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# hydrolink

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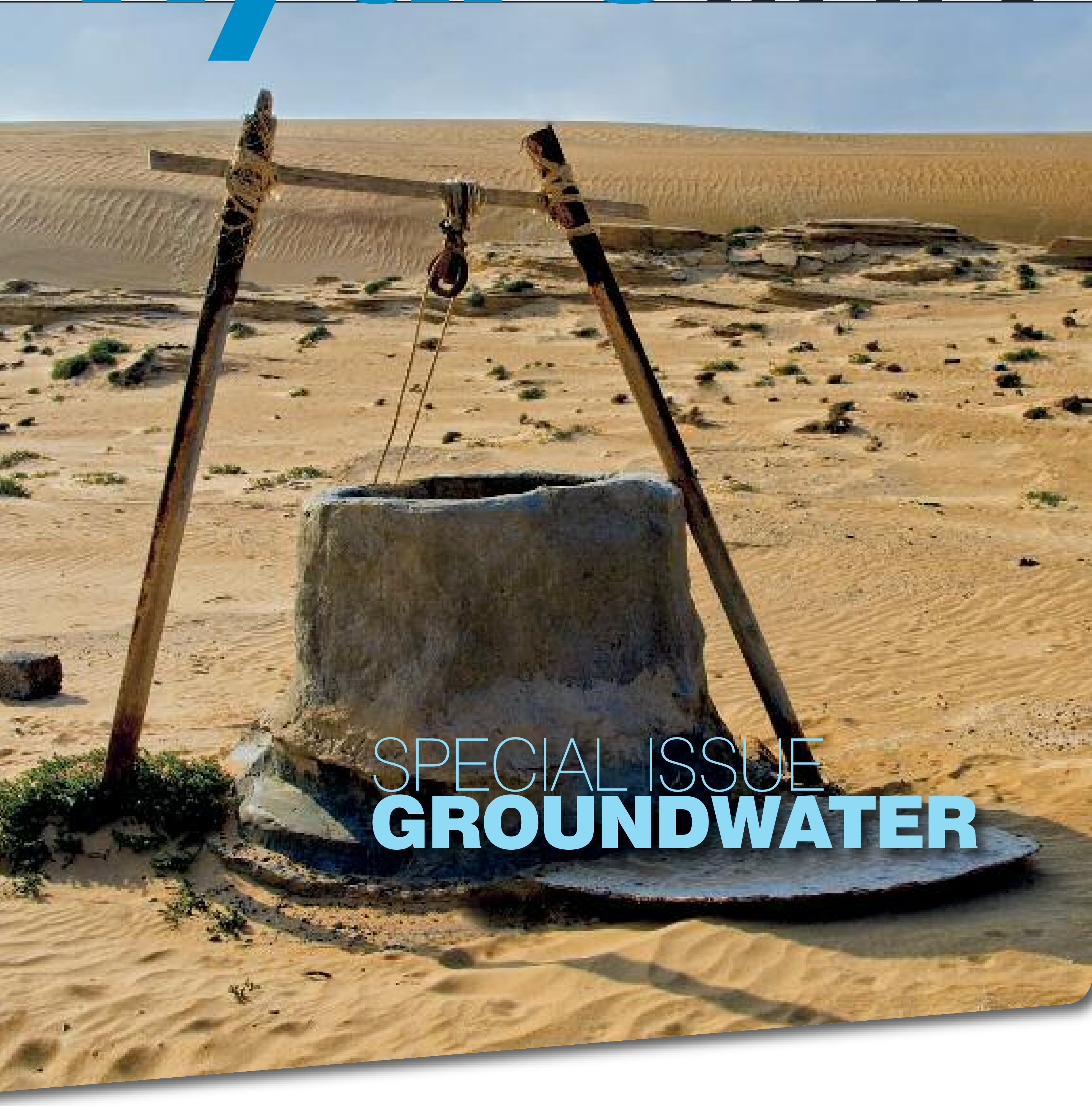

## SPECIAL ISSUE
## GROUNDWATER

IAHR AIRH
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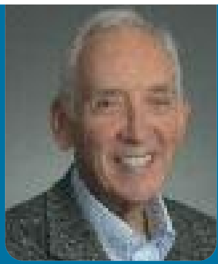
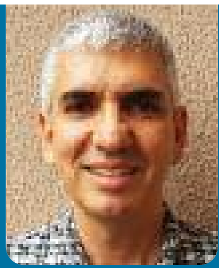
# GROUNDWATER
# THE HIDDEN WATER RESOURCE

## EDITORIAL BY ANGELOS N FINDIKAKIS AND NADIM K COPTY



**Angelos N. Findikakis**
***Hydrolink Editor***



**Nadim K Copty**
***Guest Editor***

Groundwater has been a major water resource for humans throughout history. Excluding polar ice and glaciers, groundwater stocks constitute about 97% of the useable freshwater, more than 100 times the total volume of surface water resources in lakes and rivers. Groundwater has been traditionally considered a relatively reliable water resource that is less affected by seasonal or annual variations in precipitation. As a result, the amount of groundwater extractions has increased dramatically in recent decades from about 312 $Gm^3$/year in the 1960s[1] to 982 $Gm^3$/year in 2010[2]. This sharp increase in groundwater use has led to significant economic development worldwide. These benefits however have been accompanied, in many instants, by high environmental and social costs. Adverse impacts of excessive groundwater extraction include rapid drop in groundwater levels, groundwater depletion, reduction in water levels in nearby streams and lakes, land subsidence, and high salinity levels in coastal aquifers.

Groundwater has been used for domestic use and irrigation since ancient times. Early evidence of human-dug wells going back thousands of years have been uncovered in various regions of the world including China, Egypt and India and Europe[3]. On the island of Cyprus in the Mediterranean, archaeologists have unearthed the remains of constructed wells dating back to the Stone Age almost 10,000 years ago[4]. These wells relied on various construction techniques to raise the water to the ground surface. Early examples of wood-lined and ceramic wells can be found in China. Step wells lined with blocks of stone can be found in India. The *shadoof,* which consist of a well pole for raising water, was used in Egypt, China and other parts of the world. The *qanat,* which likely originated in ancient Iran or in the southeast of the Arabian Peninsula, is comprised of a series of vertical access shafts and a sloping underground channel that allowed for the water to drain out due to gravity.

Many of these same water extraction systems remain in use to this very day. Yet, recent advances in drilling technology and the introduction of motorized pumps made possible the intensive use of groundwater, especially with the expansion of irrigation since the middle of the twentieth century. Today about 40 percent of all irrigated lands are equipped for groundwater use, and 70 percent of all groundwater is used in agriculture[5]. As a result, groundwater depletion hotspots have cropped in various regions of the world most notably the Middle East and North Africa, China, South Asia, southern Europe and the United States. There is an urgent need to develop and implement measures and policies that ensure the sustainable use of this vital resource.

This special *Groundwater* issue of Hydrolink includes articles that exemplify the wide range of issues and ongoing research activities relating to the field of Groundwater. Daniele Tonina describes the presence of hyporheic flows between stream waters and the sediments underneath. This hyporheic exchange of water brings solutes and suspended matter from stream waters into the underlying sediments where important geochemical reactions occur. These transformations can have significant impacts on microorganisms and water quality. The article by Nahed Ben-Salem et al examines the water quality of one of the main Wadis flowing into the Gulf of Tunis in the Mediterranean Basin. It investigates the main sources of the pollution and the potential contamination of underlying groundwater resources. These two articles highlight the interaction between surface and subsurface waters which in the past has often been ignored.

One of the most challenging groundwater contamination problems is the remediation of the groundwater and soil contaminated by hydrocarbons in the form of non-aqueous phase liquids (NAPLs). Because of their low solubility and low biodegradation potential, many of these compounds once released can persist in the subsurface environment for decades or even centuries, contaminating large volumes of the aquifer. The article of Geoffrey Tick provides a comprehensive review of promising technologies for the remediation of NAPL source zones.

Aquifer systems involve numerous stakeholders often with competing interests and goals. These diverging goals can lead to conflicts, particularly in water scarce regions of the world. The article by Dalila Loudyi et al discusses the water stresses that Morocco faces. It proposes the implementation of a "participative water management contract" to help alleviate some of the conflicts between different stakeholders that have arisen in the past. As noted earlier, extensive use of groundwater resources has contributed significantly to the economic development of numerous regions of the world. This however has led to sharp depletion of the water resource in many parts of the world. The article by Eduardo Cassiraga et al describes the extensive exploitation of the East Mancha Aquifer System in Spain and how stakeholders have successfully come together to better manage this system and reverse some of the adverse trends of the past years.

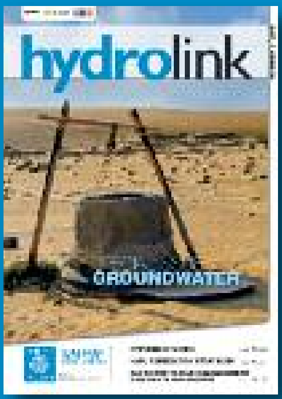
In recent decades significant progress has been made in the field of groundwater flow and contaminant transport modelling. In real life problems, a major challenge remains, the lack of data to fully describe the heterogeneous subsurface system and the complex biochemical processes occurring within. The article by Alberto Guadagnini et al. discusses the uncertainties involved in the characterization of subsurface systems and the need for the quantification of these uncertainties across various scales and for assessing the implications of these uncertainties. Christina Stylianoudaki et al present an application of artificial neural networks (ANN) for predicting nitrate concentrations in the groundwater. ANN are a class of data-driven easy to use models that attempt to quantify correlations between different variables without the need to solve complex physics-based models that are not fully characterized.

Human-induced climate change has emerged in recent years as a major threat to the well-being of our planet. As an integral component of the global water cycle, groundwater resources in many regions of the world will unequivocally be adversely impacted through diminished precipitation, extended droughts, frequent flooding and seawater rise. The article by Glen Walker et al discusses the potential adverse effects that climate change is predicted to have on groundwater resources in southern Australia. It highlights some of the adaptive measures that are being considered and that can serve as a guide for the protection of this vital resource in other water-stressed regions of the world.

The articles published in this issue provide a sample of the broad range of ongoing research in understanding the behaviour of groundwater systems and developing methods for safeguarding and managing sustainably this valuable resource. ■

### References


[1] Wada Y, van Beek LPH, van Kempen CM, Reckman JWTM, Vasak S, Bierkens MFP (2010) Global depletion of groundwater resources. *Geophys Res Lett 37(20)*, L20402.
[2] Margat, J., and J. van der Gun. 2013. Groundwater around the World. CRC Press/Balkema
[3] Tegel W, Elburg R, Hakelberg D, Stäuble H, Büntgen U (2012), "Early Neolithic Water Wells Reveal the World's Oldest Wood Architecture". *PLoS ONE. 7(12)*: e51374.
[4] BBC News (2009), "Stone Age wells found in Cyprus", http://news.bbc.co.uk/2/hi/europe/8118318.stm
[5] Hoogesteger J, and Wester P (2015), Intensive groundwater use and (in)equity: Processes and governance challenges, *Environmental Science & Policy*, 51, 117-124.

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*Cover picture:*
Water well in Oman Desert (gettyimages)

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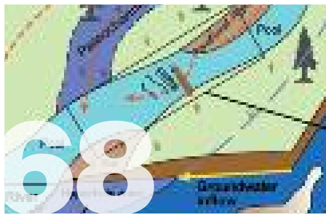
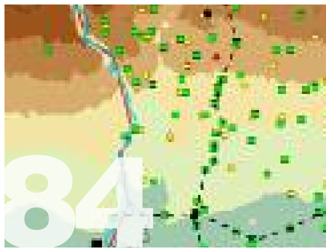
# IN THIS ISSUE

## GROUNDWATER SPECIAL










# HYPORHEIC FLOWS: THE STREAM WATER FLOWING UNDERNEATH

BY DANIELE TONINA

Stream water flows above but also within the interstitial spaces of the sediment underneath and besides streambeds. Water enters and exits streambed sediments ubiquitously in streams and rivers and forms the so called hyporheic exchange, which defines the hyporheic zone, the band of sediment mostly saturated with stream water. This zone is characterized by strong physical gradients, a unique ecotone and is responsible for most of the biogeochemical processes in streams.

There is a stream of surface water, called hyporheic zone[1], that flows underneath and besides rivers and streams (Figure 1). This “hidden stream” is formed by stream water entering streambed sediments in so called downwelling areas and emerges back into the stream in so called upwelling areas. These downwelling and upwelling fluxes form the hyporheic exchange and delineate a stream-water saturated volume of sediment that is the hyporheic zone. This exchange is ubiquitous in streams and rivers because of permeable sediments and stems from near-bed pressure gradients (induced by flow and streambed-features interaction), that are dependent on alluvium depth, alluvium lateral confinement, and streambed sediment hydraulic conductivity, and from flow turbulence, which causes pressure and velocity fluctuations at the channel bottom[2]. These mechanisms depend on stream geometry and discharge such that their relative importance on hyporheic exchange may depend on channel type and may vary along the stream network[3] (Figure 2). Thus, both the vertical and the horizontal extent of the hyporheic zone vary spatially, due to changes in stream size, morphology, alluvial bed and aquifer conditions, and temporally, due to physicochemical fluctuations, e.g., stream discharge, groundwater table, water temperature and stream solute loads.

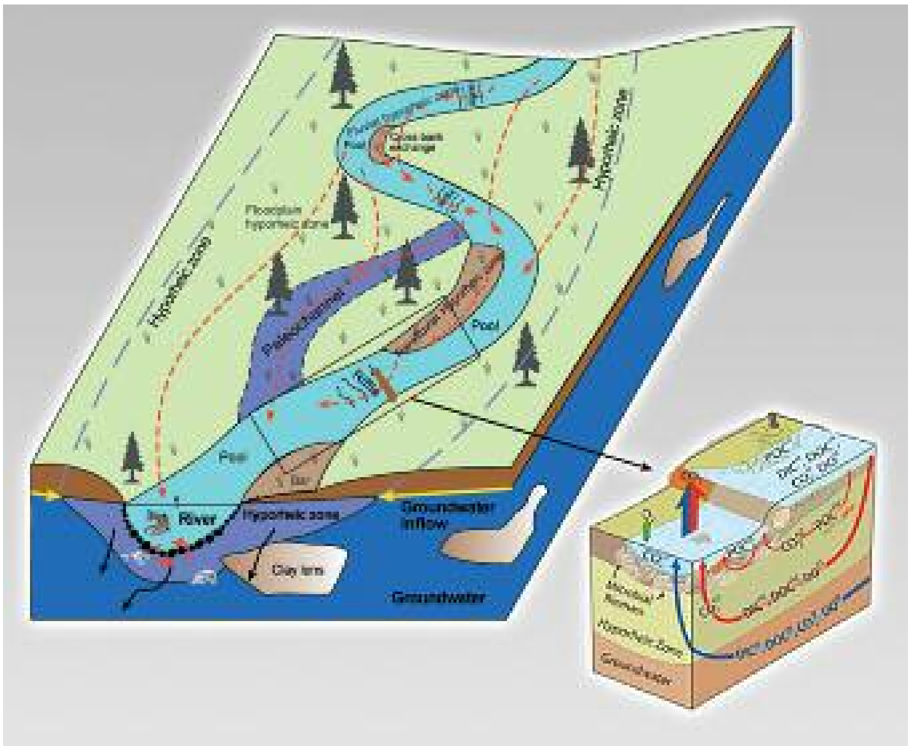



Figure 1. Hyporheic zone classification, as fluvial within the active channel, parafluvial below emerged bedforms, like bars and floodplain with its pathlines depicted in red lines, from Tonina and Buffington[1]

Hyporheic exchange brings solute and suspended particles laden surface water into the sediment, where reactive solutes undergo biogeochemical reactions due to biofilms attached to streambed particles or are uptaken by organisms dwelling within particle interstices. Products of such transformations are then carried away by hyporheic flows and brought to the surface water by upwelling fluxes. In streams (riverine systems with widths less than about 30 m), micro-organism population densities are higher in the hyporheic zone than in the water column, such that most microbially mediated transformations occur in the hyporheic zone rather than in the water column[4]. These transformations depend on reaction time, water temperature, solute concentrations, flow velocity, and length of the flow path. Consequently, this exchange affects both surface and pore water quality and the transport and fate of nutrients. For instance, field investigations and recent flume experiments on the nitrogen cycle have shown the importance of the hyporheic zone as a biogeochemical reaction zone, where both nitrification, which is an aerobic process, and denitrification, a primarily anaerobic process, occur[5]. Nitrification transforms reduced forms of nitrogen, mainly ammonium, into nitrate, whereas denitrification transforms mostly $NO_3^-$ to dinitrogen gas ($N_2$) and nitrous oxide ($N_2O$). The latter is a major destructor of stratospheric ozone and is 300 times more potent as greenhouse gas than carbon dioxide ($CO_2$) with a century long life span in the atmosphere. Thus, the hyporheic zone may have far reaching global impacts as a source or sink of greenhouse gases, which may include besides $N_2O$ also carbon dioxide and methane. The

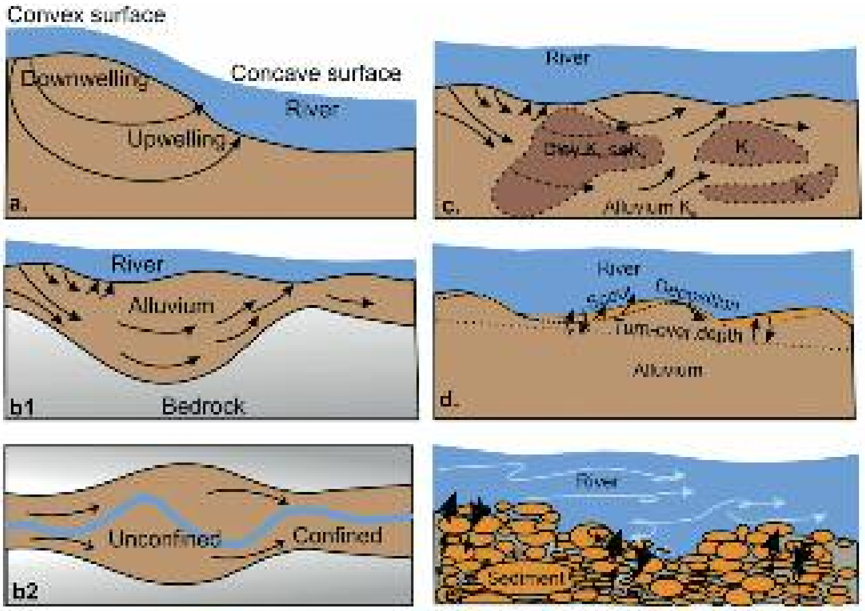



Figure 2. Major mechanisms inducing hyporheic exchange: spatial variations of (a) energy heads, (b) alluvial area, which depends on (b1) alluvial depth and (b2) valley constrain variations, (c) of hydraulic conductivity of the sediment and (d) sediment transport (turn-over) and (e) near-bed turbulence (modified from Tonina and Buffington, 2009).

**Daniele Tonina** is an Associate Professor at the Center for Ecohydraulics Research. He held post-doctoral research positions at the University of California at Berkeley (USA) and at the University of Trento (Italy). He received engineering degrees from the University of Trento (BS, MS, 2000) and the University of Idaho (PhD, 2005). His research focuses on the interaction between surface and subsurface waters, riverine aquatic habitat and use of remote sensing in monitoring stream hydraulics. He is an IAHR, ASCE and AGU member. He serves as Associate Editor for the scientific journals of Water Resources Research, Hydrological Processes and Hydraulic Engineering.

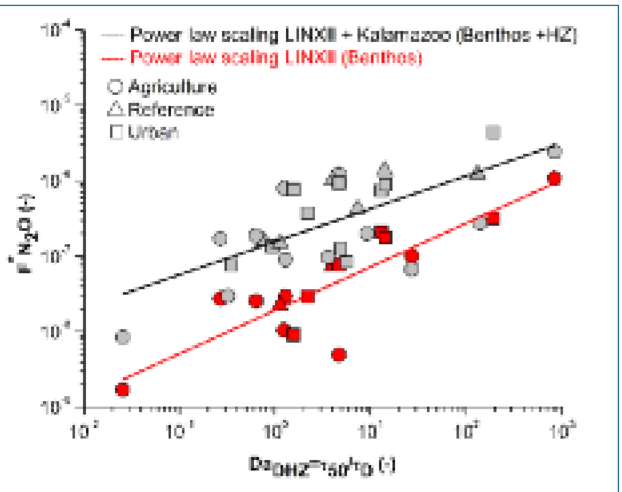
hyporheic zone production of $N_2O$ in small streams, defined with widths less than 10 m, is about one order of magnitude larger than that in benthic zone and the water column[4]. It is the dominant source of $N_2O$ in small streams regardless of land use based on data collected in streams draining urban areas, agricultural fields and natural and pristine areas (defined as reference in Figure 3).

The physical gradients, which include solute concentration, water temperature and velocity, developed within the hyporheic zone, identify a unique ecotone formed by the presence of macro invertebrates, called hyporheos, that are neither benthos nor groundwater fauna. Large organisms like fish may reside for part of, or for an entire stage of their life in this ecotone, which is an important part of the river corridor and which includes in-stream water, banks, floodplain and hyporheic zone, and its ecosystem.

Because of these three main characteristics: (i) hyporheic exchange, which can be enveloped and separated by the groundwater system, (ii) waters with different chemical signature than the groundwater, and (iii) unique macro-invertebrate populations, the hyporheic zone has been studied separately in biology, hydrology and bio-geochemistry. This resulted in three operational approaches to define it: hydraulic based on flow paths, geochemical based on solute concentrations and biological based on presence or absence of hyporheos.

Research in these fields suggests that the hyporheic zone is important for modeling and predicting biogeochemical reactions in streams, microbial and macro-invertebrate distributions and ecosystem functioning[e.g., 5][6]. Theoretical frameworks, which account for both hyporheic and surface hydraulics and for bio-geochemical reactions are necessary to study riverine systems and the impact of anthropogenic activities on riverine corridors. Whereas hyporheic models are available for dune and pool-riffle morphology, little is known about steeper systems like step-pool, cascade or plane bed systems. Advances in understanding feedback among hyporheic fluxes, biogeochemical processes and organisms' use of riverine corridors are key to better manage surface and also subsurface water in a sustainable way. ■

Figure 3. Dimensional flux, defined as the ratio of $N_2O$ flux per unit area of stream surface normalized by dissolved inorganic nitrogen mass flux (concentration of dissolved inorganic nitrogen multiplied by mean flow velocity), versus the hyporheic Damkhöler number defined as the ratio between the median residence time of surface water within the hyporheic zone and the denitrification time scale. The red points and line show the dimensionless $N_2O$ emission from the benthic zone and the gray from the combined hyporheic and benthic zone. Modified from Marzadri et al.[4].




## References

[1] Tonina, D. & Buffington, J. M. Hyporheic exchange in mountain rivers I: Mechanics and environmental effects. *Geography Compass* 3, 1063–1086 (2009).
[2] Boano, F. *et al.* Hyporheic flow and transport processes: Mechanisms, models, and biogeochemical implications. *Rev. Geophys.* 52, 603–679 (2014).
[3] Buffington, J. M. & Tonina, D. Hyporheic exchange in mountain rivers II: Effects of channel morphology on mechanics, scales, and rates of exchange. *Geogr. Compass* 3, 1038–1062 (2009).
[4] Marzadri, A., Dee, M. M., Tonina, D., Bellin, A. & Tank, J. L. Role of surface and subsurface processes in scaling N2O emissions along riverine networks. *Proc. Natl. Acad. Sci.* 114, 4330–4335 (2017).
[5] Quick, A. M. *et al.* Controls on Nitrous Oxide Emissions from the Hyporheic Zones of Streams. *Environ. Sci. Technol.* 50, 11491–11500 (2016).
[6] Tonina, D., Marzadri, A. & Bellin, A. Benthic uptake rate due to hyporheic exchange: The effects of streambed morphology for constant and sinusoidally varying nutrient loads. *Water (Switzerland)* 7, 398–419 (2015).

# IMPACT OF ANTHROPOGENIC ACTIVITIES ON PHYSIOCHEMICAL PROPERTIES OF WADI EL BEY (NORTHEAST OF TUNISIA)

BY NAHED BEN-SALEM, MAKRAM ANANE, SEIFEDDINE JOMAA & SALAH JELLALI

Wadi El Bey and its two tributaries Wadi El Maleh and Wadi Tahouna, is one of the main Wadis flowing into the Gulf of Tunis. The main pollution sources in the region of Grombalia-Soliman were investigated through intensive sampling campaigns, allowing detailed mapping of the spatial distribution of pollution in the Wadis. Industrial wastewater discharges in the region seem to be mostly responsible for water quality degradation in Wadi El Bey, and therefore represent a severe threat of groundwater pollution in the region.

Population growth and the relative improvement of the standard of living in the Mediterranean (MED) coastal cities led to the expansion of agricultural and industrial activities, resulting in large water consumption[1]. This growing anthropogenic pressure resulted in significant increase in pollutant loads discharged into the hydrographic network, which in turn impacts significantly the aquatic ecosystem. To address this problem, different national, regional and international environmental protection initiatives have been established. Among these, the Marine Strategy Framework Directive (MSFD) that was developed to achieve a good environmental status in the MED Sea by reducing the causes of eutrophication and its consequences[2]. Also, a partnership on groundwater issues was developed between EU and non-EU countries of the MED region, where actions to reduce the marine pollution in the MED Sea and solutions on surface water resources management were suggested (e.g., ENI SEIS II SOUTH Project[1]). At the national level, two big initiatives are undertaken currently in Tunisia named the Water Code and Water 2050[3],[4] which concern the establishment of mid- and long-term national strategies regarding water resources management and environmental preservation by the horizon 2050. Special attention is paid to coastal water resources management and wastewaters valorisation and reuse. Despite these numerous initiatives, some water systems are experiencing challenging conditions due to multiple interacting anthropogenic pressures. Among them are the Wadis feeding the Gulf of Tunis, which constitute the natural outlet for surface runoff, but also domestic and industrial effluents, as well as discharges from agricultural drainage. For instance, Wadi El Bey is the stream most polluted by industrial wastewaters (Figure 1). The primary sources of pollution in Wadi El Bey are tanneries, paper mills, breweries, tomato processing and slaughter products. Faced with this precarious situation and for better preservation of the receiving environment, the Tunisian government decided to support the implementation of a study for the decontamination of the Gulf of Tunis[3]. The main objectives of this study are to: (i) perform an inventory of the main pollution sources of Wadi El Bey from industrial and urban pollution activities, (ii) evaluate the degree of its pollution based on extensive physicochemical analyses of its waters at different locations from the watershed upstream until the Soliman's wetland (close to the MED sea), and (iii) to map the spatial distribution of each pollution parameter using Geographical Information Systems.

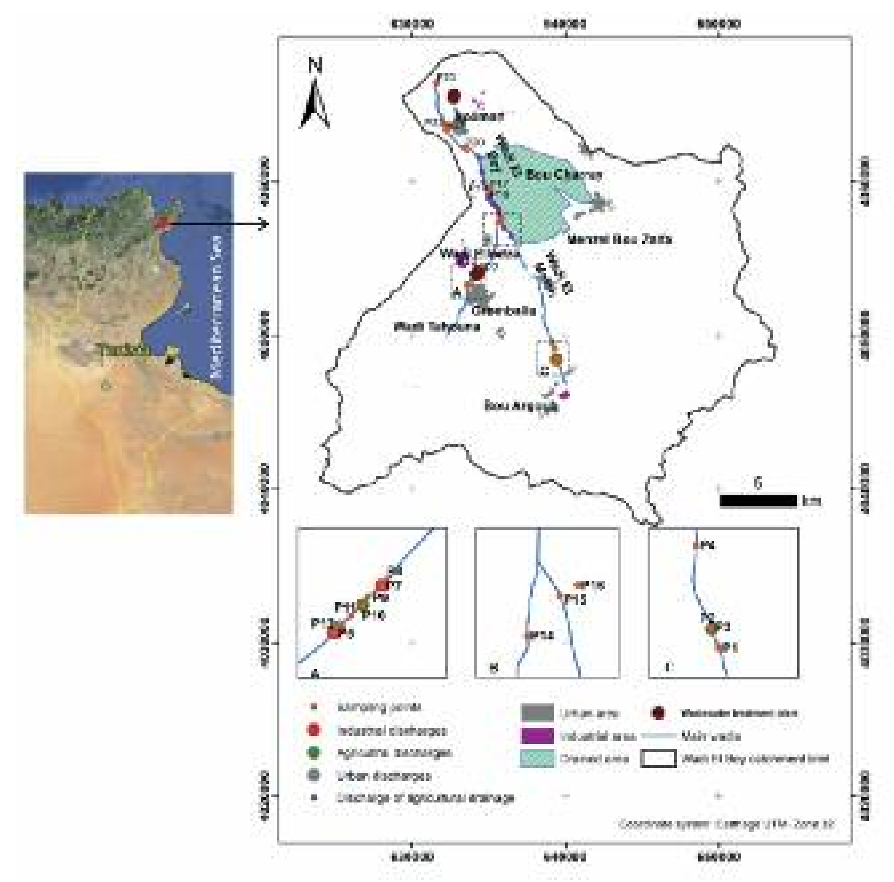



Figure 1. Location of the study area, the major sources of pollution and sampling points

**Study area description**

The Wadi El Bey watershed is located in the northeastern of Tunisia. It covers a total area of about 513 km$^2$ and includes several urban agglomerations including Soliman, Bou Argoub, Grombalia and Menzel Bouzelfa with several industrial areas (Figure 1). Rainfall in this area is characterized by temporal irregularity with a rainy season spread over a period from September to May and a dry season in summer. The lowest temperatures are observed in January with an average of 12.3 °C. The geological layers present a stratigraphic succession from the lower Eocene to the Quaternary. The study area consists of a vast plain with gentle slopes (0-3%), directed towards the MED Sea. The hydrographic network extends downstream to the wetland of Soliman and is subject to many sources of pollution mainly the Grombalia, Bou Argoub and Soliman Waste Water Treatment Plants (WWTPs), the Grombalia and Soliman industrial areas and the Bou Charray agricultural drainage network (Figure 1). This lagoon covers approximately 2.25 km$^2$ and has a depth of less than 5 m. It communicates with the sea through a small canal. The study area is mainly agricultural with the presence of some agri-food and textile industries.

A total of 17 surface water sampling points (Figure 1) distributed along the hydrographic network have been used, located upstream and downstream of the point sources of pollution from industrial, domestic and agricultural discharges. Six additional samples were taken from the sources themselves; three from urban WWTPs output, two from industrial WWTPs and one from the agricultural drainage main stream. The main physicochemical parameters (pH, Electrical Conductivity (EC), Dissolved Oxygen (DO), Turbidity, the Suspended Matter (SM), the Total Dissolved Salts (TDS), Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD5)) were analyzed in the laboratory, according to AFNOR (French Association of NORmalization) standards[4], along with the main major elements (chlorides, sulphate, orthophosphate, nitrate, ammonium and sodium)obtained by ion chromatography. For each analyzed parameter, the descriptive statistics were performed and compared to the Tunisian standards of surface water discharge in order to assess the contamination level of Wadi El Bey surface water. Then the analyzed physic-ochemical parameters were interpolated from the sampled points to the entire hydrographic network using the diffusion kernel, a method of interpolation with barriers. The obtained maps depict the spatial distribution of each analyzed parameter over the entire hydrographic network from the first sampling point to the outlet in Soliman lagoon.

**Nahed Ben-Salem** received her Master of Science in 2018 in Integrated Planning for Rural Development and Environmental Management at the Mediterranean Agronomic Institute of Zaragoza (IAMZ) - University of Lérida, in Spain. In 2016, she obtained her Engineering degree in Hydraulics from the National School of Hydraulic Engineering of Medjez El Bab (ESIER), Tunisia. She is currently a PhD student at the Helmholtz Centre for Environmental Research (UFZ), Germany. Her areas of expertise include water resources management, soil erosion monitoring and modeling, hydrology and hydrogeology.

**Seifeddine Jomaa** is a senior scientist at the Helmholtz Centre for Environmental Research (UFZ), Germany. In early 2012, he received his PhD degree in Environmental Engineering from the Swiss Federal Institute of Technology Lausanne (EPFL), in Switzerland. In 2007, he received his Master of Science in Hydrology from the National Polytechnic Institute of Toulouse (INPT), France. He started his academic career investigating fundamental gaps in understanding physical processes that govern the soil erosion at the local scale. Currently, he leads a research effort on hydrological water quality monitoring and modelling at the watershed scale, with a particular interest in water quality assessment at the regional and global scale.

**Makram Anane** received his PhD from Lérida University, Spain, in 2004, on remote sensing applications for agriculture. His current research interest includes assessment and mapping of the impact of irrigation on soil and groundwater, conception and building geodatabases on water and wastewater, application of GIS, multi-criteria decision analysis and remote sensing for agriculture and water management.

**Salah Jellali** served as a full professor in sciences and engineering for water at the Wastewaters and Environment Laboratory of the Water Research and Technologies Centre (CERTE), Tunisia. He is actually a member of the Chair for the development of industrial estates and free zones at the Centre for Environmental Studies & Research, Sultan Qaboos University, Oman. He obtained a dual Engineering diploma from the high school of rural equipment of Medjez El Bab (Tunisia) and from the National School for Water and Environmental Engineering, Strasbourg, France and MSc (1996) and PhD (2000) degrees in Water and Engineering for water from the University of Strasbourg, followed by a postdoctoral position and a brief experience in BURGEAP, Strasbourg. His research interests include wastewaters treatment by low cost materials, nutrients recovery from wastewaters and reuse in agriculture, local water management, and groundwater flow and pollutants transport modeling for a sustainable management.

## Results and discussion

### *Physicochemical elements*

The mean pH was about 7.4, varying between 6.86 and 7.86. With the exception of the discharge of the industrial area of Grombalia, all measured pH values fall within the range of values allowed by discharge standard NT-106.02 (i.e., between 6.5 and 8.5). The mean EC and salinity are relatively high (3.23 mS/cm and 4 g/L), indicating that the discharges operated in Wadi El Bey are highly mineralized. The highest values were measured at the discharge of the tannery. This can be explained by the use of several types of salts in their industrial processes such as chromium salts.
The concentration of DO in the samples ranges from 0.43 to 3.96 mg/L with an average value of 2.53 mg/L and a Coefficient of Variation (CV) of 0.49.These low values reflect a fairly steady state of microbiological activity. All measured values of DO are below the threshold acceptable for proper development of aquatic life (4 mg/L). The minimum DO contents were recorded at the sampling points downstream of the Grombalia industrial area and after the tannery discharge. At these two locations, the DO contents were less than 1 mg/L indicating a very critical state in this portion of the river. This oxygen deficiency is mainly caused by high organic matter concentrations in the discharge sources and also by low water-flow velocities. The spatial distribution of DO illustrates a striking heterogeneity between the different sections of the Wadi. At the confluence of Wadi El Maleh with Wadi Ellouza, the quality of the water, in terms of DO, improves slightly (Figure 2).

Almost all sampling points in Wadi El Bey water have turbidity values that exceed the tolerable threshold. Except for waters at some locations like Wadi El Maleh and before the outlet of lagoon El Maleh, these values do not exceed the acceptable limit specified in the Tunisian standards.

The average value of SM was 121.86 mg/L, which exceeds the Tunisian standards (i.e., 50 mg/L) and highlights the high pollutant load discharged into this river and also the fairly slow flow rate. The SM represents all the mineral and organic particles contained in the water in a suspension state. The levels of SM in Wadi El Bey waters vary from 9 to 463 mg/L, which can be rather harmful to the biological diversity of this watercourse, since these levels can cause the decrease of DO and light penetration, thus affecting the photosynthesis process. TDS values vary between 1558 and 5538 mg/L, with an average value of 2934 mg/L. This can be explained by the use of mineral salts in industrial processes, and by the contribution of drainage water, which is quite rich in salts. The surface waters of Wadi El Bey have high concentrations of TDS.

Various measured COD values were much higher than the Tunisian standards (i.e., 125 mg/L) and are indicative of very high pollutant loads discharged in Wadi El Bey. The maximum concentration of COD was measured at the disposal level of the Grombalia industrial area, which is about 16 times higher than the standard. Apart from the samples collected in Wadi El Maleh and downstream of the discharge of agricultural drainage, all other points are characterized by COD values that exceed the acceptable threshold. Similarly, the highest BOD5 values were measured at the Grombalia industrial area. This shows that the pollution contained therein is not subject to biological degradation. The measured values of BOD5 vary between 15 and 400 mg/L. The waters of Wadi El Maleh are the least loaded in terms of BOD5. Concerning Wadi El Bey, its waters are less loaded with BOD5 compared to the other streams, which is attributed to the effect of dilution.

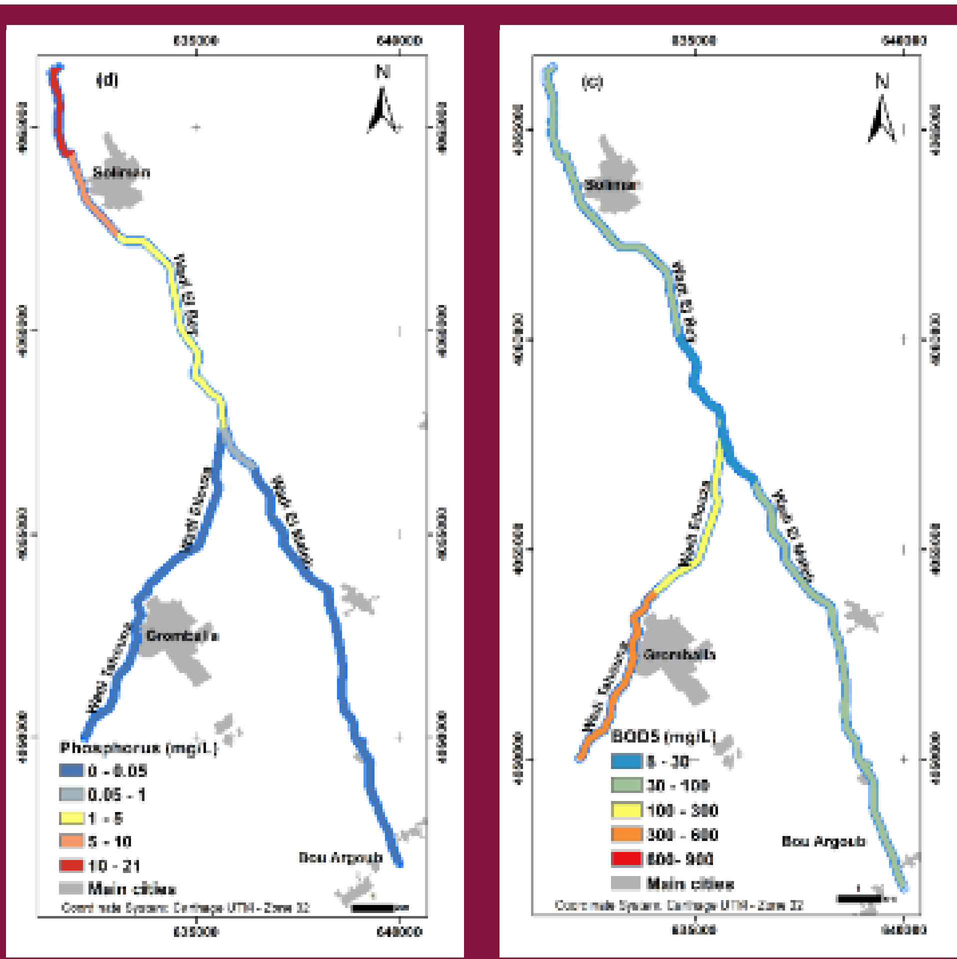



Figure 2. Spatial distribution of a) Dissolved oxygen, b) Turbidity.

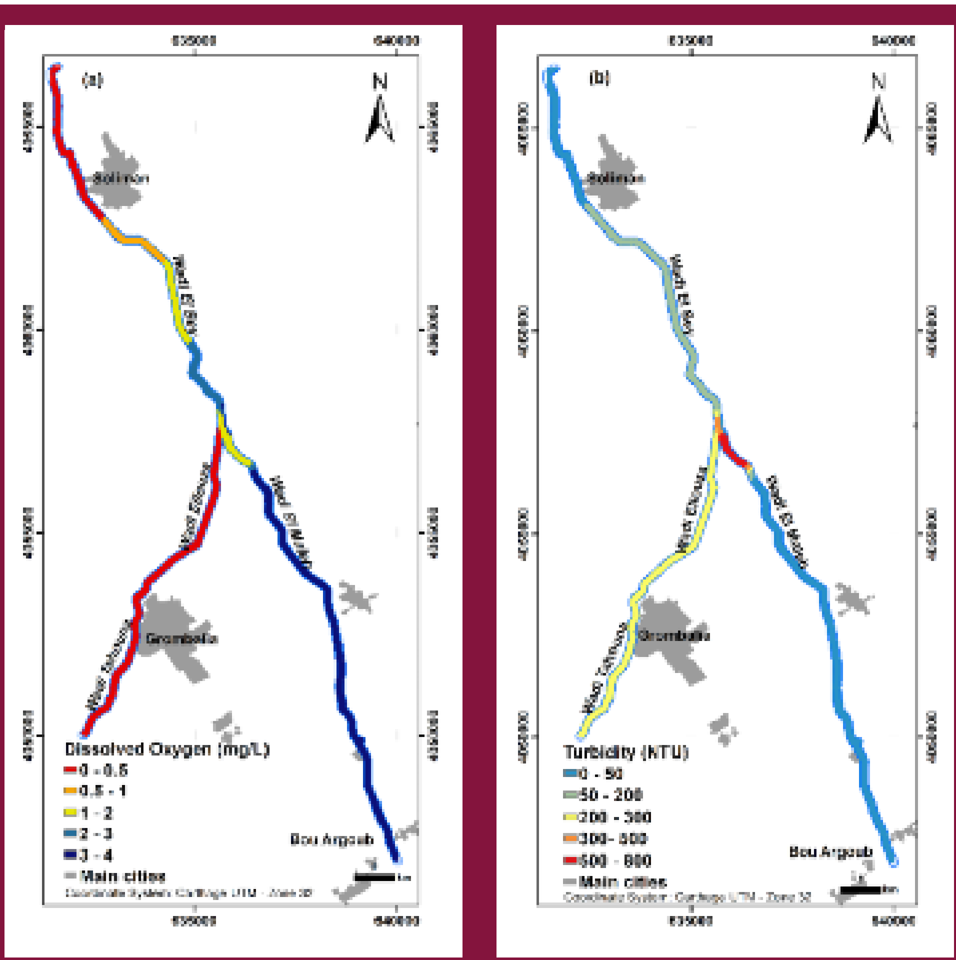
### *Major elements*

Wastewater discharges have high levels of chlorides ranging from 624 to 4005 mg/L. This latter value exceeds eight times the Tunisian standards and corresponds to the discharged wastewater by the Grombalia tannery. These high levels are mainly due to the use of NaCl in the tanning process. The industrial zone of Grombalia also discharges water with chloride content about three times higher than the corresponding standard. Wadi El Bey is characterized by an average sulphate concentration of 518.9 mg/L. The surface water of the Wadi El Bey watershed shows, also, chlorides contents that vary from 622.6 mg/L at Wadi El Maleh to 2038.6 mg/L at the discharged wastewaters from the tannery.

Orthophosphate concentrations, however, were negligible in all sampling points. Except at the outlet of Wedi El Bey Watershed, this value, even if it is relatively low in absolute, exceeds by sixteen times the Tunisian standard (i.e., 2 mg/L). With the exception of sites substantially rich in phosphates, the rest of the water samples have low or negligible concentrations. Similarly, when approaching the lagoon of Soliman, phosphate concentrations continue to increase to reach relatively high values. This could lead to eutrophication and the intense proliferation of algae in this lagoon. The high phosphate concentrations are attributed to the presence of agricultural activity on both sides of this river, where the use of nitrogen fertilizers is predominant [5].

The concentrations of nitrate ions are relatively low, varying between 1.16 mg/L and 49.74 mg/L, with an average value of 8.10 mg/L, complying with the Tunisian standard (i.e. 90 mg/L). The corresponding CV exceeds 1.0, indicating that there is a disparity in concentrations between sampled points. This disparity is reflected in the sensitivity of nitrates to the physical and chemical conditions of the environment. Virtually, all discharge waters have low nitrate concentrations. With the exception of agricultural tile drain, the measured nitrate content is about 179 mg/L. It can be explained by the agricultural activities based on the excessive use of nitrogen fertilizers in the Bou Charray agriculture area. The spatial distribution of nitrate levels shows characteristic surface water values but is also low compared to levels found in other studies [6]. The spatial distribution of nitrates along Wadi El Bey and its tributaries shows that the water sampled in the various streams has acceptable levels of nitrates.

Conversely, the ammonium ion contents are relatively high compared to the corresponding discharge standard. Several sampling points do not comply with the Tunisian Norm (i.e. 10 mg/L). All water discharged in Wadi El Bey has magnesium levels that exceed the discharge standard. As for nitrates, the highest magnesium values are recorded in the drainage network of the agricultural area of Bou Charray. Sodium levels in Wadi El Bey waters vary in a heterogeneous way, from a minimum of 300.79 mg/L at Wadi El Bey downstream to a




c) BOD5 and d) phosphorus levels in sampled waters

maximum of 913.5 mg/L at the sampling point after discharging of the tannery. This last maximum value is three times higher than the Tunisian standard.

***Relationship between pollutants transport and groundwater processes***

It is essential to underline that the attenuation of the pollution in Wadi El Bey downstream of the Grombalia watershed could be directly linked to the infiltration of pollutants through the vadose zone to the shallow groundwater[7]. This pollution transport/transfer is mainly due to the physicochemical characteristics of both the pollutants and the porous media. For instance, pollutants with small molecular size such as nitrates are generally very mobile and are not retained significantly even by soil layers rich in an organic matter[8]. However, heavy metals discharged by industrial effluents (including tanneries) could be easily and significantly adsorbed by soil layers which reduce the groundwater pollution risk[9]. On the other hand, the soil layers properties, mainly hydraulic conductivity, porosity and organic matter content, as well as oxides, could considerably influence the degree of pollutant transport to the groundwater [10]. Indeed, homogeneous soils with high hydraulic conductivity and porosity facilitate the transport of pollutants from the wadis to the shallow groundwater due to the low contact time between pollutants molecules and soil particles [11]. On the other hand, the industrial pollution discharging at specific locations could enhance the formation of a biofilm that could significantly reduce the organic pollutants transport to the groundwater [12].

## Conclusions

This work demonstrates that the wastewaters discharged by the industrial zone of Grombalia and specifically the tannery induce a serious pollution of various Wadis of Grombalia region. The main affected parameters concerned the COD, the BOD5 and the TDS contents. Moreover, the tile drain network in the agricultural area of Bou Charray has significantly increased nitrates and phosphorus content due to the use of synthetic fertilizers. Water samples analyses have shown that natural attenuation of the Wadi El Bey pollution occurs while moving downstream. This finding might be mainly the result of a dilution process and especially infiltration into the underground environment, which represents, a real danger to the shallow groundwater. The preservation of the water quality of Wadi El Bey and its tributaries depends on the improvement of the water quality discharged by the industrial zone of Grombalia and also the tannery operating in this city. Setting up WWTPs specific to the nature of the generated pollution inside these two industrial sites would be the best solution to tackle this pollution problem. Continuous monitoring of the pollution using specific sensors (i.e., [13]) by adopting the high frequency monitoring approach at these identified different pollution hotspots but also along the longitudinal direction of the Wadi flow will make it possible to better assess the spatial and temporal variation of the physicochemical quality of the Wadi and its impact on the Gulf of Tunis and groundwater system.

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## References

[1] UNEP-MAP, UNESCO-IHP (2015). Final report on Mediterranean coastal aquifers and groundwater including the coastal aquifer supplement to the TDA-MED and the sub-regional action plans. Paris: Strategic Partnership for the Mediterranean Sea Large Marine Ecosystem (MedPartnership). http://unesdoc.unesco.org/images/0023/002353/235306e.pdf
[2] European Union, 2016. Alternative assessments of large scale Eutrophication using ecosystem simulations: hind-casting and scenario modelling, Edits: Stips Adolf, Macias Moy Diego, Garcia Gorriz Elisa, Miladinova-Marinova Svetla. DOI: 10.2788/156650
[3] COMETE Engineering-BCEOM-IHE Group (2008) Ministry of the Environment and Sustainable Development-Pre-investment study on the cleanup of the golf course of Tunis.
[4] Order of the Minister of Local Affairs and the Environment and the Minister of Industry and Small and Medium Companies (OMLAE-MISMC), 26 March 2018. The limit values for discharges of effluents into the receiving environment.
[5] Mlayah, A. E. Ferreira da Silva, F. Rocha, A. Charef and F. Noronha, 2009. The Oued Mellegue: Mining activity, stream sediments and dispersion of base metals in natural environments, North-western Tunisia. Journal of Geochemical Exploration 102(1):27-36. DOI: 10.1016/j.gexplo.2008.11.016.
[6] Hamdi N. (2009). The watershed of El Bey Wadi: State of pollution and preliminary implications on the quality of the waters of the Grombalia aquifer. Master Thesis, Faculté des Sciences de Tunis, 91p.
[7] Khadhar, S., Mlayah, A., Chekirben, A., Charef, A., Methammam, M., Nouha, S., and Khemais, Z., 2013. Transport of heavy metal pollution from the Wadi El Bey basin toward the Tunisian Gulf. Hydrological Sciences Journal, 58 (8), 1803-1812. 1
[8] Ma, Z., Lian, X., Jiang, Y., Meng, F., Xi, B., Yang, Y., Yuan, Z., Xu, X. (2015). Nitrogen transport and transformation in the saturated-unsaturated zone under recharge, runoff, and discharge conditions. Environmental Science and Pollution Research, 2016, 23(9):8741-8.
[9] Mlayah, A. Jellali, S. (2015): Study of continuous lead removal from aqueous solutions by marble wastes: efficiencies and mechanisms. International Journal of Environmental Science and Technology. 12, 2965–2978
[10] Gawdzik, J., Sygaldo, M. (2010). Modelling transport of hydrocarbons in soil-water environment. Ecological Chemistry and Engineering, 17(3), 331-343.
[11] Jellali, S., Sediri, T., Kallali, H., Anane, M., Jedidi, N. (2009): Analysis of hydraulic conditions and HRT on the basis of experiments and simulations on soil column. Desalination, 246, 435-443
[12] Mitra, A., Mukhopadhyay, S. (2016). Biofilm mediated decontamination of pollutants from the environment. AIMS Bioengineering, 3(1): 44-59
[13] Rode, M., Wade, A.J., Cohen, M.J., Hensley, R.T., Bowes, M.J., Kirchner, J.W., Arhonditsis, G.B., Jordan, P., Kronvang, B., Halliday, S.J., Skeffington, R.A., Rozemeijer, J.C., Aubert, A.H., Rinke, K., Jomaa, S., (2016). Sensors in the stream: the high-frequency wave of the present. Environmental Science and Technology, 50 (19), 10297-10307.

# REMEDIATION STRATEGIES FOR NONAQUEOUS PHASE LIQUID (NAPL) CONTAMINANT SOURCES IN GROUNDWATER: INNOVATIONS AND CHALLENGES

BY GEOFFREY R. TICK

Over the last several decades, a wide variety of groundwater remediation technologies have been developed to remove or reduce sources of aquifer contamination such as nonaqueous phase liquids (NAPL). This article provides an overall review of some groundwater remediation strategies that have proven to show promise for NAPL source removal and reduction including enhanced solubilization and mobilization (i.e. surfactants, complexing sugars, and cosolvents) and in-situ destruction and stabilization (i.e. oxidation, reduction, and composition modification). Although numerous studies have shown these technologies to be quite successful for contaminant source reduction and groundwater remediation, most have been shown to be ineffective at complete source-mass removal and reducing aqueous-phase contaminants of concern (COC) concentrations to levels suitable for site closure.

## Introduction

NAPL, such as chlorinated solvents, fuels, and coal tars, are a major cause of groundwater and soil contamination. In fact, their presence is the most important factor inhibiting suitable risk assessment, characterization, and cleanup of most hazardous waste sites. Due to low aqueous solubilities, trapped residual NAPL mass can persist in the subsurface as a long-term contamination source, producing aqueous concentrations several orders of magnitude greater than maximum contaminant levels (MCL) for drinking water. Traditional groundwater extraction (i.e. pump-and-treat) methods are generally inefficient at removing contaminant source mass, and therefore remediation typically requires extremely long times to reduce groundwater concentrations to acceptable and/or regulatory levels (i.e. reach MCLs) [1,2,3,4, 5,6]. Under such scenarios, the risk to human health from toxic levels of contaminants will persist and site closure may not be feasible.

In order to deal with such challenges for effective aquifer remediation, several strategies have been developed to either enhance the removal of NAPL contaminant sources, transform or destroy the source in situ, and/or stabilize/contain or modify the source, all of which act to decrease (minimize or eliminate) contaminant mass flux to groundwater. Hence, such contaminant source reductions can further aid in allowing natural attenuation processes to occur more effectively (e.g., dilution, dispersion, microbial/biological degradation or uptake, etc.) by reducing contaminant concentrations in the environment and thus minimizing human health risks through exposure to contaminated groundwater.

## Remediation Technologies when NAPL Sources are Present

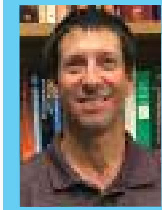
### *Enhanced Flushing*

Generally, enhanced flushing technologies rely on increasing NAPL mass removal over traditional flushing approaches via enhanced-solubilization, enhanced-mobilization, or facilitated transport processes. Enhanced-solubilization works by injecting and flushing an aqueous reagent into the NAPL contaminant source area to increase mass-transfer (dissolution) of the NAPL phase into the water phase via molecule complexation or micellar partitioning (e.g., complexing sugars, surfactants) or polarity modification of the flushing solution (e.g., cosolvents) to yield higher mass removal from the aquifer while simultaneously reducing (eliminating) source mass. Such mass transfer processes may be rate-limited (kinetically controlled) thereby decreasing removal efficiency on a per-time basis but pose less risk for NAPL (especially dense NAPL, DNAPL) mobilization and escape from remediation control. Enhanced-mobilization works by injecting and flushing a reagent (e.g., cosolvent or surfactant) that acts to reduce the interfacial tension between the NAPL, water, and solid phases within the aquifer, increasing displace-ment and capture (extraction) of the NAPL phase directly. This process reduces source mass transfer to the water phase by essentially extracting the pure NAPL source from the aquifer. Although this is generally a faster removal process with less mass-transfer constraints, greater risk of losing control of the NAPL (especially DNAPL that sinks in water) may occur. More details on enhanced solubilization and enhanced-mobilization can be found in [7,8,9,10,11,12,13,14,15].

Facilitated transport works by injecting and flushing a reagent solution with nanoparticles, colloids, and/or macromolecules that are surface reactive (e.g., minerals, zeolites, humic and fulvic acids, etc.) with the targeted chemicals of concern (COC source or plume) in groundwater. Ideally, the targeted COC will preferentially partition, react, or complex with the remedial reagent, increasing NAPL dissolution and desorption from aquifer materials in the water phase, yielding higher contaminant mass removal from the aquifer via extraction while simultaneously reducing (eliminating) source mass.

### *In Situ Destruction/Degradation/ Transformation and Stabilization*

Generally, in-situ destruction/degradation/transformation based remediation technologies rely on destroying or altering the contaminant source (i.e. NAPL in this case) in place without the need to extract contaminant mass from the subsurface. These in-situ destruction techniques typically involve injecting an amendment or reagent into or near the source zone (contaminated NAPL region) whereby oxidation, reduction, and/or biomineralization reactions occur to destroy or degrade the NAPL or dissolved phase plume emanating from the NAPL source itself. Strong oxidants such as potassium permanganate, hydrogen peroxide, or persulfate can react with NAPL or dissolved phase contaminant to break contaminant bonds yielding transformation products that are ideally benign or present much lower toxicity hazards in the environment [16,17,18,19,20,21]. Such technologies can generate unfavorable byproducts or precipitates that can clog or



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isolate the contaminant sources under certain circumstances. Another in-situ technique, not typically used to directly target NAPL but can reduce plume or source concentration, is enhanced bioremediation where an amendment is injected into the contaminated zone to transform subsurface redox conditions that can induce oxidative or reductive biodegradation (biomineralization) of the targeted COC to benign or less toxic compounds (e.g., co-metabolism, co-respiration, reductive/anaerobic dechlorination, or oxidative/aerobic degradation).

In-situ source stabilization remediation also focuses on source treatment without mass removal. Typically used for metals and inorganics, there have been few methods developed specifically for organics including NAPL. These techniques rely on the modification of the source to decrease contaminant mobility, availability, and mass flux to groundwater [22,23]. Unlike enhanced-flushing in which source mass flux is increased for greater mass removal via extraction, in-situ source stabilization techniques focus on decreasing COC concentrations and mass flux to groundwater by reducing source accessibility and aqueous permeability (via an increase in NAPL saturation) [24]. Such in-situ source stabilization relies on injecting an amendment into the contaminant source or source area (NAPL) to modify the NAPL composition (i.e. reduce COC mole fraction) while also creating a corresponding increase of NAPL saturation ($S_N$) within the source (decrease hydraulic permeability and isolate the source) in order to reduce COC mass flux, aqueous concentrations, and availability to groundwater. In other words, by injecting amendment into the NAPL source and surrounding zones, mixing of amendment into the contaminant source reduces the contaminant source ratio (i.e. mole fraction) while also creating higher amendment/contaminant mixture saturation, effectively reducing water permeability (of source zone) and limiting the contact of flowing groundwater with the source itself. This condition acts to reduce the contaminant mass discharge to groundwater, ideally allowing concentrations to be low enough by which natural attenuation processes can reduce the contaminant below regulatory action levels (i.e. MCL) and pose negligible risk to human health. The effectiveness of such techniques relies on targeted and uniform mixing of the amendment into and with the contaminant source (i.e. NAPL) [24,25,26]. It should be noted that most studies have only been tested at small-scale laboratory or pilot-scale field scenarios.

## Challenges and Limitations of Source Remediation Techniques

Although such NAPL source removal/reduction techniques have demonstrated great promise in reducing source mass and contaminant concentration/exposure in groundwater, studies have shown that complete NAPL source removal from the subsurface (groundwater or vadose zone) is extremely difficult and not generally possible. An overview study demonstrated that such NAPL source removal technologies were only effective for removing a fraction of the total NAPL source mass, ranging from ~40-90% mass removal for most flushing based methods [15]. In general, the NAPL mobilization (i.e. surfactant and cosolvent mobilization) techniques showed the most effective percent mass removal achieving up to 98% recovery under the particular conditions of the pilot-scale field demonstrations. Overall, the partial mass removal/reduction associated with these source remediation techniques may still leave significant mass in the subsurface for mass-flux and release to groundwater (and/or vadose zone), presenting potential health risks to humans via drinking water, groundwater or vapor exposure.

### *Enhanced Flushing*

Related literature has indicated that primary limitations for such enhanced flushing technologies are due to NAPL source accessibility, appropriately locating and targeting source zones, and rate-limited mass transfer constraints that are, in part or collectively, responsible for incomplete mass removal. Under such strategies, the most hydraulically accessible NAPL (source) mass is removed first and fairly efficiently, with zones of poorly-accessible mass (isolated from flushing) still remaining in the subsurface (i.e. due to bypass flow, preferential flow, and size exclusion effects) (Figure 1). Figure 1 (A and B) shows the results of pilot-scale field demonstrations of both complexing sugar (hydroxypropyl-β-cyclodetrin, HPCD) and surfactant (sodium dihexyl sulfosuccinate, AMA) enhanced flushing for DNAPL (PCE) source removal at the Dover National Test Site (Dover Air Force Base, Delaware) [15]. Both flushing scenarios (Figure 1A and 1B) show PCE concentrations declining while respective solubilization-enhancing amendment concentrations remain constant over time, indicating that a significant portion of DNAPL mass becomes hydraulically inaccessible to amendment as flushing progresses. Only 48% and 65% DNAPL mass removal was achieved for the complexing sugar and surfactant flushing applications, respectively, suggesting that significant inaccessible DNAPL mass remained hydraulically isolated from the flowing amendment.

In some cases, amendments such as surfactants and cosolvents can reduce interfacial tension of NAPL-water or NAPL-air interface and result in mobilization and escape of control of the NAPL phase source in the aquifer or vadose zone. This "escape of control" for remediation is of highest concern for DNAPL, which can sink

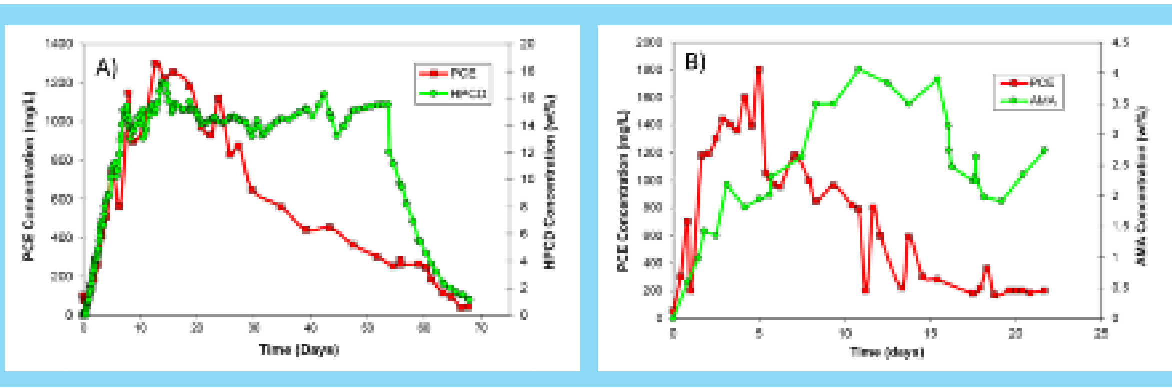



Figure 1. Pilot-scale field demonstrations of enhanced flushing for tetra-chloroethene (PCE) DNAPL source removal at Dover National Test Site, Dover Air Force Base, Delaware [15].

and spread further into the aquifer. Also, injecting reagents/amendments with varying viscosity/density can reduce (or clog) flow through the aquifer and into the targeted source-zone areas or escape the source area via unintended buoyancy or sinking effects. If source mass still remains after remediation operations, concentration rebounding (increase in aqueous phase contaminant concentrations) is likely to occur in groundwater or the vadose zone (Figure 2). Figure 2 (A and B) shows results of an in situ chemical oxidation (potassium permanganate) demonstration for the remediation (destruction/transformation) of trichloroethene (TCE) in an aquifer in which TCE mass discharge is significantly reduced pre- and post-treatment (Figure 2A), a favorable condition for treatment [20]. However, after the termination of permanganate injection TCE concentration rebound can be observed, indicating that complete TCE mass removal (destruction) was not achieved through the in situ oxidation ($KMnO_4$) process and it likely that TCE mass was hydraulically inaccessible to the permanganate injection solution (Figure 2B) [20].

*In Situ Destruction/Degradation/ Transformation and Stabilization*

Similar to enhanced flushing technologies, effectively delivering amendment/reagent to source zones is a primary challenge for such in situ destruction/degradation/transformation and stabilization techniques. Preferential flow paths and related bypass flow effects around and away from contaminant NAPL source zones can limit desired amendment/reagent delivery to the source, thereby limiting remediation effectiveness. Another potential issue with in situ NAPL source remediation is the potential to create precipitation products that can reduce or alter the source-zone permeability by clogging some of the pores of the aquifer material, so that amendment delivery cannot effectively reach the intended contaminant source. For instance, use of strong oxidants such as potassium permanganate can create the formation of manganese oxides that clog pores or isolate the NAPL source zone from further treatment [16,19] (Figure 2). Related concerns with in situ amendment delivery can include changing the environmental aqueous conditions within the subsurface (e.g., alter pH and redox conditions) initiating unintended consequences such as solubilization or enhanced release of other contaminants (i.e. metals, etc.) into groundwater. When using in situ oxidation remediation techniques (permanganate, persulfate, peroxide, etc.) it is also important to consider constituents other than the COC in the aquifer (or vadose zone) that may be oxidized (react with reagent) thereby depleting the reactivity of the desired injected reagent, leading to less efficient overall remediation (destruction) of the intended contaminant source. In other cases, transformation or incomplete transformation of the COC to more toxic compounds may occur and such effects should be considered when applying such techniques. For instance, enhanced bioremediation or oxidation methods may not completely mineralize the COC, yielding more toxic intermediates in the degradation pathway process leading to continued exposure and contamination of groundwater.

In situ stabilization techniques have unique challenges to achieve optimal reduction of COC concentrations in groundwater, in addition to many of the same challenges that exist with locating the contaminant (NAPL) source zone and delivering the stabilizing amendment/ reagent to the source zone (as mentioned above) [24]. For this technology to be optimally effective, uniform mixing of stabilization amendment into the source is desired, so that the COC mole fraction is reduced in the amended NAPL mixture thereby reducing mass flux of the COC (and aqueous phase concentration) to groundwater. Additionally, the injected stabilization amendment should remain relatively immobile with the source zone region, increasing localized immiscible-liquid/NAPL source saturation that acts to hydraulically isolate the source zone from flowing groundwater while simultaneously optimally reducing COC mole fraction in the source (Figure 3). Figure 3 shows the results of in situ NAPL stabilization to reduce the contaminant of concern (COC), TCE, concentration and mass flux from the source to groundwater [24]. Injection of different volume ratios of stabilization amendment demonstrate that vegetable oil ideally reduced COC concentration and mass flux, in addition to remaining more uniformly mixed and immobile within the source zone, compared to hexadecane stabilization amendment (Figure 3A). It can be observed that concentration (i.e. mass flux) of COC rebounds

**Figure 2. Field remediation of TCE source zones using in situ chemical oxidation (ISCO) with potassium permanganate at the TIAA Superfund Site, Tucson, Arizona [20]**

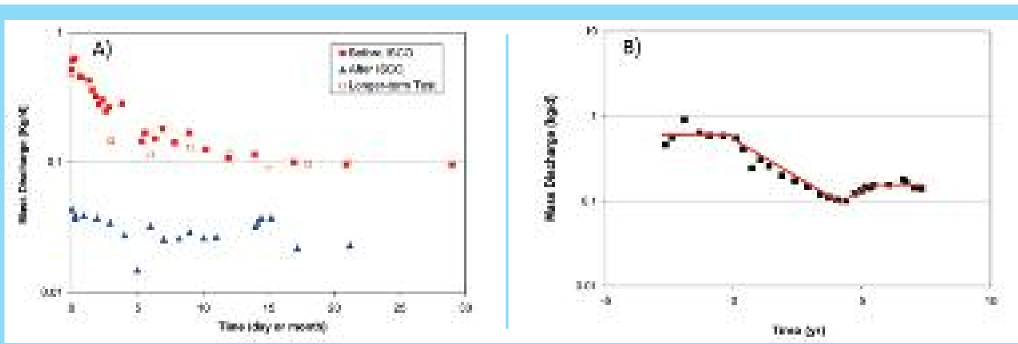
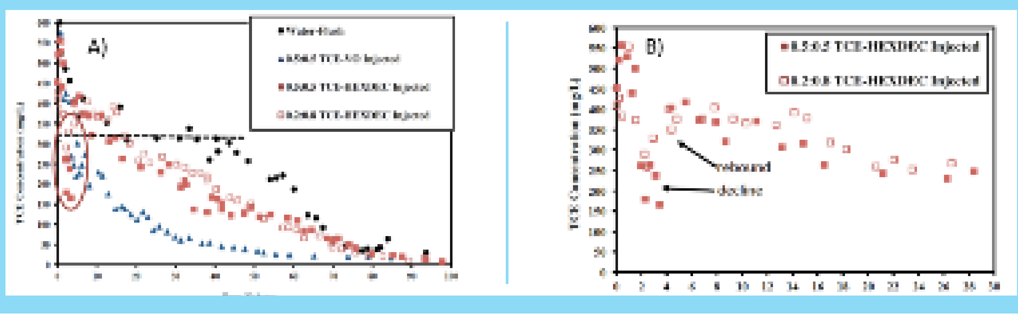



**Figure 3. In situ NAPL stabilization (hexadecane and vegetable oil) study to reduce TCE concentration and mass flux from NAPL sources to groundwater [24]**



| Primary Challenges When Implementing Source Flushing and In-Situ Remediation Techniques | |
|---|---|
| 1. Finding and targeting sources and source zones | 10. Minimizing the degradation of amendment/reagent |
| 2. Effective amendment/reagent delivery for contact with source | 11. Rate limited mass transfer constraints (flushing) causing low-concentration tailing and inefficient mass removal |
| 3. Altering permeability field and enhancing bypass flow | 12. Unintended consumption of reagents (oxidants, reductants) and limited activators |
| 4. Accessible vs. poorly-accessible source mass (hydraulically) | 13. Preferential flow and clogging (colloids and macromolecules) |
| 5. Clogging and/or precipitation of byproducts within the subsurface | 14. Altering pH or other environmental factors (aqueous chemistry and redox) leading to unintended release or precipitation of other contaminants |
| 6. Restriction of flow and flushing due to increased amendment viscosity | 15. Uncontrolled mobilization of the source leading to escape of control for remediation and removal |
| 7. Escape of amendment/reagent control due to variations in solution viscosity/density | 16. Incomplete mass removal and concentration rebounding effects |
| 8. Uniform mixing of stabilization amendment into source | |
| 9. Maintaining immobilization of stabilization amendment in the source zone | |

Table 1. Challenges to consider when implementing Source Zone Remediation Techniques.

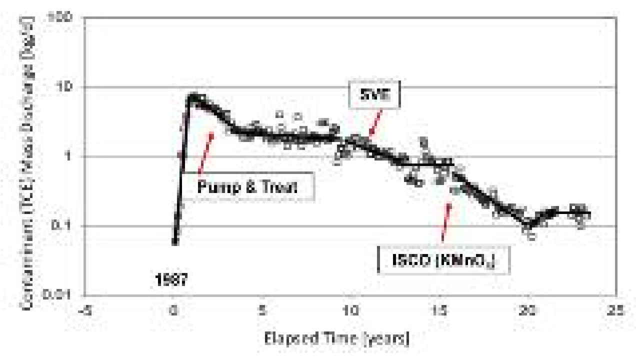



Figure 4. High resolution temporal data set of TCE mass discharge for combined extraction effluent during full-scale aquifer remediation at Tucson International Airport Authority (TIAA) Superfund site, Tucson, Arizona [27].

in groundwater when the hexadecane stabilization amendment mobilizes away from the NAPL source, a less ideal condition associated with this stabilization amendment (Figure 3B). These combined effects result in minimizing COC mass flux and concentration to the aquifer. However, challenges exist in selecting appropriate stabilizing amendments (uniform mixing and immobile in source) and effectively targeting source zone in the subsurface [24].

## Summary - Primary Limitations for NAPL Source Remediation

Whether implementing enhanced flushing and removal, in-situ destruction/transformation or stabilization techniques for NAPL remediation, some primary challenges must be considered are included in Table 1.
Primary Challenges When Implementing Source Flushing and In-Situ Remediation Techniques

## Conclusions

A wide variety of groundwater remediation technologies have been developed to remove or reduce NAPL source mass or decrease COC mass flux from such sources in groundwater and vadose zones. Enhanced flushing, in situ destruction/degradation/transformation and stabilization applications have shown promise for source remediation demonstrating significant improvement over traditional pump-and-treat methods. However, when implementing such source remediation techniques it is important to understand the limitations for source removal/depletion and that even under relatively ideal conditions, generally only partial mass removal/destruction is achievable. For example, these source remediation methods can lead to initial concentration declines but after operations cease, concentration rebounding effects are often observed indicating that contaminant source mass still exists and will continue to pollute the groundwater or surface water resources [20,27] (Figure 4). Figure 4 shows the results of a full-scale aquifer remediation effort over 25 years, demonstrating significant reductions of contaminant (TCE) mass discharge during aggressive in situ source treatment strategies (soil vapor extraction and chemical oxidation) but limited mass removal (stabilization or even rebound of mass discharge) once injections/treatment is terminated [27]. Such conditions generally indicate that only a fraction of the total source mass was removed and that the remaining source mass (i.e. complexly distributed and or hydraulically isolated from treatments) will continue to contaminate the aquifer. These results suggest that these source remediation methods may need to be modified or combined with other methods (stabilization, biodegradation, natural attenuation, etc.) and/or use other metrics in evaluating conditions necessary for site closure and effective site restoration. ■

## References

[1] National Research Council (NRC), 2000. Research Needs in Subsurface Science. National Academy of Sciences, Washington, D.C.
[2] National Research Council (NRC), 2005. Contaminants in the Subsurface: Source-zone Assessment and Remediation. National Academic Press, Washington, D.C.
[3] National Research Council (NRC), 2013. Alternatives for Managing the Nation's Complex Contaminated Groundwater Sites. National Academic Press, Washington, D.C.
[4] Interstate Technology and Regulatory Council (ITRC), 2002. Regulatory Overview: DNAPL Source Reduction. Facing the Challenge. www.itrcweb.org.
[5] U.S. Environmental Protection Agency (USEPA), 1993. Guidance for evaluating the technical impracticability of groundwater restoration. Interim final In: Directive 9234. Office of Solid Waste and Emergency Response, Washington, DC, pp. 2–24
[6] U.S. Environmental Protection Agency (USEPA), 2003. The DNAPL remediation challenge: Is there a case for source depletion? Expert panel on DNAPL remediation. Co-chairs In: Kavanaugh, M.C., Rao, P.S.C. (Eds.), EPA/600/R-03/143. Office of Research and Development, Washington, D.C.
[7] Pennell, K.D., Pope, G.A., and Abriola, L.M., 1996. Influence of viscous and buoyancy forces on the mobilization of residual tetrachloroethylene during surfactant flushing. *Environmental Science and Technology*, v. 30, no. 4, 1328-1335.
[8] Falta, R.W., Lee, C.M., Brame, S.E., Roeder, E., Wood, L., and Enfield, C., 1999a. In Innovative Subsurface Remediation: Field Testing of Physical, Chemical, and Characterization Technologies, Brusseau (Chapter 8), ed. M.L. Sabatini, D. Gierke, and J. Annable, 102–117. Washington DC: American Chemical Society.
[9] Falta, R.W., Lee, C.M., Brame, S.E., Roeder, E., Coates, J.T., Wright, C., Wood, A.L., and Enfield, C.G., 1999b. Field test of high molecular weight alcohol flushing for subsurface nonaqueous phase liquid remediation. *Water Resources Research*, 35(7), 2095–2108.
[10] Knox, R.C., B.J. Shau, D.A. Sabatini, and J.H. Harwell. 1999. In Field Demonstration Studies of Surfactant-Enhanced Solubilization and Mobilization at Hill Air Force Base, Utah. Innovative Subsurface Remediation: Field Testing of Physical, Chemical, and Characterization Technologies, ed. M.L. Brusseau, D.A. Sabatini J.S. Gierke, M.D. Annable, and A.C.S. Symposium Series, 49–63. Washington, DC: American Chemical Society.
[11] Boving, T.B., and Brusseau, M.L., 2000. Solubilization and removal of residual trichloroethene from porous media: Comparison of several solubilization agents. *Journal of Contaminant Hydrology*, 42, 51–67. doi:10.1016/S0169-7722(99) 00077-7.
[12] Wood, A.L., and C.G. Enfield. 2005. Field Evaluation of Enhanced DNAPL Source Removal, USEPA Technical Report and SERDP Report CU-368, National Risk Management Research Laboratory, Office of Research and Development, U.S. EPA, Cincinnati, Ohio.
[13] Wood, A.L., C.G. Enfield, F.P. Espinoza, M. Annable, M.C. Brooks, P.S.C. Rao, D. Sabatini, and R. Knox. 2005. Design of aquifer remediation systems: (2) Estimating site-specific performance and benefits of partial source removal. *Journal of Contaminant Hydrology*, 81, no. 1–4: 148–166.
[14] Tick, G.R. and Rincon, E.A., 2009. Effect of enhanced-solubilization agents on dissolution and mass flux from uniformly distributed immiscible liquid trichloroethene (TCE) in homogenous porous media. *Water, Air, and Soil Pollution*, 204:315-332, doi: 10.1007/s11270-009-0047-3.
[15] McCray, J.E., Tick, G.R., Jawitz, J.W., Brusseau, M.L., Gierke, J.S., Falta, R.W., Knox, R.C.,Sabatini, D.A., Harwell, J.H., and Annable, M.D., 2011. Remediation of NAPL source zones: Lesson learned from field studies at Hill and Dover AFB. *Ground Water*, 49(5): 727-744, doi: 10.1111/j.1745-6584.2010.00783.x.
[16] Seol, Y., Zhang, H., Schwartz, F.W., 2003. A review of in situ chemical oxidation and heterogeneity. *Environmental and Engineering Geoscience*, 9 (1), 37–49.
[17] Christ, J.A., Ramsburg, C.A., Abriola, L.M., Pennell, K.D., and Loffler, F.E., 2005. Coupling aggressive mass removal with microbial reductive dechlorination for remediation of DNAPL source zones: A Review and assessment. *Environmental Health Perspectives*, Vol. 113(4): 465-477.
[18] Furman, O.S., Teel, A.L., Watts, R.J., 2010. Mechanism of base activation of persulfate *Environmental Science and Technology*, 44, 6423e6428.
[19] Marble, J.C., Carroll, K.C., Janousek, H., and Brusseau, M.L., 2010. In situ oxidation and associated mass-flux-reduction/mass-removal behavior for systems with organic liquid located in lower-permeability sediments. *Journal of Contaminant Hydrology*, 117, 82-93.
[20] Brusseau, M.L., Carroll, K.C., Allen, T., Baker, J., DiGuiseppi, W., Hatton, J., Morrison, C., Russo, A., and Berkompas, J., 2011. Impact of in-situ chemical oxidation on contaminant mass discharge: linking source-zone and plume scale characterizations of remediation performance. *Environmental Science and Technology*, 45, 5352–5358.
[21] Zhong, H., Brusseau, M.L., Wang, Y., Yan, N., Quig, L., Johnson, G.R., 2015. In-situ activation of persulfate by iron filings and degradation of 1,4-dioxane. *Water Research*, 83, 104-111.
[22] U.S. Environmental Protection Agency (USEPA), 2009. Technology performance review: Selecting and using solidification/stabilization treatment for site remediation. In: EPA/600/R-09/148. Office of Research and Development, Washington, D.C.
[23] Interstate Technology and Regulatory Council (ITRC), 2011. Development of Performance Specifications for Solidification/Stabilization. Prepared by The Interstate Technology & Regulatory Council Solidification/Stabilization Team, Washington, D.C. (July 2011).
[24] Mateas, D.J., Tick, G.R., Carroll, K.C., 2017. In situ stabilization of NAPL contaminant source-zones as a remediation technique to reduce mass discharge and flux to groundwater. *Journal of Contaminant Hydrology*, 204, 40-56.
[25] U.S. Department of Defense (USDOD), 2007. In-situ substrate addition to create reactive zones for treatment of chlorinated aliphatic hydrocarbons: Cost and performance report. *Directive ER-9920*, Environmental Security Technology Certification Program (ESTCP), Washington, D.C.
[26] U.S. Department of Defense (USDOD), 2009. Edible oil barriers for treatment of chlorinated solvent and perchlorate-contaminated groundwater: Final report. *Directive ER-0221*, Environmental Security Technology Certification Program (ESTCP), Washington, D.C.
[27] Brusseau, M.L., 2013. Use of historical pump-and-treat data to enhance site characterization and remediation performance assessment. *Water, Air, and Soil Pollution*, 224:1741-1747, doi: 10.1007/s11270-013-1741-8.

# PARTICIPATIVE WATER MANAGEMENT CONTRACTS IN MOROCCO FOR SCARCITY ALLEVIATION: THE GROUNDWATER CONTRACT MODEL

BY DALILA LOUDYI, ABDELMOUNAIM EL HADDARI & AHMED FEKRI

When scarcity becomes the prevailing condition, conflicts become the enforcing rule. This is particularly true in water scarce-countries such as Morocco. In order to alleviate such tensions, some drastic measures have to be implemented for rational sharing and conservation of the scarce resource. The general acceptance of these solutions is pending on effective participative water management processes.

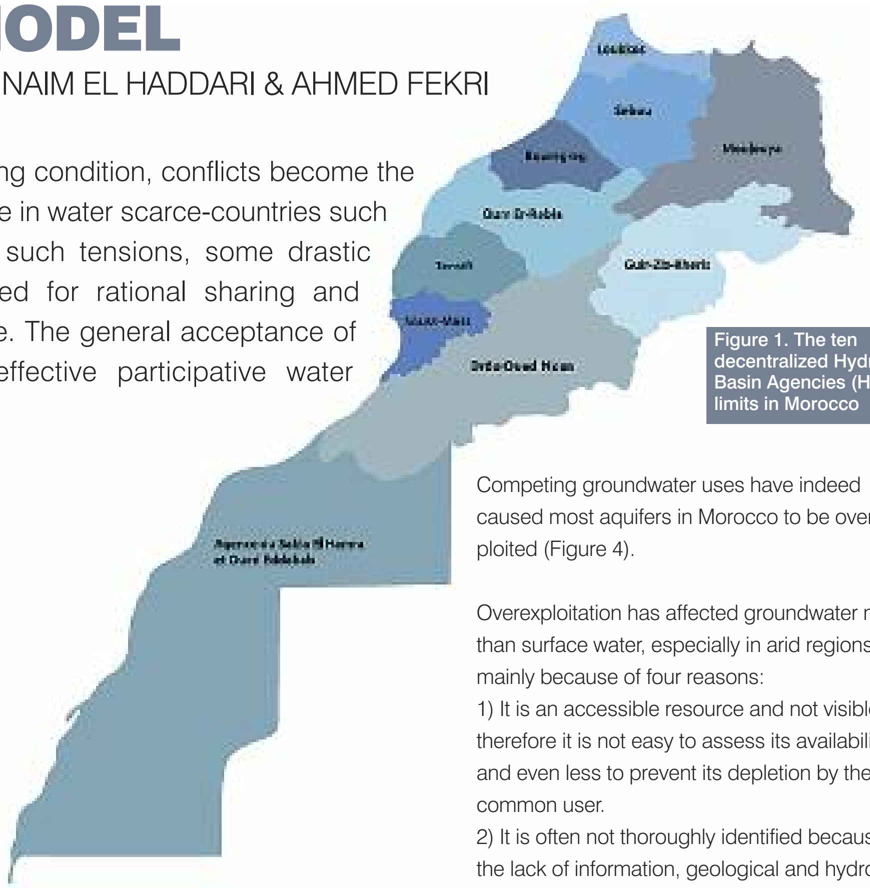


Figure 1. The ten decentralized Hydrulic Basin Agencies (HBA) limits in Morocco

### Morocco water resources facts

The potential of natural water resources in Morocco is estimated on average at 22 billion cubic meters per year. It is the equivalent of nearly 700 $m^3$/year/inhabitant, which is below the threshold of 1000 $m^3$/inhabitant/year, commonly accepted as the threshold below which water scarcity occurs. These resources are distributed unevenly over ten watersheds in Morocco managed by ten Hydraulic Basin Agencies (HBA) representing decentralized water administration (Figure 1).

Surface water resources over the entire territory are estimated on average to more than 18 billion $m^3$/year. Groundwater represents about 18% of the country's water resources potential. The irrigation sector consumes 85% of the mobilized water resources (Figure 2). The National Water Strategy (NWS) and the Moroccan Green Plan (MGP) for agriculture have emphasized the importance of enhancing, conserving and preserving water resources, especially in the irrigation sector. Thus, a global irrigation water saving program, part of the MGP, was launched to reduce by 2030 the demand for irrigation water by 2300 $Mm^3$/year[1].

### Water stress in Morocco and triggered conflicts

By 2050, the available water will drop below 500 $m^3$/year/inhabitant, mainly due to demographic factors and climate change that suggests a decrease of runoff, depending on various scenarios, over the horizons 2020, 2050 and 2080, from 5% to 34% . The expected deteriorating water stress in Morocco, for year 2020, already visible in several regions today, has led to increasing competition for water resources among the various water users, that is articulated in particular between the tourist and agricultural sectors. At local level, existing conflicts between water users have aggravated the depletion of the groundwater table.
There are six major hydrogeological domains in Morocco that contain 130 aquifers (Figure 3). Groundwater provides almost all the needs of rural populations and allows irrigation of 40% of the total irrigated area in the country [2].


Figure 2. Availability and uses of water resources in Morocco

Competing groundwater uses have indeed caused most aquifers in Morocco to be overexploited (Figure 4).

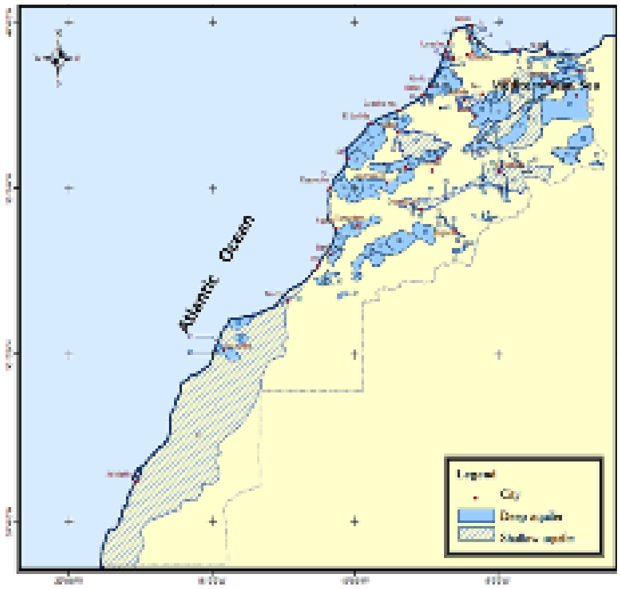
Overexploitation has affected groundwater more than surface water, especially in arid regions, mainly because of four reasons:
1) It is an accessible resource and not visible, therefore it is not easy to assess its availability and even less to prevent its depletion by the common user.
2) It is often not thoroughly identified because of the lack of information, geological and hydrogeological surveys and financial means,
3) Its exploitation control is often beyond the reach and capacity of water authorities as it lays over large territories.
4) Its over-exploitation is often irreversible, either because of lack of aquifer recharge or soil degradation.
The resulting aquifers depletion generates more water conflicts. The case of Souss, a region in the south of Morocco (Figure 1), is particularly dramatic. The scarcity of groundwater resources reached a critical level forcing many small farmers to abandon their lands and migrate to

the city of Agadir in search for better economical livelihoods.

### Emergence of participative water management in groundwater: The aquifer contract

The raise of water conflicts in Souss region urged local authorities to search for a common ground for discussion to manage the scarce resource. In fact, Souss has become the first region to implement participative groundwater management through “the aquifer contract”. The integrated and participative management of Sous-Massa basin groundwater resources was adopted in September 2007 in Agadir city, as a result of a series of meetings bringing together, for nearly a year and a half, the various concerned actors. Indeed, this initiative was prompted because of the pressing conflict that has led to significant migration of farmers who have gave up their agriculture activities and sold their land under urban expansion pressure and conditions of water scarcity. The objectives, the parties, the terms of validity and financing measures of the aquifer contract are shown in Figure 5.

Figure 3. Morocco main aquifers



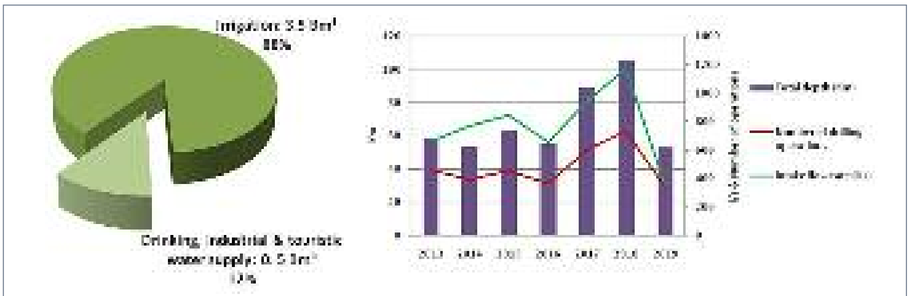




Figure 4. Groundwater uses and overexploitation in Morocco

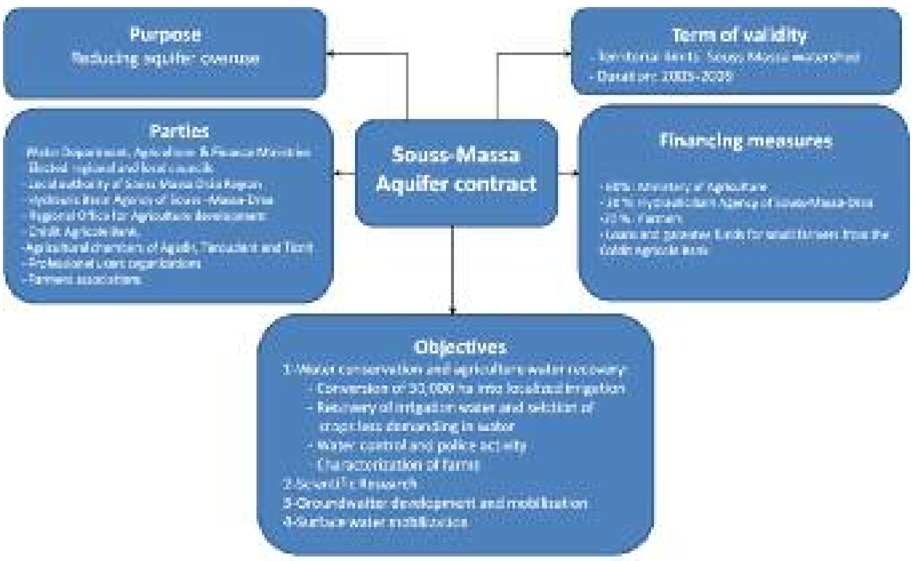



Figure 5. Souss-Massa aquifer contract elements

Other regions followed the Souss-Massa model with the support of international organizations. In 2014, the HBA of Tensift, based in the city of Marrakech, launched the ‘Haouz aquifer contract’ with the technical support of the German international cooperation agency GiZ within their program for supporting integrated water resources management in Morocco (AGIRE). The same year, the HBA of Sebou, based in the city of Fez, approved two contracts for the ‘Mnasra’ and ‘Drader’ aquifers. The total cost of operations within these contracts is 450 Millions U.S.Dollars. These operations aim to realize water economy of 263 Millions $m^3$ by year 2030 and reduce the annual depletion of groundwater resources within the basin from -100 $Mm^3$ to - 5 $Mm^3$ In Oum Er Rbia basin, the World Bank supported in the years 2015-2017 groundwater management contracts for two aquifers in Tadla. No aquifer contract template exists, but generally, the main steps followed the Souss-Massa model. The contract, established by the HBA, sets, in particular, the action plan, its objectives, its duration, its financing terms, water users rights and obligations, related administrations and various partners. It also lays down the rules and framework enabling water users to participate in the management and control of the aquifer uses.

### Shortcoming of the contract process

In Souss, large farm owners agreed to engage in the aquifer contract in 2006, provided that the State undertakes, among other things, to explore the deep groundwater resources, to continue mobilizing the surface water resources, even though they are already mobilized beyond the ecological minimum, and to consider desalination for the neighboring Chtouka area. Unfortunately, these partial conventions have not been implemented for several reasons. Delays in the legal promulgation of the royalties decree and the delimitation decree created a gap between the reality in the field and the national water authorities.

In fact, the 10-95 water Law implemented in 1995, fundamentally redesigned the management of the resource and supported the

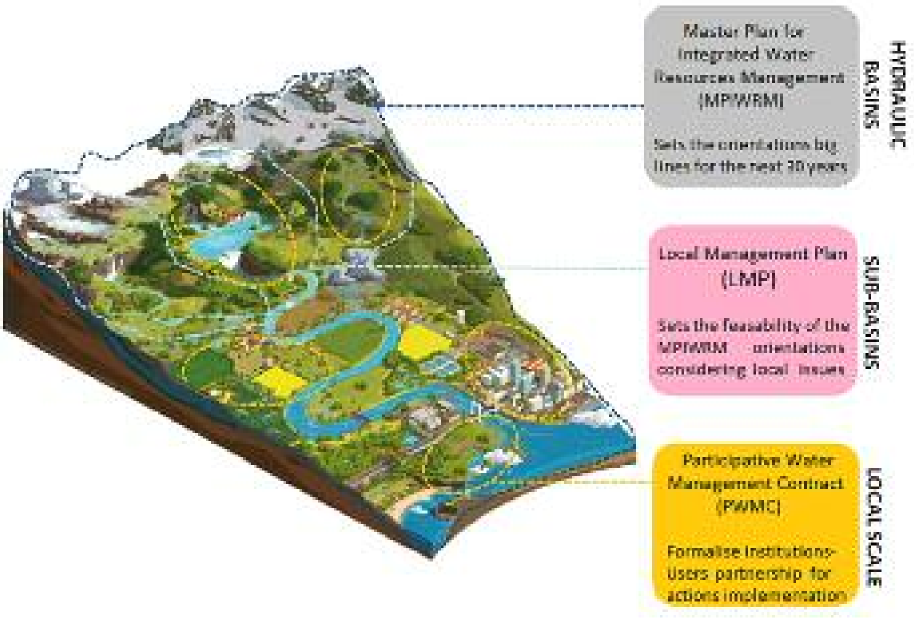



Figure 6. The planning scales of water resources management

decentralization of water management through the creation of seven HBAs initially and the involvement of Agricultural Water User Associations (AWUA) in water management at local levels and maintenance of irrigation systems infrastructure.

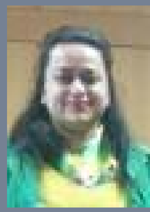
Twenty years after the promulgation of 10-95 Law, the practice of water on the ground has shown that certain aspects of water use and management have not been sufficiently treated or are simply absent in this law. Provisions regarding groundwater conservation, particularly through groundwater contracts, seawater desalination and sewage discharge into the sea were not clearly cited in this law. In fact, groundwater contracts were established as an institutional initiative from the HBAs to ensure the collective engagement of all users despite the absence of a specifically related regulatory framework.

Moreover, experience has shown that the exclusive objective of preserving resources, often based on restrictive measures, does little to promote collective work to bring actors together and ensure their commitment. In short, the aquifer contract failed to create a real dynamic of change at the local level. It did not entice water end-users to fully play their role, participating in the diagnosis of the situation, and in the identification of solutions. As a result, their commitment to actions achievement remains marginal. To make the aquifer contract an effective management tool, it is necessary to strengthen local and regional authorities in water management, but also expand the involvement of users.

## New participative water management contract

To meet the challenges faced by the implementation of aquifer contracts, the new water Law 36-15 introduced in 2016 a new approach to water management through the Participative Water Management Contract (PWMC). This approach focuses on water users and allows for a better consideration of the local scale rather than extending over large hydraulic areas. Instead of considering the issues related to the overexploitation of groundwater resources only, the PWMC governs all water bodies, whether they are surface or ground waters in the public hydraulic domain, taking into account both quantitative and qualitative issues. Water resources users - farmers, households, industrial units, touristic developers, etc - are organized within entities that have their own concerns, policies and approaches. For these users, the interest to participate in water management is mainly to be heard in order to influence the public authorities' decisions, to benefit from technical and financial support and to direct the help of the public authorities to the real needs of users.

The parties' obligations within this contract relate to: 1) rules of access and use of the water resource, 2) control of these rules, 3) measures in case of water crisis, 4) changes of practices and uses, 5) establishment of equipment maintenance 6) application of new pricing, 7) royalties collection and 8) creation of new forms of collective organization.

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The contract may be initiated by the HBA or by the users organizations. The PWMC is required to be submitted to the Hydraulic Basin Council for approval. The HBA provides secretariat resources throughout the process of developing and implementing the contract. It ensures that the PWMC complies with the requirements of the local Master Plan for Integrated Water Resources Management (MPIWRM) and the regulations in force (Figure 6). The subsequent regulatory order of this Law is still being developed by legislators to specify details about the PWMC elements (contents, parties, implementation process, duties and rights of each party, financing options, etc.).

This new governance model relies on the good will of all parties for an efficient implementation based on voluntary engagement. The state is hoping that this type of mechanism will enhance water users' engagement along with local and water authorities for a more equitable sharing and use of water resources, foster social change and make public decisions more democratic. It is guided by the search for a win-win partnership between users and the administration. ■

## References


[1] Moroccan water department, 2015, 'National Plan for Water (PNE)'
[2] Minister in charge of water declaration, National workshop on groundwater management, 26-27 March 2014, Skhirat, Morocco.

# GROUNDWATER MANAGEMENT IN SPAIN: THE CASE OF THE EASTERN MANCHA AQUIFER SYSTEM

BY EDUARDO CASSIRAGA, DAVID SANZ, JUAN JOSÉ GÓMEZ-ALDAY &
J. JAIME GÓMEZ-HERNÁNDEZ

Socio-economic development in the Eastern Mancha during the last 50 years has been made possible thanks to the intensive use of groundwater for irrigation, mostly financed by the end water users. Unfortunately, this intensive groundwater exploitation has created problems, such as the continuing lowering of piezometric levels, and the disconnection of the aquifer from some riverbeds. As a result, groundwater management became an issue with a need to control how and when groundwater could be used. Users, managers and groundwater experts have implemented a series of rules with the aim of reverting the effects of the past 50 years.

The Eastern Mancha aquifer system (EMAS) is one of the largest carbonate aquifers in the southwest of Europe, with a surface of 7260 $km^2$. Located at the Eastern part of Spain, EMAS belongs to the Júcar river water basin (Figure 1).

From a geomorphological point of view, the area is formed by big hollows from the intra-Miocene age, filled with later materials, still maintaining its original disposition, resulting in a flat high plain. This high plain, at 700 masl on average, is broken only by the Júcar river valley, that crosses the system. Surrounding the plain, the relief is smooth, becoming more abrupt, together with complex tectonics, away from it (Figure 1).

In the EMAS, there are outcrops from the Mesozoic, and large Plio-Quaternary deposits. The sedimentary sequence makes a multilayer aquifer with complex interactions between the different aquifer layers. From a hydrogeological point of view, the EMAS is made up of three hydrogeological units separated by aquitards or aquifuges [2]. At the base, there is an impermeable formation of silts, clays and chalks from the lower Jurassic. The most important permeable facies in the system, for their lateral extension and thickness are: i) dolomites and limestones from the Jurassic (partly Dogger) throughout the entire system, working as free aquifer layer towards the outer limits of the system and confined elsewhere, with transmissivities around 10,000 $m^2$/day; the average thickness is 250-350 m, with a maximum value of 400 m; ii) dolomites and limestones from the upper Cretaceous, in the northeastern side, with thicknesses of 50-150 m, as conductive as the previous ones; iii) limestones from the Miocene, in the center, working as a free aquifer layer; their transmissivity is between 1,200 and 7,200 $m^2$/day. All these aquifer units are separated by aquitards or aquifuges from the lower Cretaceous and by detritus rocks from the Tertiary (Figure 2).

The system is laterally contained by impermeable borders with the exception of: i) the northeast border, which is the water divide of the Júcar and Guadiana rivers and which does not coincide with the groundwater water divide; Sanz [3] suggests that the groundwater divide should be located 8 to 10 km to the west with

Figure 1. Plan view of the Eastern Mancha aquifer system with a simplified geological map (modified from [1])

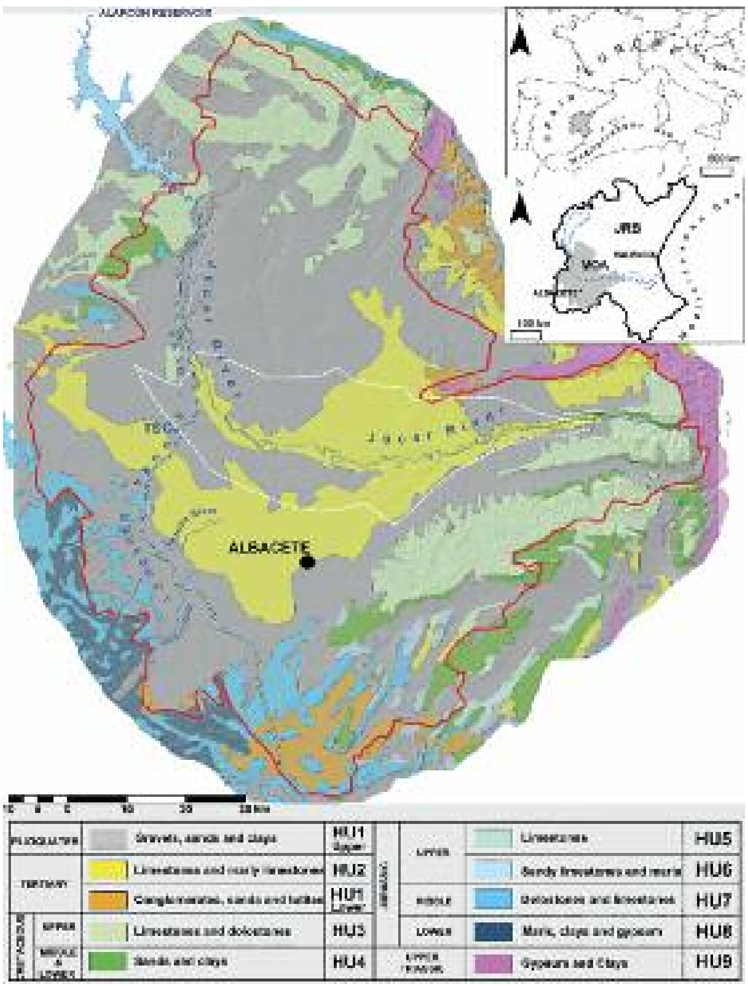


Figure 2. Geological block diagram of the EMAS. Example of piezometric head evolution in one of the control points.

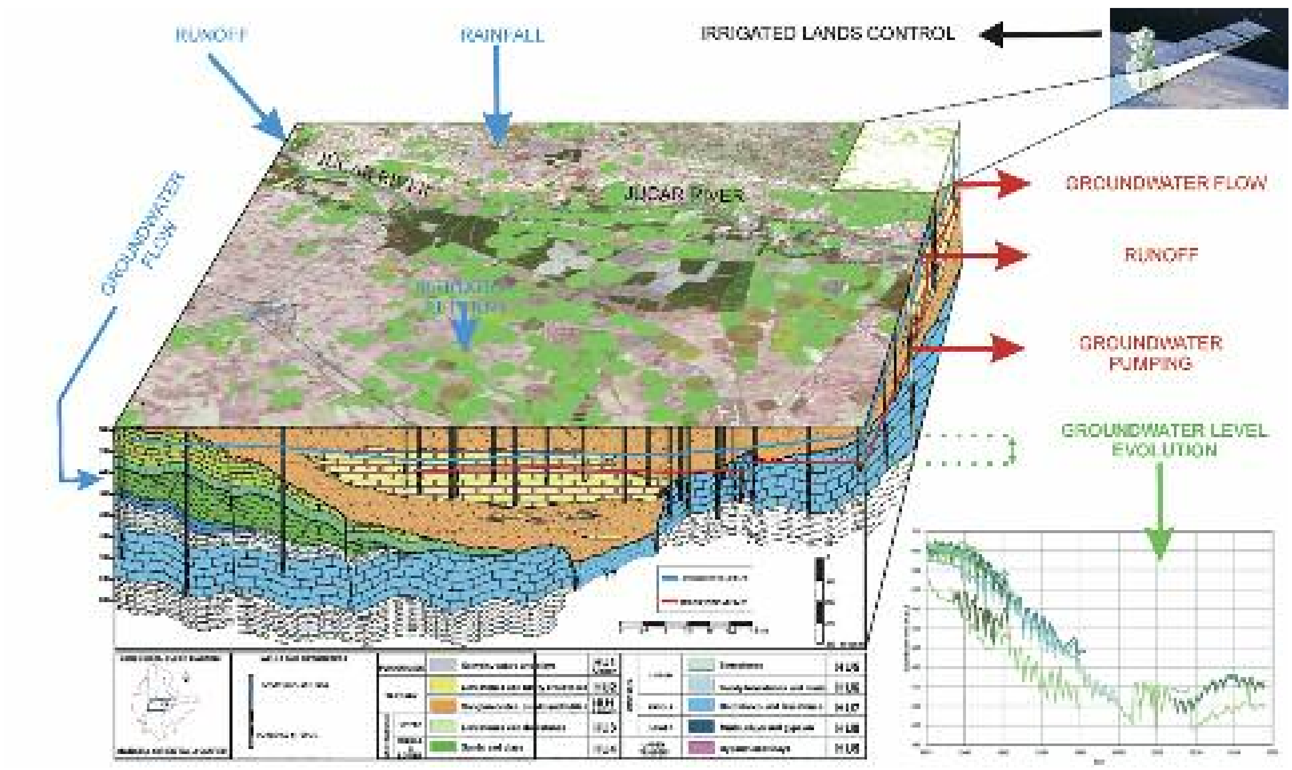



Figure 3. Differential stream-flows between gage stations at two different stretches of the Júcar river. In red observed values, in blue simulated ones.

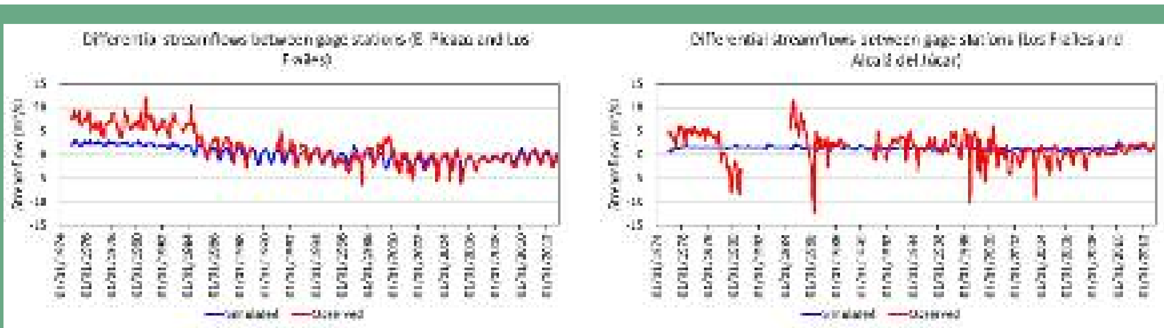
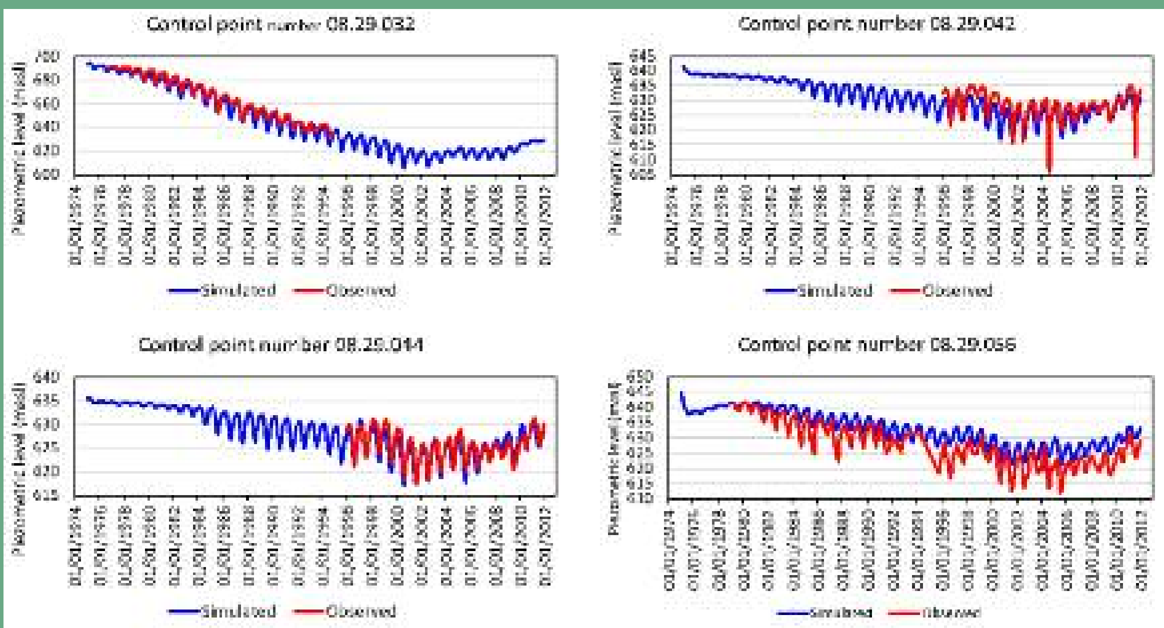



Figure 4. Observed (red) versus simulated (blue) piezometric levels at a few control points.



respect to the surface water divide, and possibly be variable in time; and ii) the southwest border, which is in contact with the aquifers of Jardín-Lezuza and Arco de Alcaraz that provide water to the EMAS. From a water balance point of view, under natural conditions, the main water input would be rain infiltration plus lateral inflow from the southwest aquifers, and the main water output would be the Júcar river.

In the middle of the 1960's, from initial hydrogeological studies, the significance of the EMAS as a water reservoir was recognized, with an estimate of exploitable resources of 350 x $10^6$ $m^3$/year. This understanding, together with the discovery of submersible water pumps, cheap energy, and the relatively high price of crops, such as corn, produced a surge of well drilling by individuals, without much control by the water authorities, and the transformation of large areas into irrigated land. In the following years, a new legislation about water rights plus the lack of enough personnel to supervise the application of the new law, made the characterization, regulation and control of new wells difficult. The Mancha Oriental region suffered an important economic transformation driven by the development of 100,000 new hectares, most of them groundwater irrigated. In the first decade of the 21$^{st}$ century, groundwater extraction from EMAS was as large as 400 x $10^6$ $m^3$/year, 98% of which went to irrigation. Extractions were not compatible with the estimated sustainable volume by [4] of 320 x $10^6$ $m^3$/year, which, together with drought periods between 1990 and 1994, induced a significant drawdown in the piezometric level (as high as 80 m in some areas) and a reduction in the water flow from aquifer to the Júcar river (Figure 2).

The hydrogeological regime of the system has clearly been modified. Groundwater is now flowing to the cones of depression created by the extraction wells and the river-aquifer interaction has changed. The Júcar river changed from being a draining stream to a recharging one; the drawdown produced by the extraction wells prompted a recharge from the river to compensate for the large extractions. But the total rate of extraction was so large that in the 1990's the Júcar river, for the first time on record, went dry during the drought of 1990-1994. The point where the water table dropped below the river bed moved 20 km downstream with respect to its position prior to the beginning of the extensive pumping.

The Júcar Water Authority and the Mancha Oriental User Community realized the unsustainability of the situation and set a number of actions to revert the situation, namely: i) harmonization of the water rights and control of extractions by an annual exploitation plan agreed by all stakeholders, ii) improving the efficiency of irrigation systems, iii) importing surface water from outside of the system to replace some groundwater extractions, iv) replacing the Albacete city urban groundwater supply with water from the Alarcón reservoir in the Júcar river, v) buying (by the water authorities) some water rights during drought periods to reduce extractions. All these actions have had the intended effect as shown by the stabilization and initial recovery of the piezometric levels in the aquifer.

The Water Authority and the User Community established a collaboration agreement with the Universities of Castilla-La Mancha (UCLM) and Politècnica de València (UPV) to develop a numerical model using MODFLOW [5] and its graphical interface ModelMuse [6]. This model, which is calibrated and updated annually, serves to i) study the historical evolution of the aquifer under natural conditions, ii) understand the water balance of the system, iii) predict the state of the system, including both streamflows (Figure 3) and piezometric levels (Figure 4), iv) perform long-term scenario analysis and v) simulate the impact that actions such as well replacement or the buying of water rights may have during drought periods.

Model results have allowed a better understanding of the functioning of the EMAS and its interactions with the Júcar river, as well as to make predictions of how the different actions considered by the stakeholders will affect the system.

The Eastern Mancha Aquifer System is a clear example in which the awareness by stakeholders of a situation of unsustainability gives rise to collaboration among all concerned parties resulting in the acceptance of a set of necessary actions to prevent the continuous deterioration, and ultimately lead to the reversal of the depletion of the system. ■

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## References


[1] Sanz D., Vos J., Rambags F., Hoogesteger J., Cassiraga E. and Gómez-Alday J. J. (2018). The social construction and consequences of groundwater modelling: insight from the Mancha Oriental aquifer, Spain, International Journal of Water Resources Development, DOI: 10.1080/07900627.2018.1495619.

[2] Sanz D., Gómez-Alday J. J., Castaño S., Moratalla Á., De las Heras J. and Martínez-Alfaro P. E. (2009). Hydrostratigraphic framework and hydrogeological behaviour of the Mancha Oriental System (SE Spain), Hydrogeology Journal, DOI 10.1007/s10040-009-0446-y.

[3] Sanz D. (2005). Contribution to the geometrical characterisation of the hydrogeological units forming the Mancha Oriental aquifer system. PhD Thesis, Univ. Complutense de Madrid, Spain.

[4] Estrela T., Fidalgo A., Fullana J., Maestu J., Pérez M. A., Pujante A. M. (2004). Júcar Pilot River Basin. Provisional Article 5 Report pursuant to the Water Framework Directive. Ministry for the Environment, Valencia, Spain.

[5] McDonald, M. y Harbaugh, A. (1984). A Modular Three-Dimensional Finite-Difference Ground-Water Flow Model. U.S. Geological Survey, Open File Report 83-875.

[6] Winston, R. B. (2009). ModelMuse-A graphical user interface for MODFLOW-2005 and PHAST. Chapter 29 of Section A, Ground Water Book 6, Modeling Techniques, U.S. Geological Survey.

# UNCERTAINTY QUANTIFICATION ACROSS SCALES IN THE SUBSURFACE WORLD

BY ALBERTO GUADAGNINI, GIOVANNI PORTA & MONICA RIVA

The research team at Politecnico di Milano (Italy) tackles conceptual, theoretical and numerical approaches to study flow, transport and chemical/biological reactions in natural subsurface porous and fractured media under uncertainty. Our approach is to recognize the importance of tracking and quantifying the impact of uncertainty across scales to ultimately identify uncertainty controls to constrain predictions. This article discusses some key aspects of our research approach and vision on current challenges associated with groundwater quality and quantity.

The Earth's subsurface hosts key resources for the development of society. Aquifers provide invaluable freshwater reserves across several regions worldwide, water demand being largely satisfied by renewable or non-renewable resources hosted by subsurface reservoirs. Similar to all natural systems, geological media exhibit an intrinsic variability of properties, which is the result of a variety of processes that have shaped their formation and current internal make-up. Sedimentary systems, for instance, are the result of the sedimentation of various materials deposited over millions of years. The properties of such geological bodies display remarkable variability across a variety of (space and/or time) scales, associated with physical, chemical or biological heterogeneities (see Figure 1).

Proper and sustainable management of these systems in the complex and ever changing modern environment requires solid scientific understanding and handling of uncertainty to address system functioning and feedbacks amongst its multiple components. Acquiring this knowledge often benefits from comprehensive and scientifically sound conceptual and theoretical frameworks, which are then translated in mathematical and numerical models. The need for modeling is especially compelling when dealing with the Earth's subsurface because of limited direct access to it. Therefore, efforts aimed at managing subsurface resources ubiquitously rely on a (model or) representation of reality. The hydraulic conductivity, a key parameter for the characterization of subsurface materials, is arguably the most heterogeneous parameter in Earth sciences, with laboratory data spanning more than ten orders of magnitude. Paucity of observations and the documented marked spatial heterogeneity in the properties of natural subsurface systems require devising strategies and tools that take into account uncertainty in management and engineering studies and decisions. In this context, stochastic approaches have been developed, motivated by recognizing both the importance of spatial variability and the impossibility of describing in an exhaustive manner the spatial distribution of variables of interest. As such, a non-deterministic framework of analysis is required. In the following we briefly review some of the main elements that we faced in our research concerning the characterization of subsurface systems under uncertainty.

Our research group is based in the Metropolitan area of Milano (Italy), residing in the Po river Valley where the vast majority of drinking water supply is associated with groundwater resources. Yet, the quality and the quantity of this invaluable resource are threatened by anthropogenic activities, including pollution resulting from industrial or agriculture activities, as well as by geogenic sources of hazardous substances (e.g., arsenic or chromium [1], [2]). Improved scientific understanding of the flow and transport processes and accounting for the uncertainty in the description of the system may be the only realistic approach to handling (qualitatively and quantitatively) integrated hydrological problems with sparse data. Original results in this sense have been recently attained through scientific outputs of the project

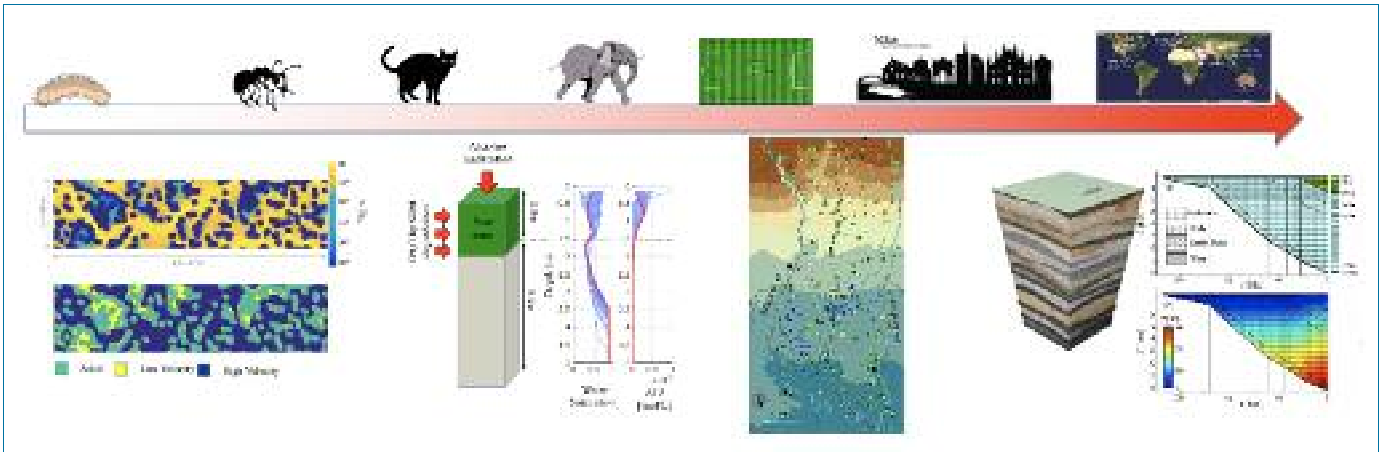

Figure 1. Flow and transport processes in subsurface porous media across scales (images modified from [7], [5], [4], [11])

WE-NEED (Water Needs, Availability, Quality and Sustainability; http://www.we-need.polimi.it [3]), funded by the European Union and the Italian Ministry for Education, University and Research under the 2015 Joint Activities developed by the Water Challenges for a Changing World Joint Programme Initiative (Water JPI). This major project was recently completed under the coordination of Prof. M. Riva and with participation of international partners from Israel (Weizmann Institute of Science), Portugal (University of Aveiro), and Spain (Polytechnical University of Catalunya). A significant challenge underpinning the set-up of a model of an aquifer system is how to address optimal data acquisition, system monitoring, and use of the ensuing information content. This important issue is addressed in detail by WE-NEED upon considering two major aquifer systems in Italy, associated with the areas of Bologna and Cremona, respectively. While the former is a key source of water for the metropolitan area of Bologna (the lower aquifer provides 80% of all groundwater used for drinking and industrial purposes), the strategic importance of the latter is related to the presence of a high number of natural high-quality water springs constituting the main supply to agriculture and key environmental drivers, with significant social, historical and touristic value. As an example, Figure 2 depicts the location of the study area and some aspects of the available dataset, which constitutes a unique source of information for the characterization of the system functioning. These data have enabled us to assess the effect on groundwater flow simulations of alternative conceptual models used to represent the spatial arrangement of geomaterials characterizing the internal architecture of the aquifer [4], a result which constitutes one of the major outputs of WE-NEED and is available to local water companies.

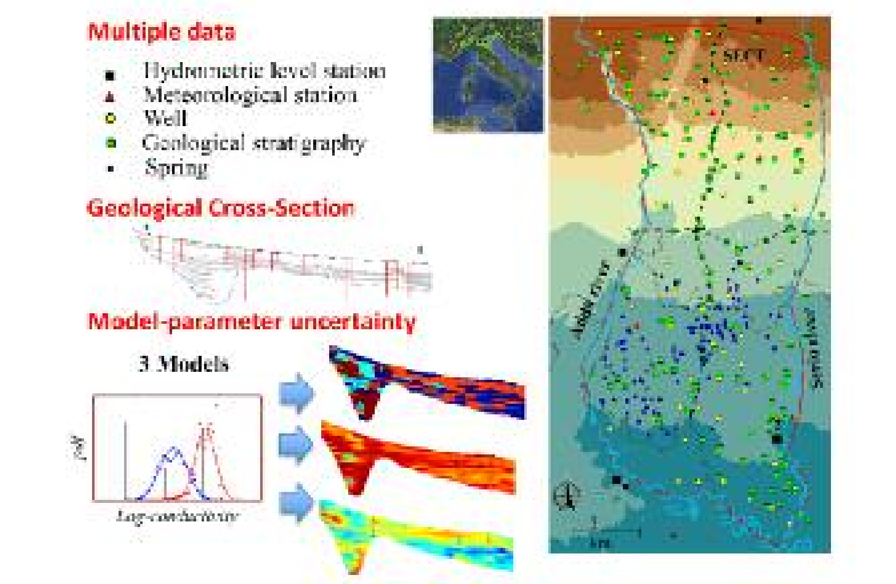

Figure 2. Integration of multiple sources of information and modeling approaches for field scale numerical simulation of groundwater flow in the Cremona area (Italy) (modified from [4])

Assessing quality of groundwater resources requires joint analysis of fluid flow and transport of chemicals dissolved in the water phase. The complexity of the problem is exacerbated by the observation that chemically- and biologically-driven reactive processes typically affect transport in subsurface environments due to a variety of processes, including, e.g., mineral-water interactions or processes such as sorption-desorption or microbial activity. As the ability to fully describe these processes in the natural environment is still very limited, there is a need for approaches that can assist to improve the understanding of system behavior and quantify the implications of the uncertainty associated with model structure and the ensuing parameters. Some of our recent work shows that relying on appropriate sensitivity analysis tools is critical to enhance the knowledge of these complex processes. As an example, the geochemical response of groundwater to environmental factors, such as temperature or redox conditions, may display a strongly nonlinear behavior (see Figure 3). In such cases the sensitivity of the model can be only measured by combining local sensitivity analysis (that can detect local nonlinearities) with global sensitivity analysis approaches (enabling us to measure the overall impact of uncertain model parameters on key statistics of modeling goals). When considering a simplified geochemical model of hexavalent chromium (Cr(IV)) geogenic release [2], we show that combining various techniques is key to obtain a proper assessment of parameter uncertainty and the way it propagates to modeling targets as well as to identify which model inputs should be further subject to improved observation to enhance the ability to meet given modeling goals.

An increase of the complexity of the modeled processes may yield more pronounced

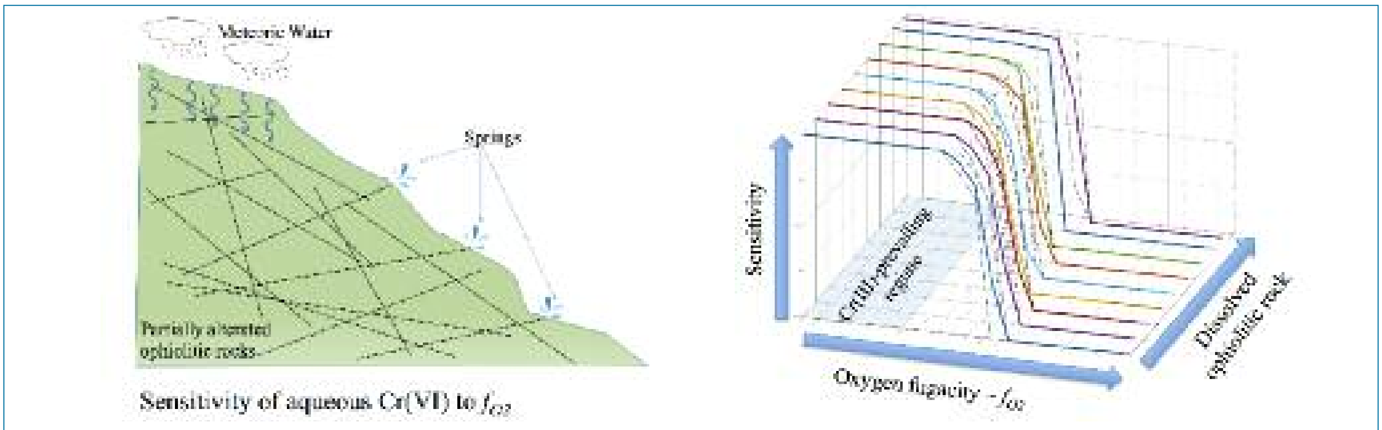

Figure 3. Groundwater quality under uncertain environmental conditions: conceptual and geochemical modeling of hexavalent chromium (Cr(VI)) at hillslope scale. Sensitivity of Cr(VI) to oxygen fugacity exhibits highly nonlinear patterns (modified from [2])

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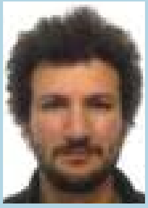


**Giovanni Porta** is currently associate professor at Politecnico di Milano (Italy). The core of his recent and ongoing research is focused on developing models and methods to improve the current understanding of reactive transport processes in porous media. Target applications span across a wide range of disciplines, processes and scales from micron/seconds scales to geological settings involving km-scale domains evolving across millions of years. He is 2019 Junior Marchi Lecturer (award by the Italian Hydraulic Engineering Society (GII)).

**Monica Riva** is Professor of Groundwater and Fluid Mechanics at Politecnico di Milano (Italy). She has managed several research projects focusing on groundwater resources and stochastic approaches. She is currently the coordinator of the Water JPI project WE-NEED. Chair of the EGU Groundwater Division and Vice-chair of the Interpore Council (International Society for Porous Media). Her key areas of expertise include data assimilation, stochastic inverse modeling, uncertainty quantification, and multiphase flows.

couplings and nonlinearities, as suggested by our recent analysis of atrazine biodegradation in agricultural soils [5]. This issue was considered through the implementation of a coupled reaction network of biogeochemical processes, including aerobic and anaerobic processes activated by various functional microbial groups. The model requires specifying 74 biochemical parameters to describe the kinetic response of the system. The results show that uncertainty propagation across these processes taking place in natural soils and under transient conditions yields multimodal distributions for prescribed modeling goals. This implies that diverse and mutually exclusive final system states are likely to occur. Available prior information may not be strongly informative to allow discriminating amongst such states. In this context, sensitivity analysis tools such as those developed in [6] can be used to improve process understanding through model diagnosis, as well as guide environmental monitoring investments, enabling one to prioritize the acquisition of data that can potentially assist to identify parameters that are actually informative on the system response.

While characterizing state-of-the-art models under uncertainty is of utmost importance, existing models and tools need to be improved to resolve the limitations that hamper state-of-the-art modeling strategies. A key research question in this context concerns the ability to transfer information and uncertainties across scales. Even seemingly homogeneous porous media may display a relatively complex geometry and structure at the microscale. This implies, for instance, that fluid velocity at the scale of the pore space is characterized by relevant spatial variability, with spatial distributions entailing preferential flow regions (fast channels) and stagnant areas. These features are well known and explored in the recent literature which documents pore-scale data sets and modeling efforts. In this context, there is still the need for approaches that are capable to transfer such richness of pore-scale information to larger scales, where it could be used to strengthen the interpretation of laboratory or field scale observations. Local fluctuations and gradients, due to pore-scale features such as channeling or segregated stagnant regions [7], [8], may heavily impact nonlinear reaction rates. In general, these features may be critically important for the parameterization of processes that take place unevenly in space (e.g., surface reactions such as adsorption-desorption). Our work in this context provides original operational frameworks and quantitative tools to transfer and upscale available information.
With reference to scaling aspects, our research team has developed and applied a theoretical framework and model enabling us to transfer information on statistics of system parameters across diverse spatial scales in real scenarios, and to quantify the way they impact the statistics of the system states (e.g., fluxes and concentrations).

Relying on the mounting evidence that many spatially varying quantities exhibit non-Gaussian behavior over a multiplicity of scales, we have developed a theoretical model that captures documented scalable non-Gaussian geostatistics of Earth and environmental variables [9], [10]. Such a model allows blending observations of hydraulic parameter distributions into a new framework and is adaptable to diverse spatial/temporal scales. In this context, a key innovative aspect is the development of methodologies and algorithms to generate random fields, exploiting available data to be employed in computational analyses of groundwater flow and transport, with the aim of quantifying risk in realistic (non-Gaussian) environments.

In summary, the core activity of the team is focused on a variety of scientific/application-oriented objectives with the aim of providing quantitative understanding and process-based models of hydrogeological systems and the geochemical behavior of reactive chemical species in environmentally and industrially relevant subsurface settings under model and parametric uncertainty. Our studies aim at identifying and developing methods to incorporate uncertainty quantification and its propagation across observation scales, as grounded on direct observations at diverse scales of interest. The research emphasizes the consequences of human actions on groundwater resources by coupling field and laboratory evidence with original theoretical and modeling concepts. This can support the evaluation of the sustainability of current and future plans for aquifer development with the aim of providing an effective governance perspective while maximizing ecosystem preservation. ■

## References


[1] Molinari, A., Guadagnini, L., Marcaccio, M., Guadagnini, A. (2019) Geostatistical multimodel approach for the assessment of the spatial distribution of natural background concentrations in large-scale groundwater bodies *Water Research*, 149, 522-532. doi: 10.1016/j.watres.2018.09.049 [2] Ceriotti, G., Guadagnini, L., Porta, G., Guadagnini, A. (2018) Local and Global Sensitivity Analysis of Cr (VI) Geogenic Leakage Under Uncertain Environmental Conditions Water Resources Research, 54 (8), 5785-5802.

[3] http://www.we-need.polimi.it

[4] Bianchi Janetti E., L. Guadagnini, M. Riva, A. Guadagnini (2019) Global sensitivity analyses of multiple conceptual models with uncertain parameters driving groundwater flow in a regional-scale sedimentary aquifer, *J. of Hydrology,* 574, 544-556, doi: 10.1016/j.jhydrol.2019.04.035

[5] Porta, G., la Cecilia, D., Guadagnini, A., Maggi, F. (2018) Implications of uncertain bioreactive parameters on a complex reaction network of atrazine biodegradation in soil *Advances in Water Resources*, 121, 263-276, doi: 10.1016/j.advwatres.2018.08.002.

[6] Dell'Oca A., M. Riva, A. Guadagnini (2017) Moment-based Metrics for Global Sensitivity Analysis of Hydrological Systems, *Hydrol. Earth Syst. Sci.*, 21, 6219–6234, doi:10.5194/hess-21-6219-2017.

[7] Ceriotti, G., Russian, A., Bolster, D., Porta, G. (2019) A double-continuum transport model for segregated porous media: Derivation and sensitivity analysis-driven calibration *Advances in Water Resources*, 128, 206-217, doi: 10.1016/j.advwatres.2019.04.003

[8] Siena M., O. Iliev, T. Prill, M. Riva, A. Guadagnini (2019) Identification of Channeling in Pore-Scale Flows, Geophysical Research Letters, 46(6), 3270-3278, doi: 10.1029/2018GL081697.

[9] Riva M., M. Panzeri, A. Guadagnini, S. P. Neuman (2015), Simulation and analysis of scalable non-Gaussian statistically anisotropic random functions, *J. Hydrol.*, 531 Part 1, 88-95, doi:10.1016/j.jhydrol.2015.06.066.

[10] Guadagnini A., M. Riva, S.P. Neuman (2018) Recent Advances in Scalable Non-Gaussian Geostatistics: The Generalized Sub-Gaussian Model, *J. of Hydrology*, 562, 685-691.

[11] Cerotti G., G.M. Porta, C. Geloni, M. Dalla Rosa, A. Guadagnini (2017) Quantification of CO2 generation in sedimentary basins through Carbonate/Clays Reactions with uncertain thermodynamic parameters, *Geochimica et Cosmochimica Acta*, 213, 198-215.

# ARTIFICIAL NEURAL NETWORKS FOR THE PREDICTION OF GROUNDWATER NITRATE CONTAMINATION

BY CHRISTINA STYLIANOUDAKI, IOANNIS TRICHAKIS & GEORGE P. KARATZAS

An artificial neural network (ANN) model is proposed for the determination of groundwater nitrate contamination, based on an approach of easily measurable and cost-effective water quality parameters (pH, electrical conductivity, $HCO_3^-$, $Cl^-$, $Ca^{2+}$, $Mg^{2+}$, $Na^+$, $K^+$, $SO_4^{2-}$). The data used, derived from the chemical analyses of groundwater samples, from wells located in the Kopaidian Plain, Greece. The results of the model described in this article indicate that ANNs could be used as an alternative method for the estimation of groundwater contamination problems.

The rapid increase in population, as well as industrialization and intensification of agricultural activities, have led to significant quantitative and qualitative degradation of groundwater resources worldwide [1]. The situation is expected to be burdened by climate change, which will cause changes in rainfall patterns and an increase in average surface temperature, especially in drought-prone areas [2]. Surface and groundwater contamination due to the presence of nitrates ($NO_3^-$) is considered as one of the most common environmental problems [3], [4]. The major anthropogenic source of nitrogen in the environment is the application of nitrogen fertilizer [5]. Other anthropogenic sources include industrial wastes, deforestation (leading to conversion to agricultural land) domestic wastewater and septic tanks [6]. According to Greek and EU legislation, nitrate concentration shall not exceed 50 mg /l for nitrates ($NO_3^-$), or 11 mg /l for nitrate-nitrogen ($NO_3^-N$) [7]. On a global scale, concentrations of nitrate in groundwater exceed the limits that have been set, and it is estimated that in the last three decades nitrate pollution has increased by 36%. In the eastern Mediterranean and Africa, the situation is even more worrying, as nitrate levels seem to have more than doubled [6].

Nitrate ions have a toxic effect with proven effects on human health and have been statistically associated with various forms of cancer [8], [9], [10]. In addition, increased indices of thyroid diseases have been recorded in areas with high nitrate levels in water supplies [5]. In order to protect public health, sustainable management of groundwater resources is required. However, techniques for detecting and measuring nitrate concentrations in water can be characterized by high cost and high time demand [11], while the portable devices used for this purpose are not of sufficient accuracy. Furthermore, in the various methods used for chemical analysis of water, the detection of nitrates is affected by the presence of other ions, especially $Cl^-$ [12]. Physics-based models for the analysis of groundwater contamination problems have developed significantly in recent years. However, these models require extensive data that are often not available. In many applications, there is a need to develop surrogate, easy to use models that can rapidly analyze groundwater contamination, without the limitations of more complex models. The scope of this study is to describe a model for the easy estimation of nitrate groundwater contamination based on easily measurable and cost effective input parameters.

Artificial neural networks are data driven models that treat the system being studied as a 'black box' [13]. ANNs have the ability to correlate variables whose relationship is not known or is very complex [14], [15], without the use of physical data, such as porosity or hydraulic conductivity [16].

> **"It is estimated that in the last three decades nitrate pollution has increased by 36%"**

ANNs have widely found applications in hydrology and have been successfully used in groundwater quality modeling [16],[17]. Several studies have presented ANNs for the estimation of the water level by using water budget variables as input parameters [15], [18], [19], [20], [21]. Regarding nitrate contamination, ANN models using water quality parameters or/and water budget variables as input parameters, have been proposed [22], [23], [24]. Comprehensive reviews over the applications of ANNs in hydrology can be found in [17], [25] and [26].

An ANN consists of a number of fully connected processors called neurons, which accept, analyze, and exchange information over a network of weighted connections [26]. The feed-forward neural network was the first type of ANN, where information moves in a forward direction. The information is processed at different layers, divided in three categories: input, output and hidden layers. Each input $x_i$ presented in a neuron, is weighted by a synaptic weight $w_i$ and the results are summed. The sum is introduced in an activation function; in the case it exceeds a certain threshold value,. The basic function of an ANN is the training process, which is performed by a learning rule that modifies the weights of the connections in order to minimize the difference between the calculated output of the network and the desired output (real value) [25]. The efficiency of the model is evaluated by its generalization ability, i.e. its ability to give the correct output even for examples not included in the training set. More information regarding ANNs and their operation is presented by [16] and [27].

In this study, a feed-forward neural network consisting of three layers was developed, using MATLAB R2010 software. A Levenberg-Marquardt regularization algorithm was

Table 1: Maximum, minimum and mean values of the input and output parameters used in the model

| | $NO_3^-$ (mg/l) | pH | COND | $Ca^{2+}$ (mg/l) | $Mg^2$ (mg/l) | $Na^+$ (mg/l) | $K^+$ (mg/l) | $HCO_3^-$ (mg/l) | $Cl^-$ (mg/l) | $SO_4^{2-}$ (mg/l) |
|---|---|---|---|---|---|---|---|---|---|---|
| Min | 5 | 6.4 | 234 | 2.4 | 4.4 | 2.3 | 0.4 | 49 | 5.3 | 10 |
| Max | 167 | 9.1 | 2750 | 236 | 121.1 | 303.5 | 13 | 585 | 560.2 | 148.9 |
| Mean | 27.31 | 7.66 | 791.03 | 66.66 | 38.40 | 36.62 | 1.94 | 324.10 | 58.01 | 38.04 |

Figure 1. Model's results - R coefficient

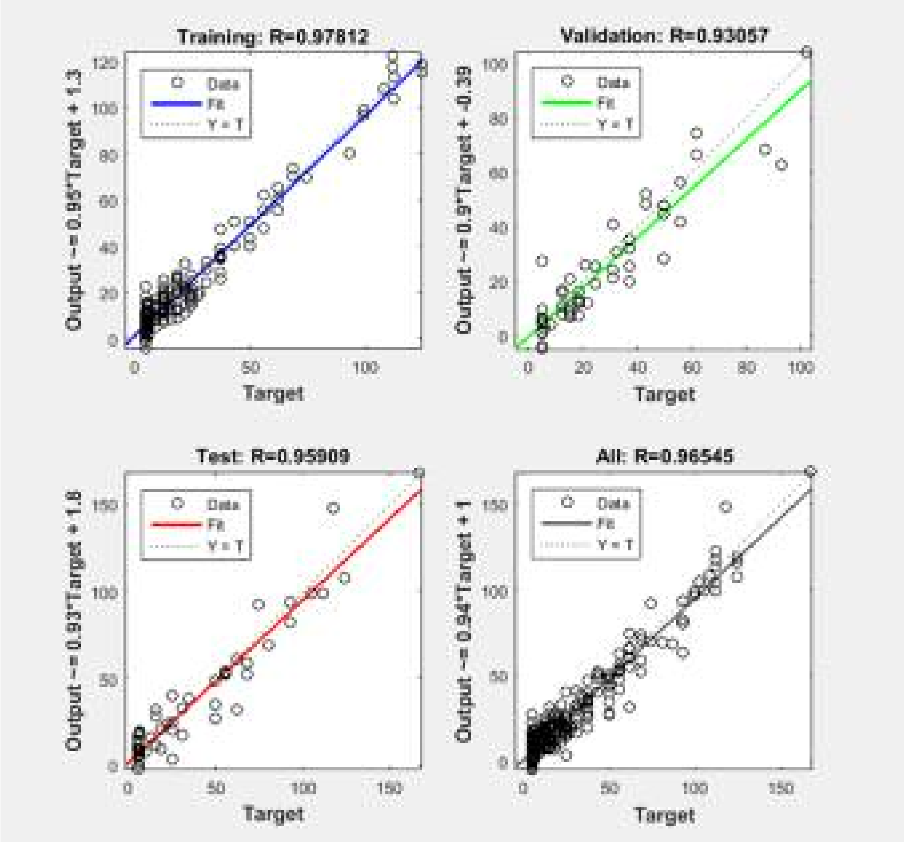



Figure 2. Observed versus simulated values

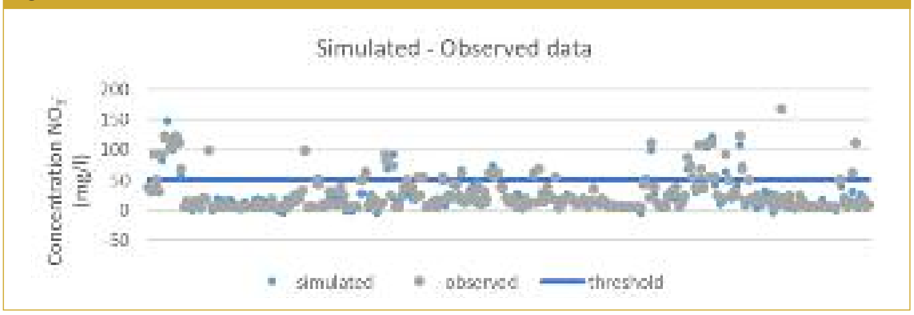



Table 3. Calculated statistical indicators

| Index | All | Test | Validation |
|---|---|---|---|
| RMSE (mg/l) | 7.75 | 10.94 | 9.14 |
| MAE (mg/l) | 5.70 | 8.07 | 6.90 |
| Bias (mg/l) | -0.65 | -0.77 | -3.07 |
| NSE | 0.9878 | 0.9193 | 0.9969 |
| St. Deviation | 29.70 | 38.90 | 23.64 |

employed for the training procedure. The ANN's architecture was determined through a trial and error procedure, based on the correlation coefficient (R) between the real data and the simulated values by the model. The best architecture came out to be that of one hidden layer with 10 nodes, a sigmoid function in the first layer and a linear in the output layer as activation functions. Of the available data, 60% was used in the training process, 20% in the testing process and the remaining 20% was used for the evaluation of the model's performance (generalization ability).
For the analysis of the results, four error indicators were calculated: the Root Mean Square Error RMSE), the Mean Absolute Error (MAE), the bias (mean error) and the Nash-Sutcliffe Model Efficiency (NSME).

The data used for the ANN's training and validation, were derived from a set of 263 measurements of typical water quality parameters, obtained from sampling in the Kopaidian Plain. The area has been designated as a vulnerable zone with respect to nitrogen pollution from agricultural runoff, according to the requirements of the European Union Directive 91/676/EEC [28], due to the intensive agricultural, livestock and industrial activities that take place in it. The input parameters to the model were pH, electrical conductivity, bicarbonate ($HCO_3^-$) and $Cl^-$, $Ca^{2+}$, $Mg^{2+}$, $Na^+$, $K^+$, $SO_4^{2-}$.
Table 1 presents the maximum, minimum and the mean value of the $NO_3^-$ concentrations and of all the parameters used as inputs to the model.

The simulation results are presented in Figure 1. The Pearson coefficient (R index) is shown on top of each chart, respectively, for the training data in the top left part of the figure, for the validation set in the top right, for the testing data set in the bottom left and for the full data set in the bottom right. In the plot, the simulated values (output – vertical axis) are plotted against the observed values (target – horizontal axis), and their best-fit equation, which describes the solid line in the graph, is shown on the vertical axis title. As shown in the charts of Figure 1 the ANN has delivered very good results.

A high correlation is observed, between the simulated and actual values, for every data set, with a correlation index for the full data set of 0.96545. In the test set, R is equal to 0.95909 and in the validation set, R= 0.93057. Considering that these data have not been used

in the training process, R values signify a very good generalization ability of the model.
In Figure 2, the simulated values by the model, together with the real data are presented, along with the threshold value of 50 mg/l for nitrate concentration. As already expected from the R value, Figure 2 confirms that the simulated values are very close to their observed counterparts.

The calculated indicators for an additional evaluation of the model's performance are shown in Table 3.

For the full data set, the NSE is equal to 0.9878, for the test set, $NSE_{test} = 0.9193$ and for the validation set, $NSE_{vald}=0.9969$. As shown by the indicators, the model has produced remarkably satisfactory results. NSE values in the range ($0.75 < NSE < 1$), indicate very good performance of the model being assessed [29]. Therefore, taking into account the NSE index, the simulation can be characterized successful. Moreover, according to [30], RMSE and MAE values less than half of the standard deviation of the observed data are considered low. In addition, the small difference between RMSE and MAE (7.754874 mg/l - 5.702638 mg/l) indicates the absence of extreme errors. Lastly, it is worth pointing out that according to the Bias index, the model tends to underestimate the observed values but not by much. The calculated indices suggest that the model is highly accurate.

## Conclusions

In hydrological applications, there is a need to develop simple models that can capture the main relationships between parameters without the need to develop complex physics-based models that are difficult to solve. ANNs have the essential advantage that they can track the hidden relationship between variables – without the need to assume linearity- and so, available data that are not usually used in conventional techniques can be exploited. The results of the study described in this article demonstrate that ANNs are a potentially powerful modelling method, more economical and less time consuming. ■

## References


[1] Kazakis, Nerantzis & Voudouris, K. (2015). Groundwater vulnerability and pollution risk assessment of porous aquifers to nitrate: Modifying the DRASTIC method using quantitative parameters. Journal of Hydrology. 525. 13-25.

[2] IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

[3] Spalding R.F., Exner M.E. (1993) Occurrence of nitrate in groundwater—a review. Journal of Environmental Quality 22. 392–402.

[4] Rivett, Michael & R Buss, Stephen & Morgan, Philip & Smith, Jonathan & D Bemment, Chrystina. (2008). Nitrate Attenuation in Groundwater: A Review of Biogeochemical Controlling Processes. Water research. 42. 4215-32.

[5] Ward, Mary & Jones, Rena & Brender, Jean & de Kok, Theo & J Weyer, Peter & T Nolan, Bernard & M Villanueva, Cristina & Breda, Simone. (2018). Drinking Water Nitrate and Human Health: An Updated Review. International Journal of Environmental Research and Public Health. 15. 1557.

[6] Shukla, Saurabh & Saxena, Abhishek. (2018). Global Status of Nitrate Contamination in Groundwater: Its Occurrence, Health Impacts, and Mitigation Measures. *Handbook of Environmental Materials Management*. Springer, Cham.

[7] WHO (World Health Organization) (2011) Guidelines for drinking-water quality, 4th edn. World Health Organization, Geneva. Available: Accessed 10 Apr 201.

[8] Hunder.H. Comly. (1945). Cyanosis in infants caused by nitrates in well water. JAMA. 129. 112-116

[9] Morales-Suárez-Varela, Maria & Llopis-Gonzalez, Augustin & L. Tejerizo-Perez, María. (1995). Impact of nitrates in drinking water on cancer mortality in Valencia, Spain. European Journal of Epidemiology. 11. 15-21.

[10] Palli, Domenico & Saieva, Calogero & Coppi, Claudio & Del Giudice, Giuseppe & Magagnotti, Cinzia & Nesi, Gabriella & Orsi, Federica & Airoldi, Luisa. (2001). O 6 -Alkylguanines, Dietary N-Nitroso Compounds, and Their Precursors in Gastric Cancer. Nutrition and cancer. 39. 42-9.

[11] Azmi, Aizat & Amsyar Azman, Ahmad & Ibrahim, Sallehuddin & Md Yunus, Mohd Amri. (2017). Techniques in advancing the capabilities of various nitrate detection methods: A review. International Journal on Smart Sensing and Intelligent Systems. 10. 223-261.

[12] Alahi, Md & Mukhopadhyay, S.C. (2018). Detection methods of Nitrate in water: A Review. Sensors and Actuators A: Physical. 280. 210-221. https://doi.org/10.1016/j.sna.2018.07.026

[13] Tapoglou, Evdokia & Karatzas, George & Trichakis, Ioannis & Varouchakis, Emmanouil. (2014b). A spatio-temporal hybrid neural network-Kriging model for groundwater level simulation. Journal of Hydrology. 519. 3193-3203.

[14] Almahallawi, Khamis & Jacky, Mania & HANI, Azzedine & Shahrour, Isam. (2011). Using of neural networks for the prediction of nitrate groundwater contamination in rural and agricultural areas. Environmental Earth Sciences. 65. 917-928.

[15] Ghose, Dillip & Das, Umesh & Roy, Parthajit. (2018). Modeling response of runoff and evapotranspiration for predicting water table depth in arid region using dynamic recurrent neural network. Groundwater for Sustainable Development. 6. 263-269, https://doi.org/10.1016/j.gsd.2018.01.007

[16] ASCE, 2000a. Artificial neural networks in hydrology. I: Preliminary concepts. Journal of Hydrologic Engineering. 5. 115-123.

[17] ASCE, 2000b. Artificial Neural Networks in hydrology. II: Hydrologic applications. Journal of Hydrologic Engineering. 5. 124-137.

[18] Nayak, P., Rao, Y. & Sudheer, K. (2006) Groundwater level forecasting in a shallow aquifer using artificial neural network approach. Water Resources. Management. 20. 77–90.

[19] Trichakis, I., Nikolos, I., & Karatzas, G. (2009). Optimal selection of artificial neural network parameters for the prediction of a karstic aquifer's response. Hydrological Processes. 23. 2956-2969. https://doi.org/10.1002/hyp.7410

[20] Trichakis, I.C., Nikolos, I.K. & Karatzas, G.P. (2011). Artificial Neural Network (ANN) Based Modeling for Karstic Groundwater Level Simulation. Water resources management. 25. 1143-1152. https://doi.org/10.1007/s11269-010-9628-6

[21] Tapoglou, Evdokia & Trichakis, Ioannis & Dokou, Zoi & Nikolos, Ioannis & Karatzas, George. (2014a). Groundwater-level forecasting under climate change scenarios using an artificial neural network trained with particle swarm optimization. Hydrological Sciences Journal. 59. 1225-1239. https://doi.org/10.1080/02626667.2013.838005

[22] Yesilnacar M.I., Sahinkaya E., Naz M. and. Ozkaya B. (2008). Neural network prediction of nitrate in groundwater of Harran Plain, Turkey. Environmental Geology. 56. 19–25. DOI: 10.1007/s00254-007-1136-5

[23] Ehteshami, M., Farahani, N.D. & Tavassoli, S. (2016). Simulation of nitrate contamination in groundwater using artificial neural networks. Modeling Earth Systems and Environment. 2: 28. https://doi.org/10.1007/s40808-016-0080-3

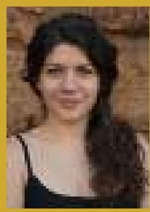
[24] Wagh, Vasant & Panaskar, Dipak & Muley, Aniket & Mukate, Shrikant & Gaikwad, Satyajit. (2018). Neural Network Modelling for Nitrate Concentration in groundwater of Kadava River basin, Nashik, Maharashtra, India. Groundwater for Sustainable Development. DOI: 10.1016/j.gsd.2017.12.012

[25] Tanty, Rakesh & S. Desmukh, Tanweer. (2015). Application of Artificial Neural Network in Hydrology- A Review. International Journal of Engineering Research. 4. DOI: 10.17577/IJERTV4IS060247

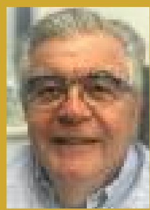
[26] Oyebode, O, Stretch, D. (2019). Neural network modeling of hydrological systems: A review of implementation techniques. *Natural Resource Modeling*. 32:e12189. https://doi.org/10.1111/nrm.12189

[27] Haykin, S.S., (1999). Neural networks: a comprehensive foundation. Prentice Hall

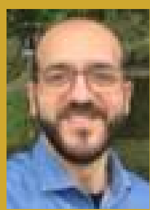
[28] Ministry of Environment and Energy, Nitrate Pollution, http://www.ypeka.gr/Default.aspx?tabid=250&locale=en-US&language=el-GR

**Christina Stylianoudaki** is an Environmental Engineer, MSc and Ph.D. student in the School of Environmental Engineering at the Technical University of Crete, in the field of water resources management and climate change. Currently she works in the Geoenvironmental Engineering Laboratory, as a researcher on the development of a flood risk management system. Her research interests include hydrological processes modelling, environmental systems optimization, simulation of flood events and assessment of climate change's impact on water resources.



Professor **George P. Karatzas** received his Ph.D. in 1992 from the Department of Civil Engineering of Rutgers University, USA in a combined Ph.D. with Princeton University, USA. He is a Professor in the School of Environmental Engineering at the Technical University of Crete, Greece since 2006. His area of research includes numerical simulation in porous media, groundwater management, subsurface remediation techniques, global optimization techniques, surface water modeling and floods. He has been principle investigator and co-investigator in several national and international research projects related to groundwater flow and contaminant transport, groundwater management, surface hydrology and floods.



**Ioannis Trichakis** studied Environmental Engineering at the Technical University of Crete (1999-2004) and did postgraduate studies at the National Technical University of Athens (2004-06). He received his PhD from the University of Crete (2006-2011) where he worked on research projects on water resources and flood management. He has worked as a researcher at the European Commission's Joint Research Center on water resources management programs in European and developing countries and on simulating groundwater and surface water systems. He has extensive experience in technical support for research programs, writing progress reports and coordinating a large number of partners from different countries.


[29] Moriasi, Daniel & Gitau, Margaret & Pai, Naresh & Daggupati, Prasad. (2015). Hydrologic and Water Quality Models: Performance Measures and Evaluation Criteria. Transactions of the ASABE (American Society of Agricultural and Biological Engineers). 58. 1763-1785. DOI: 10.13031/trans.58.10715

[30] Singh, J., H. V. Knapp, and M. Demissie. 2004. Hydrologic modeling of the Iroquois River watershed using HSPF and SWAT. ISWS CR 2004-08. Champaign, Ill.: Illinois State Water Survey. DOI:10.1111/j.1752-1688.2005.tb03740.x

# CLIMATE CHANGE AND GROUNDWATER: AN AUSTRALIAN PERSPECTIVE

BY GLEN WALKER, RUSSELL CROSBIE, FRANCIS CHIEW, LUK PEETERS & RICK EVANS

The annual streamflow into water storages of Perth, a major city in south-western Australia, has fallen from 338 giga liters (GL) in the period 1911–1974 to 134 giga liters (GL) during 1975–2017[1]. This loss of surface water inflow, due to a drying climate, has raised awareness amongst water planners of climate change and potential water shortages affecting cities and irrigation areas across southern Australia. This article discusses the role of groundwater in response to climate change in Australia.

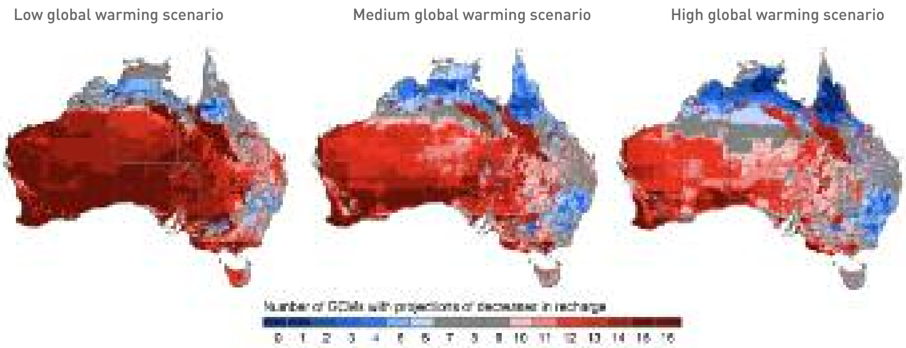



Figure 1. Number of GCMs (general circulation model) under which a decrease in recharge is projected (from the 16 GCMs for each global warming scenario). The consistency of darker reds in the very south for all climate scenarios means greater likelihood of recharge reducing. From [7]

Groundwater is an important source of water in Australia. It represents about 30% of total water use [2] in Australia, and it is the main source or only source of water in drier regions. It is particularly important during droughts, when surface water resources are limited [3]. However, most of the groundwater in the drier areas of southern Australia is too saline for human consumption or irrigated agriculture. Fresher groundwater is heavily used, but is spatially patchy.

Since 1990, water planning in Australia has undergone national reform [4]. Responsibility for water allocation has shifted towards regional authorities, supported by state agencies. There, water planning can be integrated with other planning (such as regional development) and all sources of water (surface water, groundwater, desalination, recycled, imported) considered in relation to changing demands. As demand increases to near the limit of the current water availability alternative and more (climate) resilient supplies and contingency plans need to be developed.

Under the water reform, groundwater is managed to balance consumptive and environmental water requirements to protect important ecosystems. The process differs across and within jurisdictions, but usually involves either the setting of an extraction limit and associated rules for consumptive use, and/or adjustment of allocations based on the state of the ground water system.

The climate projections for Australia [1] show further increases in temperatures; decreases in cool-season rainfall across southern Australia, with more time spent in drought; and more intense heavy rainfall throughout Australia. These predictions reflect recent climatic and hydrological trends. For example, the May to July rainfall across south-western Australia has been reduced by 20% since 1970[1], while projected June to August rainfall in 2090 is 32 ± 11% lower than in 1990[5]. Similarly, the April to October rainfall for 1999-2018 over south-eastern Australia has fallen 11% compared to the 1900-1998 period[1], while the projection for June to August rainfall for the Murray-Darling Basin in south-eastern Australia in 2090 is less than that in 1990 by 16 ± 22%[5]. Despite the large uncertainty in the future projections, there is consistency with respect to the direction of change across the different global climate models for southern Australia. The observed long-term reduction in rainfall has led to even greater reductions in stream flows in southern Australia, as illustrated by the reduced inflows for Perth dams [1] and runoff across Victoria and the southern Murray-Darling Basin [6].
Climate change can affect groundwater directly by changing inflows (such as diffuse recharge, i.e. recharge that occurs by percolation below the rooting zone across the landscape) and outflows (such as evapotranspiration). Climate can also affect groundwater indirectly through changes in other sources of water, land use and demand.

Southern Australia has winter-dominant rainfall, run-off and recharge. Reductions in winter rainfall lead to amplified reductions in diffuse recharge. This means that a reduction of cool season rainfall by 10% may possibly reduce diffuse recharge or recharge by 30% [7]. This causes water planners most concern as moderate reductions in rainfall cause major reductions in groundwater availability.

Diffuse recharge under future climate can be informed by rainfall outputs (long-term averages and high rainfall intensity driving recharge) of global climate models. Since such models do not estimate local climate well, the results need to be downscaled to better reflect future local climate. Physically based soil-vegetation-atmosphere models have been used in Australia to estimate the change in recharge due to changed climate using this down-scaled climate data as input. The uncertainty in each of the models from global climate model to downscaling to recharge models combine to generate a large predictive uncertainty. Recharge outputs will have generally greater

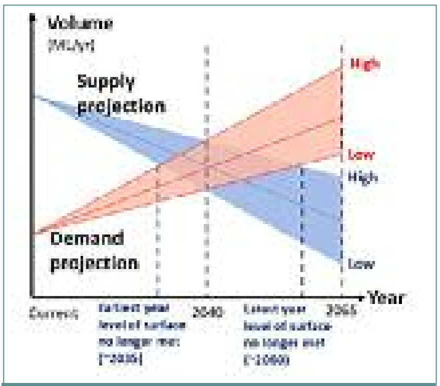



Figure 2. The inclusion of uncertainty into supply and demand predictions, showing how this translates to uncertainty in timing once demand matches supply

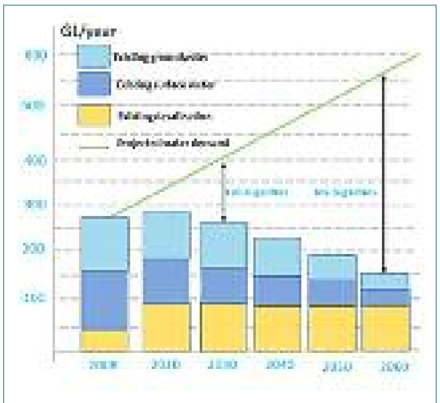



Figure 3. The projections of supply and demand for Perth and surrounding areas until 2060. The supply is separated into groundwater, surface water and existing desalination. Adapted from [12].

variability than surface water run-off outputs due to the nature of the models and the underlying data. There may also be biases introduced by assumptions in the models (e.g. free drainage) which may not hold in the future, and the impact of climate on other recharge factors such as land use and streamflow. While further investigations may reduce this uncertainty, it will never be eliminated. This means that any water resource planning will need to incorporate uncertainty in managing climate change risk.

Information on changes in recharge is available in Australia via maps and associated databases at various scales, for different climate scenarios and using different downscaling and recharge models [7],[8],[9]. Outputs are often depicted in a way to reflect the degree of consistency in predictions (Figure 1). The outputs of both global circulation models and downscaling techniques are also available for use in other models [10].

Despite the large uncertainty in the predictions of recharge, these estimates provide useful inputs to the water planning process [11]. Most planning processes use a risk approach and need to consider a range of recharge outputs, including the worst-case scenarios.
The extraction limit is more directly relevant to the water planning process than recharge, as it represents the maximum volume of groundwater that can be pumped. While a reduction in recharge will generally lead to a reduction in the extraction limit, the relative reduction is likely to be amplified, due to minimum environmental water requirements. The degree of amplification will be influenced by decisions regarding the trade-offs between the economic benefits of consumptive use and adverse impacts on the environment. The extraction limit also depends on the hydrogeology which introduces more uncertainty in its determination.

Because predictions for southern Australian are for further drying into the future, any uncertainty in the extraction limit will affect the rate of decline of supply and hence the time lag before any action to be taken. Uncertainty in recharge can therefore be conceptualized as uncertainty in timing for actions to be required (Figure 2). The timing of any actions also depends on the climate impacts on demand. Demands for water will depend on a range of factors, including population growth and regional development. Climate change may affect demand through changes in the types of irrigated crop, crop water use and dryland vegetation in response to higher temperatures, lower rainfall, lower water reliability and higher water prices. As it could be costly to implement appropriate management responses either long before they are needed or too late, some form of adaptive management is required once demand approaches supply. Even though climate change will have a widespread impact on recharge across southern Australia, the impact on groundwater management will be locally variable. Factors affecting groundwater vulnerability to climate change [7] include ratio of groundwater use to extraction limit and effective aquifer storage. Relatively higher groundwater use means that not only is there minimal opportunity for greater use of groundwater, but a risk that entitlements of groundwater may need to reduce to fall below sustainable levels of extraction. This applies almost exclusively to fresh groundwater systems.

A low aquifer storage means that there is only a small buffer (and hence time) to manage any reduction in recharge. Groundwater systems tend to be more resilient to climate variability because the large aquifer storage can be much greater than dam storages. For this reason, groundwater systems have often been used as a drought contingency measure. Small groundwater systems, such as fresh-water lenses, can be vulnerable during droughts. The effective storage is also influenced by high-value groundwater-dependent ecosystems that require higher water tables to avoid degradation of health and to protect baseflow.

**Dr Glen Walker** is a groundwater hydrologist, who worked with CSIRO in Adelaide for over 30 years before setting up his own consultancy, Grounded in Water. He specializes in salinity and groundwater sustainability.

**Dr Francis Chiew** is a science leader and leads the hydroclimate and hydrological modelling group in Australia's Commonwealth Scientific and Industrial Research Organisation (CSIRO). Francis has published widely on hydroclimate and hydrological sciences and has a long history of leading climate-water research and working with the industry in Australia and overseas on integrated basin management and water resources challenges.

**Dr Crosbie** is a Principal Research Scientist at CSIRO and is currently the Team Leader of the Regional Scale Groundwater Analysis team. Throughout his research career he has worked in several areas including water resources, climate change, salinity and water in the resources sector. He is best known for his expertise in estimating groundwater recharge, particularly under climate change. He is the author (or co-author) of over 100 scientific publications.

**Dr Luk Peeters** research focuses on modelling groundwater dynamics at hte regional to continental scales with a focus on the uncertainly of model predications both quantitatively and qualitatively. Luk led the developmentt and application of the uncertainty analysis, including propagation of uncertainty between model, for the Bioregional Assesments Proggram.

**Dr Richard Evans** is Principal Hydrogeologist with Jacobs. Rick has 40 years experience in all aspects of water resource development with a focus on groundwater resource management and managed aquifer recharge. He has worked on numerous water resource projects throughout Australia and Asia. His strong interest is on the potential for conjunctive water management and managed aquifer recharge to secure both urban and irrigation development throughout Australia.

Should groundwater extraction approach the extraction limit, management responses are similar to those for any stressed system, namely some combination of redistributing extraction through trade; reducing demand; augmenting recharge (through managed aquifer recharge, water sensitive urban design and changing land use) and seeking alternative sources. All of these are used or planned in Australia. The aim of these management responses is to develop a water supply that is more resilient to climate change.

Perth has advanced most with their planning process for climate change impacts. While Perth has declining rainfall, surface water and groundwater, its demand is increasing. A gap has been identified between supply and demand of 120 GL/yr by 2030 and 365 GL/yr by 2060 (Figure 3)[12]. As part of the Water Forever Plan, the gap is to be met by reduction in water use (74 GL/yr by 2030 and another 102 GL/yr by 2060), increased water recycling (39, 48 respectively) and new sources 218, 335 respectively). The new sources of water include managed aquifer recharge, further desalination and deeper groundwater systems [10]. Since its initial release, the plan has continued to change.

## Conclusions

While there are potentially other impacts of climate change on Australian groundwater, the drying climate and longer droughts in southern Australia is the most immediate threat. The amplification of reductions in rainfall into reductions in recharge, and then further amplification to reductions in extraction limit mean that groundwater supplies can diminish quickly.

The large uncertainties in climate prediction, together with those of hydrogeology, means that predictive uncertainty is high. An adaptive management strategy is required both for the groundwater supply as part of an integrated water supply, but also the management of the groundwater supply itself. Adaptive management will, in turn, require monitoring of extraction, piezometric responses and land use and more attention to forecasting demand. There needs to be a general shift to more climate-resilient sources of water such as desalination, managed aquifer recharge and water use efficiency measures.

The experience in Perth provides a guide for other regional water authorities in planning for climate change. ■

## References


[1] Bureau of Meteorology and CSIRO (2018). State of the Climate (2018). Commonwealth of Australia, 24pp.

[2] Harrington N and Cook P, 2014, Groundwater in Australia, National Centre for Groundwater Research and Training, Australia.

[3] CSIRO (2008). Water availability in the Murray-Darling Basin. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. CSIRO, Australia. 67pp.

[4] Australian governments (2004). Intergovernmental Agreement on a National Water Initiative. http://agriculture.gov.au/water/policy/nwi

[5] Reisinger, A., R.L. Kitching, F. Chiew, L. Hughes, P.C.D. Newton, S.S. Schuster, A. Tait, and P. Whetton, 2014: Australasia. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi,Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White.(eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1371-1438.

[6] Chiew FHS, Potter NJ, Vaze J, Petheram C, Zhang L, Teng J, Post DA (2014) Observed hydrologic non-stationarity in far south-eastern Australia: implications and future modelling predictions. Stochastic Environmental Research and Risk Assessment, 28, 3–15.

[7] Barron OV, Crosbie RS, Charles SP, Dawes WR, Ali R, Evans WR, Cresswell R, Pollock D, Hodgson G, Currie D, Mpelasoka F, Pickett T, Aryal S, Donn M and Wurcker B (2011) Climate change impact on groundwater resources in Australia: summary report. CSIRO Water for a Healthy Country Flagship, Australia.

[8] Crosbie RS, Pickett T, Mpelasoka FS, Hodgson GA, Charles SP, Barron O 2011. Diffuse recharge across Australia under a 2050 climate: Modelling results. CSIRO: Water for a Healthy Country National Research Flagship. 64 pp

[9] Littleboy M, Young J and Rahman J (2015) Climate change impacts on surface runoff and recharge to groundwater. NSW Office of Heritage, NSW government, 75pp. https://climate-change.environment.nsw.gov.au/Impacts-of-climate-change/Water-resources/Groundwater-recharge-and-surface-runoff

[10] CSIRO and Bureau of Meteorology 2015, Climate Change in Australia Information for Australia's Natural Resource Management Regions: Technical Report, CSIRO and Bureau of Meteorology, Australia https://www.climatechangeinaustralia.gov.au/en

[11] DEWLP (2016). Guidelines for Assessing the Impact of Climate Change on Water Supplies in Victoria. Department of Environment, Land, Water and Planning, Victorian government, 75pp.

[12] Water Corporation (2009). Water Forever: Towards Climate Resilience. 109pp. https://www.watercorporation.com.au/-/media/files/residential/about-us/planning-for-the-future/water-forever-50-year-plan.pdf

# Erich J. Plate (1929 – 2019)

Erich J. Plate, past president and honorary member of IAHR, passed away on Monday, July 22, 2019, shortly after his 90. birthday. Erich Plate was a creative researcher and promoter of hydrology and water resources development, an interdisciplinary and internationally active networker and integrator, a motivating teacher and an always openminded and reliable colleague with a strong impact, and also a good friend.

Erich Plate was born 1929 in Hamburg, Germany. From 1950, he studied Civil Engineering at Universität Stuttgart, Germany where he earned his Dipl. Ing – degree in 1958 and his Dr.-Ing. degree in 1966. In 1954, he received a Fulbright scholarship for studies in USA at Colorado State University (CSU), where he received an MS in Irrigation Engineering. Consecutively, he was employed at CSU until 1969 as research engineer and professor. After a year as Visiting Scientist at the Argonne National Laboratory (USA), he returned to Germany in 1970, where he was appointed full professor at the Universität Karlsruhe (by now KIT: Karlsruhe Institute of Technology), which remained his professional base for several decades until retirement.

In Colorado, he got engaged in basic hydraulic research and in experimental investigations using wind tunnel studies exploring wind forces as a basis for building aerodynamics. From the beginning, his interests were farsighted and oriented towards a good balance of basic research investigations, applied research and model studies and engineering application.

In Karlsruhe, he started forming a new institute with focus on addressing the future needs of the profession. His chair was named „Institute for Hydrology and Water Resources Management", which soon took up a leading position in these fields both in Germany and also internationally. With a broad range of basic research projects (60 doctoral dissertations) and a large number of application oriented projects and consulting, Erich Plate and his team had a growing impact on German water research and management practice.

The teaching obligations were demanding, and Erich Plate was an engaged and inspiring teacher. He understood to motivate his students, involve them in model studies and research projects and thus to recruit the top students as assistants and doctoral candidates. He managed to communicate with students on a very personal level and thus to generate a good team. And he loved to play tennis. So during busy times often lunch was skipped and replaced by a tennis match – much to the liking of his assistants (and also myself), who got the chance for a double at the center court of the university, which was reserved for professors, thus avoiding long waiting times.

From the beginning, Erich Plate was engaged in national and international organizations. Soon he was elected as a member of the DFG KOWA (DFG: German Research Foundation: main sponsor of basic research in Germany. KOWA: Commission on Water Research).He acted as chairman of the DFG KOWA from 1975 until 1989. The goal of KOWA was to overcome traditional barriers between the various diciplines in natural sciences and engineering involved in water research and to initiate new programs for future-oriented basic research. And Erich Plate was the ideal candidate for this task.

Under his chairmanship, the DFG KOWA has also considered international programs and has actively promoted them. In particular, this concerned the UNESCO – IHP (International Hydrological Program) and the IDNDR (International Decade of Natural Disaster Reduction). In addition to his IAHR activities, Erich Plate acted in several international functions, e.g. as director of the IWRA (International Water Research Association), as president of COWAR (Committe on Water Research) of the ICSU (International Council of Scientific Unions) and as Advisory Council member for the International Research and Training Center on Erosion and Sedimentation (IRTCES).

For six decades, Erich Plate has been actively engaged in IAHR – originally the International Association for Hydraulic Research, in the meantime renamed into International Association for Hydro-Environment Engineering and Research. This redefinition reflects the developments in the association in the course of time, which to some extent have been foreseen and supported by Erich Plate and others. He served as council member and vice president from 1973 to 1979 and as president from 1985 until 1989. In 1993, he was awarded honorary membership of IAHR. Finally, he served 1998 until 2000 as "Acting Secretary General" to help manage the personal changes in the IAHR secretariat

The IAHR Congress 1977 in Baden-Baden, Germany was a highlight for both IAHR and for German water research. Erich Plate and Eduard Naudascher succeeded with their application, and Erich Plate became chairman of the Local Organizing Committee. The 3 Karlsruhe water institutes provided the organisation – and faced a very high financial risk with worst case deficit expectations far beyond their possibilities. But Erich Plate found a good solution: he organized a meeting of all German water institutes at which the 25 partners agreed to bear a possible deficit in equal parts – a good option of risk management in the preparatory phase, which finally was not needed since the highly successful Congress did not produce any deficit. But Erich Plates initiative was the start of a series of annual meetings among the German water institutes for cooperation and networking over decades.

Erich Plates contributions in research , international and interdisciplinary cooperation and networking were recognized by numerous honors and prizes. To mention just one example: In 2000 he was awarded the Henry Darcy MedaL of the EGS (European Geophysical Society) in recognition of his fundamental contributions in water resources research and applied hydrology. In these fields, he was the leading scientist in Germany for several decades. He opened the eyes of many water scientists and engineers on the need for a sustainable development and use of water resources. And he was one of the first to urge the scientific community for a multi-disciplinary approach in order to cope with the future challenges in water resources management

After his retirement in 1997, he remained still very active as „Professor Emeritus" for another two decades. And the inspiring and forward looking program of Erich Plates institute was the source for many of his doctoral students for a good start into careers in leading functions in research, environmental administration and as consulting engineers.

Erich Plate will be remembered as a highly valued and admired colleague, mentor and good friend. He is survived by his wife Gabriele and their 3 children with families.

*By Helmut Kobus,*
*Institute of Water and Environmental Systems Modelling, Universität Stuttgart, Germany*

# NEW IAHR PRESIDENT'S MESSAGE

I am very honoured and humbled by the outpouring of trust in me by IAHR colleagues – my sincere thanks to all those who have supported me. I would also like to congratulate the success of the other elected members of the Council. I would like to express my deep appreciation to the outgoing Executive Committee and Council led by Professor Peter Goodwin who have built an excellent foundation for the new Council to work from.

IAHR has always been a community of top scholars and researchers in different fields of water science and engineering, a truly international organization that produces high quality knowledge products that lead the industry; and it has a unique family spirit that ties young members to more experienced mentors. It is an excellent platform for hydraulics researchers. We are a prestigious organization with a proud history.

However, to stay successful as an organization we have to align ourselves with the needs of the users of water engineering needs of the users of water engineering research. We cannot afford to stand still and be complacent while the world changes. We have to race for relevance. We have to work more coherently as a team to capture global R&D and knowledge transfer opportunities and to contribute more visibly to solving the grand challenges of our times.

Water and environment rank high on the policy agenda of many governments in the coming decade. Population growth, urbanization and climate change give rise to many water, energy and food security issues – and many infrastructure needs. During the recent Panama Congress IAHR hosted a global forum on "Adaptive management in the face of climate change" – just three weeks ahead of the looming threats profusely propounded by many political leaders in the United Nations Climate Action Summit on September 23, 2019. The "Second Machine Age" (MIT press 2014) is also bringing many opportunities for the next generation of water environment research and practice: e.g. smart water management systems for climate resilient cities. Nature-based solutions to many environmental problems will offer a continuing stimulus to exciting developments at the interface of ecology, hydraulics and hydrology, and system science – besides job opportunities for young engineers and professionals.

In the past few months IAHR leadership has deliberated extensively on a new Strategic Plan which will be refined and announced at the beginning of the 2020. I will work with our new Executive Director, Tom Soo, and Executive Committee colleagues to implement the vision. Some of the messages in my election statement echo the Strategic Plan:

(i) **Increase global presence** – enhance connection with major regions of the world that are under-represented – including Africa - and with other global forums. This requires the development of new international collaborative initiatives and business models to create and market the value of the collective synergy of our members. Mount dedicated efforts to work with institute members to create more impact.

(ii) **Inspire, disseminate and catalyze state of the art knowledge** - IAHR can provide a platform for us to redefine the field and in so doing create stimulus to our thinking, foster innovations and create new water industries. We will actively engage IAHR scholars and practitioners in high level agenda setting inter-disciplinary forums and publications. An inter-disciplinary task force will be created to enhance oversight, efficient direction and impact of the high quality monographs and knowledge products produced by our technical committees. In an era where subjects like molecular neuroscience, Big data and robotics, and climate change command the attention of policy makers, we need to re-invent ourselves to utilize the powerful research of our Committees to address grand challenges like climate change and water sustainability. We can do this if we share this new vision, and look beyond our own narrow interests.

(iii) **Promote diversity and enhance international collaboration.** In achieving the above goals, we will maintain our policy of actively promoting diversity and international collaboration. We welcome ideas and suggestions from young members and will build a sustainable strategy to increase IAHR membership.

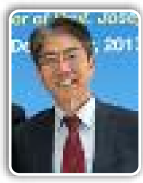
**New beginnings:**
With the support of our host institutes in Beijing and Madrid, I believe all the above is possible. IAHR is about "water connecting the world". In this age where we see many conflicts due to globalisation and the "Clash of Civilisations" (as expounded by the Harvard scholar Samuel Huntington), IAHR can do a lot to help the world achieve sustainability through providing a connection and network, a flow of global expertise, and capacity building. Perhaps what I can offer most is to take the good ideas from colleagues and make them work.
I always remember this saying: "Ideas don't work unless we do." I look forward to working with all of you to move this great organization forward.

*Joseph Hun-wei Lee, IAHR President*

# NEW IAHR COUNCIL FOR 2020-2021

(NB R* means eligible for re-election for a second term)

**President**



**Prof. Joseph Hun-wei Lee**
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HONG KONG, CHINA

(elected until 2021 R*)

**Vice President Americas**

**Prof. Robert Ettema**
Colorado State University
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**Prof. Dalila Louydi**
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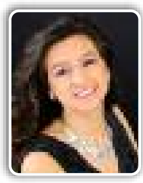
MOROCCO

(elected until 2023)



**Dr. Veronica Minaya**
Escuela Politécnica Nacional
ECUADOR

(elected until 2023)

**Prof. Vladimir Nikora**
University of Aberdeen
UK

(elected until 2021)

**Prof. Ioan Nistor**
University of Ottawa
CANADA

(elected until 2023)

**Dr. Ioana Popescu**
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# Invitation for Nominations

# 9th Award (2020)

## Nominations open online until 31 December 2019


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