However, comprehensive and consistent data are hardly available and the use of ground- and surface water resources may be determined by other factors than large scale and local scale physio-geographic features. In addition, existing infrastructure, technological trends, socioeconomic factors, legal and institutional framework come into play. Water demands may be satisfied using water from remote catchments, aquifers or reservoirs and groundwater may substitute surface water and vice versa. An overview on total water abstractions and the relative role of consumptive losses is given in Figure 22 and Figure 23. Please note that water abstractions were calculated excluding abstractions for cooling purposes. This was not only done for methodological reasons but also because cooling water abstractions have typically little consumptive losses and are generally directly returned to the surface waters they were taken from (see Wriedt et al., 2009d).

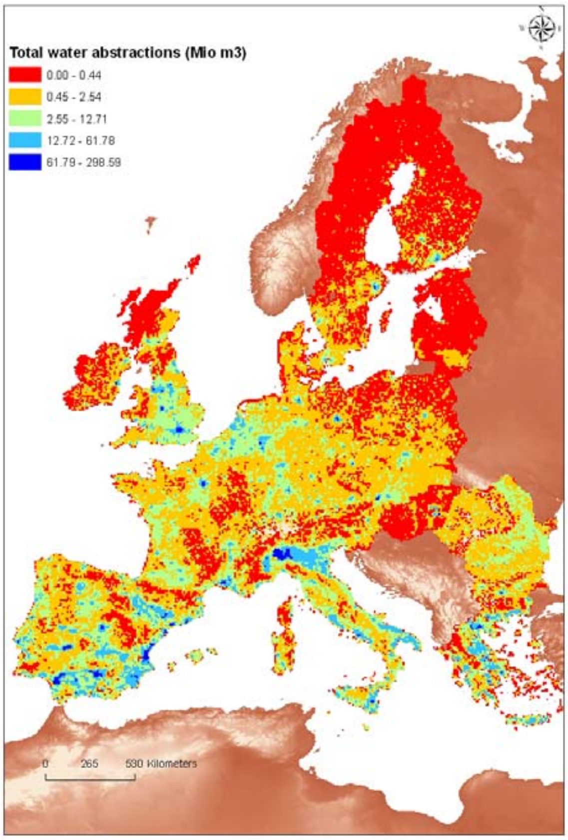

Figure 22: Water abstractions (excluding abstractions for cooling water in energy generation)

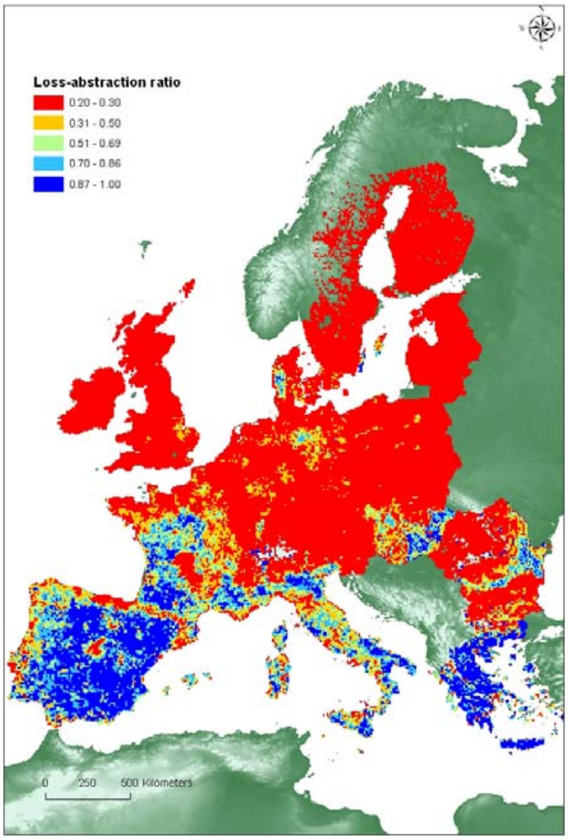

Figure 23: Ratio of consumptive losses to water abstractions

2.6 Assessing pressures on water quantity

A direct comparison of water availability (water surplus) and water use (as water abstractions or losses) gives a first indication of quantitative pressures on water resources.
Two indicators were applied to evaluate quantitative pressure on water resources at catchment and river basin level. The water exploitation index (WEI) is the ratio of abstractions and excess water available for runoff and groundwater recharge (HXS):

$$WEI = \frac{Abstraction}{HXS} \qquad [22]$$

The water loss index WLI is defined as the ratio of losses and excess water (HXS). In contrast to the WEI, the WLI accounts for the losses of water, taking into account potential re-use of abstracted water after return.

$$WLI = \frac{Loss}{HXS} \qquad [23]$$

Interpreting these indices requires considering that for sustainable water management the hydrological excess water available for groundwater recharge and runoff can not fully be exploited. A certain amount of water must be left to guarantee ecological functioning of ecosystems, and to support other human related services of water bodies such as transport, power generation (Minimum flow, *MinFlow*). In addition, part of the water may not be available for human consumption, as topographical and geological conditions limit accessibility for exploitation (*non exploitable water*). This would require rewriting the above equations into:

$$WEI = \frac{Abstraction}{HXS - MinFlow - NonExploitableWater}, \qquad [24]$$

and *WLI* re-defined analogously

The minimum flow and non exploitable water are difficult to determine in a straightforward way and local conditions must be considered. Therefore this extended concept was not applied in this assessment. This consideration implies, however, that the threshold for both indices indicating where water resources must be considered under pressure is not at one, but potentially **far below** one.
The water exploitation index and the water loss index were calculated at catchment level (Figure 24 and Figure 25) and at river basin level (Figure 26 and Figure 27). While the catchment based indices give insight into the local imbalance of water supply and demand, they do not account for satisfying water demands using water from other catchments or even other basins. As indicated in Wriedt et al. 2009d, the water abstraction data were allocated to the location of use and not the location of water abstraction. However, in water rich countries and in regions with dominant groundwater use, it can be assumed that water is abstracted and used at the local scale and long distance transport is of minor importance (unless to supply water to large cities and agglomerations).
The river basin based index partly takes this problem into account, as the aggregation of abstractions and available water to river basin level can smooth out local imbalances. However, as a level for balancing of supply and demand the river basin level may not be appropriate, as water supply and transport infrastructure are not necessarily organised by natural river basins, but may relate to administrative and organisational issues.
In terms of quantitative pressure on water resources, the water exploitation index WEI does not account for water returns and potential re-use of water. The water loss index WLI takes returns and re-

use into account, considering only physical loss of water. However, each unit of water abstracted can result in quantitative pressure to the specific water body, as the water is likely to be returned into another water body (for example pumping from groundwater, drainage to surface water). Here, problems may occur as the returned water can be polluted or unusable for further use. While the water loss index gives information on the mere water quantity effect at the level of spatial unit, the water exploitation index is a more general tool assessing the quantitative and ecological consequences of water abstractions.

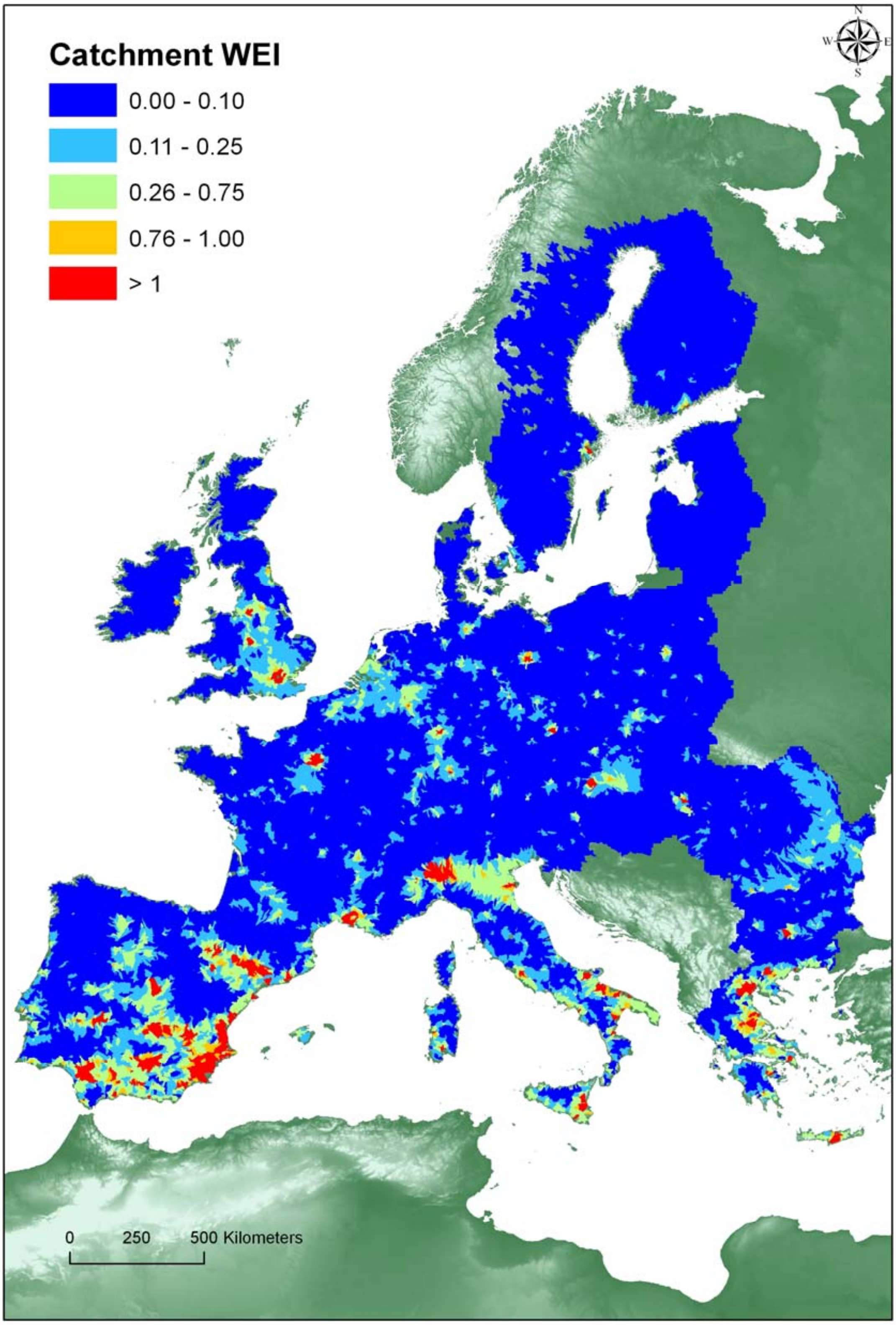

Figure 24: Water exploitation index at catchment level, excluding abstractions for electricity generation cooling.

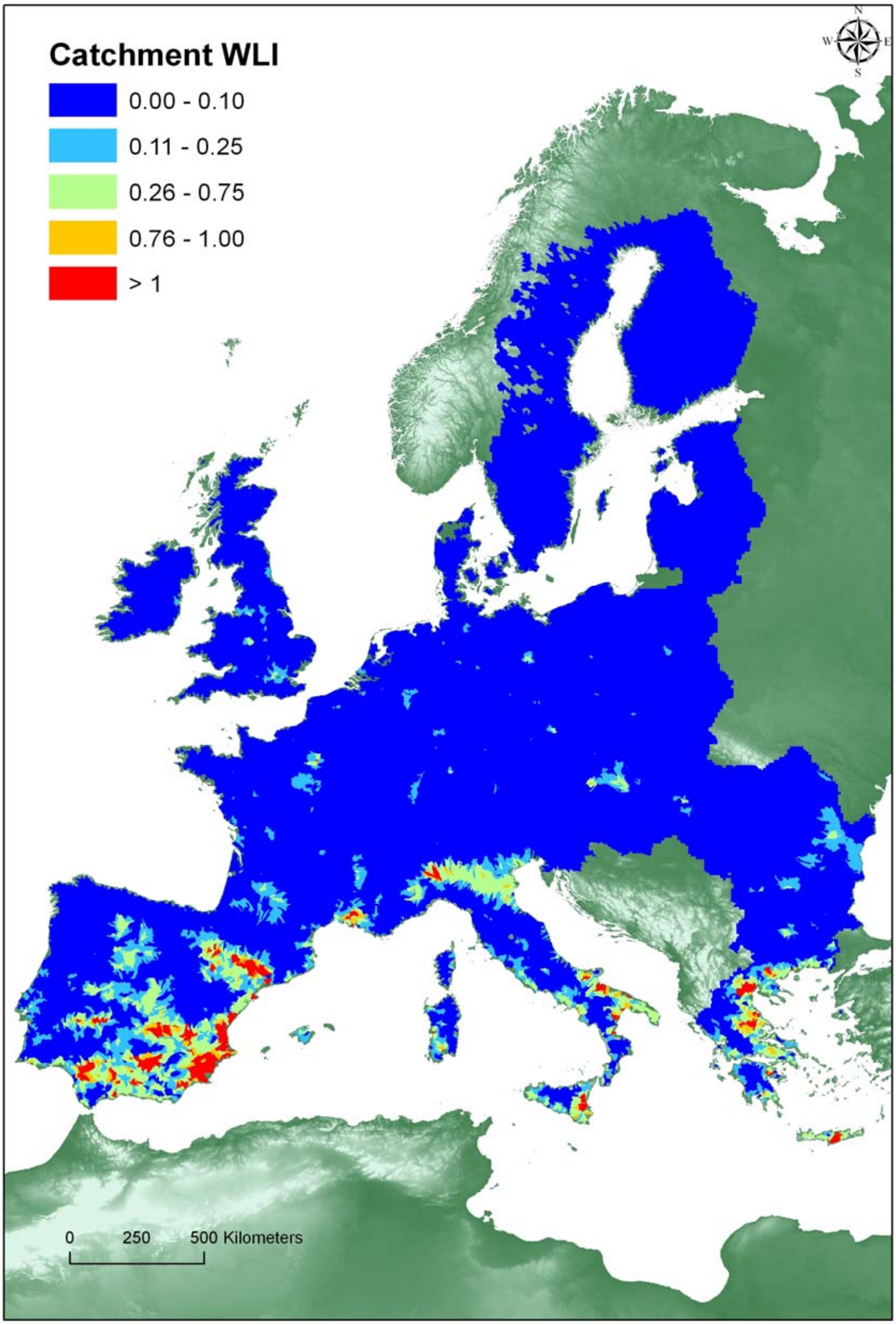

Figure 25: Water loss index at catchment level, excluding abstractions for electricity generation cooling.

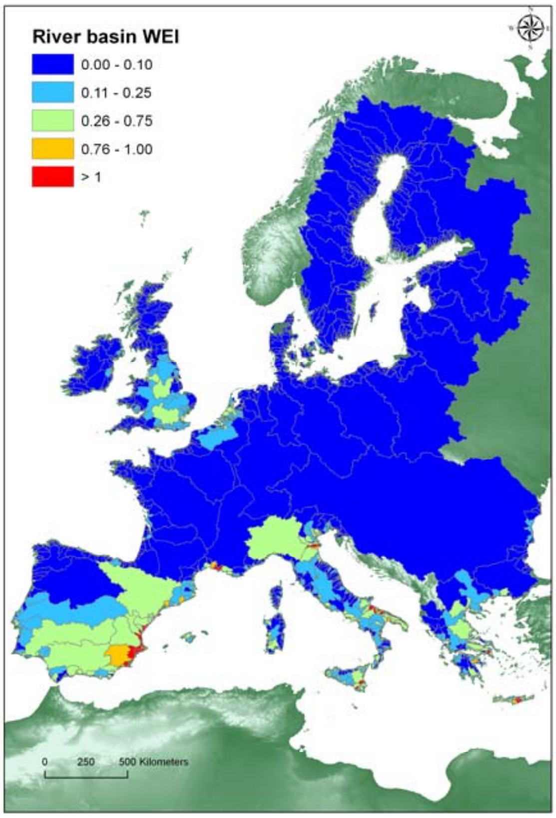

Figure 26: Water exploitation index at river basin level.

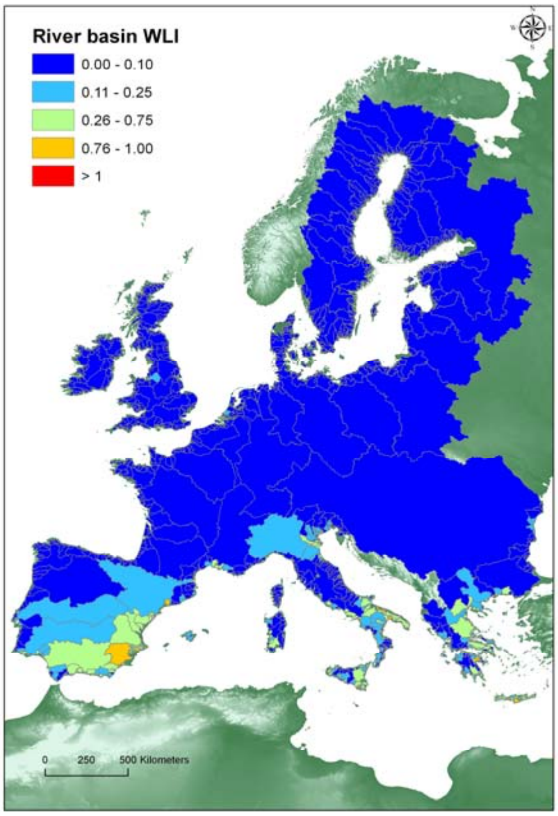

Figure 27: Water loss index at river basin level.

2.7 Heuristic baseflow index estimation

The baseflow index approach assumes that baseflow can be expressed as a constant fraction of total runoff. Thus subsurface and surface flow can be separated according to the equations:

$$Q_{SSF} = bfi \cdot Q_{Total} \quad [25]$$

$$Q_{SF} = (1 - bfi) \cdot Q_{Total} \quad [26]$$

where *bfi* = Baseflow index, Q_{Total} = total runoff, Q_{SF} = surface runoff, Q_{SSF} = Subsurface runoff (baseflow).

We calculated an a priori baseflow index using a heuristic approach based on available catchment properties. Large scale assessments of groundwater runoff using the baseflow index approach have been presented by Bogena et al. (2005), Döll et al. (2002), de Witt (1999) and Meinardi et al. (1994).

The baseflow index represents the combined effect of all factors affecting the separation of surface and subsurface runoff. Following the before mentioned authors, the most relevant factors are slope, soil permeability, rock permeability, groundwater depth and contributing area or land use. The main factors considered in the generic approach are slope, soil, hydrogeology and contributing area (unsealed area). In contrast to Bodena et al. (2005) and de Witt (1999) and Meinardi (1994) we do not consider groundwater depth as a factor. Neglecting surface-near groundwater tables is owed to the fact that continental scale data do not allow to identify areas with shallow groundwater tables with sufficient accuracy.

The slope factor f_{sl} was calculated following de Witt (1999), Wendland (1992) and Dorhoefer and Josopait (1980) using the equation:

$$f_{sl} = 1 + \left(3 \cdot (10 \cdot slope)^{0.6}\right) \quad [27]$$

The slope factor implements a negative relation between slope and baseflow: the steeper the slope, the smaller is the fraction of baseflow and the higher is the fraction surface runoff.

The original correction for gleysols and histosols given in the literature was omitted, as locations with high groundwater tables could not be identified in this analysis and were treated differently in this analysis.

The soil factor f_{soil} (Meinardi et al. 1994) ranges from 0.05 on clay soils to 0.95 on sandy soils, reflecting the qualitative ranking of the soils with respect to infiltration capacity.

Table 3: Soil factors (Meinardi et al., 1994)

Soil texture	Soil factor f_{soil}
Sand	0.95
Loam/Sand	0.75
Loam	0.50
Loam/Clay	0.25
Clay	0.05
Groundwater near surface level	0.10

The permeability of the rock material determines the amount of water that can be transported to the streams by groundwater flow. When recharge exceeds the aquifer transport capacity groundwater levels rise. This results either in higher gradients increasing groundwater flow or in high groundwater levels with reduced recharge and increased surface runoff.

The hydrogeology factor f_{hdg} (Meinardi et al. 1994) takes into account rock type and hydraulic permeability, ranging from 0.1 (Igneous and metamorpheous rocks with poor permeability) to 0.95 (unconsolidated sedimentary aquifers with good permeability).

Table 4: Hydrogeology factors (Meinardi et al. ,1994)

Rock type	Permeability	Hydrogeology factor f_{hdg}
Unconsolidated sedimentary aquifer	Good	0.95
Unconsolidated sedimentary aquifer	Poor	0.80
Consolidated sedimentary aquifer	Good	0.70
Consolidated sedimentary aquifer	Poor	0.50
Igneous and metamorphic rocks	Modest	0.40
Igneous and metamorphic rocks	Poor	0.10

Not all land uses within a certain region contribute equally to groundwater recharge. On bare rock, ice-covered areas and sealed areas infiltration into the soil can be neglected. A contributing area factor f_{ca} was calculated as:

$$f_{ca} = 1 - \left[\frac{1}{A_{Total}} \cdot \left(\omega_{Settlement} \cdot A_{Settlement} + \omega_{bare\ rock} \cdot A_{Bare\ rock} + \omega_{Ice} \cdot A_{Ice} \right) \right] \qquad [28]$$

where A_{Total} = Total area of spatial unit, A_i = Area of land use class i, ω_i = sealed or non-contributing fraction of land use class i. The non-contribution fractions for land use classes were set to 0.5xxcheck! for settlements and to 1.0 for bare rock and ice.

Different approaches can be chosen to combine the various partial factors. Döll et al. (2002) multiplied the individual factors to derive a general baseflow index, whereas Bogena et al. (2005) selected only one decisive factor following a decision tree accounting for the specific factor combinations within each catchment. In our approach, we used a mixture of the two concepts, determining a decisive factor as the minimum value of the slope, hydrogeology and soil factor and correcting for the contributing area within each spatial unit. The equation to calculate the baseflow index can thus be written as:

$$bfi = f_{ca} \cdot Min\left(f_{sl}, f_{soil}, f_{hdg} \right) \qquad [29]$$

Figure 28 shows the resulting baseflow indices calculated for the catchments defined in the HydroEurope database. At the pan-European scale, slope is clearly a dominating factor, while substrate properties have a secondary role. The baseflow index clearly distinguishes the alpine areas with low baseflow index (steep slopes, fractured rock with local groundwater in valley deposits), the mountainous areas with moderate baseflow index (moderate slope, fractured rock, consolidated and unconsolidated sediments, larger valleys) and the plains and basins with high baseflow index (low slopes, unconsolidated sediments). The general landscape features are well reflected at European scale and consistent with heuristic expectations. This gives some confidence that the BFI estimates are reasonable in a qualitative sense, although the specific values require further verification.

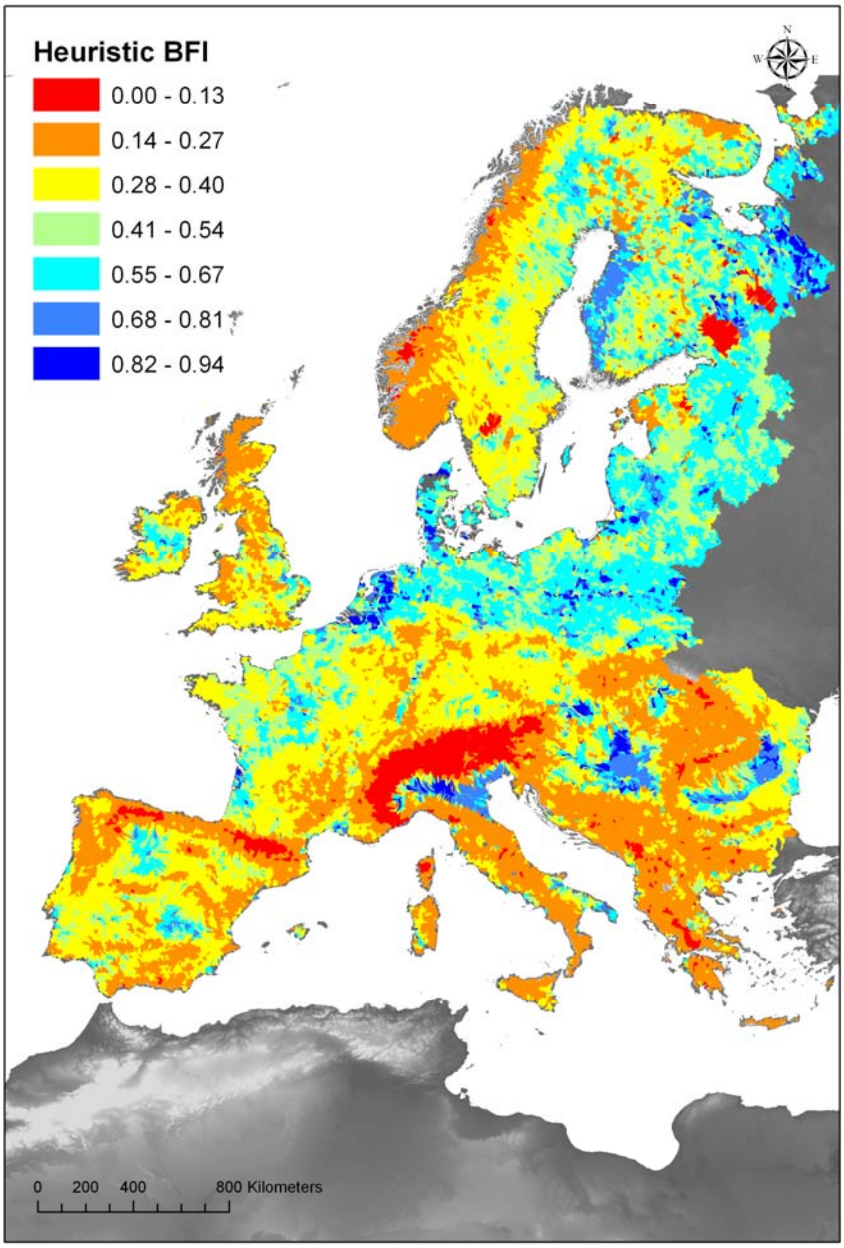

Figure 28: Baseflow index (% of total discharge)

2.8 Hydrograph based approaches to estimate baseflow and recharge estimation

2.8.1 Baseflow separation based on hydrograph filtering

2.8.1.1 The USGS-PART algorithm analysing daily runoff time series

The PART software developed by the United States Geological Survey (USGS) was applied to estimate baseflow from daily runoff time series. The methodology is described in detail by Ruthledge (1998). The program identifies days where stream low equals groundwater discharge based on periods of antecedent recession. The antecedent recession period is defined by:

$$N = A^{0.2} \qquad [30]$$

where A = Catchment area [mi^2], N = antecedent recession period [d].
During the recession period, groundwater discharge is calculated from the recession curve. Groundwater discharge during periods of surface runoff is calculated by linear interpolation.
The PART method is distributed by USGS as ready-to-run software package. A model interface was implemented as a VB.NET software package (USGSTools). The interface generates the required model input files, controls model execution and supports post-processing of model results for multiple monitoring stations. The USGS PART software was designed for American units (cube feet, square miles) and not SI-Units. This required multiple conversion of input data and model results.
The PART software requires the user to visually analyze the time series and enter the appropriate time period for each analysis. Automating the program runs time periods where defined a priori, therefore some catchments were not considered for incompatibility with the predefined time periods. The pre-defined time period was initially set to 1990-2005. Missing values at the end or the beginning of the time series are automatically omitted by the PART software. For many stations there was a data gap in the year 2001. Therefore additional time periods from 1990-2000 and 2002-2005 were analysed and added to the result set where appropriate. Stations where gaps occurred at different time intervals where not considered.

2.8.1.2 A monthly hydrograph filter (MoHyFi) analysing monthly runoff time series

The MoMLR-Approach and the PART software require availability of daily data or information on monthly variability of runoff to define mean low runoff values. Smakhtin (2001) proposed a methodology to derive monthly baseflow estimates from filtering a monthly runoff time series.
The filter equations are given by

$$q_m = \alpha \cdot q_{m-1} + 0.5 \cdot (1+\alpha) \cdot (Q_m - Q_{m-1}) \qquad [31]$$

and

$$QB_m = Q_m - q_m \qquad [32]$$

where q_m = direct runoff (m^3/month), α = coefficient, Q_m = monthly total runoff (m^3/month), QB_m = monthly baseflow (m^3/month).
In the following, this approach is referred to as 'Monthly Hydrograph Filter' (MoHyFi). The main problem of the MoHyFi approach is the parameterization of the filter parameter α, as it can not be derived from heuristic considerations or from the time series data themselves. Smakthin (2001) calibrated the filter parameter α to monthly baseflow time series generated from running a baseflow separation technique on daily flow records. Doing so he could establish a default value for Australian catchments, but also points out that regionalisation of the filter parameter α may be required.
The basic algorithm can be formulated as follows:

1. Run a baseflow separation technique on a daily streamflow record and calculate monthly baseflow accumulating the daily baseflow values.
2. Run the filter technique on the monthly streamflow record and compare the resulting baseflow record with the record obtained from daily data. Evaluate goodness of fit and adjust filter parameter α.
3. Repeat step 2 until both baseflow estimates correspond.

In our case the daily baseflow estimates of the USGS-PART method were aggregated to monthly baseflow values and the MoHyFi parameter α was optimized to reproduce the USGS-PART results. The resulting α values were further analyzed to establish (regionalized) default values that can be applied to basins where daily data were not available. The MoHyFi approach and the optimization algorithm were implemented using a series of MATLAB scripts, optimizing α by a linear search algorithm.

2.8.2 Baseflow index based on the MoMLR-Methodology

Monthly low-water runoff values have been used to estimate groundwater recharge and baseflow in unconsolidated rock areas (Wundt et al. 1958). In consolidated rock areas, where interflow plays a significant role in runoff generation, a modified approach known as the MoMLR-method (Kille, 1970, Demuth, 1993) provides more accurate results of groundwater recharge and baseflow. For this study the methodology described by Bogena et al. (2005) was applied. The monthly low runoff values are sorted in ascending order creating a distribution curve. A linear trend is fitted to the linear zone of the distribution curve. Values diverging from the linear trend are affected by interflow. The linear zone is identified iteratively, rejecting first the upper and then the lower values of the dataset, until the highest correlation between data and the linear trend is achieved. Recharge values are then interpolated into the diverging zone, separating the interflow component.
The linear trend equation is given by:

$$MoMLR = a \cdot x + b \qquad [33]$$

where MoMLR = Monthly Mean Low Runoff, a = regression coefficient, b = regression coefficient, x = dependent variable (index of sorted value). The monthly low runoff is then determined from the linear regression equation by

$$MoMLR = a \cdot \frac{nc}{2} + b \qquad [34]$$

where nc = number of values.
In contrast to Bogena et al. (2005), the European discharge data contain not only perennial but also intermittent streams. We therefore modified the original approach calculating the MoMLR value as the positive integral of the linear trend divided by the number of values.

$$MoMLR = \frac{1}{nc} \cdot \int_{nc-x0}^{nc} (ax + b)\,dx \quad where\ x_0 > 0 \qquad [35]$$

For perennial streams (x0 < 0) this equation is equivalent to the previous equation.
The baseflow index can be determined as ratio of MoMLR and mean monthly runoff:

$$BFI = \frac{MoMLR}{\frac{1}{nc} \cdot \sum_{nc} MR} \qquad [36]$$

Where BFI = baseflow index, MR = mean monthly runoff.
For the analysis we excluded all incomplete months and all years with less than 10 months of data. In total 228 stations located in the UK, Spain, Portugal, Italy, Benelux and Germany were included. The resulting baseflow index and groundwater recharge are displayed in Figure 29 and Figure 30.

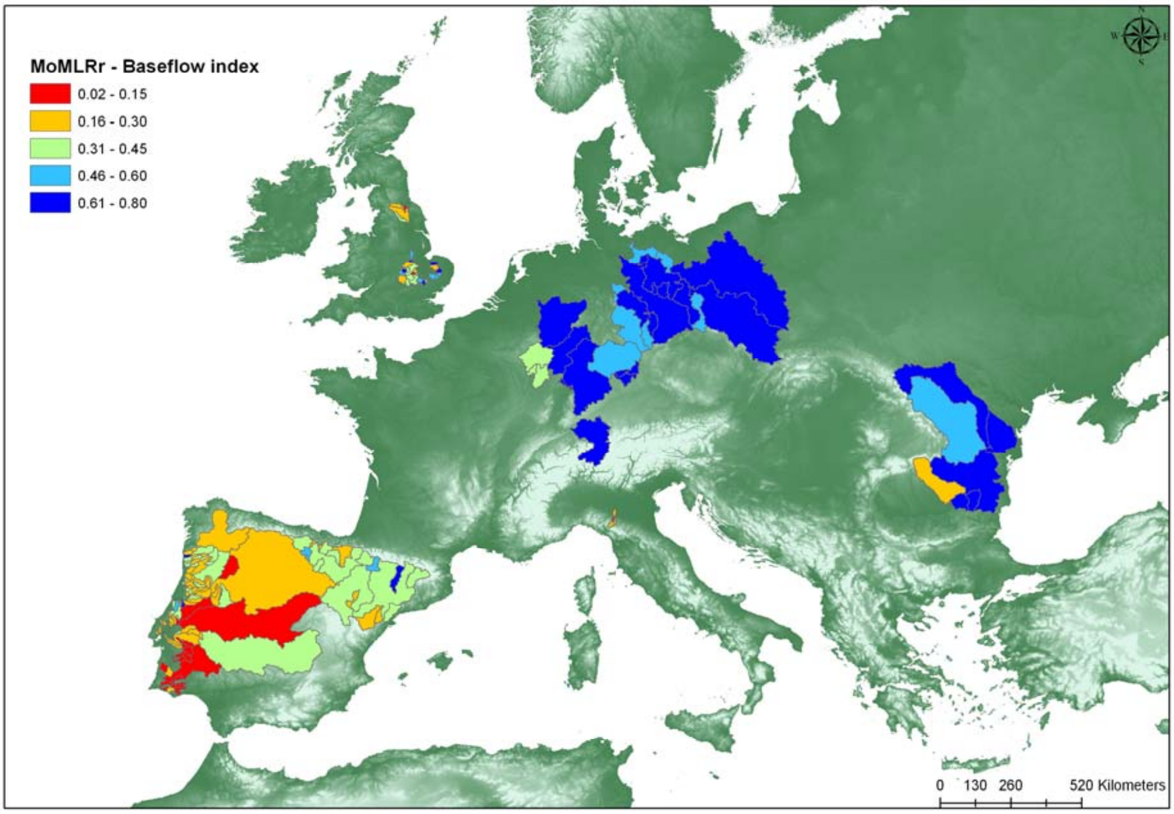

Figure 29: MoMLR-Analysis - Baseflow index

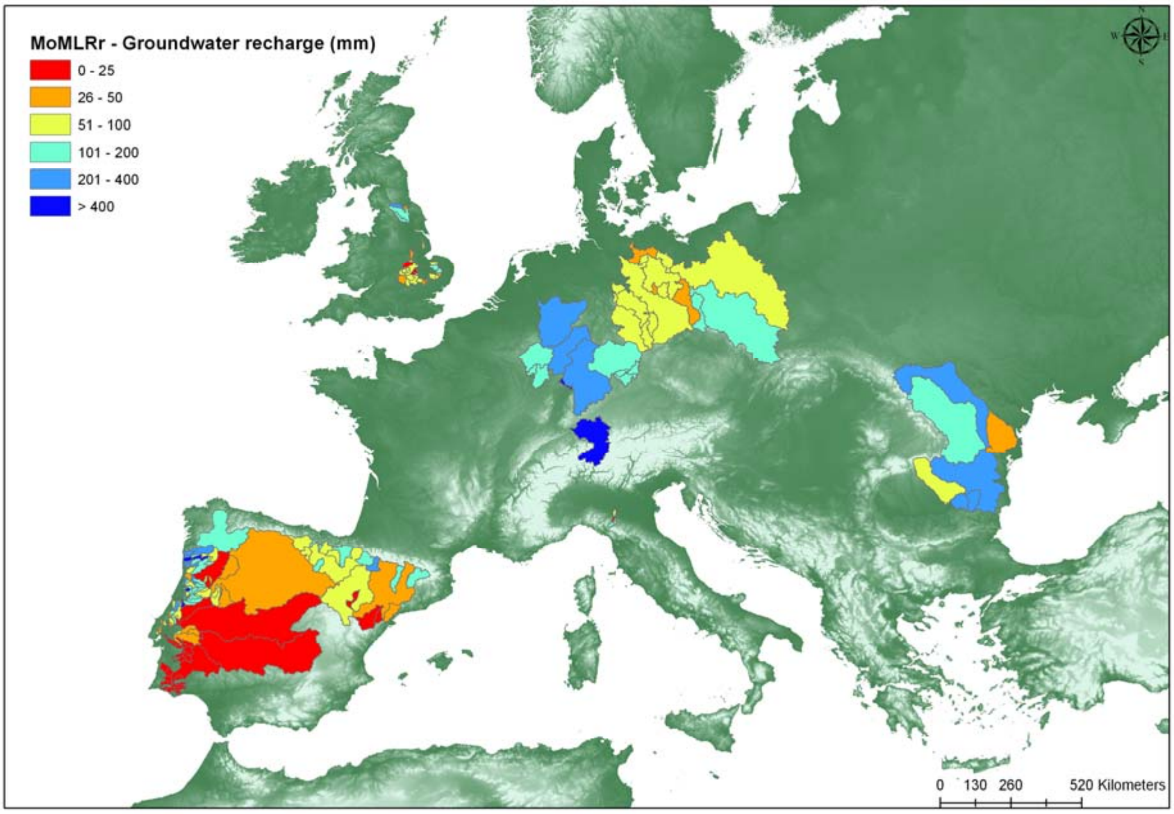

Figure 30: MoMLR-Analysis - Groundwater recharge (mm)

2.9 Comparison of baseflow indices obtained by different hydrograph separation methods

For the USGS-PART/MoHyFi approach discharge time series of 122 monitoring stations were evaluated. This lower number resulted from the fact that the PART software requires the user to visually analyze the time series and enter the appropriate time period for each analysis. Automizing the program runs time periods where defined a priori, therefore some catchments were not considered for incompatibility with the predefined time periods.
The MoHyFi approach can reproduce the results obtained from the USGS-PART method with reasonable accuracy (Figure 31). Annual baseflow values correspond well to each other. A similar relation was obtained for the monthly baseflow values with slightly higher variations.

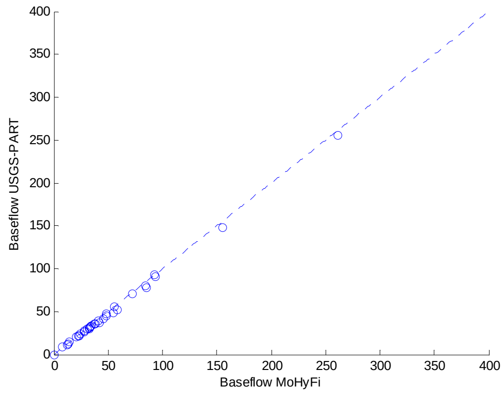

Figure 31: Comparison of annual baseflow calculated using the MoHyFi and the USGS-PART approach (dashed line indicates 1:1 relation)

Comparing the MoMLR-baseflow estimates with the MoHyFi/PART results, the scatterplot suggests two different linear relations, one where MoMLR overestimates MoHyFi/PART results and another where MoMLR underestimates MoHyFi/PART results (Figure 32). Differences between various baseflow separation techniques can be considerable (Eckhardt, 2008), it is therefore not possible to judge which technique provides more accurate results.
The values obtained for the filter parameter α cover the entire possible range (0...1) (Figure 33). It is therefore not possible to assign a default parameter providing reasonable results throughout Europe. Further relations with basin characteristics could not yet be established. There is also no obvious relation between the filter parameter alpha and the MoMLR baseflow index (Figure 33), that could have added some confidence in the consistency of both methods.
Baseflow separation techniques are typically applied in small watersheds, where the runoff dynamics are dominantly controlled by the runoff generation processes. In meso- and macro-scale river basins (to which most monitoring stations refer), hydrograph dynamics are in addition modified by the interaction of multiple watersheds (with different climate, hydrological response, and position within the network) and the translation and dispersion of the runoff peaks in the river system. This causes an 'apparent' behaviour no longer associated with runoff processes. Given the considerable size of most basins included in the study, it can therefore be questioned, if hydrograph separation techniques can provide reasonable results at all. This poses severe limitations to any validation of baseflow assessments.
We must therefore conclude that generic approaches to calculate total runoff, baseflow and groundwater recharge are currently the only feasible solutions to address groundwater issues. Such

approaches have a cost in terms of local accuracy, but can provide reasonable estimates for a large scale assessment.
The heuristic baseflow index has no obvious relation to the baseflow index calculated from the MoMLR approach. This is not surprising considering the size of basins that were used for runoff separation. All available hydrograph data refer to medium size and large river basins (or subbasins). The average catchment size is already in the order of 100 km^2. Hydrograph based approaches work relatively well, when applied to small catchments. In large basins, however, the hydrograph is influenced not by the reaction of a single catchment, but aggregates the reactions of a multitude of catchments, each reacting differently and responding to different climatic conditions and with translation and dispersion of the hydrograph, depending on distance to the monitoring station. The impact of artificial structures and especially reservoirs can additionally distort runoff regime with respect to runoff processes. Consequently, the hydrograph of larger basins is no longer appropriate to derive baseflow indices.

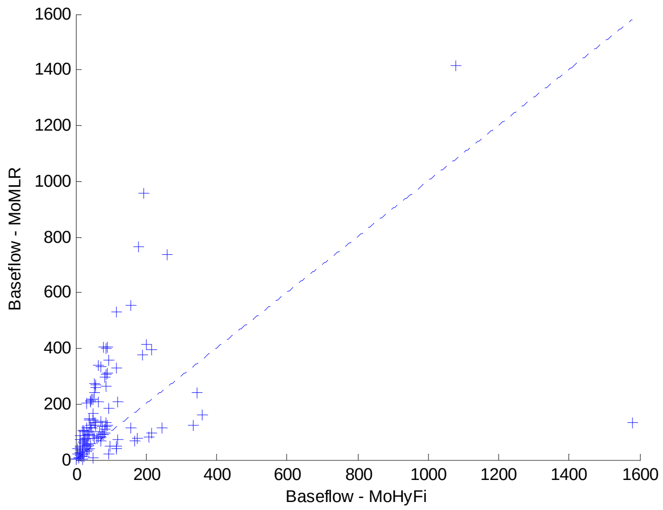

Figure 32: Comparison of baseflow calculated using the MoHyFi and MoMLR approach (dashed line indicates 1:1 relation)

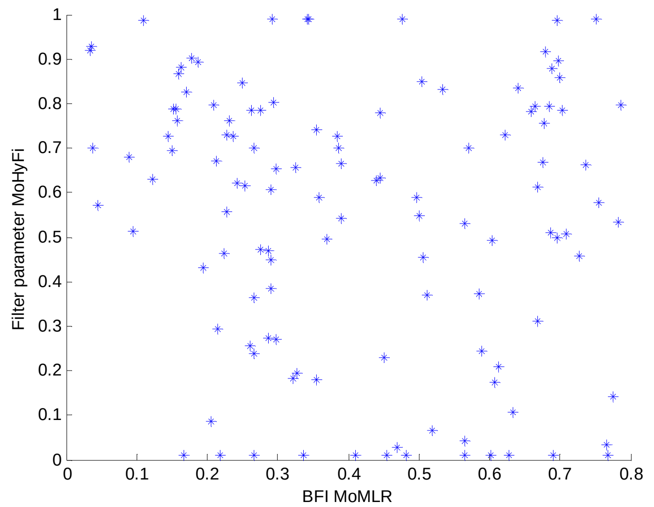

Figure 33: Relation of filter parameter and MoMLR baseflow index

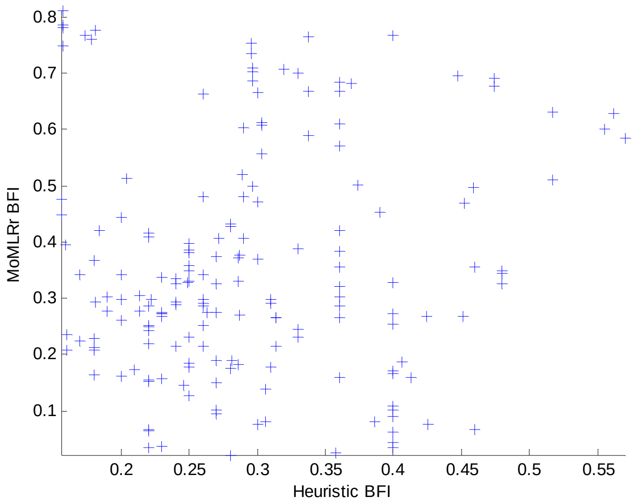

Figure 34: Comparison of heuristic and MoMLR-baseflow indices

2.10 Travel distance and characteristic travel time

The travel time characterizes the time scale required for water (and solutes) to pass an aquifer and being discharged into surface waters. Strictly speaking, groundwater travel times are characterised by a travel time distribution, resulting from long and short flow distances and heterogeneity in hydraulic conductivity and gradients as well.
An indicative assessment of travel distance and characteristic travel time was based on a simplified geometrical model of each spatial unit and its river network. The river density γ [1/m] is given by the ratio of river length L [m] to the catchment area A [m^2]:

$$\gamma = \frac{L}{A} \qquad [37]$$

The average flow distance ς [m] can be calculated from the river density γ assuming a simple geometrical model of the catchment based on a quadrangular shape of equal area and equidistant parallel streams with a total length equal to the river length:

$$\varsigma = 0.25 \cdot \frac{1}{\gamma} \qquad [38]$$

Groundwater flow velocity v_f [m/d] can be estimated applying the Darcy-Equation, using the dominant hydraulic conductivity K [m/d] of each spatial unit and slope ΔH [m/m] as a surrogate for the hydraulic gradient [Step 1]. Groundwater flow velocity can be transformed into transport velocity v_a [m/d] multiplying with the inverse of the effective porosity θ_{eff} [m^3/m^3] [Step 2]. Transport velocity v_a and flow distance ς can then be used to calculate the average characteristic travel time t [Step 3].

Step 1 $$v_f = -K \cdot \frac{\Delta H}{\Delta \varsigma} \qquad [39]$$

Step 2 $$v_a = \frac{1}{\theta_{eff}} \cdot v_f \qquad [40]$$

Step 3 $$t = \frac{\varsigma}{v_a} = \theta_{eff} \cdot \frac{\varsigma}{v_f} \qquad [41]$$

River density was calculated for each catchment using the length of rivers derived from the CCM2 river network and the catchment area using the catchments defined in the HydroEurope Database.

Slope was calculated as average slope of each catchment using the SRTM digital elevation model. Hydrogeological substrate properties where based on the association of parent materials defined in ESDB and substrate data as described in 2.2.4.

The indicative travel time applied here is based on the average transport distance and must be interpreted as an order of magnitude rather than an exact value. In addition to the conceptual problem having a single value representing in reality a distribution function, the calculation procedure is subject to various uncertainties. The main impact factors are the slope (as a surrogate for hydraulic gradient), the river density (defining flow distances) and hydraulic conductivities of the substrate. In many environments, slope can be a poor estimator for hydraulic gradient. Hydraulic conductivities refer to surface near substrate rather than to deep aquifers. Considerable heterogeneity within spatial units may exist. The river density depends on the scale of the river dataset used. The CCM dataset provides a high resolution network (Strahler Order 1) and may therefore be considered appropriate for an indicative assessment. Artificial drainage networks, however, may change river density regionally.

Fractured rock areas with low hydraulic conductivity can be represented with short travel times, as slope and sediments close to the surface (infiltration capacity and interflow generation) determine runoff response. This is consistent with the heuristic baseflow index concept, where surface runoff and interflow and local groundwater flow in valley bottoms dominate runoff in mountainous fractured rock areas.

The European scale patterns are reasonable with respect to general geological and territorial features. A clear distinction is made between mountainous areas dominated by bedrock and consolidated sediments (Alps, Scandinavian shield, Ireland-Scotland, Pyrenees, Appenine, etc.) having short travel times (high slope, high river density) and the sedimentary basins with long travel times (high slope, lower river density, medium to low conductivity). Surprisingly the Weichselian deposits (Denmark, North- and East Germany) are also characterised by very short travel times although medium conductivities let us expect a higher order of magnitude. This possibly results from the interplay of a high river density (short flow distances) and a hilly landscape where slopes only poorly represent true hydraulic gradients. This observation requires further verification.

The large scale assessment can not appropriately account for local characteristics altering runoff response. Also the geological information included can severely bias estimates, as the dominant geological strata distinguished in the map may not be identical to the geological strata actually containing the aquifers.

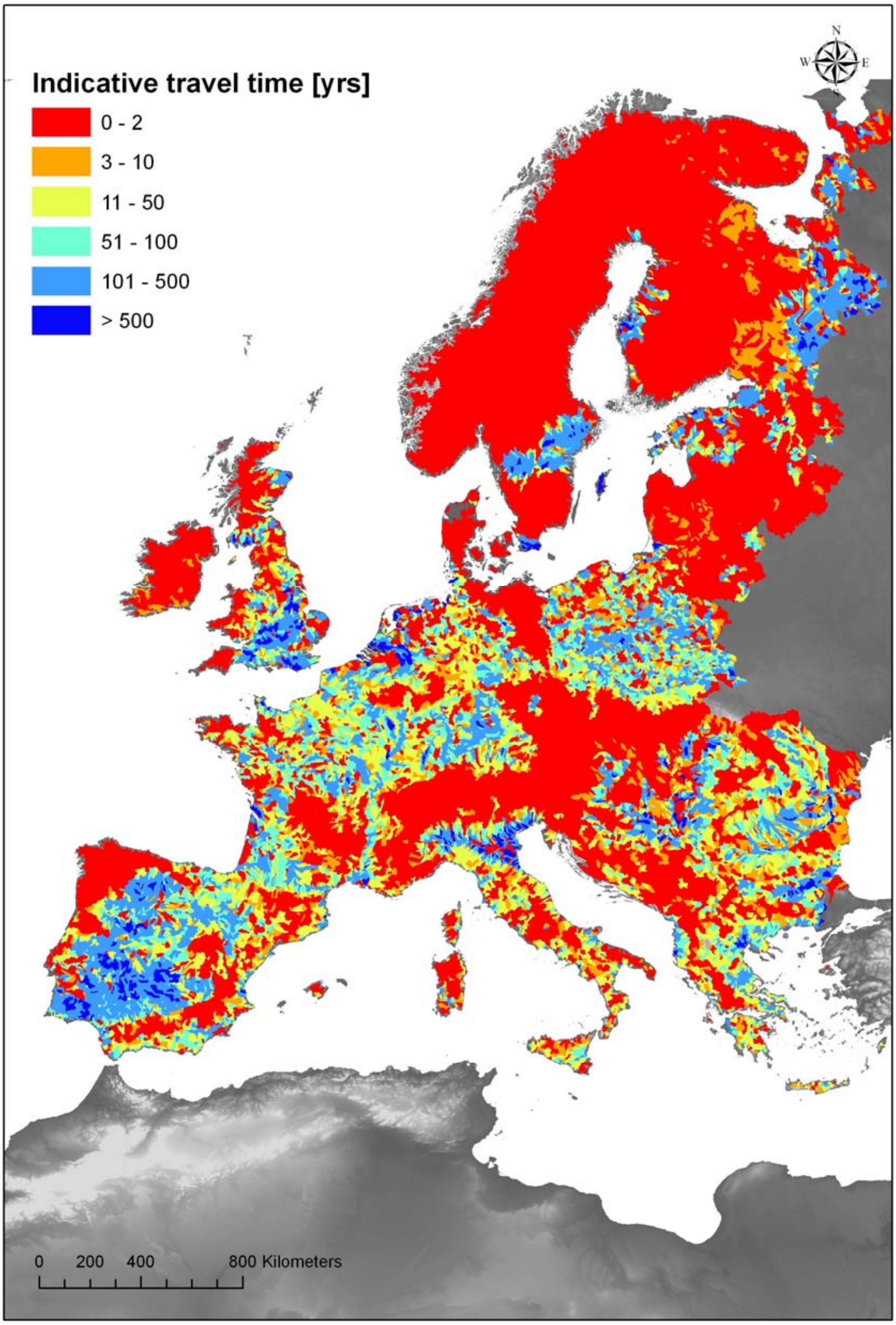

Figure 35: Indicative groundwater travel time [years]

2.11 Screening of groundwater recharge

In section 2 we applied a generic water balance model (Pistocchi et al., 2008), calculating hydrological excess water as sum of direct runoff and infiltration. The water balance model also calculates a direct runoff and an infiltration component, that could be associated to surface and groundwater runoff. These runoff components result from climatic factors (precipitation and evapotranspiration) and the soil water holding potential on a plain soil. Topographical and geological features were not taken into account in the calculation of direct runoff and infiltration, which can result in considerable miscalculation of the flow components requiring further correction. Only the calculation of hydrological excess water is unbiased by the separation of direct and infiltration flow as actual evapotranspiration is calculated independently.

The actual groundwater recharge is different from the infiltration flow, which enters different subsurface flow paths which may be described as groundwater runoff and interflow, fast and slow subsurface runoff or local and regional groundwater runoff (depending on the scale of interest and underlying hydrological concepts)(Barthel, 2006). Thus only part of the infiltrating water becomes groundwater recharge, the fraction of water entering the groundwater flow path is mainly determined by topography (interflow generation) and water transport capacity of the substrate (conductivity). Also the calculation of direct runoff is misleading, as slope is not explicitly taken into account and especially in flat areas or areas with considerable surface roughness the calculated direct runoff may not run off but re-infiltrate in the soil. Slope and hydrogeological conditions as the relevant factors at this level are already considered in the calculation of the heuristic baseflow index, thus allowing separation of a groundwater runoff component (baseflow) and a combined direct/interflow component.

We calculated the infiltration ratio of the water balance model as the ratio of infiltrating water to total runoff and compared it with the heuristic baseflow index. The spatial comparison (as ratio of both indices) is shown in Figure 36. Assuming that the heuristic baseflow index reasonably represents the true groundwater flow component, values greater than one indicate that the infiltration component overestimates groundwater flow as expressed in the baseflow index. This is especially the case in mountainous areas, where slope is a dominant runoff control. Values smaller than one indicate that the infiltration component underestimates the groundwater component and the direct runoff component is overestimated.

Combining the heuristic baseflow index with the water balance simulation results allows recalculating groundwater runoff or recharge. The hydrological excess water (as sum of the direct runoff and infiltration component) was combined with the heuristic baseflow index, providing a groundwater component and a combined direct-runoff/interflow component. The resulting groundwater recharge is displayed in Figure 37.

The entire procedure was applied using a priori settings without further calibration or validation. For a screening approach, this is acceptable but more specific analysis require further validation of the underlying models and information.

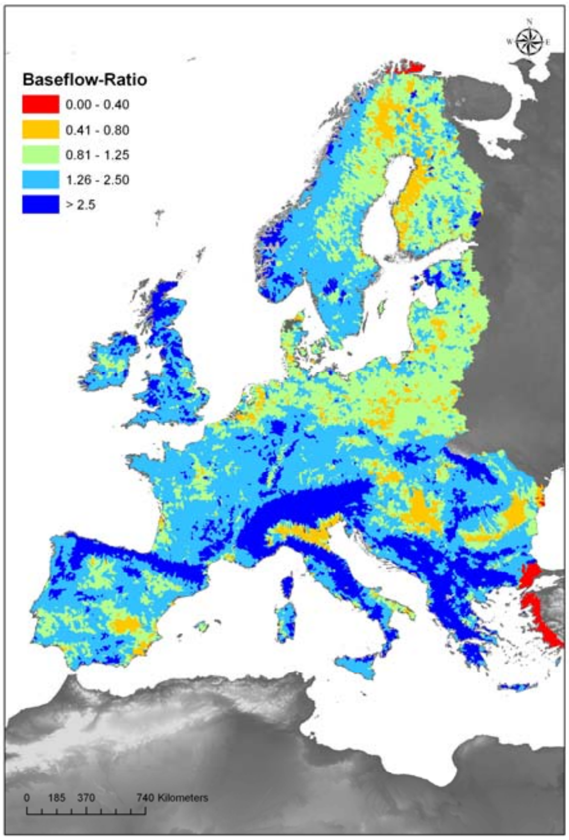

Figure 36: Ratio of the infiltration index (i.e. ratio of the infiltration component to total runoff calculated with the generic water balance model) to heuristic baseflow index (i.e. ratio of groundwater runoff to total runoff)

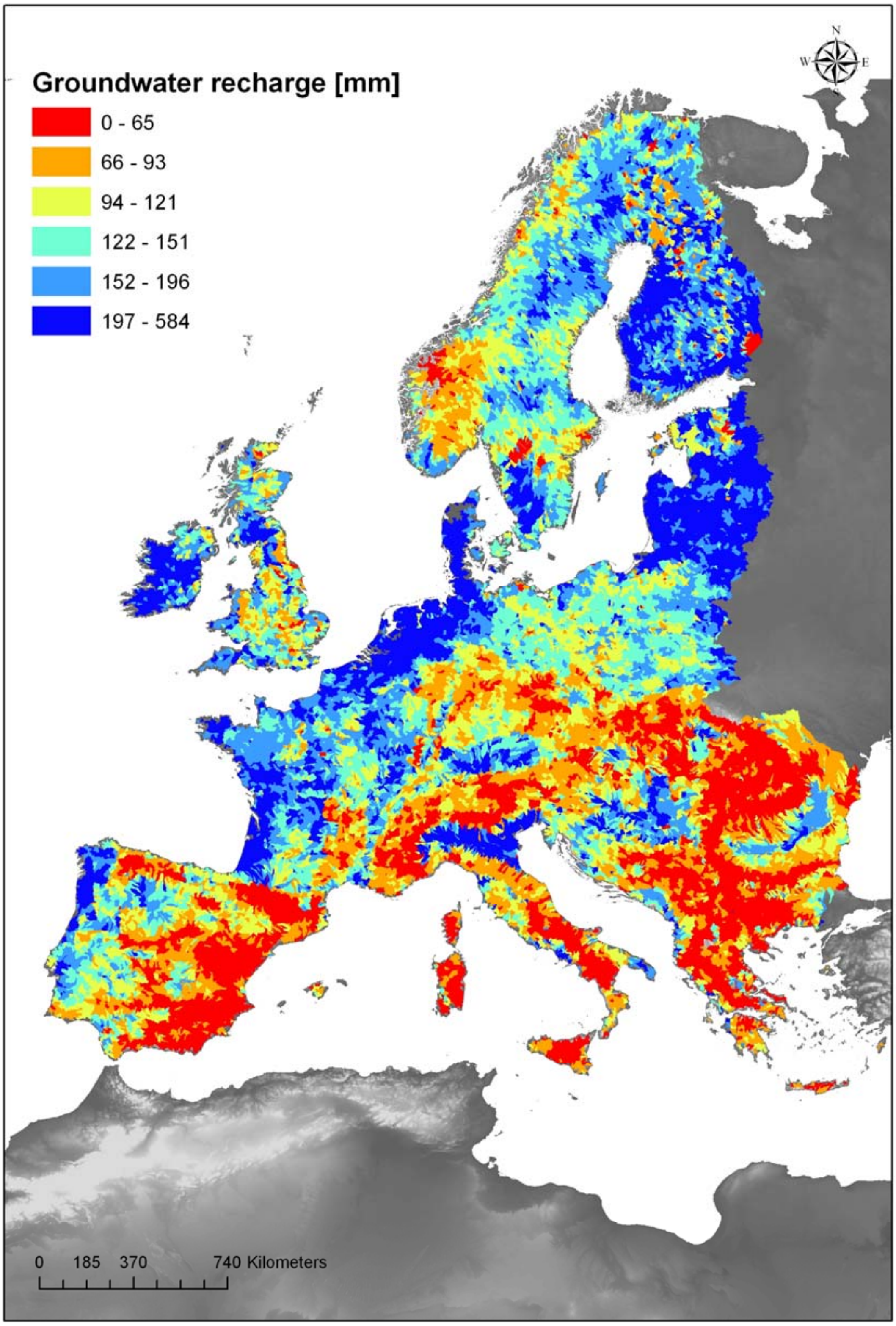

Figure 37: Average groundwater recharge potential (mm) combining the generic water balance model results with the heuristic baseflow index.

3 Synthesis

In this report we documented different studies targeted towards a groundwater balance assessment in Europe. General water balance simulations and hydrograph analyses were applied to estimate hydrological excess water and groundwater recharge and discharge. Using European scale information on human water abstractions, losses, and returns (Wriedt, 2009d), also indicators of pressures on water quantity and a groundwater recharge map were developed.

The work presented in this report relates to other research carried out during the last three years. Previous works aimed at assessing the human water use (demand) (Wriedt et al. 2009d). Spatially distributed estimates of water abstractions (as totals and by sectors) but also of consumptive water losses and water returns where based on available national statistics. The assessment of irrigation requirements in Europe (Wriedt et al. 2009a,b) allowed replacing the statistical information by a more consistent modelling approach, using harmonized approach across the EU. The assessment so far gives an overview on water use in Europe, but various improvements can be made supporting an integrated water balance assessment. Methods for spatial allocation of water abstractions and to allocate water abstractions to specific water resources are necessary but require considerable efforts in terms of method development and data collection (see Wriedt et al. 2009d). In addition, a useful option would be to replace the analysis of statistical information by a generic water use model, that allows developing scenarios of future consumption and water savings to be used in continuous simulation studies. In this report, we have focused on developing tools and applying methods for a general water balance assessment at European scale, with a more specific focus on estimating groundwater recharge. Given the various limitations at European scale regarding data availability and suitable model approaches, we did not derive a final picture of the supply side and the available water resources in Europe.

An assessment of seawater intrusion risk has been a first approach to develop water pressure indicators based on the assessment of water supply and demand. The assessment has outlined some potentially useful approaches, but did not go beyond a pilot assessment, as various limitations could not be overcome at that time. Given the achievements made thereafter (Wriedt et al. 2009d and this report), a revision of the approach and re-assessment at European scale have become possible.

The individual studies have not yet been integrated into a common framework. Rather, they show various restrictions that require further research on various specific issues relevant for water management at European scale. The main concerns laid out in this report are:

- Groundwater assessment must be integrated into a general hydrological assessment explicitly including human water use.
- At European scale data availability, data handling and simulation capacity set limitations to distinction and simulation of individual groundwater systems.
- For screening purposes generic approaches not requiring specific calibration should be developed. The general applicability of such models should be validated independently.
- Alternative approaches for model validation should be developed and applied. Remote sensing methods could allow independent measurements of precipitation, evapotranspiration, and soil water balance, overcoming some problems related to the use of rainfall-runoff time series.
- The allocation of human water abstractions, losses and returns can have considerable impacts on the pressures exerted on water resources. Further research is required to collect suitable data and to develop allocation rules that can be implemented into a water consumption model.

The approaches laid out in this report and related reports provide a starting point for further development of screening approaches to be integrated in a common water resources assessment framework.

References

- Barthel, R. (2006). Common problematic aspects of coupling hydrological models with groundwater flow models on the river catchment scale. Advances in Geosciences 9, 63-71.
- Bear, J., Cheng, A., Sorek, S., Ouazar, D., Herrera, I. (Eds, 1999): Seawater Intrusion in Coastal Aquifers Concepts, Methods and Practices (Theory and Applications of Transport in Porous Media). Berlin, 640 p.
- BGR (2007): International Hydrogeological Map of Europe 1500. http://www.bgr.de/app/ihme1500/
- Bogena, H., Kunkel, R., Schoebel, T., Schrey, H.P., Wendland, F. (2005). Distributed modeling of groundwater recharge at the macroscale. Ecological Modelling 187, 15-26.
- Bouraoui, F., Aloe, A., 2007. European Agrochemicals Geospatial Loss Estimator: Model development and applications. EUR – Scientific and technical research series, EUR 22690 EN.
- Bredehoeft, J.D., 2002. The Water Budget Myth Revisited. Why Hydrogeologists Model. Ground Water 40(4), 340-345.
- De Witt, M., 1999. Nutrient fluxes in the Rhine and Elbe basins. Netherlands Geographical Studies 259. 160pp. University of Utrecht. Utrecht.
- Doherty, J. (2002). PEST – Model independent parameter estimation. Watermark Numerical Computing. http://www.parameter-estimation.com/
- Döll, P., Lehner, B., Kaspar, F. (2002). Global Modelling of Groundwater recharge. In Schmitz, G.H. (ed.), Proceedings of Third International Conference on Water Resources and the Environment Research, Technical University of Dresden, Germany, ISBN 3-934253-17-2, Vol. 1, 27-31.
- Eckhardt, K., 2008. A comparison of baseflow indices which were calculated with seven different baseflow separation methods. Journal of Hydrology, 352, 168-173.
- Ecologic – Institute for International and European Environmental Policy, Final report: EU Water saving potential (Part 1-Report) ENV.D.2/ETU/2007/0001r, Brussels/Berlin 19.07.2007, http://www.ecologic.de/download/projekte/900-949/917/917_water_saving_1.pdf (20.12.2007).
- EEA (1999): Groundwater quality and quantity in Europe. Environmental assessment report No 3. European Environment Agency. Copenhagen.
- ETC, European Topic Centre on Terrestrial Environment, 2005. Corine land cover database (Version 05/2005).

- Figuères, C.M., Tortajada, C., Rockström, J. (Eds.), 2003. Rethinking Water Management. London, Sterling VA., 242p.
- Goubanova, K., Li, L., 2006. Extremes in temperature and precipitation around the Mediterranean in an ensemble of future climate scenario simulations. Global and Planetary Change, doi:10.1016/j.globaplacha.2006.11.012.
- Hiscock, K., 2005. Hydrogeology. Principles and Practice. Oxford, 349pp.
- IPCC, 2007. Climate Change 2007: The Physical Science Basis – Summary for Policymakers. Contribution of WGI to the 4th Assessment Report of the IPCC. Geneva.
- Jones, R.J.A., Hiederer, R., Rusco, E., Loveland, P.J., Montanarella, L., 2004. The map of organic carbon in topsoils in Europe, Version 1.2, September 2003: Explanation of Special Publication Ispra 2004 No. 72 (S.P.I.04.72) European Soil Bureau Research Report No. 17, EUR 21209 EN, 26pp. and 1 Mao in ISO B1 format. Office for Official Publications of the European Communities, Luxembourg.
- Kemper, K.E., 2003. Rethinking groundwater management. In: Figuères et al. (2003).
- Meinardi, C.R., 1994. Groundwater recharge and travel times in the sandy regions of the Netherlands. National Institute of Public Health and Environment Protection (RVIM). Report No. 715501004. Bilthoven, RIVM.
- MGWWG Mediterranean Groundwater Working Group of the Mediterranean Joint Process WFD/EUWI Process, 2007. Mediterranean Groundwater Report. http://www.semide.net/topics/groundwater/Mediterranean_Groundwater_Report_final_150207.pdf (9.7.2009)
- Micale, F., Genovese, G., 2004. Methodology of the MARS Crop Yield Forecasting System. Vol. 1: Meteorological Data Collection, Processing and Analysis, ISBN 92-894-8130-3, EUR 21291/EN/1.
- Milnes, E., Renard, P. (2004). The problem of salt recycling and seawater intrusion in coastal irrigated plains: an example from the Kiti aquifer (Southern Cyprus). Journal of Hydrology 288, 327-343.
- Morris, D.A., Johnson, A.I. (1967): Summary of hydrologic and physical properties of rock and soil materials, as analyzed by the hydrologic laboratory of the U.S. Geological Survey 1948-1960. U.S. Geological Survey Water Supply Paper 1839-D. Internet: http://onlinepubs.er.usgs.gov/lizardtech/iserv/browse?cat=WSP&item=/wsp_1839-D.djvu&style=simple/view-dhtml.xsl&wid=750&hei=600&props=img(Name,Description)&page=0

- Mouelhi, S., Michel, C., Perrin, C., Andreassian, V. (2006). Stepwise development of a two-parameter monthly water balance model. Journal of Hydrology 318, 200-214. doi:10.1016/j.jhydrol.2005.06.014
- Mulligan, D., Bouraoui, F., Grizzetti, B., Aloe, A., Dusart, J., 2006. An Atlas of Pan-European Data for Investigating the Fate of Agrochemicals in Terrestrial Ecosystems. JRC-Institute for Environment and Sustainability, EUR Scientific and Technical Research Series, EUR 22334 EN.
- Pawlewicz, M.J., Steinshouer, D.W. and Gautier, D.L. (2003): Map Showing Geology, Oil and Gas Fields, and Geologic Provinces of Europe including Turkey. USGS Open File Report 97-470I
- Pistocchi A., Bouraoui, F., Bitelli, M., 2008. A simplified parameterization of the monthly topsoil water budget. Water Resources Research, 44, W12440, doi:10.1029/2007WR006603.
- Rawls, W.J., Brakensiek, D.L., 1985. Prediction of soil water properties for hydrological modelling. In: E.Jones and T.J. Ward (eds.): Proceedings of the ASCE Symposium Watershed Management in the Eughties, 30. Apr. – 2. May, Denver, CO, 293-299.
- Rodriguez Diaz, J.A., Weatherhead, E.K., Knox, J.W., Camacho, E., 2007. Climate change impacts on irrigation water requirements in the Guadalquivir river basin in Spain. Regional Environmental Change 7, 149-159.
- Rutledge, A.T., 1998, Computer programs for describing the recession of ground-water discharge and for estimating mean ground-water recharge and discharge from streamflow data – update: U.S. Geological Survey Water-Resources Investigations Report 98-4148, 43 p.
- Schulla, J. & K. Jasper (2007): Model Description WaSiM-ETH. *Technical report*, pp. 181, http://homepage.hispeed.ch/wasim/products/wasim_description.htm.
- Smakhtin, V.U., 2001. Estimating continuous monthly baseflow time series and their possible applications in the context of the ecological reserve. Water SA, 27(2), 213-217.
- Wendland, F., 1992. Die Nitratbelastung in den Grundwasserlandschaften der alten Bundeslaender. Berichte aus der öekologischen Forschung,, Bd. 8. Juelich, 150pp.
- Wriedt, G., Bouraoui, F., 2009c. Large scale screening of seawater intrusion risk in Europe - Methodological development and pilot application along the Spanish Mediterranean coast. JRC Scientific and Technical Reports, EUR 23840, 47p.
- Wriedt, G., Bouraoui, F., 2009d. A pan-European atlas of water abstractions, losses and returns A screening for spatially distributed hydrological modelling at European scale. JRC Scientific and Technical Reports. In preparation.

- Wriedt, G., Van der Velde, M., Aloe, A., Bouraoui, F., 2008. Water use for irrigation in the European Union. JRC Scientific and Technical Reports, 70p, EUR 23453 EN. http://ies.jrc.ec.europa.eu/documentation/scientific-reports/rural-water-and-ecosystem-resources-unit-2.html
- Wriedt, G., Van der Velde, M., Aloe, A., Bouraoui, F., 2009a. A European irrigation map for spatially distributed agricultural modelling. Agricultural Water Management 96, 771-789, doi:10.1016/j.agwat.2008.10.012
- Wriedt, G., Van der Velde, M., Aloe, A., Bouraoui, F., 2009b. Estimating irrigation requirements in Europe. Journal of Hydrology, 373, 527-544, doi:10.1016/j.jhydrol.2009.05.018.

European Commission

EUR 23966 EN – Joint Research Centre – Institute for Environment and Sustainability
Title: Towards a General Water Balance Assessment of Europe
Author(s): Gunter Wriedt, Fayçal Bouraoui
Luxembourg: Office for Official Publications of the European Communities
2009 – 57 pp. – 21 x 29 cm
EUR – Scientific and Technical Research series – ISSN 1018-5593
ISBN 978-92-79-12979-7
DOI 10.2788/26925

Abstract
Large proportions of water supply in European countries rely on groundwater resources, and many aquifers in water scarce regions are overexploited. Water management relies on reasonable information on water availability as well as on water demands by different sectors. Information on water availability and water needs are crucial to identify hot spots of quantitative pressures on water resources. This report focuses on estimating natural water availability across Europe. Simple water balance models were applied for an assessment of available water and potentials and limitations of their application are shown. Special emphasis is given to the role of groundwater in the water cycle and we explore ways to derive groundwater balance terms for large scale assessments. We further develop indicators of water quantity pressure relating water availability to water use and losses at different spatial scales. A short overview on the functioning of groundwater systems is given, highlights properties, processes and problems relevant for groundwater quantity and quality assessment. Some concepts to address groundwater issues at large scale are derived. The methodological part combines a general water balance assessment at large scale with more specific approaches to characterize groundwater systems and to quantify groundwater balance terms at large scale. Two different water balance modelling approaches are applied estimating the amount of water available for direct and subsurface runoff. The modelling approaches are compared to observed values and to each other. The available water is compared to water abstractions developing to indicators for human pressures on water resources. Focusing more specifically on groundwater systems, different methods to calculate baseflow and groundwater recharge are applied and compared and a prototype groundwater recharge map of Europe is presented. The report concludes with a synthesizing discussion of methods and results and an outlook on possible future studies. The individual studies have not yet been integrated into a common framework. Rather, they show various restrictions that require further research on various specific issues relevant for water management at European scale. The approaches laid out in this report and related reports provide a starting point for further development of screening approaches to be integrated in a common water resources assessment framework.

The mission of the JRC is to provide customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies. As a service of the European Commission, the JRC functions as a reference centre of science and technology for the Union. Close to the policy-making process, it serves the common interest of the Member States, while being independent of special interests, whether private or national.

LB NA 23966 EN C

EUROPEAN COMMISSION

Publications Office
Publications.eu.int

ISBN 978-92-79-12979-7
9 789279 129797