Fate and Behavior of Pharmaceuticals in Groundwater of Arid and Semi-Arid Climates - Examples from the Lower Jordan Valley

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Abstract

The presented thesis about the accumulation potential of pharmaceutical traces in groundwater of arid and semi-arid climates evolved from the detection of elevated pharmaceutical concentrations in the groundwater of the Jordan Valley in comparison to the contributing surface water. This result in the principle question whether persistent trace substances, e.g. X-ray contrast media, might enrich in groundwater in the long term under such conditions and essentially whether they ultimately reach ecotoxicologically relevant levels. Field investigations were conducted in two areas: The Lower Jordan Valley and the Wadis Shueib and Kafrein.

The extensive agriculture on the floor of the Jordan Valley is irrigated with "blended water", a mixture of locally pumped groundwater and treated wastewater which is conveyed to the area. This results in a continuous external input of wastewater-borne trace substances. A portion of the irrigation water is therefore used in circulation and subject to evaporation during each irrigation cycle. The hydromorphology of the Lower Jordan Valley comprises low precipitation and high evaporation rates and the Dead Sea as the final sink without an outlet. Groundwater in the area is mainly present in the quaternary sediments.

Wadi Shueib and Wadi Kafrein are two steep side wadis along the eastern escarpment of the Lower Jordan Valley. Urban areas are found especially in the upper parts of the wadis. Anthropogenic pollution contaminates almost all groundwater sources due to the inefficient or damaged wastewater infrastructure. Both catchments are characterized by karstic limestone and dolomite formations. Shallow groundwater flows in two interconnected aquifer systems in depths of 50 to 75 m.

Within the framework of a five year sampling campaign, the temporal and spatial occurrence of wastewater-borne contaminants like different pharmaceuticals, *E.coli*, and nitrate were evaluated in the study areas.

In addition to the field studies, lab scale experiments on unsaturated columns were conducted in order to investigate the effect of evaporative accumulation of the two pharmaceuticals bezafibrate and carbamazepine. Both substances showed accumulation

under microbiologically inhibited conditions. Accumulation rates corresponded to the volume of evaporated water and were similar to conservative species like chloride and bromide. These experiments indicate the accumulation potential of pharmaceuticals with high persistence against biodegradation. Thereby, the general potential for evaporative enrichment of pharmaceuticals could be demonstrated for the first time. This potential should be included in risk assessments in the future. In respect to the Lower Jordan Valley however, the experiments under near to natural conditions did not indicate any health risks arising from the application of treated wastewater in agriculture for the near future.

Within the scope of the field investigation, the accumulation effect could not be proven on a statistically evident level (e.g. by continuous increasing concentrations over time). However, essential information could be gathered about contamination dynamics of the local groundwater by evaluating the spatial and temporal distribution of contaminants and associated concentrations. The X-ray contrast media diatrizoic acid, a standard diagnostic until its ban in 2000, showed a widespread occurrence in the groundwater while it was almost completely absent in treated wastewater and surface water. At the same time, iopamidol, a potential substitute for diatrizoic acid, showed increasing detection rates over the sampling period. These results give important indications on changes in prescription and application practice.

This effect has been consistently described in a conceptual model for contamination, transport, and leaching pathways for Wadi Shueib and Wadi Kafrein. The correlation of nitrate and pharmaceutical concentrations in both wadis strongly indicate that the nitrate contamination of groundwater originates from leaking sewers and cesspits. The number of detected substances at the sampling locations was also correlated with increasing concentrations of wastewater indicators such as nitrate and *E. coli*. Eventually, both field studies demonstrate the excellent potential of pharmaceuticals as tracers for anthropogenic contamination.

Kurzfassung

Ausgangspunkt der vorliegenden Arbeit zur potentiellen Anreicherung von pharmazeutischen Spurenstoffen im Grundwasser semi-arider und arider Gebiete war ein Spurenstoffscreening im Grundwasser, Oberflächenwasser und Abwasser des Jordantals. Erhöhte Konzentrationen von Pharmazeutika im Grundwasser verglichen zum Oberflächenwasser warfen dabei die Frage auf, ob sich persistente Spurenstoffe, wie z.B. Röntgenkontrastmittel, unter den gegebenen Bedingungen über längere Zeiträume im Grundwasser anreichern und ob hierbei ökotoxikologisch relevante Konzentrationen entstehen können. Die daran anknüpfenden Untersuchungen dieser Arbeit wurden in zwei Regionen durchgeführt: Dem Unteren Jordantal und den Wadies Shueib und Kafrein.

Am Talboden des Untere Jordantals wird extensive Bewässerungslandwirtschaft betrieben, wozu eine Mischung aus lokalem Grundwasser und importiertem Klärwasser genutzt wird. Dies bedeutet einen kontinuierlichen externen Stoffeintrag, der durch die zusätzliche Grundwasserentnahme zur Bewässerung zyklisch der Verdunstung ausgesetzt wird. Die Hydromorphologie ist charakterisiert durch geringen Niederschlagsraten, hohen Verdunstungsraten und dem Toten Meer als abflussloser lokaler Senke. Grundwasser befindet sich hauptsächlich in quartären Sedimenten.

Die Wadis Shueib und Kafrein sind Seitentäler an der östlichen Kante des Jordantals die nach Westen hin steil abfallen und im oberen Drittel stark urbanisiert sind. Das Grundwasser ist durch anthropogene Schadstoffeinträge häufig kontaminiert, was sehr wahrscheinlich auf eine z.T. unzureichende oder defekte Abwasserinfrastruktur zurückgeführt werden kann. Grundwasser findet man in der durch verkarstete Kalkstein- und Dolomitformationen geprägten Gegend in zwei Aquifersystemen in Tiefen von 50 bis 75 m.

Im Rahmen einer mehrjährigen Beprobungskampagne wurde die zeitliche und räumliche Verteilung verschiedener verschmutzungsrelevanter Parameter, wie Pharmakarückstände oder die Konzentration von Nitrat und coliformer Bakterien erfasst und ausgewertet.

Parallel zu den Untersuchungen im Gelände wurde der Anreicherungsprozess im Labormaßstab anhand von ungesättigten Säulenversuchen untersucht. Im Laborversuch

konnte hierbei für zwei Substanzen (Bezafibrat und Carbamazepin) eine evaporative Anreicherung unter biologisch inhibierten Bedingungen nachgewiesen werden, welche sich in der Größenordnung der verdunsteten Wassermenge bewegt. Die Akkumulationsraten waren ebenfalls in der gleichen Größenordnung wie die der konservativen Stoffe Chlorid oder Bromid. Die Experimente zeigten das evaporative Anreicherungspotential von abbauresistenten Pharmaka welches hier zum ersten Mal nachgewiesen werden konnte. Dieser Prozess sollte bei zukünftigen Risikobetrachtungen berücksichtigt werden. Eine Übertragung der Laborergebnisse unter naturnahen Bedingungen auf das Untere Jordantal zeigt auf absehbare Zeit keine Gesundheitsrisiken durch die Verwendung von Klärwasser in der Landwirtschaft.

Im Rahmen der Beprobungskampagne konnte der Anreicherungseffekt im Untersuchungsgebiet durch über die Zeit stetig ansteigende Schadstoffkonzentrationen im Grundwasser nicht verifiziert werden. Dennoch lassen sich anhand der räumlichen und zeitlichen Spurenstoffverteilung und Konzentrationsentwicklung wichtige Informationen über die lokale Kontaminationsdynamik im Grundwasser ableiten. Die großflächige Verbreitung des Röntgenkontrastmittels Amidotrizoesäure (Standardsubstanz in der Diagnostik bis zur Vermeidungsempfehlung seit 2000) bei fast völliger Abwesenheit im Klär- und Oberflächenwasser, sowie über den Untersuchungszeitraum steigende Detektionsraten von Iopamidol (ein möglicher Ersatzstoff) geben Hinweise auf Änderungen in der Verschreibungspraxis.

Anhand eines konzeptionellen Models konnte die Eintrags-, Transport und Auswaschungsdynamik der beiden Röntgenkontrastmittel für das Wadi Shueib beschrieben werden. Die Korrelation der Pharmaka- und Nitratkonzentration im Grundwasser der urbanen Einzugsgebiete Wadi Shueib und Wadi Kafrein weist auf Leckagen im Kanalsystem und in Klärgruben als Quelle der Nitratkontamination hin. Die Anzahl der an den Messstellen gefunden pharmazeutischen Substanzen korrelierte ebenfalls mit den Konzentrationen anderer typischer Abwasserinhaltsstoffe wie Nitrat und *E. coli* und kann somit als Abwasserindikator genutzt werden. Beide Feldstudien zeigen, dass persistente Spurenstoffe als nützliche Tracer für anthropogene Kontamination eingestuft werden können.

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Table of Contents

1. Introduction

1.1. Thesis structure

The present study is a cumulative PhD thesis enclosing three peer reviewed publications:

1. Zemann, M., Wolf, L., Pöschko, A., Schmidt, N., Sawarieh, A., Seder, N., Tiehm, A., Hötzl, H., Goldscheider, N., (2014): *Sources and processes affecting the spatio-temporal distribution of pharmaceuticals and X-ray contrast media in the water resources of the Lower Jordan Valley, Jordan*. Science of the Total Environment 488-489, 100-114.
2. Zemann, M., L. Wolf, F. Grimmeisen, A. Tiehm, J. Klinger, H. Hötzl and N. Goldscheider (2015): *Tracking changing X-ray contrast media application to an urban-influenced karst aquifer in the Wadi Shueib, Jordan.* Environmental Pollution **198**: 133-143.
3. Zemann M., Majewsky M., Wolf L. (2016): *Accumulation of pharmaceuticals in groundwater under arid climate conditions – Results from unsaturated column experiments*. Chemosphere **154**, pp 463-471.

They first focus on pharmaceutical fate and occurrence in an arid and a semi-arid environments, namely the Lower Jordan Valley (LJV) (chapter 2) and the Wadi Shueib (chapter 3). Second they deal with the potential of pharmaceuticals to accumulate in groundwater in an arid climate which is demonstrated in laboratory experiments (chapter 4). The introduction (chapter 1) contains the thesis framework, the methodology and the geology and hydrogeology of the area investigated.

Chapter 2 provides the main aspects of spatial and temporal occurrence of pharmaceuticals in the LJV. Against the background of intense wastewater reuse in agriculture, the evolution of pharmaceutical detections and concentrations in groundwater, surface water and treated wastewater is discussed with a special focus placed on the contaminant origin and the transfer and diversion systems of raw water and wastewater. Furthermore, a small literature study presents degradation rates, sorption and elimination rates for two pharmaceuticals derived from multiple laboratory batch and column experiments.

Chapter 3 particularly deals with the fate and occurrence of X-ray contrast media in an urban influenced limestone aquifer in the Wadi Shueib. By correlating well known pollution parameters such as nitrate and *E. coli* and the number of detected pharmaceutical substances at each sampling location, this parameter established a suitable tool to assess pharmaceutical pollution on a semi-quantitative level. Also, a conceptual model presents the flow and storage of diatrizoic acid in the karst system and the subsequent release of residual concentrations from the matrix.

Chapter 4 investigates the potential of evaporative pharmaceutical accumulation by column studies. Within this chapter the matter of relative, evaporative accumulation is discussed. First, the general flow mechanics and water mass balances are presented to demonstrate the overall successful and uniform operation of the experimental setup. The relevant physical, biological and chemical processes for pharmaceuticals are then explained in the context of the experimental results. Further on, the accumulation of pharmaceuticals is demonstrated mainly under biologically inhibited conditions while results for one persistent substance indicate the potential under near to natural conditions. Accumulation factors of pharmaceuticals were calculated and compared successfully to the evaporation rates and other conservative substances like chloride and bromide. The relevance of the results regarding their transferability to natural settings is discussed as well.

1.2. Funding framework – SMART & BOMOCIS

The presented work was initiated and realized within the framework of two projects. Firstly, the SMART-project (SMART = **S**ustainable **M**anagement of **A**vailable Water **R**esources with Innovative **T**echnologies) which focuses on the topic of "Integrated Water Resources Management" (IWRM) of the Lower Jordan Basin and was funded by the BMBF (German federal Ministry of Education and Research; grant numbers of project phases I and II: 02WT9724 & 02WM1079). Secondly the BOMOCIS-project (**B**ehavior **o**f **M**obile **O**rganic **C**ompounds **i**n the **S**ubsurface), a KIT startup program to facilitate inter facultative cooperation.

The main goal of the SMART-project was thereby the development of transferable solutions for an integrated water management in the LJV (see Figure 1). Priority was set to the sustainable improvement of water quality and the available quantities. Within this semi-arid to arid region, water has been scarce for centuries and different countries need

to share this limited resource. Due to the political situation, this has often been complicated. While the different religious backgrounds and past belligerence already generate a tense political environment, the recent political development of the Arabic spring increased pressure on the water resources due to refugees from surrounding countries. The SMART-project aims to increase the water availability and quality in the LJV by an integrative approach, including new treatment techniques, the use of formerly unused or unavailable water sources like brackish water, the optimization of usage and storage of seasonal flashfloods and by the raising of awareness for conservation and sustainable water usage in schools. The project is aiming to consider not only single topics, but to include all relevant aspects from available and innovative techniques, different water resources, decision support for the local decision makers, awareness raising and capacity building by an integrative approach.

The research objective of the BOMOCIS-project mainly focuses on the transport behavior of pharmaceuticals in the unsaturated porous underground. A special focus was put thereby on accumulation processes induced by evaporation as it may take place in arid environments. The investigations were mainly drawn from laboratory experiments, in particular by column studies, simulating percolation of treated wastewater (TWW) through the unsaturated zone after application e.g. in irrigation.

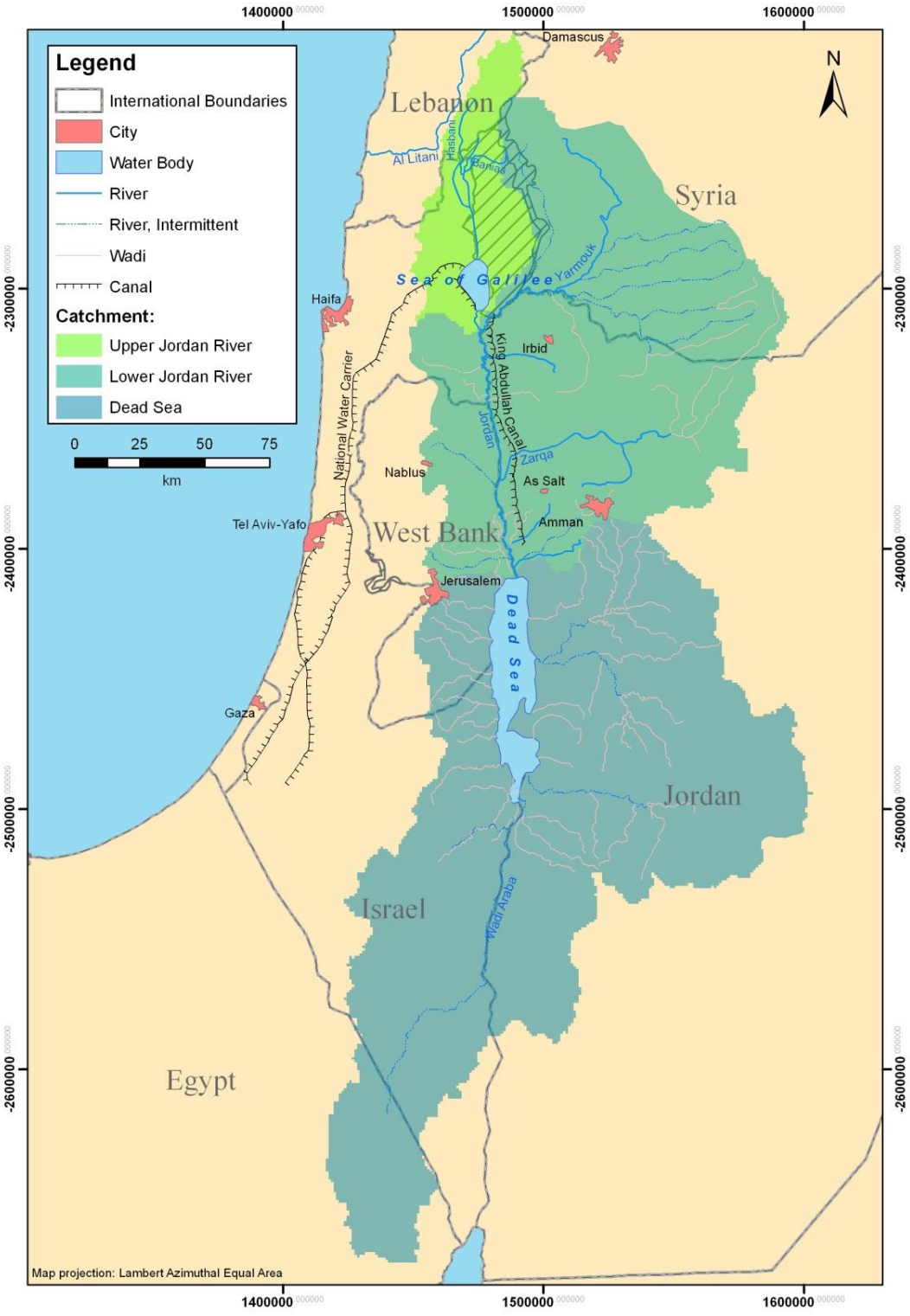


Figure 1: Overview of the Jordan and Dead Sea Basin with the investigated area and catchment of the Lower Jordan River in the middle part (Riepl, 2012).

1.3. Motivation of the thesis

Within the last decade, the country of Jordan has been faced with a severe population growth. This was mainly due to the high natality but additionally boosted by several refugee flows starting with the Iraq War in 2003. The overall population number therefore increased from 5.9 Mio in 2008 to above 8 Mio in 2014 (MercyCorps, 2014). Due to the arid and semi-arid climate, water resources were already scarce in the past and have been overused for years, as is documented by the decline of groundwater levels and the water level of the Dead Sea (Salameh & El-Naser, 2000). The increasing water demand within the recent years is an enormous challenge for the local authorities (i.e. the Ministry of Water and Irrigation) since drinking water supply is not only a matter of demand, but as well needs to meet quality aspects. Like many other developing countries, Jordan suffers from insufficient water management and defective water infrastructures. High rates of losses in the supply system come along with lacking or damaged wastewater infrastructure as leaking sewers or inefficient and sometimes even missing wastewater treatment. Infrastructural losses were quantified around 30 % of the total production (MercyCorps, 2014). Due to continuous water scarcity, treated wastewater is used for irrigation in agriculture in the LJV as a supplement for formerly used drinking water. However, locations of water abstractions were often closely spaced to e.g. infiltrating wastewater especially as in Jordan drinking water is abstracted almost solely (80 %) from groundwater (Properzi, 2010). As a consequence, the water quality has been deteriorating for years whereas the demand is further growing.

While most of the problems were apparent, any approach towards a sustainable solution seems complex as it needs to consider aspects of all kinds of different fields of expertise. Though, if different experts add their knowledge on a specific field to a shared base it might later help any decision maker in the water sector towards an optimized planning or realization of improvement activities. The main motivation of this thesis was, therefore, to investigate, understand and describe the interconnected problem of groundwater contamination, drinking water supply and wastewater treatment for the area of the LJV in a sufficient way. Besides, it was tried to evaluate the risk arising from the intensive use of treated wastewater in agriculture for the local catchment. This was furthermore based on the evaluation of tempo-spatial occurrence of several pharmaceutical trace substances. Pharmaceuticals here provide the opportunity of well-defined sources and therefore can

help to distinguish the origin and flow paths of groundwater contamination and improve the overall understanding of the local contamination dynamics.

1.4. Objectives and hypotheses

The evaluation of initial pharmaceutical screening results in groundwater, surface water and final effluent of several wastewater treatment plants (WWTP) in the Jordan Valley in 2007 showed, that at that time several of the detected substances had higher concentrations in the groundwater than in the contributing surface water and treated wastewater (Wolf et al., 2009). Amongst them were different X-ray contrast media, namely iopamidol and diatrizoic acid, which both were known to be rather persistent against biodegradation. Due to the local conditions with high temperatures, high evaporation rates and almost no precipitation, an evaporative enrichment of persistent substances seemed one possibility besides others, like seasonal effects or changes in medication. As soil salting is also a known problem in such areas, it was hypothesized that pharmaceuticals might accumulate in groundwater under such specified conditions which are described as follows:

- A closed catchment with no or negligible outflow, i.e. the Jordan Valley with the Dead Sea acting as the final sink.
- High temperatures and high evaporation rates, i.e. arid climate.
- A continuous replenishment of organic trace substances or pharmaceutical residues; here, the transfer and use of treated wastewater for irrigation in agriculture.

Due to the limited amount of available data a multi-step approach was done to investigate the problem on different scales. First, unsaturated column experiments were chosen to investigate the theoretical potential of evaporative enrichment under near to natural conditions. Results were later interpreted under the aspect of human hazards and long term impacts. The following objectives were thereby investigated (Zemann et al., 2016):

- Can we simulate the accumulation of pharmaceuticals in a laboratory experiment under near to natural conditions?
- What can we predict from laboratory results?
- Is the accumulation process capable to become a dangerous fact in groundwater quality studies?

In parallel to the laboratory investigations, field studies were conducted in the LJV to increase the data amount and obtain long term data. They were interpreted on the temporal and spatial level regarding long term accumulation effects and pharmaceutical mass fluxes in the area. The following objectives were hereby investigated (Zemann et al., 2014; Zemann et al., 2015):

- What substances occur in the different water types of the LJV and in the urban karst aquifer of Wadi Shueib and Wadi Kafrein? What concentration levels do they reach?
- Is there any proof of an accumulation processes in the groundwater?
- How can we make use of pharmaceutical occurrence and concentrations above the qualitative level?
- Are those substances suitable anthropogenic tracers?
- Is it possible to distinguish tempo-spatial trends or characteristic distribution patterns regarding concentrations and occurrence?
- Can trends and distribution patterns be related to the administration or known persistence of pharmaceuticals and do findings match with local infrastructures for wastewater distribution in irrigated agriculture?

1.5. Methodology

1.5.1. Field studies

As mentioned in chapter 1.4, occurrence and fate of pharmaceuticals were investigated in two different aquifer systems:

- The shallow alluvial aquifer in the LJV with an intense reuse of treated wastewater in agricultural irrigation.
- The karstic limestone aquifer system in the Wadi Shueib which is heavily influenced by effluents from leaking sewers and cesspits of the upstream urban area of the city Salt.

To determine possible pollution sources and pharmaceutical flow paths, samples were taken from groundwater as well as from the contributing surface water and treated wastewater. The sampling strategy was thereby roughly oriented towards previous studies from Wolf et al. (2009). The area between the Dead Sea and the Lake Galilee was divided into three sections according to the local diversion of treated wastewater to the

farmers. The assumed degree of pollution, in terms of the amount of treated wastewater mixed to the irrigation water hereby was increasing from north to south, with highest rates close to the Dead Sea. The idea mainly follows the course of the national water carrier King Abdullah Canal (KAC) which is an open canal along the LJV (Figure 2). In the northern part, the KAC is conveying water from the Lake of Galilee and the Yarmouk River to a pumping station in Deir Alla. Further on, most of the water is transferred to Amman where it is used for drinking water after purification. Parts of the water are already diverted for irrigation along its course. After Deir Alla, water from the King Talal Dam, which is collecting the effluent of Ammans biggest WWTP As Samra, is lead into the canal (Figure 3). From here, this water, together with the remaining fresh water is delivered further south towards the Dead Sea. In this southern part, the canal is refilled and mixed with water from several dams. During the dry summer months, those dams were mainly fed by the effluent of upstream WWTP.

Figure 2: King Abdullah Canal in the southern LJV.

Figure 3: Inflow of King Talal Dam effluent into the King Abdullah Canal at Deir Alla.

Within four field campaigns (autumn 2008, spring 2011, autumn 2011, spring 2012) different water quality parameters were surveyed at a total of 24 sampling locations in the Wadi Shueib, the Wadi Kafrein and the LJV. The sampling (Figure 4) and analysis (Figure 5) procedure are described in detail in chapter 2.2.2. and chapter 3.2.3. The investigated parameters were:

- Pharmaceuticals, i.e. X-ray contrast media
- Major ions (K, Ca, Mg, Na, PO_4, Cl, SO_4…)
- Pysico-chemical parameters (electric conductivity, pH, temperature, oxygen demand, redox)
- *E. coli* and total coliforms

For the LJV, samples were taken from open groundwater, springs and wells. In each of the three areas, at least 3 groundwater samples and one contributing surface water sample were taken during each campaign. For Wadi Shueib and Wadi Kafrein, samples were collected from the main springs (in terms of discharge), some wells and the WWTPs. While most of these springs and wells (Shoreia, Baqquriah, Azzraqu, Um Attija, Yesidia) are still used for drinking water supply, water quality is a crucial matter, as any single detection of a pharmaceutical gives a hint of anthropogenic pollution by wastewater (leaking sewers and cesspits) or treated wastewater (infiltration of final effluent via wadi streams).

Figure 4: Sampling at Um Attija Well.

Figure 5: Field laboratory at Hazzir spring.

1.5.2. Laboratory studies

Beside the field investigations, the hypothesis of evaporative accumulation of pharmaceuticals was also examined on the laboratory scale. To illustrate the sole effect of evaporative accumulation and later evaluate possible risks and hazards, unsaturated column studies were conducted from March 2012 until August 2012.

Within the experiments, treated wastewater dotted with pharmaceuticals was trickled on sand filled columns (Figure 6). With the laboratory temperature set to 30°C, the evaporation of water should then lead to a relative accumulation of the pharmaceuticals in the outflow of the columns. Effects of biodegradation should be considered by the comparison with parallel experiments under microbiologically uninhibited and inhibited conditions. The amount of adsorbed substances was investigated at the end of the experiments by dividing each column into segments and analyzing the eluate of the soil samples.

The inflow water to the columns consisted of final effluent of the Karlsruhe Neureut WWTP. Stable inflow conditions were obtained by taking all treated wastewater at one time and stabilizing it by autoclave treatment. All experiments were conducted inside a climatic chamber (Figure 7), where 30 °C and 60 % relative humidity simulated a semi-arid environment.

Figure 6: Trickling of treated wastewater via cannulas on the column surface.

Figure 7: Climatic chamber.

1.6. Geology and hydrogeology of the investigated area

1.6.1. Structural geology of the Lower Jordan Valley

The Lower Jordan Valley stretches between the Lake Tiberias at -200 m bsl down to -415 m bsl at the Dead Sea with 8 to 15 km width extending over a length of 100 km. The elevation at its escarpments reaches 1200 m asl (Hötzl, 2009).

The Arava Dead Sea Jordan Rift Valley is the plate boundary which separates the Arabian continental plate from the African plate (Hötzl, 2009). It has developed since the pre-middle Miocene (~13 Mio years) as a left lateral (sinistral) transformation fault zone (McKenzie et al., 1970). The left lateral movement is postulated to be 107 km (Bender, 1974). Vertical displacements at faults are accompanying the eastern flanks of the rift system (Marcus and Slager, 1985). The whole Dead Sea Transform (DST) is approximately 1000 km long and connects the sea floor center of the Red Sea with the Alpine convergence zone in Turkey. Along the DST are several basins (see Figure 8) with a lengths range from 15 – 150 km, widths from 5 – 20 km and depths > 10km, especially along the southern half (Hötzl, 2009). They are interpreted as pull-apart basins (Garfunkel, 1981). The biggest amongst them shown in Figure 8 are: Gulf of Aqaba-Elat (A), Dead Sea (C), Sea of Galilee (D), Hula (E) and Ghab (F) basins (Ben-Avraham et al., 2012).

During the Quaternary, several lakes occupied this tectonic depression along the DST: Lake Amora (mid to late Pleistocene), the last interglacial Lake Samra, the last glacial Lake Lisan and the Holocene to modern Dead Sea (Waldmann et al., 2009). During the Miocene, the Dead Sea Basin (DSB) was filled by fluvio lacustrine deposits (Garfunkel, 1981). During the Pliocene, the Mediterranean Sea intruded into the DSB, forming the Sedom lagoon and deposited thick sequences of salts. After the disconnection of this lagoon, terminal lacustrine bodies (Amora, Samra, Lisan, Dead Sea) successively occupied the basin (Waldmann et al., 2009).

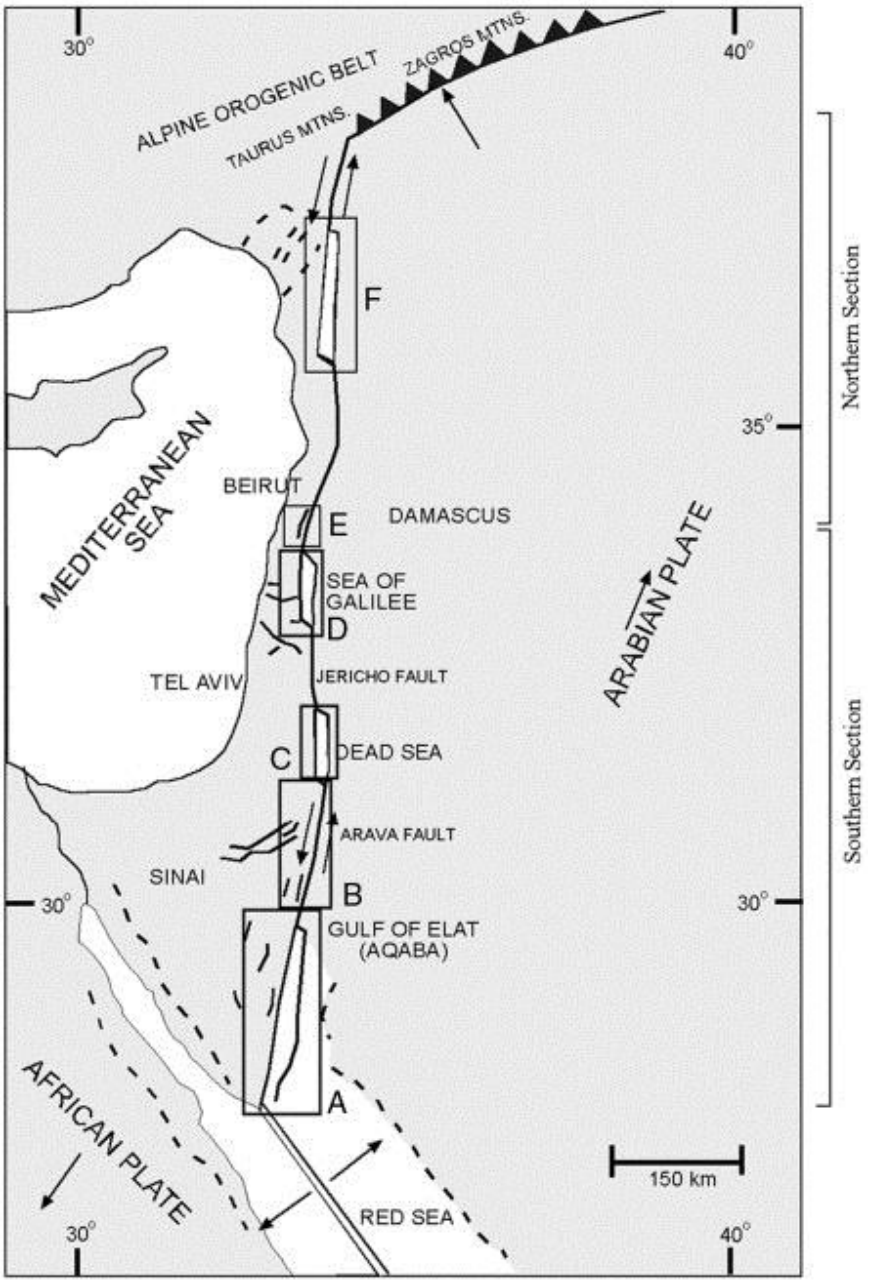


Figure 8: Pull apart basins along the DST (Ben-Avraham et al., 2012).

1.6.2. Stratigraphy in the Wadis Shueib and Kafrein

The taxonomy of the following chapter is used according to the Jordan 1:50,000 Geological Mapping Project of the National Resources Authority (NRA) which is based on Masri (1963). The following paragraphs are mainly based on Lenz (1999), Werz (2006), AlKhoury (2011), Sahawneh (2011).

The study area of Wadi Shueib and Wadi Kafrein are both located next to each other at the eastern escarpment of the LJV (see Figure 10). Wadi Kafrein is embedded between the two main fault systems of the area: The Amman Halabad structure and the Wadi Shueib

structure both originate from the early Tertiary Syrian folded arc system and later were additionally narrowed and squeezed out during the transform movement started from the Upper Miocene and continuing until today. Wadi Shueib borders Wadi Kafrein and the Wadi Shueib structure to the north.

The outcropping rocks in the Wadi Shueib area are of Jurassic age (Azab Group) to Upper Cretaceous age (Balqa Group) (Hahne, 2008). Outcropping rocks in Wadi Kafrein are from Crataceous ages (Aarda Formation) and younger (Lenz, 1999). The stratigraphy of both areas is rather similar comprising the main groups Kurnub, Ajlun and Belqua. At the end of the wadi towards the LJV, sediments of the Jordan Valley Group were deposited on the valley floor. The different groups, formations and notations for all of Jordan are given in Table 1. The outcrops of the important hydrogeological formations are given in the map in Figure 9.

1.6.2.1. Kurnub Group

The term Kurnub was introduced by Damesin around 1930 for sandstones in the northern Negev. This group is of the Lower Cretaceous age and consists mainly of fine to coarse grained, partly carbonaceous sandstones. They were intercalated by sandy dolomite, dolomitic limestone, siltstone and shale (Margane, 2002). Bender (1974) subdivided the group into a white (Aarda) and a variegated (Subeihi) sandstone unit, however today they are mainly mapped as one. The thickness of the Kurnub Group in Wadi Sheuib is around 250 m (Powell, 1989) while it is up to 150 m in Wadi Kafrein (Lenz, 1999).

1.6.2.2. Ajlun Group

The Ajlun Group developed in the Upper Cretaceous (upper Albian to Coniacian) and consists of five formations. Those were named by Masri (1963) from base to top: Naur (A1/2), Fuheis (A3), Hummar (A4), Shueyb (A5/6) and Wadi as Sir (A7). It consists of interbedded marl, marly limestone, limestone/dolomite units with a thickness in the investigated area of approx. 350 – 400 m (Powell, 1989). Most of the outcropping rocks in both wadis belong to the Ajlun Group.

1.6.2.3. Belqa Group

The term Belqa was first introduced by Quennel (1951). This group forms the most Upper Cretaceous to the early Tertiary age, namely from the Santonian to the Eocene. It mainly consists of chalk and limestone with partly high amounts of marl, chert, phosphate and siliclastic sand (Quennel, 1951; Parker, 1970). Wolfart (1959) subdivided into five

subunits named from base to top: Umm Ghudran (B1), Amman Al Hisa (B2), Muwaqqar (B3), Rijam (B4) and Wadi Shallala (B5) Formation. In Wadi Shueib and Kafrein only outcrops of the Umm Ghudram and the Amman Al Hisa Formations could be found (Werz, 2006; AlKhoury, 2011).

Table 1: Geological groups of Jordan (modified after Sahawneh (2011) and Ministry of Water and Irrigation (2004) (*taxonomy after McDonald (1965)).

System	Epoch	Group	Formation	Symbol*	Aquifer Unit
Quaternary	Holocene	Jordan Valley	Alluvium	Qal	Alluvium (Aquifer)
	Pleistocene		Lisan & Samra	JV3	
Tertiary	Pliocene			JV1-2	
	Miocene				
	Oligocene		Neogene		
	Eocene	Belqa	Wadi Shallala	B5	B4/5 (Aquifer)
			Umm Rijam	B4	
	Paleocene		Muwaqqar	B3	B3 (Aquitard)
Upper Cretaceous	Maastrichtian		Al Hisa Phosphate	B2b	A7/B2 (Aquifer)
	Campanian		Amman Silicified limestone	B2a	
			Wadi Umm Ghudran	B1	
	Santonian	Ajlun	Wadi as Sir	A7	
	Coniacian				
	Turonian		Shuayb	A5/6	A1-6 (Aquitard)
	Cenomanian		Hummar	A4	
			Fuheis	A3	
			Naur	A1/2	
Lower Cretaceous		Kurnub	Subeihi	K2	K1/K2 (Aquifer)
			Aarda	K1	
Jurassic		Azab	(Zarqua Group*)	Z2	Z1/Z2 (Aquifer)
Triassic		Ramtha		Z1	
Permian		Hudayb			

1.6.3. Quaternary sediments in the Lower Jordan Valley

The sediments in the JV between the Dead Sea and the Lake Galilee which are younger than the Belqa Group form the Jordan Valley Groups. However, even today the early history of the Jordan Valley Group is not fully known, due to the regularly changing sedimentation regimes. According to their local occurrence, the sediments in the JV were divided into the Jordan Valley Group 1 (JV1), Jordan Valley Group 2 (JV2) the Jordan Valley Group 3 (JV3) and the alluvial fans (Salameh, 2001). An overview comparing the Quaternary stratigraphy of Jordan and Israel according to their deposition environment is given in Table 2.

Table 2: Comparison of Quaternary stratigraphic tables from different authors for the Lower Jordan Valley.

<table>
<tr><th colspan="2">Geological age</th><th>Deposition environment (sea/lake)</th><th>Group</th><th>Formation</th><th>Group</th><th>Formation</th><th>Formation</th></tr>
<tr><td colspan="2"></td><td></td><td colspan="2">Israel (DSB)</td><td colspan="2">Southern LJV</td><td>Northern LJV, Lake Kinneret</td></tr>
<tr><td colspan="2"></td><td></td><td colspan="2">(Waldmann et al., 2009)</td><td colspan="2">(Al-Amoush et al., 2012, Bender, 1968)</td><td>(Hazan et al., 2005, Bender, 1968)</td></tr>
<tr><td rowspan="2">Holocene</td><td rowspan="2">< 15000 y</td><td rowspan="2">Dead Sea</td><td rowspan="11">Dead Sea Group</td><td rowspan="2">Zeèlim</td><td rowspan="11">Jordan Valley Group</td><td>Fluviatile and lacustrine sediments & Alluvial fans</td><td rowspan="2">Zeelim</td></tr>
<tr><td>Damya</td></tr>
<tr><td rowspan="3">Late Pleistocene</td><td rowspan="3">125000 – 15000 y</td><td rowspan="3">Lake Lisan</td><td rowspan="3">Lisan</td><td>Lisan</td><td rowspan="3">Lisan</td></tr>
<tr><td>Samra</td></tr>
<tr><td rowspan="3">Middle Pleistocene</td><td rowspan="3">800000 – 125000 y</td><td rowspan="2">Lake Samra</td><td rowspan="2">Samra</td><td rowspan="3">Ghor el Katar Basalt & Kufranja</td><td>Naharayim</td></tr>
<tr><td rowspan="2">Ubeidiya</td></tr>
<tr><td>Lake Amora</td><td>Amora</td><td>Abu Habil</td></tr>
<tr><td rowspan="2">Early Pleistocene</td><td rowspan="2">0.8 – 2.6 Mio</td><td rowspan="3">Sedom lagoon</td><td rowspan="3">Sedom</td><td>Ghor el Katar</td><td rowspan="3">Erk el Ahmar</td></tr>
<tr><td rowspan="2">Pliocene</td><td rowspan="2">5.5 -2.6 Mio</td><td rowspan="2">Shagur</td></tr>
<tr><td>Miocene</td><td></td><td>Pre-spreading river system</td><td></td><td>Hazeva</td><td></td><td></td></tr>
</table>

1.6.3.1. Hazeva Formation

Even though it is not present in the investigated area, the Miocene Hazeva formation marks the stage before the beginning development of the DSB (Niemi et al., 1997). The formation is deposited under an fluviatile freshwater environment and is present inside the DSB as well as on its shoulders. Thus it indicates that the Rift Valley was not developed and the sediments could be transported over the later rift. Parts of it contain quartz which is most likely delivered from southern Sinai and northwestern Arabia towards the Mediterranean Sea (Niemi et al., 1997).

1.6.3.2. Jordan Valley Group 1 (JV1)

These formations were deposited between the Upper Miocene until the Middle Pleistocene. Its deposition was accompanied by the transform movement with faulting and formation of the JV. The Shagur Formation consists of well cemented fluvio-limnic conglomerates with interbedded marls, travertines or claystone (Toll, 2007). It covers the older rock sequences unconformable and was strongly deformed by structural movement (Bender, 1974). The Ghor el Katar Formation is composed of alternating conglomerates, sandstones, marl, shale and fossil red soil and equivalent to the Grain Sabt series of Ionides and Blake (1939). The series shows deformation as well but to a lower degree (Bender, 1974). Both formations only occur in the southern part and middle of the LJV (Bender, 1968). Abu Habil Formation consists of hard conglomerate with partly limestone. It may be partly correlative to the Erk El-Ahmar series (Horowitz, 1979). The Kufranja gravel Formation consists of poorly consolidated gravel, which were correlated to the later mentioned Naharayim Formation south of Lake Galilee by Bender (1968). Both formations only show small outcrops in the investigated area (Al-Amoush et al., 2012). The Jordan Valley Group 1 features overall low porosities and permeability (Al-Amoush et al., 2012). It may have secondary porosities in some restricted locations which, however, may contain salt water. Their thickness may reach up to 350 m (Salameh, 2001).

1.6.3.3. Jordan Valley Group 2 (JV2)

The Jordan Valley group 2 consists of the Ubediya and the Samra Formation. Both were formed in the middle Pleistocene. They consist of conglomerates, sands, silt and clayey marls but no evaporates and covers the JV1 with a thickness of some 100 m (Salameh, 2001). The Ubediya Formation occurs only in the area southern to Lake Tiberias (Bender, 1968). It is rich in prehistoric remains, representing the oldest human implements ever found in the Middle East (Horowitz et al., 1973). It is covered by the Naharayim and the

Lisan Formation and was faulted and tilted by the middle Pleistocene tectonic movement of the central Jordan Valley (Horowitz, 1979). The Samra Formation is the lower, clastic part of the Lisan formation (Horowitz, 1979) which corresponds to the Hamamar member of Langozky (1961) (Niemi et al., 1997) as later referred to in the JV3. At its edges, it is overlain, underlain and inter-fingered with the later described Lisan Formation (Toll, 2007). Water inside the formation is generally salty, however in areas close to the slope there may emerge some fresh water due to lateral flows which substitutes the saline water (Salameh, 2001).

1.6.3.4. Jordan Valley Group 3 (JV3)

The Lisan Formation was deposited in the Lisan Lake (younger Pleistocene). It consists of thin marl layers (some mm thickness). The total thickness ranges from 30 m to several hundred meters. The deposit is highly gypsiferous and its salt content is very high (Salameh, 2001). Several members of this formation have been defined, based on lithological changes and erosional unconformities. As Langorzky (1961) divided it into the lower, clastic Hamamar member and the upper evaporitic Amiaz member, the Lisan of the Jordan Valley Group refers to the latter (Niemi et al., 1997). The Damya Formation, consists of brownish thin to medium bedded silty limestone and calcareous mudstone and overlies the Lisan Formation with a thickness of 0 to 14 m (Hötzl, 2009). Groundwater of the JV3 is typically saline. Older alluvial fans inter-finger with the JV3 and may contain fresh water in a few areas. The primary permeability is very low. The secondary may be high in some places due to gravel intercalations and dissolution of salts (Salameh, 2001).

1.6.3.5. Alluvial Fans

The Jordan Valley Group is overlain by alluvial sediment fans which are from post Lisan age. They are composed from uncemented gravel and sand at their apexes and silt and fine sand at their toes, so they fine up towards the Jordan River. They possess high permeability in the upstream areas and low in the downstream areas. Their thickness may reach up to 20 m and thin out totally towards the Jordan River. At the mouth of the wadis towards the LJV, the fans are reduced to rather small coarse channel fillings towards the wadis. Their deeper parts are contemporaneous with the Lisan Formation and replace the Lisan marls completely. Here, the aquifers of the Jordan Valley below the Lisan marls can be recharged from the wadis. Slope debris originating from escarpment foothills overlap with these fans (Salameh, 2001). They originate from the exposed formations at the rift side and were brought from the eastern side wadis with the winter floods.

1.6.4. Hydrogeology

The hydrogeological units in the research area can be subdivided into three main units:

- The Deep Sandstone Aquifer Complex (Ram-Zarqua-Kurnub)
- The Upper Cretaceous Carbonat Aquifer Complex
- The shallow sandy-gravelly Aquifer Complex (Tertiary - Quaternary)

Their classification, as well as the interbedded aquitards, is given in Table 1.

1.6.4.1. The deep sandstone aquifer complex (Ram-Zarqua-Kurnub)

This aquifer system includes the Ram, Zarqua and Kurnub groups. The Ram (Disi) aquifer is the deepest aquifer in Jordan und underlies the entire country (Ministry of Water and Irrigation, 2004). It consists mainly of sandstone and is not exposed in the catchment area (Margane, 2002). Its average thickness is about 1000 m (Ministry of Water and Irrigation, 2004). The Zarqua and Kurnub aquifers are mainly composed of variegated sandstone (Margane, 2002). The Zarqua aquifer can be characterized as a multi-layer bedrock aquifer with layers of different permeability and storability (Ministry of Water and Irrigation, 2004). The Kurnub aquifer is a fractured rock aquifer. Its thickness decreases from northwestern to southeastern Jordan (Ministry of Water and Irrigation, 2004). Kurnub and Zarqua aquifer both receive direct and indirect recharge at their outcrops on the eastern escarpment of the Jordan Valley and the northern part of the Dead Sea (Ministry of Water and Irrigation, 2004).

1.6.4.2. The Upper Cretaceous carbonate aquifer complex

This complex consists of an alternating sequence of limestones, dolomites, marl stones and chert beds. The total thickness in central Jordan is about 700 m. It can be divided in the Lower Ajlun aquitards and aquifers and in the Upper Ajlun A7/B2 aquifer (Ministry of Water and Irrigation, 2004). The Lower Ajlun contains different aquifers, while some of them (A1/2, A4) are mainly of local importance (Ministry of Water and Irrigation, 2004). Within this study two springs, Baqquriah in Wadi Shueib and Tujabyl in Wadi Kafrein, emerge from the A1/2 aquifer (Hahne, 2008; AlKhoury, 2011). All wells in Wadis Shueib extract groundwater only from these two aquifers as well (Riepl, 2012).

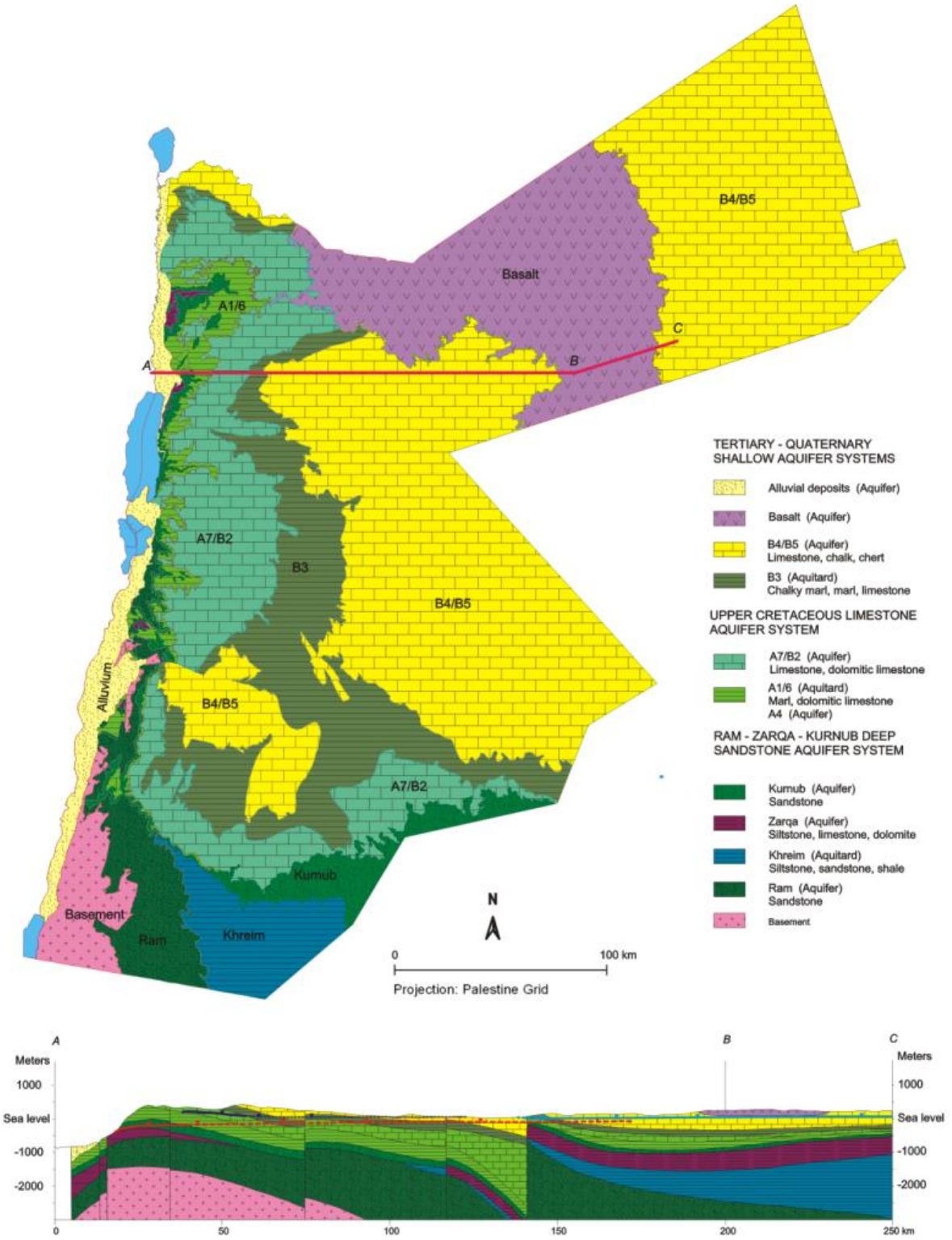


Figure 9: Outcrops of the main aquifer systems of Jordan (Ministry of Water and Irrigation, 2004)

The uppermost unit of the Ajlun Group (A7) and the lower part of the Belqa Group (B2) are considered as one hydrogeological unit. Due to multiple cracks and faults, both aquifer systems are interconnected and therefore named the Wadi Sir Limestone aquifer (A7/B2). It forms the most important aquifer for all of Jordan because of its vast extent and its relatively high permeability (Margane, 2002). Within the investigated area of Wadi Shueib and Kafrein, this aquifer complex is of major importance, as a great share of the local drinking water supply, is abstracted from it by wells or drained by springs (Azzraqu, Hazzir, Shoreia) (Hahne, 2008). The natural groundwater replenishment takes place at the outcrops of this group along the escarpment and in the Jordan highlands (Ministry of Water and Irrigation, 2004).

1.6.4.3. The shallow aquifer complex (Tertiary - Quaternary)

This aquifer system includes the B3 aquitard, the B4/5 aquifer, the Basalt aquifer and alluvial deposits. The B3 aquitard overlies the A7/B2 aquifer. It has a low permeability and forms a confining layer to the A7/B2 aquifer and B4/5 aquifer and is therefore regarded as an aquitard (Margane, 2002). The B4/5 aquifer consists of the Umm Rijam and the Wadi Shallala Formation. The Basalt aquifer is mainly outcropping in the eastern plateau. Both were not present in the investigated areas (Ministry of Water and Irrigation, 2004).

The groundwater in the shallow LJV aquifer is mainly found in the Quaternary deposits and is replenished by flood waters originating from the highlands. The deposition environment with decreasing gradients from the mountain foothills to the Jordan River course led to rapidly declining permeability towards the Jordan River. Therefore, infiltrating water at the foothills flows very slowly towards the Jordan River (Salameh, 2001). Freshwater is only present in the uppermost layers (gravel, sand, limestone and fluviatile deposits) as groundwater becomes salty when it comes into contact with the salt enriched deposits of the ancestors of the DS. They mostly inter-finger with the salty deposits and are therefore not extensive. Lens type groundwater bodies are also present in the alluvial fans of the side wadis (Salameh, 2001). Many wells in the Jordan Valley are slightly brackish with EC > 1,500 µS/cm. Groundwater salinity in the alluvial aquifers varies widely from 500 to 8000 mg/l. Pumping tests have indicated high transmissivities in alluvial fans and the Lisan gravel deposits. Low transmissivities have been encountered in the Lisan marls (Ministry of Water and Irrigation, 2004).

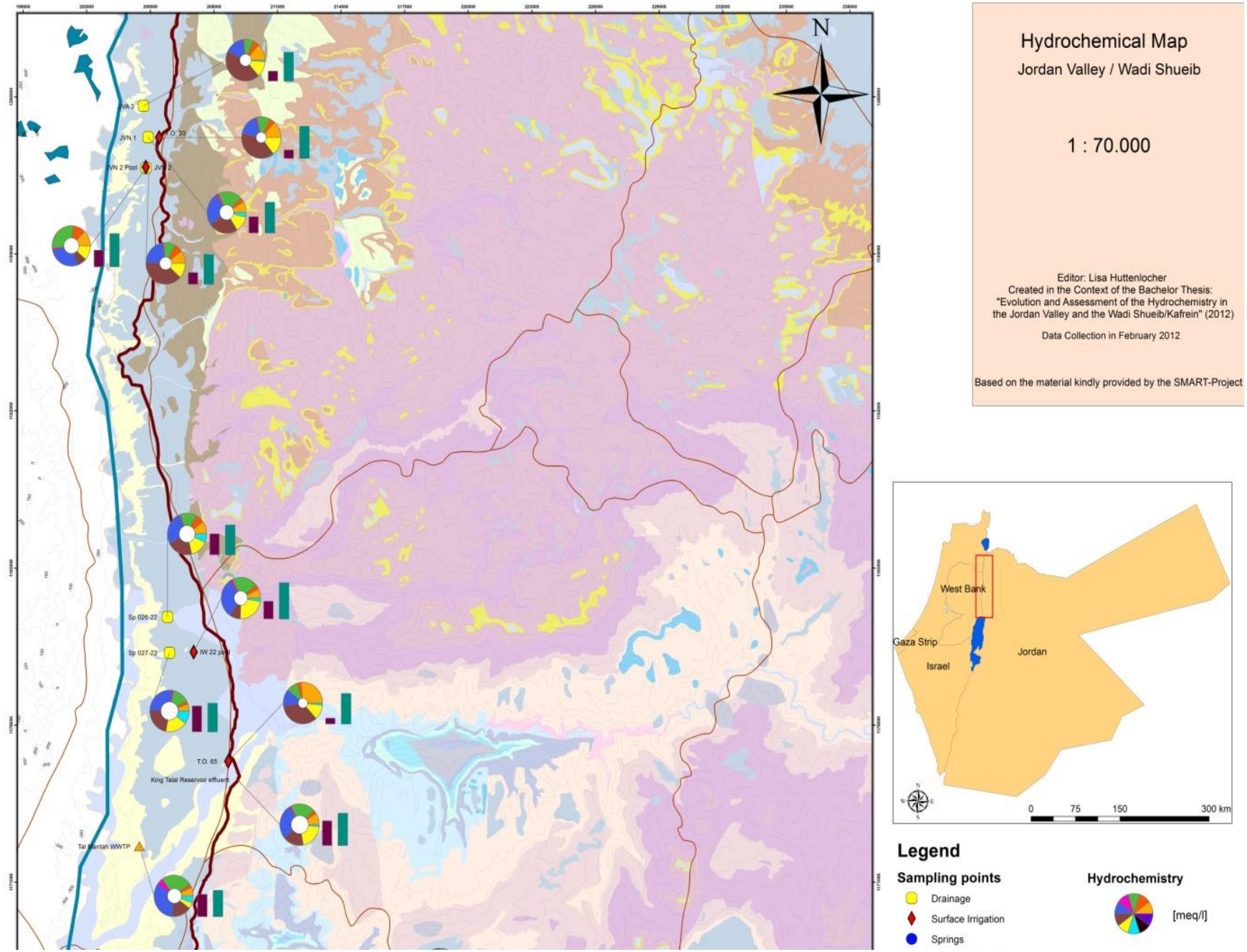
1.6.5. Hydrochemistry

Along with the pharmaceutical analyses, hydro-chemical standard parameters were collected during all sampling campaigns. A collection of these data was already evaluated within the framework of an unpublished bachelor thesis from Huttenlocher in 2012. A spatial plot of the data for February 2012 is given in Figure 10.

The groundwater in the LJV is almost solely used for irrigation. It is dominated by ions introduced via the treated wastewater like chloride and sodium. The groundwater sampling points show increasing amounts of chloride (EC likewise) from north to south and from east to west. This goes along with the increasing import of treated wastewater towards the valley floor which is later documented in chapter 2. It reflects the reuse of TWW in irrigation and the subsequent percolation towards groundwater. The surface water samples taken from King Abdullah Canal (KAC) show this pattern as well. Similar observations for the KAC were made also from Alkhoury et al (2010).

For Wadi Shueib, many of the sampled springs and wells were used to supply the cities Salt and Fuheis with drinking water. Full analysis for major ions were performed here on 30 groundwater samples (6 in 2008 and each 8 for each other sampling campaign). Twenty of these samples showed ion balances within a 5 % confidence interval and were plotted in a piper diagram (Figure 11). Eight of the residual samples had an error between 5 % and 10 % and 2 samples had an ion balance error of > 10 %, both were not considered within this study.

The samples from most springs are of Ca-Mg-HCO_3-(Cl) water type and are clustered in the piper diagram. The water of the Hazzir spring contains more sodium and is classified as Ca-Na-Mg-HCO_3-Cl type. Water from the Farkha spring is of Ca-Mg-Na-HCO_3-Cl type. Samples from Yesidia wells can be easily differentiated due to their abundance of sulfate. They are classified as Ca-Mg-HCO_3-SO_4 type water. This corresponds to the lithology (compare Figure 18) where all springs originate from As Sir limestone. Only Yesidia well is drilled down to the dolomitic limestone of the Hummar Formation.


N
JVN 3
JVN 1
T.O. 33
JVN 2 Pool
JVN 2
Sp 026-22
Sp 027-22
IW 22 pool
T.O. 65
King Talal Reservoir effluent
Tal Mantah WWTP
Hydrochemical Map
Jordan Valley / Wadi Shueib
1 : 70.000
Editor: Lisa Huttenlocher
Created in the Context of the Bachelor Thesis:
"Evolution and Assessment of the Hydrochemistry in the Jordan Valley and the Wadi Shueib/Kafrein" (2012)
Data Collection in February 2012
Based on the material kindly provided by the SMART-Project
West Bank
Gaza Strip
Israel
Jordan
0
75
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Legend
Sampling points
Drainage
Surface Irrigation
Springs
Hydrochemistry
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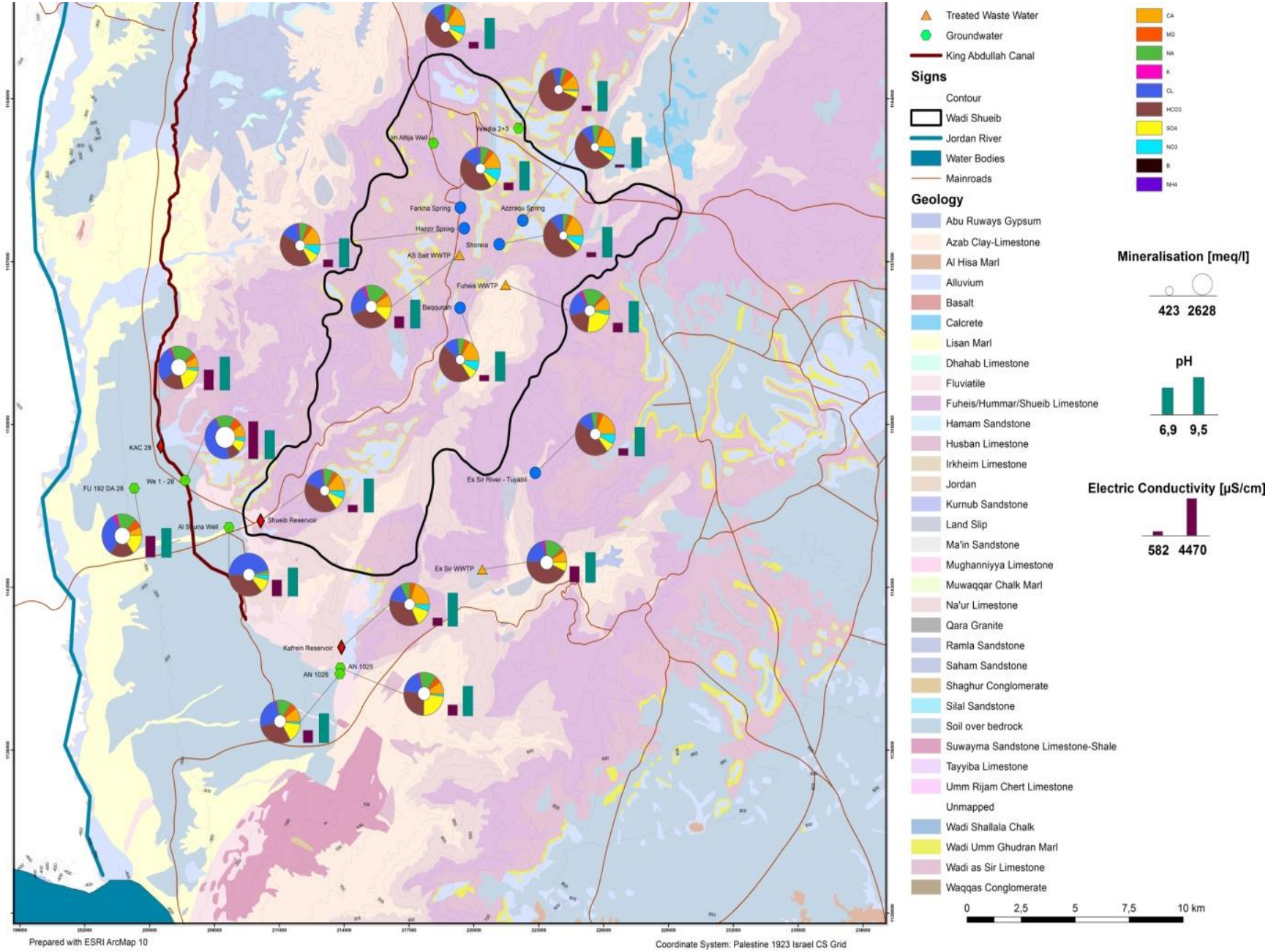

Figure 10: Hydrochemical and geological map for the LJV and the Wadis Shueib and Kafrein in February 2012 (Huttenlocher, 2012).

Mean nitrate concentrations range from 9.5 mg/l at Yesidia wells and 26.5 mg/l at Azzraqu spring up to between 30 and 40 mg/l at Shoreia spring, Um Attija well and Baqquriah spring (Table 11). Highest mean concentrations were found at Hazzir spring (53.0 mg/l) and Farkha spring (60.9 mg/l). Those values were reflected in a similar order by mean chloride concentrations and EC measurements.

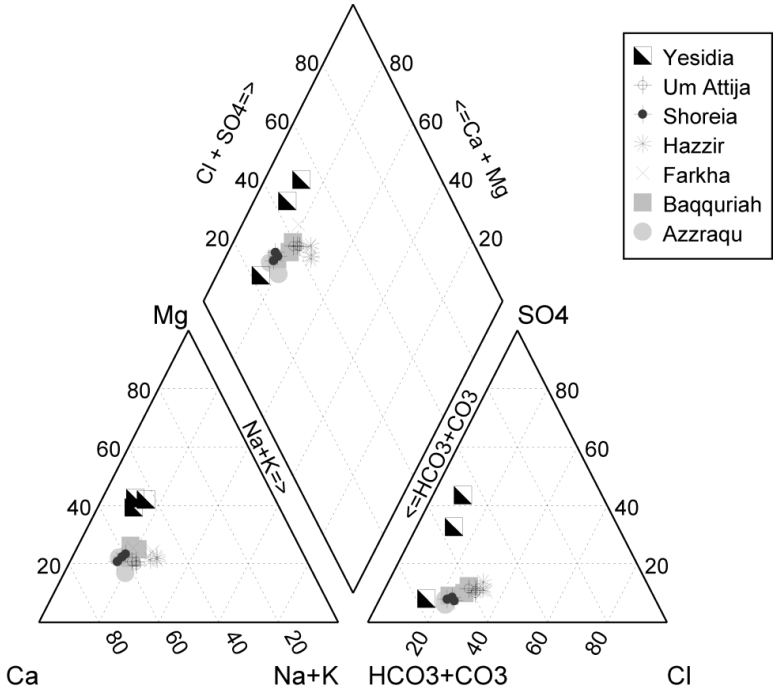

Figure 11: Piper diagram for groundwater samples from 2008 until 2012 with a charge balance error < 5 %.

The water sources in Wadi Shueib are known to be subject of groundwater pollution since several years (Werz, 2006, Al-Kharabsheh et al., 2013, Margane et al. 2009). This results from the urban areas which are located in the upper parts of the wadi on top of the vulnerable limestone aquifer. The concentration level of the investigated contaminants (nitrate, *E. coli*) follow the hydrogeological setup, whereas wells inside and springs downstream to the city showed the highest rates, indicating the influence of sewage or treated wastewater. Several nitrate concentrations (see Figure 10) at springs were in the range of 50 mg/l and partly even above the threshold of 50 mg/l in the national drinking water standard. However drinking water quality is always maintained by mixing with waters with lower nitrate concentrations. Due to the ongoing nitrate and coliform contamination, Hazzir spring was disconnected from the supply system in September 2012. Recent studies delineated the intermitted drinking water supply combined with the impact of exfiltrating sewage as the main reason for nitrate changes at Hazzir spring (Grimmeisen et al. 2016).

1.7. References

Al-Amoush, H., E. Salameh and M. Al-Raggad (2012). "Hydro-Geoelectrical Investigation for the Potential of Underground Water Storage along the Lower Reaches of King Abdullah Canal - Deir Alla Area/Jordan." Journal of Water Resouce and Protection **4**: 545-559.

Al-Kharabsheh, N. M., A. Al-Kharabsheh and O. M. Ghnaim (2013). "Effect of Septic Tanks and Agricultural Wastes on Springs` Water Quality Detoriation in Wadi Shuèib Catchment Area-Jordan." Jordan Journal of Agricultural Sciences **9**(No.1): 15.

AlKhoury, W. (2011). Hydrological modelling in the meso scale semiarid region of Wadi Kafrein / Jordan -The use of innovative techniques under data scarcity. Dissertation, Georg-August-Universität Göttingen.

Alkhoury, W., M. Ziegmann, F. H. Frimmel, G. Abbt-Braun and E. Salameh (2010). "Water quality of the King Abdullah Canal/Jordan–impact on eutrophication and water disinfection." Toxicological & Environmental Chemistry **92**(5): 855-877.

Ben-Avraham, Z., M. Lazar, Z. Garfunkel, M. Reshef, A. Ginzburg, Y. Rotstein, U. Frieslander, Y. Bartov and H. Shulman (2012). 17 - Structural styles along the Dead Sea Fault. Regional Geology and Tectonics: Phanerozoic Passive Margins, Cratonic Basins and Global Tectonic Maps. Boston, Elsevier**:** 616-633.

Bender, F. (1968). Geologie von Jordanien. Berlin,, Gebr. Borntraeger.

Bender, F. (1974). Geology of Jordan. Berlin, Gebrüder Borntraeger.

Garfunkel, Z. (1981). "Internal Structure of the Dead-Sea Leaky Transform (Rift) in Relation to Plate Kinematics." Tectonophysics **80**(1-4): 81-108.

Grimmeisen, F., M. Zemann, N. Göppert, N. Goldscheider (2016): Weekly variations of discharge and water quality caused by intermittent water supply in an urbanized karst catchment. Journal of Hydrology **537**: 157-170.

Hahne, K., Margane, A., Borgstedt, A. (2008). Geological and Tectonic Setting of the Upper Wadi Shuayb Catchment Area, Jordan. Jordan Technical Cooperation Project: Groundwater Resources Management. Amman; Hannover, MWI. Special Report. B.

Horowitz, A. (1979). The Quaternary of Israel. A. Horowitz. New York/London, Academic Press**:** 394.

Horowitz, A., G. Siedner and O. Bar Yosef (1973). "Radiometric Dating of The Ubeidiya Foramtion, Jordan Valley, Israel." Nature **242**: 186-187.

Hötzl, H. (2009). The water of the Jordan Valley : scarcity and deterioration of groundwater and its impact on the regional development. Berlin, Springer.

Hutelocher, L. (2012):Entwicklung und Beurteilung der Hydrochemie im Jordantal und im Wadi Shueib/Evolution and Assessment of the Hydrochemistry in the Jordan Valley and the Wadi Shueib. Bachelor Thesis, Karlsruhe Institute of Technology (KIT) (unpublished).

Lenz, S. (1999). Hydrogeological Investigation along the Wadi al Kafrein and the Kafrein Reservoir, Jordan. Diploma Thesis, University of Karlsruhe (TH).

Marcus, E. and J. Slager (1985). "The sedimentary-magmatic sequence of the Zemah-1 well (Jordan–Dead Sea Rift, Israel) and its emplacement in time and space." Isr. J. Earth. Sci **34**: 1–10.

Margane, A. (2002). Contributions to the hydrogeology of Northern and Central Jordan. Stuttgart, Schweizerbart.

Margane, A., A. Subah, I. Hamdan, Z. Hajali and T. Almomani (2009). Delineation of groundwater protection zones for the springs in Wadi Shuayb. Groundwater Resources Management - Technical Report. **No. 14:** 98.

Masri, M. (1963). Reports on the geology of the Amman-Zarqua area. Amman, Jordan, Central Water Authority**:** 1-74.

McKenzie, D. P., D. Davies and P. Molnar (1970). "Plate tectonics of the Red Sea and East Africa." Nature **226**: 243-248.

MercyCorps (2014). Tapped Out: Water Scarcity and Refugee Pressures in Jordan. Report, 40p. Oregon.

Ministry of Water and Irrigation (2004). National Water Master Plan. Amman.

Niemi, T. M., Z. Ben-Avraham and J. R. Gat (1997). The Dead Sea - The Lake and its setting. New York, Oxford, Oxford University Press.

Parker, D. H. (1970). The hydrogeology of the Mesozoic-Cenozoic aquifers of the western highlands and plateau of East Jordan. Rome, UNDP/FAO**:** 424pp.

Powell, J. H. (1989). Stratigraphy and Sedimentation of the Phanerozoic rocks in Central and South Jordan. Amman, Natural Resources Authority. Geology Directorate. Geological Mapping Division. **11b:** 130p.

Properzi, F. (2010). Rapid assessment of drinking water quality in the Hashemite Kingdom of Jordan: Country report of the pilot project implementaiton in 2004-2005. Country report, 72p. ISBN 9789241500579. World Health Organization and UNICEF 2010. Geneva, Switzerland.

Quennel, A. M. (1951). The geology and mineral resources of (former) Transjordan. Colon. Geol. Min. Resourc. 2. 2.

Riepl, D. (2012). Knowledge based decision support for integrated water resources management with an application for wadi Shueib, Jordan. Dissertation, Karlsruhe Institute of Technology (KIT).

Sahawneh, J. (2011). Structural Control of Hydrology, Hydrogeology and Hydrochemistry along the Eastern Escarpment of the Jordan Rift Valley, Jordan. Dissertation, Karlsruhe Institute of Technology (KIT).

Salameh, E. (2001). "The potential of groundwater artificial recharge in the Jordan Valley area / Jordan." Freiberger Forschungshefte **C494** (Selected Contributions to Applied Geology in the Jordan Rift Valley): 63-81.

Salameh, E. and H. El-Naser (2000). "Changes in the Dead Sea level and their impacts on the surrounding groundwater bodies." Acta Hydrochimica Et Hydrobiologica **28**(1): 24-33.

Toll, M. (2007). An integrated approach for the investigation of unconsolidated aquifers in brackish environment - A case study on the Jordanian side of the Lower Jordan Valley Dissertation, University of Göttingen.

Waldmann, N., M. Stein, D. Ariztegui and A. Starinsky (2009). "Stratigraphy, depositional environments and level reconstruction of the last interglacial Lake Samra in the Dead Sea basin." Quaternary Research **72**(1): 1-15.

Werz, H. (2006). The use of remote sensing imagery for groundwater risk intensity mapping in the Wadi Shueib, Jordan. Dissertation, University of Karlsruhe.

Wolf, L., A. Pöschko, M. Zemann, H. Werz, A. Sawarieh, N. Seder, M. Stieber, F. Sacher and A. Tiehm (2009). "Understanding occurrence of xenobiotics in closed river basins, examples from the Jordan Valley." International Conference on Xenobiotics in the Urban Water Cycle (XENOWAC).

Wolfart, R. (1959). Geology and hydrogeology of the Irbid district, north Jordan. Hannover, Bundesanstalt für Bodenforschung

Zemann, M., M. Majewsky and L. Wolf (2016). "Accumulation of pharmaceuticals in groundwater under arid climate conditions - Results from unsaturated column experiments." Chemosphere **154**: 463-471.

Zemann, M., L. Wolf, F. Grimmeisen, A. Tiehm, J. Klinger, H. Hotzl and N. Goldscheider (2015). "Tracking changing X-ray contrast media application to an urban-influenced karst aquifer in the Wadi Shueib, Jordan." Environmental Pollution **198**: 133-143.

Zemann, M., L. Wolf, A. Pöschko, N. Schmidt, A. Sawarieh, N. Seder, A. Tiehm, H. Hötzl and N. Goldscheider (2014). "Sources and processes affecting the spatio-temporal distribution of pharmaceuticals and X-ray contrast media in the water resources of the Lower Jordan Valley, Jordan." Science of the Total Environment **488–489**(0): 100-114.

2. Sources and Processes Affecting the Spatio-Temporal Distribution of Pharmaceuticals and X-ray Contrast Media in the Water Resources of the Lower Jordan Valley, Jordan

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Abstract: The closed basin of the Lower Jordan Valley with the Dead Sea as final sink features high evapotranspiration rates and almost complete reuse of treated wastewater for irrigation farming. This study focuses on the water transfer schemes and the presence, spreading, and potential accumulation of pharmaceutical residues in the local water resources based on findings of a five-year monitoring program. Overall 16 pharmaceuticals and 9 iodinated X-ray contrast media were monitored in groundwater, surface water, and treated wastewater. A total of 95 samples were taken to cover all geographical settings and flow paths from origin (wastewater) to target (groundwater). Nine substances were detected in groundwater, with concentrations ranging between 11 ng/L and 33,000 ng/L. Sometimes, detection rates were higher than in comparable studies: Diatrizoic acid 75 %, iopamidol 42 %, iopromide 19 %, iomeprol 11 %, carbamazepine and iohexol 8 %, ibuprofen 6 %, fenofibrate and iothalamic acid 3 %. Concentrations in groundwater generally increase from north to south depending on the application of treated wastewater for irrigation. Almost all substances occurred most frequently and with highest concentrations in treated wastewater, followed by surface water and groundwater. As exception, diatrizoic acid was found more frequently in groundwater than in treated wastewater, with concentrations being similar. This indicates the persistence of diatrizoic acid with long residence times in local groundwater systems, but may also reflect changing prescription patterns, which would be in accordance with increasing iopamidol findings by surveys at local hospitals. Trend

analyses confirm this finding and indicate a high probability of increasing iopamidol concentrations, while other substances did not reveal any trends. However, no proof of evaporative enrichment could be found. The high spatial and temporal variability of the concentrations measured calls for further systematic studies to assess the long-term evolution of organic trace substances in this reuse setting.

2.1. Introduction

Pharmaceutical residues and metabolites were detected in all aquatic compartments in the last two decades. Their presence in aquatic environments worldwide as well as their degradation under variable conditions in laboratory studies and wastewater treatment plants (WWTP) have been subjects of many studies so far. Extensive environmental screenings were conducted on groundwater (Sacher et al., 2001; Barnes et al., 2008; Loos et al., 2010; Teijon et al., 2010; Fram and Belitz, 2011; Cabeza et al., 2012; Wolf et al., 2012), surface water (Nakada et al., 2007; Sacher et al., 2008; Loos et al., 2009), and treated wastewater (Drewes et al., 2002; Andreozzi et al., 2003; Zwiener and Frimmel, 2003; Vieno et al., 2007; Matamoros et al., 2008; Loos et al., 2013; Du et al., 2014). Publications so far have focused on substances used in large quantities or on components suspected of being persistent (Schulte-Oehlmann et al., 2007), such as X-ray contrast media (ICM), which can be detected in surface water, groundwater, and drinking water all over the world, with the concentrations ranging from ng/L up to lower µg/L levels (Sacher et al., 2001; Cabeza et al., 2012; Wolf et al., 2012). Duirk et al. (2011) even detected iopamidol (IPA), iopromide (IPR), iohexol (IHE), and diatrizoic acid (DIA) in drinking water sources in the U.S. with IPA detection rates of 60 % and concentrations of up to 2700 ng/L. Most ICM behave conservatively in the environment due to their hydrophilic character and their structural design (Drewes et al., 2001). They are resistant against biochemical degradation processes and do not adsorb to sewage sludge (Kalsch, 1999; Haiss and Kummerer, 2006). Originally designed as contrast agents for X-ray diagnosis, these substances were administered in high doses (up to 200 g/treatment) and excreted almost non-metabolized due to their inert character (Perez and Barcelo, 2007). Subsequently, the ICM pass WWTPs without any significant reduction, with typical effluent concentrations reaching up to the µg/L level (Ternes and Hirsch, 2000; Drewes et al., 2002; Carballa et al., 2004; Perez and Barcelo, 2007). Other studies classified selected ICM as suitable wastewater tracers, as removal rates for DIA (0 %) and IPA

(17 %) are small compared to removal rates between 83 and 89 % for IHE, IPR, and IME (Ternes et al., 2007). WWTP effluents are the major entrance pathway, other possible sources are leaking sewers, sewage sludge or animal manure (Jekel and Reemtsma, 2006). Some techniques like photocatalytic degradation (Doll and Frimmel, 2004), advanced oxidation, and reduction processes (Jeong et al., 2010), activated powdered carbon (Lipp et al., 2012; Margot et al., 2013) or reverse osmosis (Busetti et al., 2010) seem to be suitable tools for ICM removal. Nevertheless, they are not state of the art in (waste-)water treatment and might not always be affordable, especially for developing countries. In most cases, degradation of pharmaceutical substances generally depends on environmental conditions, e.g. redox conditions (Massmann et al., 2008).

Mean differences in influent and effluent concentrations during wastewater treatment are taken from literature, stating 0 % for DIA, -1.1 % for the ICM iothalamic acid (ITA), and 17.4 % for IPA. IHE (59.6 %), IME (73.5 %), and IPR (78.1 %) showed much higher rates. The antiepileptic carbamazepine (CBZ) showed a mean difference of -5.7 %, the analgesic ibuprofen (IBU) 74.2 % (Deblonde et al., 2011).

Although trace concentrations of pharmaceuticals and ICM are measured only, concerns exist with regard to long-term exposure to low doses or potential toxic effects of mixtures of different substances due to interaction or synergetic effects (Jekel and Reemtsma, 2006). Their uptake in plants (Herklotz et al., 2010) and aquatic organisms (Nakamura et al., 2008; Paterson and Metcalfe, 2008; Meredith-Williams et al., 2012) has already been verified. Negative effects of organic trace concentrations on different animals were reported, e.g. vulture disease due to the analgesic diclofenac (DIC) in India (Taggart et al., 2007), Pakistan (Oaks et al., 2004), and Africa (Naidoo et al., 2009), collapse of fish population (Kidd et al., 2007) or changes in the social behavior of European perch (Perca fluviatilis) due to psychotropic drugs (Brodin et al., 2013). The increased formation of genotoxic disinfection byproducts in chlorinated drinking water in the presence of X-ray contrast media (ICM) was found (Duirk et al., 2011).

For the Lower Jordan Valley (LJV), the occurrence of several organic trace substances, including pesticides, pharmaceuticals, and ICM, in different water sources was described previously by Tiehm et al. (2012), Tiehm et al. (2011), and Wolf et al. (2009). The removal efficiencies of three local WWTP and the release concentrations of two hospitals in Amman were described for four pharmaceuticals by Alahmad and Alawi (2010).

Due to the huge amounts of treated wastewater used for irrigation in the LJV (Alfarra, 2010), a close relationship to the quality of groundwater is assumed, as this is a source known to introduce persistent pharmaceuticals into the groundwater (Ternes et al., 2007; Siemens et al., 2008; Chávez et al., 2011). A study of irrigation with treated wastewater in China revealed increased pharmaceutical concentrations in the soil (Chen et al., 2011). Increasing concentrations of the antiepileptic carbamazepine (CBZ) in soils and the groundwater underneath caused by irrigation with treated wastewater were also found in Tunisia (Fenet et al., 2012). Elevated concentrations of the ICM diatrizoic acid (DIA) in groundwater compared to surface water or treated wastewater (Wolf et al., 2009) led to the hypothesis of evaporative enrichment of persistent organic micropollutants in this area. Other authors observed elevated pharmaceutical concentrations in groundwater compared to surface water: A study in Berlin detected higher DIA concentrations in groundwater (4 µg/L) than in surface water (2 µg/L) after bank filtration of water from a river into which treated wastewater was discharged (Putschew et al., 2000). Monitoring the injection of treated wastewater into a confined aquifer close to Barcelona (Spain) showed 13.4 % of iopamidol (IPA) (157 ng/L) and 6.9 % of iopromide (IPR) (574 ng/L) in groundwater samples, while these substances could not be detected in the raw water or the WWTP effluent (Teijon et al., 2010).

Based on the previous findings, the aim of this study was to investigate groundwater quality dynamics of the LJV using pharmaceuticals as pollution indicators and in particular different ICM, the antiepileptic CBZ, and the analgesic ibuprofen (IBU). The following questions were to be answered:

- What substances occur in the different water types and which concentration levels do they reach? Are they suitable anthropogenic tracers?
- Are there spatio-temporal trends or characteristic distribution patterns regarding concentrations and occurrence?
- Can trends and distribution patterns be related to the administration or known persistence of pharmaceuticals or do findings match with local infrastructures for wastewater distribution in irrigated agriculture?
- Is there any proof of accumulation processes in the groundwater?

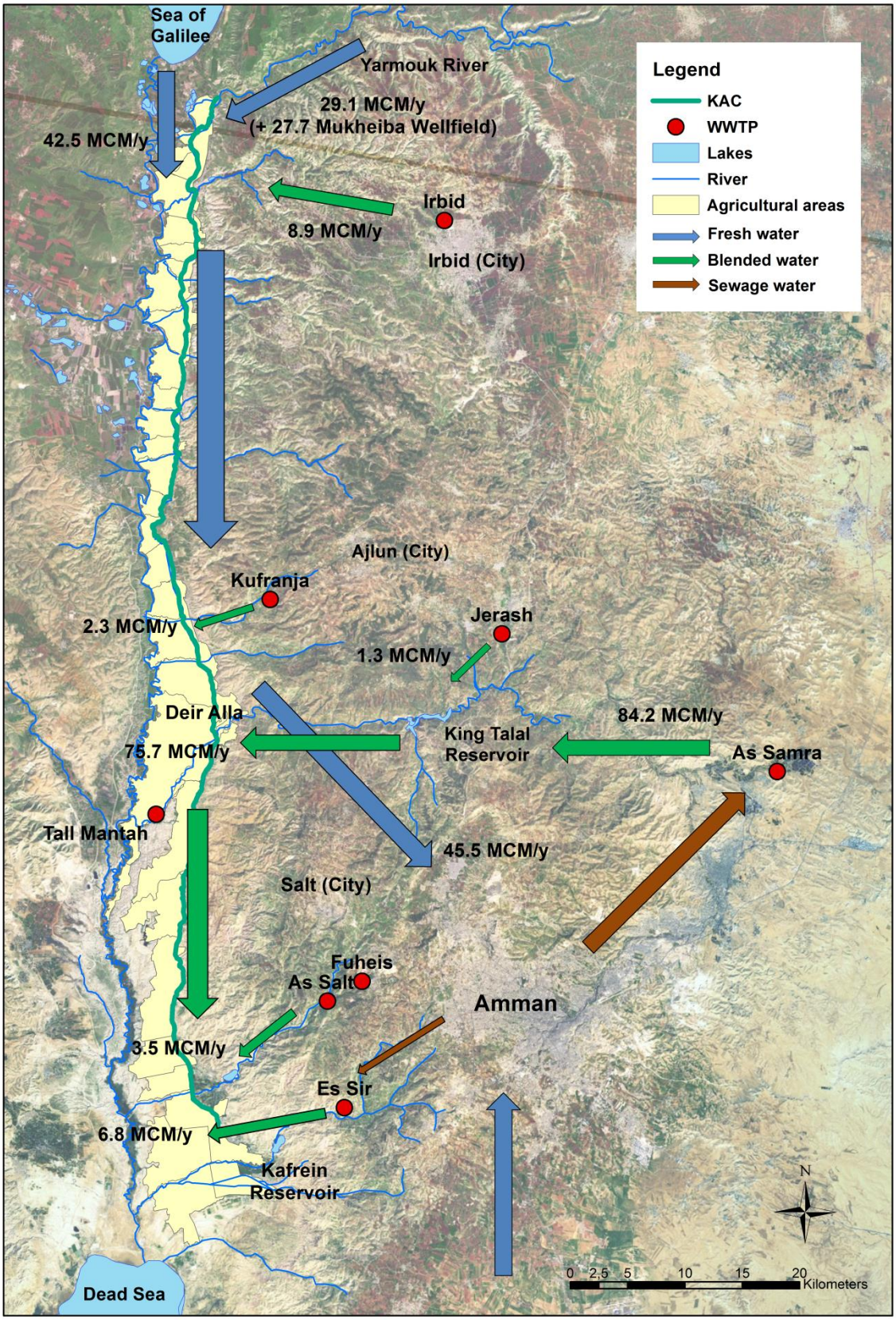

Figure 12: Main water transfers of the LJV. Annual volumes in million cubic meters (MCM), KAC = King Abdullah Canal, figures taken from the Ministry of water and Irrigation (2010) (background map source: ESRI ArcGIS basemap).

2.2. Materials and methods

2.2.1. Local introduction

The LJV is a closed river basin with its deepest point being the Dead Sea (416 m bsl) acting as a final sink for all surface water and groundwater flows. Precipitation ranges between 300 and 400 mm in the northern part, but drops to 100 to 200 mm in the southern part, just north of the Dead Sea (Ministry of Water and Irrigation, 2004). The average annual temperature ranges between 23 and 28 C°. The climate is classified to be arid to semi-arid (Hötzl, 2009). Potential evapotranspiration rates are calculated to reach 2600 mm/a. Natural groundwater recharge is low with a minimum annual safe yield of 15 to 20 MCM (Ministry of Water and Irrigation, 2004). As almost all water running towards the LJV is used in agriculture, irrigation is assumed to be the main source of recharge. To allow for extensive agriculture in the LJV despite water scarcity, huge amounts of water are transferred to this area (Figure 12). Yet, groundwater levels have been declining since the mid 1990s due to pumping and overexploitation of aquifers (Hötzl, 2009). The water level of the Dead Sea is dropping with a rate of almost 1 m/a (Salameh and El-Naser, 2000).

The main share of irrigation water is transferred by the King Abdullah Canal (KAC), diverting water from the Yarmouk River, which is then mixed with water from the Sea of Galilee according to the peace treaty of 1994. Along the KAC, the water is pumped into agricultural development areas at several turnout stations and diverted to the farm units for irrigation. Additionally, many farmers operate their (mostly unregistered) private wells to supplement the allocated KAC water (Alfarra, 2010). Ponds are used to mix groundwater with the KAC share or store water from the KAC for later application. Along the flow path down the LJV to Deir Alla, water from several dams is led into the KAC, as a result of which its water quality decreases continuously (Alkhoury et al., 2010). These dams, in turn, intercept significant amounts of treated wastewater from the highland settlements, together with base flow and flash floods, thus introducing water of poorer quality into the KAC. The main input of treated wastewater enters halfway down the LJV by the King Talal Reservoir (KTR), where the wastewater of the capital Amman (1.9 Mio inhabitants, compared to 6.3 Mio inhabitants of Jordan) is impounded after treatment in the As Samra WWTP. These transfers are illustrated schematically in Figure 13.

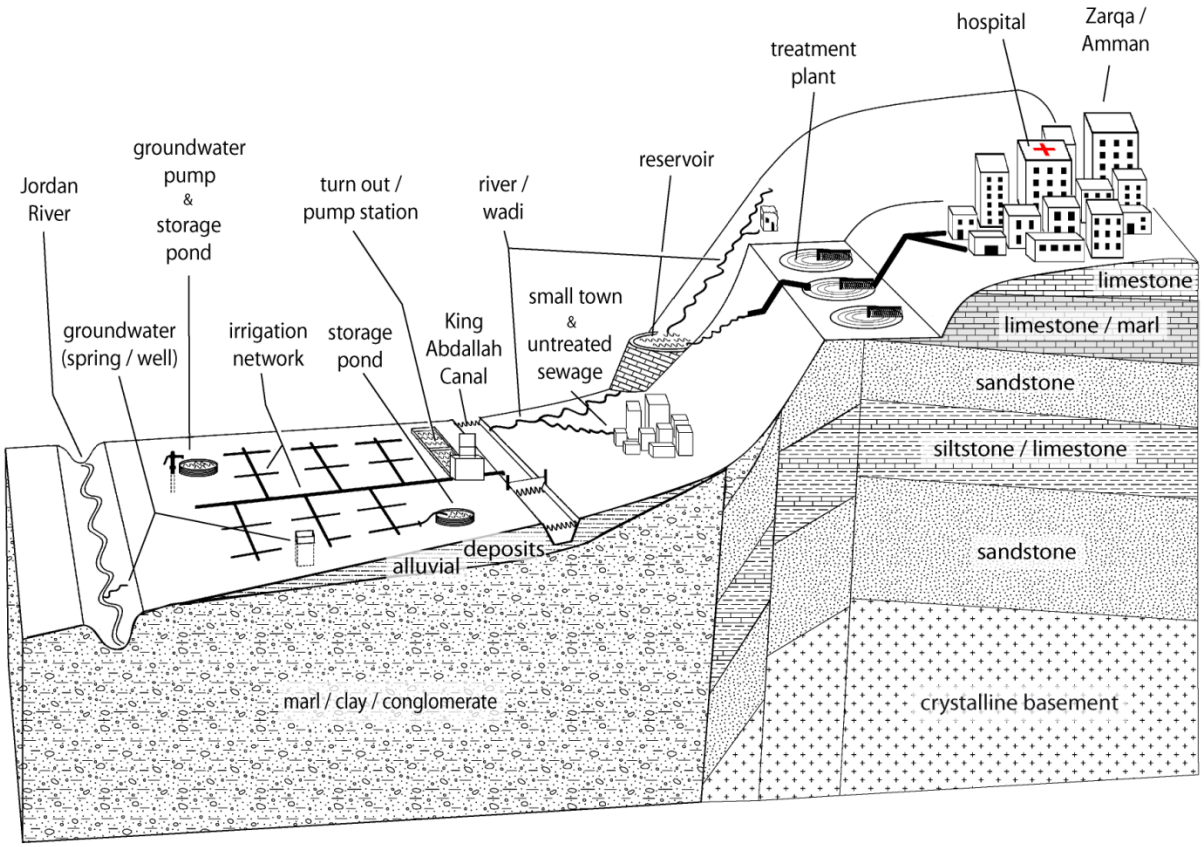

Figure 13: Schematic block diagram illustrating all relevant water sources, local distribution processes for irrigation, and a simplified geological setting (not to scale).

Just before the KAC receives the KTR, approximately 45 MCM/a of water are pumped towards Amman, where it is treated to reach drinking water quality. After the inflow of the KTR into the KAC, the proportion of treated wastewater in the canal ranges between 50 and 70 % and is classified as "blended water". At the canal's end, unused water is discharged into the Dead Sea.

The groundwater in the LJV is found in Quaternary deposits of the Jordan Valley group (Figure 13). At the borders of the valley, groundwater flow is directed from the eastern and western escarpments towards the Jordan River (Salameh, 2001) and from north to south at the bottom of the valley towards the Dead Sea as the lowest point. Due to small hydraulic gradients, groundwater movement is very slow (Toll, 2007).

2.2.2. Data collection

2.2.2.1. Sampling campaigns and site selection

From 2008 to 2012, four water quality sampling campaigns were conducted in the LJV. Additionally, a first screening had already been performed in 2007 at some locations. In 2008, the sample at Tal Mantah WWTP was taken in November, while all other samples had been taken in April 2008. Sampling was originally chosen to take place before and at the end of the rainy season (November to March), although no seasonal effect was visible within this study. For all campaigns, samples were taken within a period of 10 days. In total, 95 samples were taken to monitor 25 trace substances (Table 3), including 16 pharmaceuticals and 9 ICM, as well as major ions and physico-chemical parameters in groundwater, surface water, and treated wastewater. The work focused on areas in the northern, middle, and southern parts of the LJV (see Figure 14), corresponding to the distribution of irrigation water by the Jordan Valley Authority. All water types were investigated from origin to target for each area. Samples were always taken from WWTPs, the KAC (and KTR in the middle LJV), groundwater and irrigation pools. Groundwater site selection was driven mainly by an equal distribution in each area, although the wells were not always accessible, as farms are usually fenced and farmers sometimes were not willing to provide access. However, there was no particular search for heavily polluted sites. Due to the dominating share of KAC water in the pools, they were classified as surface water.

Apart from the wells AN 1025, AN 1026, and Shuna well no.5, which were screened in the limestone aquifer, all remaining groundwater samples were taken from the alluvial aquifer. Groundwater depths range from 5 – 10 m in the northern and middle parts and down to ~ 60 m in the southern part.

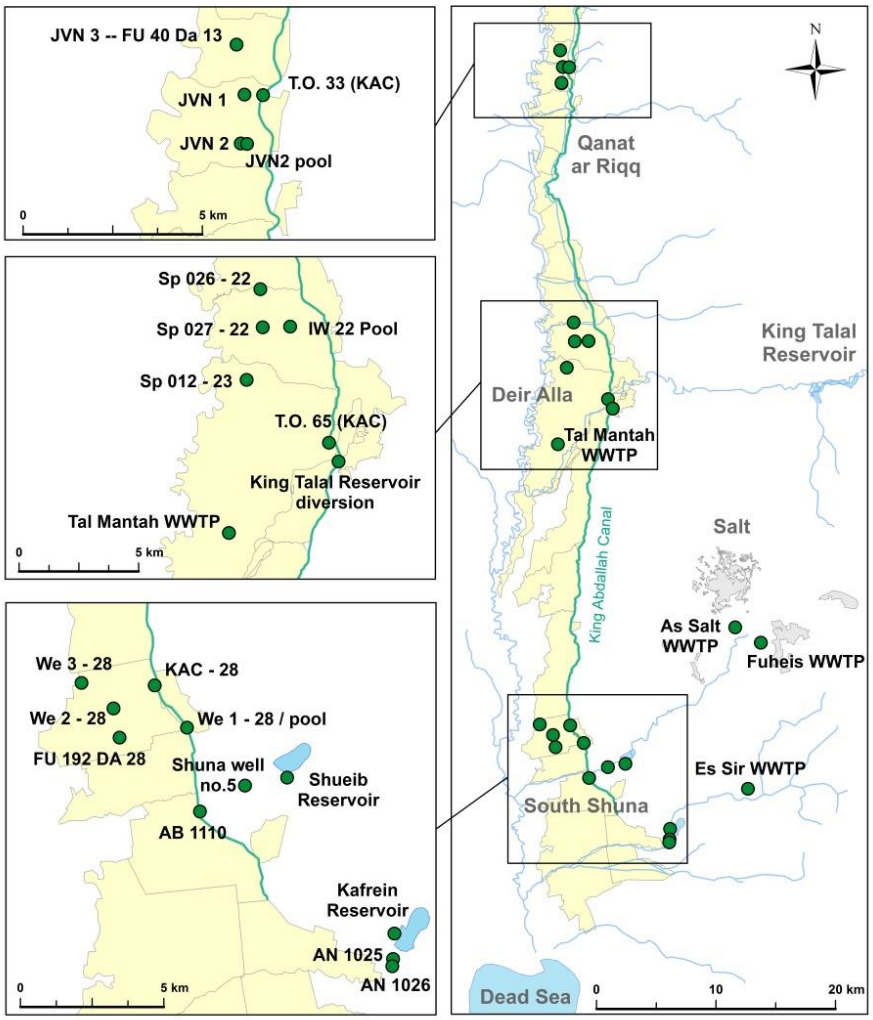

Figure 14: Location of King Abdullah Canal (KAC), King Talal Reservoir (KTR), and all sampling sites in the LJV. Areas featuring irrigated agriculture are marked yellow.

2.2.2.2. Sampling and analysis

At each location, grab samples were taken for the analysis of pharmaceuticals and major ions. In-situ measurements of electric conductivity, redox, temperature, and oxygen were carried out using a WTW Multi 3430 device. Samples for the analysis of pharmaceuticals were collected in 1 l, rinsed PE bottles and acidified using 1.5 ml hydrochloric acid (HCl, 33 %) per liter sample. They were stored refrigerated and dark at 4 to 8°C until dispatch by air. The cold chain was always interrupted for some days.

At the laboratory, extraction was performed within 24 h after arrival. Samples were never filtrated, but suspended particles were decanted before extraction. The analyses were done using an HPLC–ESI-MS–MS, the method is described in Sacher et al. (2008). Analyses were conducted within one week after arrival. The internal quality management at the

laboratories included the accuracy of measurement methods and results as well as possible matrix effects. Blank values, checking samples, and deuterated standards were used for validation during all analyses. The overall measurement inaccuracy for ICM and pharmaceuticals is around 30 %. The limit of detection (LOD) was determined in accordance with DIN-32645.

Table 3: Screened substances (and acronyms used in the text) sorted by generic product category

Analgetics / antiphlogistics	X-ray contrast media (ICM)	Lipid lowering agents
Pentoxifylline	Diatrizoic acid (DIA)	Bezafibrate (BEZ)
Diclofenac (DIC)	Iodipamid	Clofibric acid (CFA)
Ibuprofen (IBU)	Iohexol (IHE)	Etofibrate
Indomethacin	Iomeprol (IME)	Fenofibrate (FFI)
Naproxen (NAP)	Iopamidol (IPA)	Fenofobric acid
Phenacetin	Iopromide (IPR)	Gemfibrozil (GEM)
Anti-inflammatory drugs	Iotalamic acid (ITA)	**Antipsychotic drugs**
Fenoprofen	Ioxaglic acid (IXA)	Diazepam
Ketoprofen	Ioxithalamic acid (IXI)	**Antiepileptic drugs**
		Carbamazepine (CBZ)

2.3. Results

The results for the pharmaceutical substances contained in all groundwater, surface water, and treated wastewater samples are listed in Table 4 and SM1. While Table 4 summarizes the results over the whole sampling period, SM1 gives detailed concentrations at selected sampling sites for the most frequently detected substances of DIA, IPA, CBZ, IPR, and IHE.

2.3.1. Detection frequencies

The substances most frequently found in groundwater (Table 4) of the LJV are the X-ray contrast media DIA (in 79 % of all groundwater samples), IPA (49 %), IPR (16 %), and IHE (14 %), followed by the antiepileptic CBZ (14 %). The lipid lowering agent FFI, the analgesic IBU, and the ICM IME and ITA were found in less than 10 % of the groundwater samples. Detection frequency decreased from treated wastewater towards groundwater for almost all substances. Only DIA showed lower frequencies in treated wastewater than in groundwater samples. Substances found in treated wastewater were usually found in the receiving surface water and groundwater body as well. However, BEZ, CFA, DIC, GEM,

and NAP were only present in surface water and treated wastewater, but could not be detected in any groundwater sample.

2.3.2. Concentration ranges

Similar to the detection frequency, the median concentration decreased from treated wastewater to groundwater for all substances except DIA (see Table 4). DIA showed almost the same concentrations in all water types. IME and IHE reached higher concentrations in surface water than in treated wastewater, but showed the smallest concentrations in groundwater. Typical concentrations in groundwater ranged between 10 ng/L and several 100 ng/L for all substances. Typical concentrations in surface water were between 100 ng/L and 2000 ng/L.

Treated wastewater showed the widest range of concentrations, with median concentrations ranging from 28 ng/L for IXA up to 6600 ng/L for IPA. Especially surface water and treated wastewater concentrations showed a high variability, often featuring concentration differences by an order of magnitude for two subsequent samplings at the same site. For instance, CBZ concentration in the northern KAC (at KAC-28) rose from 76 ng/L in March 2011 to 1700 ng/L in September 2011, whereas IPR decreased from 280,000 ng/L to 2100 ng/L at Es Sir WWTP within the same period. Of all substances, highest concentrations were found for the X-ray contrast medium IPA reaching 36,000 ng/L in groundwater, 78,000 ng/L in surface water, and up to 680,000 ng/L in treated wastewater. All these maximum concentrations were detected in the southern LJV.

Table 4: Concentrations of pharmaceuticals and X-ray contrast media in three water types of the Jordan Valley. GW = groundwater, SW = surface water, TWW = treated wastewater. Total number of samples: 95, total number of sampling points: 26. Numbers of GW samples: 43, SW: 34, and TWW: 18. Limit of detection (LOD) for ICM is 10 ng/L in GW and SW and 50 ng/L in TWW. LOD for pharmaceuticals is 20 ng/L in GW and SW and 50 ng/L in TWW. Bold: Substance present in GW.

		Above limit of detection			Median of positives			Maximum concentration			Median ratios	
		GW	SW	TWW	GW	SW	TWW	GW	SW	TWW	SW/GW	TWW/GW
			[%]			[ng/L]			[ng/L]		[-]	[-]
Pharmaceuticals	Bezafibrate	0	21	56		89	195		390	480		
	Carbamazepine	14	71	89	74	800	3500	500	2100	17,000	10.8	47.3
	Clofibric acid	0	9	6		31	150		33	150		
	Diazepam	0	3	11		13	404		13	720		
	Diclofenac	0	18	28		78	270		160	430		
	Etofibrate	0	0	0								
	Fenofibrate	2	0	6	74		260	74		260		3.5
	Fenofibric acid	0	0	6			160			160		
	Fenoprofen	0	0	0								
	Gemfibrozil	0	62	61		230	1200		2100	4800		
	Ibuprofen	9	41	72	56	80	250	59	1400	750	1.4	4.5
	Indomethacin	0	0	0								
	Ketoprofen	0	0	11			64			64		
	Naproxen	0	15	17		71	95		550	240		
	Pentoxifylline	0	0	0								
	Phenacetin	0	0	0								
X-ray contrast media	**Diatrizoic acid**	79	47	17	120	140	120	940	850	300	1.2	1.0
	Iodipamid	0	0	0								
	Iohexol	14	65	61	31	645	270	180	1600	9000	20.8	8.7
	Iomeprol	9	59	50	39	2200	1400	790	6900	360,000	56.4	35.9
	Iopamidol	49	82	94	59	1100	6600	36,000	78,000	680,000	18.6	111.9
	Iopromide	16	65	72	24	650	860	250	4500	280,000	27.1	35.8
	Iotalamic acid	2	6	6	10	21	42	10	23	42	2.1	4.2
	Ioxaglic acid	0	0	6			28			28		
	Ioxithalamic acid	0	12	0		19			51			

2.3.3. Detection of single substances as indicators of the influence of treated wastewater

As the composition of substances was variable at most sites during this study, the respective samples did not always feature the same cluster of pharmaceuticals. Although some of the frequently found substances like DIA or IPA were usually present, the total spectrum changed. Consequently, the detection rate, i.e. the number of detected

substances at each sampling site, was used to indicate the degree of pollution of the water source by (treated) wastewater. This method had already been verified by Schaider et al. (2014), who correlated substance numbers, pharmaceutical concentrations, boron and nitrate concentrations with the extent of unsewered households to identify the probability of a polluted well.

Highest numbers of substances in this study were found in treated wastewater. Of the 25 screened single substances, six up to eleven were detected. Hence, unpolluted groundwater was indicated by zero detected substance. Most groundwater samples ranged between one and two detected substances. As shown in Table 5, the number of substances detected in groundwater increased from north to south, except for September 2011, where detection rates in groundwater decreased slightly from north to south. In general, increasing detections in groundwater from north to south were accompanied by increasing detections in surface water as well. Throughout the period investigated, no temporal trend was visible between years or over the whole investigation period.

Table 5: Total number of detections in groundwater (GW) and surface water (SW) samples for each sampling campaign in the sampling areas. No: number of samplings, GW = groundwater, SW = surface water.

	North JV				Middle JV				South JV			
	GW	no.	SW	no.	GW	no.	SW	no.	GW	no.	SW	no.
2008 April	0	3	0	2	4	3	13	3	12	7	30	4
2011 March	1	3	1	1	3	2	20	3	6	5	19	3
2011 September	15	3	6	2	2	2	14	3	10	5	18	3
2012 February	2	3	11	2	6	2	21	3	19	5	25	3
Total	18	12	18	7	15	9	68	12	47	22	92	13
Average detection per sample	1.5		2.6		1.7		5.7		2.1		7.1	

2.3.4. Diatrizoic acid (DIA)

The spatial distribution of DIA concentrations in the LJV is given in Figure 15. DIA is the most frequently found substance within this study, reaching detection rates in groundwater between 73 % (2008) and 90 % (2012). Only low amounts of DIA were found in groundwater in the northern part of the valley, with concentrations between 13 and 22 ng/L. In this area, DIA could not be detected in the source water of the KAC and was found in surface water in 2012 only, when its concentration in an irrigation pool reached 140 ng/L. In the middle LJV, high concentrations of DIA ranging between

160 ng/L and 490 ng/L were found in all groundwater samples during the investigation period from 2008 to 2012. While the corresponding surface water from the KAC showed DIA concentrations in March 2011 (140 ng/) only, DIA was present in the KTR with a maximum concentration of 850 ng/L. This was the highest DIA concentration in surface water throughout this study. The southern part showed medium to high concentrations with a maximum contamination of 760 ng/L in groundwater. There, DIA concentrations exhibited a broad distribution in groundwater. Just one negative sample was found in 2008. DIA was detected in all reservoirs and the KAC in September 2011 and February 2012. In 2008 and March 2011, it was present only at the Kafrein reservoir and the Shueib reservoir, respectively. In surface waters, DIA reached medium concentrations of up to 270 ng/L. The southern WWTPs showed only minor DIA concentrations with 110 ng/L in 2008 at As Salt WWTP, 120 ng/L in March 2011 at Fuheis WWTP, and 300 ng/L in 2012 at Es Sir WWTP.

DIA was most frequently detected in groundwater (79 % of all samples; LOD: 10 ng/L), followed by surface water (47 %; LOD: 10 ng/L) and treated wastewater (17 %; LOD: 50 ng/L). Even when taking into account the differing LOD of DIA depending on the water type, still 61 % of the groundwater samples and 42 % of the surface water samples revealed concentrations higher than 50 ng/L. Hence, the sequence of occurrence does not change.

Just two out of eight samples from the WWTPs close to Salt and Fuheis contained DIA. This is in strong contrast to the IPA occurrence shown in Figure 16. Median DIA concentrations for all three water types were in the same order of magnitude, ranging from 55 to 226 ng/L in groundwater, 26 to 280 ng/L in surface water, and 110 to 300 ng/L in treated wastewater throughout the investigation period. Especially in surface water and treated wastewater, variations in DIA concentrations were much smaller than for other substances. Additionally, DIA concentrations did not decrease along the flow path from the origin to the target, as was observed for all other substances.

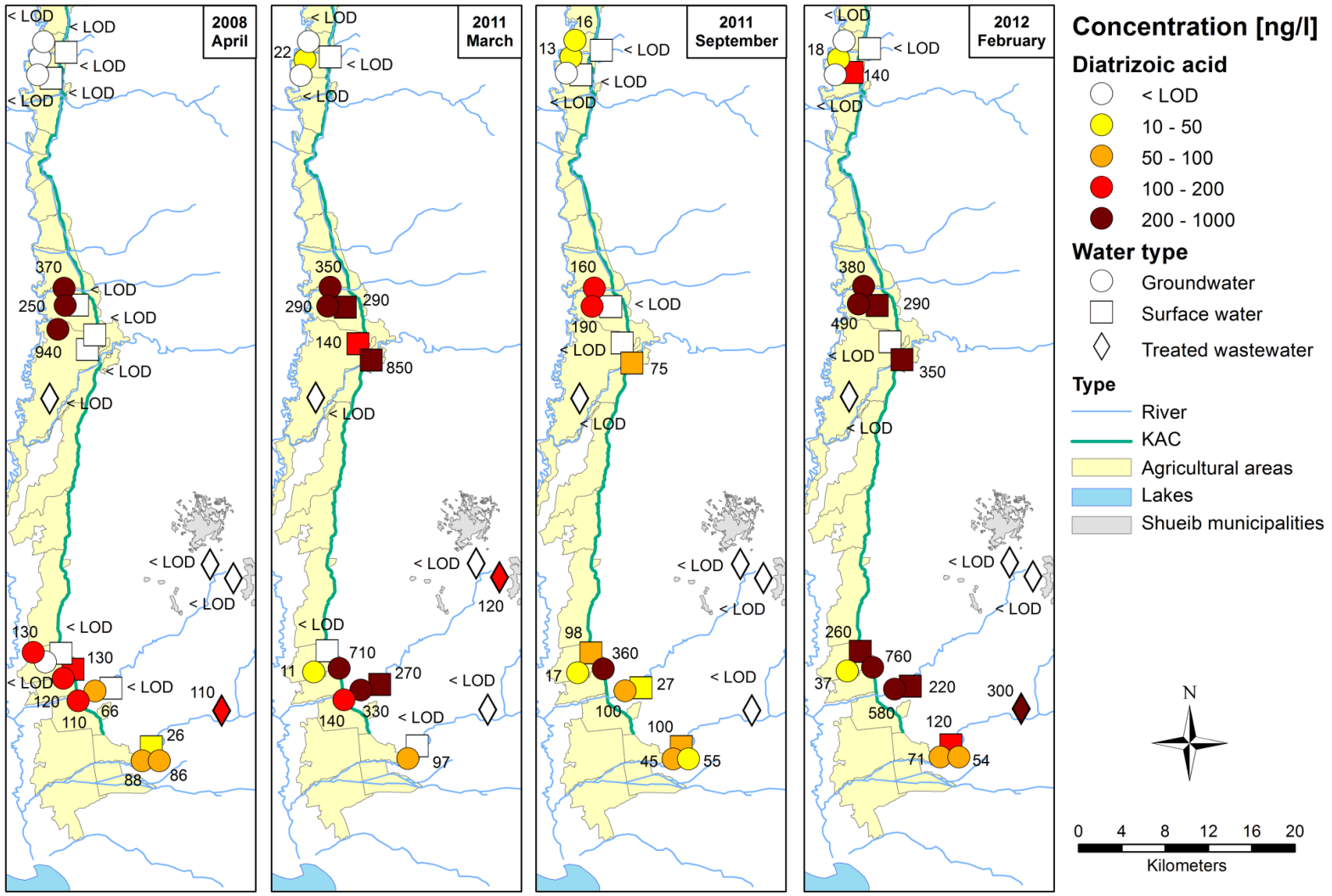

Figure 15: Spatial distribution of diatrizoic acid concentrations [ng/L] in the Lower Jordan Valley between 2008 and 2012 in different water types. LOD for groundwater and surface water: 10 ng/L, for treated wastewater: 50 ng/L. Numbers in the maps are absolute concentrations.

2.3.5. Iopamidol

With regard to the three sampling compartments, IPA showed highest concentrations in wastewater and lowest concentrations in groundwater, as it had been expected. The most important observations are the increasing IPA detection rate and the generally increasing IPA concentrations since the first sampling event in 2008. For the southern part of the Jordan Valley, Figure 16 shows that IPA was found in just two groundwater samples each taken in 2008 and March 2011. During the subsequent years, however, IPA was present in all groundwater samples, with the concentrations increasing at most sampling sites. In general, median and mean concentrations increased from 2008 to 2012 (see SM1). The maximum concentration in groundwater was found in the southern LJV (36,000 ng/L) in 2012. In the same area, maximum concentrations were measured in surface water (78,000 ng/L) and in treated wastewater (680,000 ng/L at Es Sir WWTP) in September 2011. Nevertheless, no clear temporal concentration trend is visible due to the high variability especially in surface water and treated wastewater (see SM1).

2.3.6. Carbamazepine and ibuprofen

During this study, CBZ had a total detection frequency of 14 % for groundwater samples, corresponding to six positive detections, four of which were found in the southern LJV (2008, 2012) and two in the northern LJV (Sept 2011), including the sample with the maximum CBZ concentration found in groundwater (500 ng/L). Detections in surface water ranged between 63 % and 75 % for each sampling campaign (compare SM1), with a maximum concentration of 2100 ng/L found in the KTR in March 2011. All KAC samples in the northern and middle parts were below the LOD. CBZ reached a maximum concentration of 17,000 ng/L in treated wastewater at the Tal Mantah WWTP and was found in all wastewater samples except for two samples at As Salt WWTP in 2008 and 2012.

IBU was found in one groundwater sample in the middle LJV and in three groundwater samples in the southern area in 2012, with the maximum concentration reaching 59 ng/L. This finding is similar to that in surface water, where positive detections were found in samples of the KTR (85 ng/L), middle KAC (58 ng/L), and one pool of the northern LJV in 2012. WWTPs showed a detection rate of 72 %, with the maximum concentration being 750 ng/L. This means that the maximum concentration of surface water (1400 ng/L) is higher than that of treated wastewater.

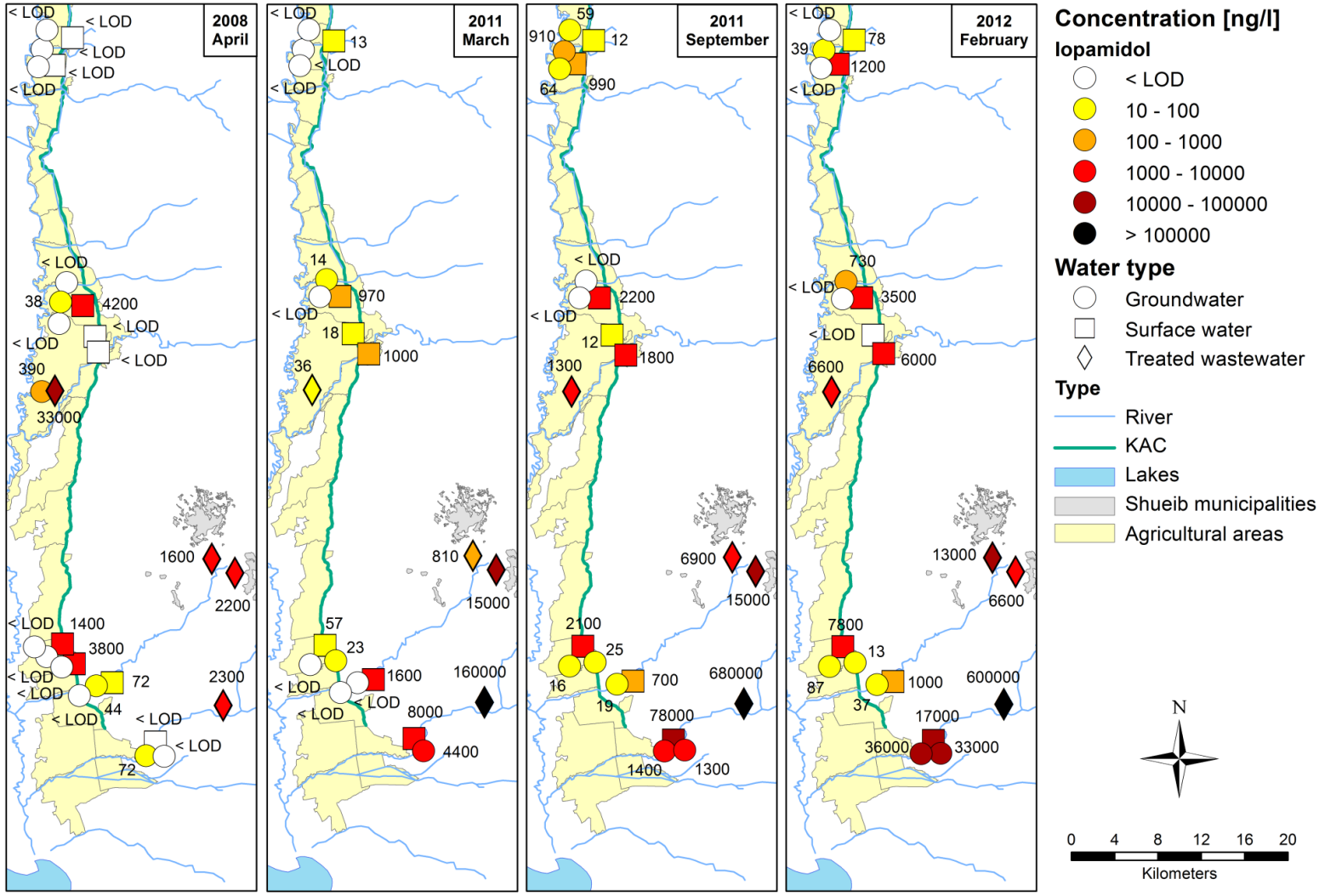

Figure 16: Spatial distribution of iopamidol concentrations [ng/L] in the Lower Jordan Valley between 2008 and 2012 in different water types. LOD for groundwater and surface water: 10 ng/L, for treated wastewater: 50 ng/L. Numbers in the maps are absolute concentrations.

Median concentrations values and detection rates decreased from treated wastewater to surface water to groundwater (see Table 4 and Table 5). The median relation coefficient is 1.4 for surface water to groundwater and 4.5 for treated wastewater to groundwater concentrations.

2.4. Discussion

2.4.1. Occurrence, frequency, concentrations

The maximum concentration of IPA in this study is the highest concentration of all pharmaceuticals found in groundwater by the studies considered (Table 7) in this context. Detection frequencies for IPA and DIA in the Jordan Valley are much higher than in most other studies, which is probably caused by climate conditions and wastewater reuse in irrigation. Maximum DIA concentrations in groundwater are similar to other studies (e.g. Sacher et al., 2001), but still do not reach the values measured in aquifers influenced by urban settlements (Wolf et al. 2012). DIA sources other than irrigation seem unlikely, as the area is sparsely populated and many farms are uninhabited. The only hospital in the LJV is located in South Shuna, whereas highest DIA concentrations were found several km northwards. However, single pollution events like infiltrating domestic sewage cannot be excluded due to the broad study area.

Altogether, it must be mentioned that each of the comparative studies (see Table 7) is strongly influenced by different site selection and sampling strategies. Except for IPA, pharmaceuticals and ICM showed similar concentration ranges (upper ng/L to lower µg/L level) in the effluents of wastewater treatment plants as other studies, e.g. Busetti et al. (2008), Ternes et al. (2007), Loos et al. (2013), and Wolf et al. (2012). Frequencies and maximum concentrations found in groundwater in this study are similar to those found in other studies, e.g. for CBZ, IBU, and CFA (see Table 7).

IBU occurrence and behavior also were in agreement with results of laboratory studies (Table 6), according to which IBU was reported to be highly mobile with almost no retardation and soil organic matter (SOM) concentrations were low (Rauch-Williams et al., 2010; Banzhaf et al., 2012; Hoppe-Jones et al., 2012). High SOM concentrations resulted in 100 % retardation during column experiments according to the OECD guideline 312 (Oppel et al., 2004). The dependency of retardation on SOM concentration was confirmed by Xu et al. (2009).

Table 6: Retardation factor, sorption coefficients, elimination rates, and half-lives of ibuprofen (IBU) and carbamazepine (CBZ) found in literature on laboratory studies. b = batch, c = column, l = lysimeter, m = microcosm, SOM = soil organic matter, h = high, m = medium, l = low, sl = sludge.

Author	Retardation	Sorption coefficient	Elimination	Half-lives	Study type
Banzhaf et al. (2012)	IBU: < 1.5 | CBZ: 8.5		IBU: 47 % | CBZ: 26 %		c
Chefetz et al. (2008)	CBZ: 5.3 - 6.8 (hSOM), 1.6 - 3.3 (lSOM)	CBZ: kf = 11.46 - 12.63 (hSOM), 0.87 - 1.21 (lSOM)	CBZ: 54 % (hSOM/TWW), 25 % (hSOM/tap)		c
Drillia et al. (2005)		CBZ: kf = 57 (hSOM), 0.51 (lSOM), 49 (activated sl), 396 (anaerobic sl). Kd = 37 (hSOM), 0.49 (lSOM)			b & l
Fono et al. (2006)				IBU: 8.6 - 13.8 d	m
Hoppe-Jones et al. (2012)	IBU: 1.0		IBU: 86 - 100 %		c
Hua et al. (2003)			IBU: 100 % (c), 96.3 % (segments)		c
Joss et al. (2006)			IBU: > 90 % (in MBR)	IBU: 0.006 - 0.02 d	b
Kunkel and Radke (2008)				IBU: 2.5 - 5.1 d	m
Lam et al. (2004)				CBZ: 69.7 - 92.6 d	m
Loffler et al. (2005)		IBU: sorption = 9 - 17 % | CBZ: sorption = 40 %	IBU: mineralization = 77 %	IBU: < 6 d | CBZ: 328 d	b
Maeng et al. (2011)			IBU: 49 - 93 % (c), 94 – 98 % (b), 2 % (abiotic) | CBZ: 0 - 6.7 % (c), 0 - 15 % (b)		b & c
Mersmann et al. (2002)	CBZ: 2.8		IBU: 100 % | CBZ: 17 %		b & c
Muller et al. (2013)	CBZ: 1.06 - 1.37			CBZ: 18990 y	c
Onesios and Bouwer (2012)			IBU: > 99 % (low acetate), > 99 % (high acetate)		c
Oppel et al. (2004)	IBU: 100 % | CBZ: 100 %				c
Patterson et al. (2010)	CBZ: 12		CBZ: < 1 µg/l and day	CBZ: > 100 d	c
Preuß et al. (2001)			IBU: 60 - 80 % (sand filtration), 0 - 40 % (aerob, anaerob, sand, gravel). Lag time ~ 5 d | CBZ: 20 - 40 % (sand filtration), 0 - 20 % (areob, anaerob, sand, gravel). Lag time 15 - 17 d		c
Rauch-Williams et al. (2010)	IBU: 1.1 | CBZ: 1.9	IBU: Kd = 0.03 mL/g (abiotic) | CBZ: Kd = 0.24 mL/g	IBU: > 83 % (anoxic), > 85 % (aerobic) | CBZ: 0 %		c
Schaffer et al. (2012)	CBZ: 5.3 (pH 4), 3.6 (pH 8)				c
Scheytt et al. (2004)	IBU: 3.00 | CBZ: 1.84		IBU: 54 % | CBZ: 0 %		c
Stevens-Garmon et al. (2011)		IBU: Kd < 30 L/kgss | CBZ: Kd = 36 - 65 L/kgs			b
Xu et al. (2009)		IBU: 0.27 - 3.42 L/kg		IBU: 0.91 - 31.2 d	b
Yu et al. (2009)		IBU: Kf = 0.1 (sand), 2.1 (mSOM), 14.3 (hSOM). Kd = 0.17 (sand), 0.18 (mSOM), 11.3 (hSOM)			b

In addition, IBU shows high elimination rates under both aerobic and anaerobic conditions (Hua et al., 2003; Joss et al., 2006; Rauch-Williams et al., 2010; Maeng et al., 2011; Hoppe-Jones et al., 2012; Onesios and Bouwer, 2012). Calculated half-lives are in the range of several days to up to two weeks (Loffler et al., 2005; Fono et al., 2006; Joss et al., 2006; Kunkel and Radke, 2008). In combination, these attributes indicate a negligible potential for IBU as an environmentally relevant substance. Subsequently, there have been only minor detections of IBU in different groundwater monitoring studies (compare Table 7) as well as in this study (see supplementary materials (SM) 1). Removal efficiencies above 70 % for IBU were reported for biofilters, sand filters, different constructed wetlands, and conventional wastewater treatment (Ternes, 1998; Llorens et al., 2009; Matamoros et al., 2009; Zorita et al., 2009; Gros et al., 2010; Bueno et al., 2012). A general increase in biological degradation with the sludge retention times (e.g. DIC, bezafibrate (BEZ), IBU) was found in lab-scale pilot plants. (Kreuzinger et al., 2004).

CBZ, by contrast, was present in higher concentrations and more frequently in all water types (see SM 1). According to laboratory studies (see Table 6), CBZ was more retarded than IBU. Retardation and sorption coefficients in these laboratory studies were closely connected to the SOM. A high SOM led to high retardation and sorption, while low amounts resulted in faster transport with minor sorption (Drillia et al., 2005; Chefetz et al., 2008; Yu et al., 2009; Muller et al., 2013). Elimination rates during different laboratory studies usually were very small (Preuß et al., 2001; Scheytt et al., 2004; Rauch-Williams et al., 2010; Maeng et al., 2011). Higher removal rates can be attributed to sorption processes (Chefetz et al., 2008). However, biodegradation rates between 17 % (Mersmann et al., 2002) and 26 % (Banzhaf et al., 2012) were observed in column experiments. Calculated half-lives ranged from 70 to 100 days (Lam et al., 2004; Patterson et al., 2010) up to years (Loffler et al., 2005; Muller et al., 2013). The poor biodegradability, combined with the SOM-bound sorption rates, indicated the potential for high environmental residence times. Although sorption of CBZ by activated sludge seemed to be high in batch experiments (Drillia et al., 2005; Stevens-Garmon et al., 2011), removal rates by WWTP all over the world are poor to non-existent (Castiglioni et al., 2006; Ternes et al., 2007; Vieno et al., 2007; Arye et al., 2011; Bueno et al., 2012). Subsequently, CBZ was frequently present in groundwater (see Table 7) and may be considered a persistent substance with a high potential for groundwater pollution.

Almost all pharmaceutical and ICM concentrations decreased from treated wastewater to surface water to groundwater, following the flow path from origin to target. In accordance with other studies, the pharmaceuticals are subject to dilution processes (Heberer et al., 2004; Arye et al., 2011), retardation (Chefetz et al., 2008), sorption (Ternes et al., 2002; Scheytt et al., 2005; Xu et al., 2009) or biodegradation processes (Preuß et al., 2001; Xu et al., 2009; Onesios and Bouwer, 2012).

2.4.2. Spatial distribution

Rising pharmaceutical detection frequencies from the north to the south of the LJV were observed for every sampling campaign except that of September 2011. Surface water concentrations also increased from the north to the middle, e.g. for CBZ or IPA, and decreased slightly (CBZ) or even increased further (IPA) towards the southern LJV. This corresponds to the geographical setting of Jordanian drinking water and wastewater infrastructure facilities and in particular to the increasing influence and import of treated wastewater and the decreasing availability of fresh water along the LJV from the north to the south (see Figure 12). The increased values in September 2011 may be explained by seasonal effects, as all other campaigns were conducted during or short after the rainy season, therefore featuring higher dilution rates of the irrigation water. However, an individual pollution event cannot be excluded, as the input of pharmaceuticals into the aquatic environment is controlled by human activity and may occur via untreated wastewater, treated wastewater or seepage from landfills receiving pharmaceuticals. In the north, the feeding streams into the KAC as the main source of irrigation water are fresh waters from the Yarmouk River and the Sea of Galilee. Further south, there is a widespread and unmonitored input of leaky or missing sewage systems as well as a large number of cesspits of smaller settlements. Nevertheless, the major input comes from the capital Amman. At Deir Alla, water from the KTR, which is catching the effluent of Amman's largest WWTP, As Samra, is led into the KAC. From here on, an increased occurrence of pharmaceuticals in groundwater reflects the influence of wastewater on the aquatic environment. Therefore, the occurrence of pharmaceuticals in the LJV follows a distinct geographical pattern induced by anthropogenic modifications of the water cycle.

Table 7: Comparison of the pharmaceuticals and ICM detection frequencies and maximum concentrations in groundwater given by different screening studies. Absolute numbers are given in brackets. TWW = treated wastewater, GW = groundwater, LOD = limit of detection

Study	**This study**		**Wolf et al. (2012)**		**Sacher et al. (2001)**		**Teijon et al. (2010)**		**Hanke et al. (2007)**		**Ternes and Hirsch (2000)**		**Focazio et al. (2008)**		**Loos et al. (2010)**	
Environment setting and location	TWW application in agriculture, Jordan Valley (Jordan)		Leaky sewers, GW influenced by urban settlements, Rastatt (Germany)		GW monitoring, Baden-Württemberg (Germany)		TWW injection, Llobregat delta (Spain)		National GW monitoring (Switzerland)		Infiltration of TWW-affected rivers, Hessian Ried (Germany)		National GW (US)		EU-wide survey on GW	
Substances	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]	[%]	[ng/L]
Bezafibrate (BEZ)	0		8	19	0										0	
Carbamazepine (CBZ)	14	500	33	35	13	900	24.5	118	19	45			21.6	190	42.1	390
Clofibric acid (CFA)	0		4	1350	0				2	41						
Diclofenac (DIC)	0		2	129	3.8	590	3.7	477							4.9	24
Fenofibrate (FFI)	2	74	0		0											
Gemfibrozil (GEM)	0		2	23	1	14	11.3	574					0		0	
Ibuprofen (IBU)	9	59	2	104	0		1.9	185					1.4	270	6.7	395
Indomethacin	0		0		1	22										
Naproxen (NAP)	0				0		3.8	263							0	
Diatrizoic acid (DIA)	79	940	27 [3]	4240	20	1100			28	92	80 (10)	170				
Iohexol (IHE)	14	180	4 [1]	187												
Iomeprol (IME)	9	790	2 [3]	1655	0											
Iopamidol (IPA)	49	36,000	4 [2]	79	5	300	13.4	396	17	130	76	2400				
Iopromide (IPR)	16	250	2 [1]	39	0		6.9	687			47	210				
Iotalamic acid (ITA)	2	10	7 [3]	238							19 (16)	49				
Ioxithalamic acid (IXI)	0		9 [0]	204							7 (14)	10				
Number of samples	43		51 (0 = 113, 1 = 114, 2 = 115, 3 = 165)		105		77		47		17 (differing number)		74		164	
LOD [ng/L]	20 (10 for ICM)		10 (25 for IME)		10		10		10		10		11 (15 for GEM, 18 for IBU)		0.2 (0.5 for CBZ)	

2.4.3. Spatio-temporal relationships

The highest concentrations in groundwater were detected for IPA in February 2012 at the wells AN 1025 (36 µg/L) and AN 1026 (33 µg/L) which are located about 350 m downstream of the Kafrein reservoir. These high concentrations can be explained by a medical center which is located in a suburb of Amman at the upper edge of Wadi Kafrein. Wastewater from this center is treated at Es Sir WWTP and subsequently released into the Kafrein reservoir. Another possible explanation might be the fact that the two biggest pharmaceutical companies pass their wastewater on to El Sir WWTP (Alahmad and Alawi, 2010). The hydraulic connection of these two wells to the reservoir has already been confirmed by a tracer test (Lenz, 1999). The high IPA concentrations found by this study in 2012 also correspond to the maximum concentrations found in surface water at the Kafrein reservoir and the treated wastewater concentrations at Es Sir WWTP in September 2011. Both wells were found to respond quickly to increasing concentrations in the Kafrein reservoir despite their depth of approximately 100 m. Additionally, they provide evidence of the spatial and temporal relationship between surface water and groundwater at this site.

2.4.4. Trend analyses

In some parts of the LJV, irrigation triggers a local reuse cycle of groundwater. Local groundwater from shallow aquifer systems and blended wastewater, including a relevant load of pharmaceuticals, from the KAC are used for irrigation. Most of the irrigation water evaporates before reaching deeper parts of the soil, leaving behind dissolved substances, including salts and pharmaceuticals which can precipitate in the soil. The surplus irrigation water seeps down towards the groundwater table, along with persistent trace organics. This groundwater is then pumped up again for irrigation. The water is exposed to evaporation in every cycle. The process was described e.g. by Milnes and Perrochet (2006). Thus, persistent substances like ICM can leave this cycle only via precipitation, plant uptake or evaporation. The two latter seem implausible, as plant uptake is only very small (Herklotz et al., 2010) and volatilization is assumed negligible for most ICMs (Joss et al., 2006). Due to the degree of irrigation, any water-soluble precipitated substance is expected to resolve and reach the groundwater with a delay only. Similar to salinity, which has already been verified for the LJV by Ammari et al. (2013), the concentrations of persistent substances are expected to increase over time under the given local conditions. These are:

- High temperatures and high evaporation rates, combined with hardly any precipitation.
- A substantial, long-term input of treated wastewater used for irrigated agriculture.
- Low natural groundwater recharge which is mainly supplemented by irrigation water.
- All surface flow remains in the LJV, with the Dead Sea being the deepest point and acting as a final sink.

However, there will be some degradation over time. Consequently, accumulation will not be as rapid as for chloride.

Considering all data collected in the Jordan Valley, the increasing detection frequencies and mean concentrations e.g. for IPA already exhibit an upward trend. Due to the absence of normal distributed data, linear regression and correlation coefficients could not be calculated. Performing the Mann-Kendall test (rank correlation, inclusive of values < LOD which were set to 0.5*LOD) for median IPA concentrations for all surface water samples proved an increasing trend hypothesis with a significance level of 4.2 %. The significance level for groundwater and as well for mean concentrations in both water types reached at least 16.7 %. DIA, on the other hand, resulted in a significance level of 37.5 % for the mean groundwater concentrations, while median groundwater concentrations showed a 37.5 % level for decreasing concentrations. Since those numbers are much higher than the usually applicable level of 5 %, most of the trend assumptions do not seem to be sustainable. Similar results are calculated for CBZ or IME, but do not show any clear trend due to the huge data gaps. Results of the Mann-Kendall test for individual sampling sites (S-values) are listed in SM 2. Although there generally is a higher probability of upward trends in IPA than in DIA concentrations regarding the S-values, only surface water coming from the KTR is clearly fulfilling the 5 % significance requirement which indicates an increasing trend in IPA concentrations. The KTR, however, features one of the largest mass flux in the area, bringing approx. 80 MCM blended water down to the LJV per year. Therefore, it has a major impact on the irrigation water quality.

However, the lowest acceptable number to perform the Mann-Kendall test is four. This also was the minimum number used in this study to calculate the mean and median values at each sampling site. Additionally, a minimum of 50 % measurements > LOD is recommended. In groundwater, this requirement was met for DIA and IPA only. Rank

correlations for constant values included a 30 % measurement inaccuracy to consider errors of the laboratory.

Trend analyses did not confirm any accumulation, as either no trend or both increasing trends in groundwater and surface water were detected. Any kind of enrichment would postulate increasing groundwater concentrations and corresponding stable concentrations in surface water. However, accumulation might still be a relevant process under the given conditions.

2.4.5. Comparing the occurrences of DIA and IPA

Both DIA and IPA are present in the groundwater of the LJV with similar median concentration ranges (compare SM1). The input of IPA corresponds well to the known sources of treated wastewater that enters the LJV via KTR, KAC, and adjacent reservoirs. Although not much DIA originates from the monitored WWTPs, concentrations at Kafrein and in the Shueib reservoir suggest at least one unmonitored input, maybe from unconnected households. Further downstream, both substances were present in the surface water of KAC and KTR. The mean and median IPA concentrations were about one order of magnitude higher than those of DIA. As already mentioned, IPA concentrations decreased from treated wastewater to surface water to groundwater, while DIA concentrations stayed the same in all water types.

The similar concentrations of DIA and IPA in groundwater and the different concentrations in the source water might be caused by a number of reasons. The possibility of accumulation was discussed above and ruled out, which is why it will not be considered any further. Firstly, the substances may be subject to different environmental behaviors, i.e. sorption capacity and biodegradation, and secondly, IPA and DIA may be subject to differing application and utilization patterns of the ICM.

Both substances are reported to be very stable. Their low sorption and high resistance to biodegradation during wastewater treatment was reported many times, showing poor to no degradation from influent to effluent (Ternes and Hirsch, 2000; Ternes et al., 2007; Margot et al., 2013). Batch studies and column tests confirmed the resistance of DIA against biodegradation in a simulated WWTP (Kalsch, 1999; Hua et al., 2003; Haiss and Kummerer, 2006) and poor sorption rates on activated sludge (Kalsch, 1999). A k_{ow} of -2.5 for IPA indicates a similar poor sorption behavior (Hebig et al., 2014). Degradation rate constants in a conventional activated sludge simulation showed a k_{biol} of < 0.1 day^{-1}

for DIA and < 0.36 day^{-1} for IPA (Joss et al., 2006). Aerobic soil-water batch systems showed no biotransformation of DIA, while IPA was transformed with a k_{biol} of 0.27 – 0.29 day^{-1} at 20 to 22°C and 2.48 day^{-1} at 30°C. Total concentrations of iopamidol and its transformation products were found to be 80 to 120 % of the initial concentration after 160 days (Kormos et al., 2010). Infiltration of river water led to a decrease of IPA from 0.5 nmol/L to 0.1 nmol/L after bank filtration. In contrast to three other ICM investigated, however, no transformation products of IPA could be detected in groundwater (Kormos et al., 2011).

These results reveal a high environmental persistence especially of DIA and IPA. Some authors presume an adaptation of degrading organisms during the tests, as different test setups led to different metabolites (Hua et al., 2003; Haiss and Kummerer, 2006). Hence, the degradability found might not be the same for all substances.

2.4.6. Shifting application patterns

The high persistence of DIA and IPA therefore suggests different application and utilization patterns of the respective ICM. A very high number of adverse reactions after prescriptions of so-called "ionic" or high osmolar ICM like DIA compared to non-ionic (low osmolar) ICM (e.g. IME, IPA) was found in one application study (Katayama et al., 1990). This led to the prohibition of DIA for intravascular administration in Germany in 2000 (Arztneimittelkommission der deutschen Ärzteschaft, 2000). Nevertheless, DIA might still be used for other than intravascular applications (Internationale Kommission zum Schutz des Rheins, 2010), as it is still present in wastewater in Jordan and Germany, but at lower concentrations than in the 1990s. In this context, Wolf et al. (2012) could not identify significant temporal trends of DIA concentrations in groundwater between 2002 and 2008 in the city of Rastatt (Germany).

According to a Jordan University Hospital (JUH) official, the prescription at JUH changed in the period from 1990 to 1994 (Table 8) from high to low osmolar contrast media. The National Jordan Drug Formulary (Jordan Food and Drug Administration, 2006) recommends to change from high osmolar to low osmolar ICM to reduce adverse effects caused by osmolality.

Table 8: ICM used in the Jordanian University Hospital (Amman) before and after 1994. HOCM = high osmolar contrast media, LOCM = low osmolar contrast media.

HOCM/ ionic used before1994		LOCM/ non-ionic used since 1994	
Agent	**Brand name**	**Agent**	**Brand name**
Diatrizoic acid	Urografin/Renografin	Iopromide	Ultravist
Metrizoate	Isopaque/Trios	Iohexol	Omnipaque
Iodamide	Uromiro	Iopamidol	Iopamiro
Iocarmate	Dimer-X/Meglumine Iocarmate	Iodixanol	Visipaque

Three of the four non-ionic contrast agents listed were detected by the sampling campaigns. The remaining agent iodixanol was not part of the analytical program. Consequently, it is assumed that the use of DIA as an ICM was significantly reduced within the last 15 years. Nevertheless, it is still present in groundwater of the LJV, which is probably due to the long residence time of the groundwater which acts as a memory of past contamination events. Similar mechanisms have also been reported for CBZ and the anticonvulsant primidone. They were prescribed at different times in Berlin, Germany, and applied as groundwater age markers recently (Scheytt, 2008). IPA concentrations in treated wastewater, surface water, and groundwater reflect the increasing application of IPA in the past decades.

2.5. Conclusions

Within the framework of the study reported here, pharmaceuticals and X-ray contrast media were screened in the water resources of the Lower Jordan Valley to assess the degree of groundwater pollution caused by irrigation farming with mixtures of treated wastewater and fresh water. Overall, the following conclusions can be drawn:

- Pharmaceutical residues, especially X-ray contrast media, are spread widely in all water types of the Jordan Valley (groundwater, surface water, treated wastewater). This indicates that the local water resources are strongly affected by anthropogenic influences.
- Increasing concentrations and positive detections of pharmaceutical residues along the flow path of the King Abdullah Canal and the King Talal Reservoir effluent

reflect the environmental impact of the discharge of Amman`s biggest wastewater treatment plant As Samra.

- Median iopamidol concentrations in treated wastewater exceed those in groundwater by two orders of magnitude. Diatrizoic acid reaches similar concentrations in groundwater, surface water, and treated wastewater. Often, concentrations in groundwater are even higher than in treated wastewater. These clearly different environmental distributions of non-ionic and ionic iodinated X-ray contrast media might be caused either by the more recent introduction of the non-ionic X-ray contrast medium iopamidol or by the lower environmental persistence of iopamidol. Considering both, the similar environmental persistence reported in current literature and the reported changes in prescription practice in Jordan, the latter is suggested to be responsible for the observed patterns.
- The data obtained suggest ongoing groundwater quality deterioration with time, but the variability and rising concentrations in surface water do not provide any proof of evaporative accumulation processes.

Due to the shifting usage patterns of ionic and non-ionic X-ray contrast media, diatrizoic acid and iopamidol might be used as age markers in groundwater. Also spatio-temporal relationships at single sampling sites prove the suitability of X-ray contrast media as groundwater tracers. Furthermore, a long-term monitoring program and systematic studies are suggested to assess the long-term evolution and potential enrichment of persistent trace organics in groundwater of the Lower Jordan Valley.

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2.7. Literature

Alahmad, W. R. and M. A. Alawi (2010). "Hplc/Uv/Fluorescence Detection of Several Pharmaceuticals in Sewage Treatment Plant Wastewaters of Jordan." Fresenius Environmental Bulletin **19**(5): 805-810.

Alfarra, A. (2010). Treated Wastewater for Irrigated Agriculture in the Jordan Valley - Analysing Water Allocation and Willingness to Pay for Reused Water Doctoral Thesis, University of Karlsruhe (TH).

Alkhoury, W., M. Ziegmann, F. H. Frimmel, G. Abbt-Braun and E. Salameh (2010). "Water quality of the King Abdullah Canal/Jordan–impact on eutrophication and water disinfection." Toxicological & Environmental Chemistry **92**(5): 855-877.

Ammari, T. G., R. Tahhan, S. Abubaker, Y. Al-Zu'Bi, A. Tahboub, R. Ta'Any, S. Abu-Romman, N. Al-Manaseer and M. H. Stietiya (2013). "Soil Salinity Changes in the Jordan Valley Potentially Threaten Sustainable Irrigated Agriculture." Pedosphere **23**(3): 376-384.

Andreozzi, R., R. Marotta and N. Paxeus (2003). "Pharmaceuticals in STP effluents and their solar photodegradation in aquatic environment." Chemosphere **50**(10): 1319-1330.

Arye, G., I. Dror and B. Berkowitz (2011). "Fate and transport of carbamazepine in soil aquifer treatment (SAT) infiltration basin soils." Chemosphere **82**(2): 244-252.

Arztneimittelkommission der deutschen Ärzteschaft (2000). "BfArM ordnet Widerruf von ionischen Röntgenkontrastmitteln (Amidotrizoesäure, Iotalaminsäure) zur intraversalen Anwendung an." Deutsches Ärzteblatt **Jg. 97**(Heft 33).

Banzhaf, S., K. Nödler, T. Licha, A. Krein and T. Scheytt (2012). "Redox-sensitivity and mobility of selected pharmaceutical compounds in a low flow column experiment." Science of the Total Environment **438**(0): 113-121.

Barnes, K. K., D. W. Kolpin, E. T. Furlong, S. D. Zaugg, M. T. Meyer and L. B. Barber (2008). "A national reconnaissance of pharmaceuticals and other organic wastewater contaminants in the United States — I) Groundwater." Science of the Total Environment **402**(2–3): 192-200.

Brodin, T., J. Fick, M. Jonsson and J. Klaminder (2013). "Dilute Concentrations of a Psychiatric Drug Alter Behavior of Fish from Natural Populations." Science **339**(6121): 814-815.

Bueno, M. J. M., M. J. Gomez, S. Herrera, M. D. Hernando, A. Agüera and A. R. Fernández-Alba (2012). "Occurrence and persistence of organic emerging contaminants and priority pollutants in five sewage treatment plants of Spain: Two years pilot survey monitoring." Environmental Pollution **164**(0): 267-273.

Busetti, F., K. L. Linge, J. W. Blythe and A. Heitz (2008). "Rapid analysis of iodinated X-ray contrast media in secondary and tertiary treated wastewater by direct injection liquid chromatography-tandem mass spectrometry." Journal of Chromatography A **1213**(2): 200-208.

Busetti, F., K. L. Linge, C. Rodriguez and A. Heitz (2010). "Occurrence of iodinated X-ray contrast media in indirect potable reuse systems." Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering **45**(5): 542-548.

Cabeza, Y., L. Candela, D. Ronen and G. Teijon (2012). "Monitoring the occurrence of emerging contaminants in treated wastewater and groundwater between 2008 and 2010. The Baix Llobregat (Barcelona, Spain)." Journal of Hazardous Materials **239**: 32-39.

Carballa, M., F. Omil, J. M. Lema, M. Llompart, C. Garcia-Jares, I. Rodriguez, M. Gomez and T. Ternes (2004). "Behavior of pharmaceuticals, cosmetics and hormones in a sewage treatment plant." Water Research **38**(12): 2918-2926.

Castiglioni, S., R. Bagnati, R. Fanelli, F. Pomati, D. Calamari and E. Zuccato (2006). "Removal of pharmaceuticals in sewage treatment plants in Italy." Environmental Science & Technology **40**(1): 357-363.

Chávez, A., C. Maya, R. Gibson and B. Jiménez (2011). "The removal of microorganisms and organic micropollutants from wastewater during infiltration to aquifers after irrigation of farmland in the Tula Valley, Mexico." Environmental Pollution **159**(5): 1354-1362.

Chefetz, B., T. Mualem and J. Ben-Ari (2008). "Sorption and mobility of pharmaceutical compounds in soil irrigated with reclaimed wastewater." Chemosphere **73**(8): 1335-1343.

Chen, F., G. G. Ying, L. X. Kong, L. Wang, J. L. Zhao, L. J. Zhou and L. J. Zhang (2011). "Distribution and accumulation of endocrine-disrupting chemicals and pharmaceuticals in wastewater irrigated soils in Hebei, China." Environmental Pollution **159**(6): 1490-1498.

Deblonde, T., C. Cossu-Leguille and P. Hartemann (2011). "Emerging pollutants in wastewater: A review of the literature." International Journal of Hygiene and Environmental Health **214**(6): 442-448.

Doll, T. E. and F. H. Frimmel (2004). "Kinetic study of photocatalytic degradation of carbamazepine, clofibric acid, iomeprol and iopromide assisted by different TiO2 materials—determination of intermediates and reaction pathways." Water Research **38**(4): 955-964.

Drewes, J. E., P. Fox and M. Jekel (2001). "Occurrence of iodinated X-ray contrast media in domestic effluents and their fate during indirect potable reuse." Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering **36**(9): 1633-1645.

Drewes, J. E., T. Heberer and K. Reddersen (2002). "Fate of pharmaceuticals during indirect potable reuse." Water Science and Technology **46**(3): 73-80.

Drillia, P., K. Stamatelatou and G. Lyberatos (2005). "Fate and mobility of pharmaceuticals in solid matrices." Chemosphere **60**(8): 1034-1044.

Du, B., A. E. Price, W. C. Scott, L. A. Kristofco, A. J. Ramirez, C. K. Chambliss, J. C. Yelderman and B. W. Brooks (2014). "Comparison of contaminants of emerging concern removal, discharge, and water quality hazards among centralized and on-site wastewater treatment system effluents receiving common wastewater influent." Science of the Total Environment **466–467**(0): 976-984.

Duirk, S. E., C. Lindell, C. C. Cornelison, J. Kormos, T. A. Ternes, M. Attene-Ramos, J. Osiol, E. D. Wagner, M. J. Plewa and S. D. Richardson (2011). "Formation of Toxic Iodinated Disinfection By-Products from Compounds Used in Medical Imaging." Environmental Science & Technology **45**(16): 6845-6854.

Fenet, H., O. Mathieu, O. Mahjoub, Z. Li, D. Hillaire-Buys, C. Casellas and E. Gomez (2012). "Carbamazepine, carbamazepine epoxide and dihydroxycarbamazepine sorption to soil and occurrence in a wastewater reuse site in Tunisia." Chemosphere **88**(1): 49-54.

Focazio, M. J., D. W. Kolpin, K. K. Barnes, E. T. Furlong, M. T. Meyer, S. D. Zaugg, L. B. Barber and M. E. Thurman (2008). "A national reconnaissance for pharmaceuticals and other organic wastewater contaminants in the United States — II) Untreated drinking water sources." Science of the Total Environment **402**(2–3): 201-216.

Fono, L. J., E. P. Kolodziej and D. L. Sedlak (2006). "Attenuation of wastewater-derived contaminants in an effluent-dominated river." Environmental Science and Technology **40**(23): 7257-7262.

Fram, M. S. and K. Belitz (2011). "Occurrence and concentrations of pharmaceutical compounds in groundwater used for public drinking-water supply in California." Science of the Total Environment **409**(18): 3409-3417.

Gros, M., M. Petrović, A. Ginebreda and D. Barceló (2010). "Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes." Environment International **36**(1): 15-26.

Haiss, A. and K. Kummerer (2006). "Biodegradability of the X-ray contrast compound diatrizoic acid, identification of aerobic degradation products and effects against sewage sludge micro-organisms." Chemosphere **62**(2): 294-302.

Hanke, I., H. Singer, C. McArdell-Buergisser, M. Brennwald, D. Traber, R. Muralt, T. Herold, R. Oechslin and R. Kipfer (2007). Arzneimittel und Pestizide im Grundwasser. Sonderdruck Nr. 1501 aus gwa 3/2007 des Schweizerischen Vereins des Gas- und Wasserfaches (SVGW). Zürich, Schweizer Vereins des Gas- und Wasserfaches (SVGW). **3/2007**.

Heberer, T., A. Mechlinski, B. Fanck, A. Knappe, G. Massmann, A. Pekdeger and B. Fritz (2004). "Field studies on the fate and transport of pharmaceutical residues in bank filtration." Ground Water Monitoring and Remediation **24**(2): 70-77.

Hebig, K. H., K. Nödler, T. Licha and T. J. Scheytt (2014). "Impact of materials used in lab and field experiments on the recovery of organic micropollutants." Science of the Total Environment **473-474**: 125-131.

Herklotz, P. A., P. Gurung, B. Vanden Heuvel and C. A. Kinney (2010). "Uptake of human pharmaceuticals by plants grown under hydroponic conditions." Chemosphere **78**(11): 1416-1421.

Hoppe-Jones, C., E. R. V. Dickenson and J. E. Drewes (2012). "The role of microbial adaptation and biodegradable dissolved organic carbon on the attenuation of trace organic chemicals during groundwater recharge." Science of the Total Environment **437**: 137-144.

Hötzl, H. (2009). The water of the Jordan Valley : scarcity and deterioration of groundwater and its impact on the regional development. Berlin, Springer.

Hua, J. M., P. L. An, J. Winter and C. Gallert (2003). "Elimination of COD, microorganisms and pharmaceuticals from sewage by trickling through sandy soil below leaking sewers." Water Research **37**(18): 4395-4404.

Internationale Kommission zum Schutz des Rheins (2010). Auswertungsbericht Röntgenkontrastmittel. 56002 Koblenz, Germany, Internationale Kommission zum Schutz des Rheins. **Nr. 187**.

Jekel, M. and T. Reemtsma (2006). Organic pollutants in the water cycle : properties, occurrence, analysis and environmental relevance of polar compounds. Weinheim, Wiley-VCH.

Jeong, J., J. Jung, W. J. Cooper and W. H. Song (2010). "Degradation mechanisms and kinetic studies for the treatment of X-ray contrast media compounds by advanced oxidation/reduction processes." Water Research **44**(15): 4391-4398.

Jordan Food and Drug Administration (2006). Jordan National Drug Formulary. Q. L. Nuseirat A., Ghanem M. http://www.jfda.jo/jndfbook/index.htm, Drug Directorate Rational Drug Use Department.

Joss, A., S. Zabczynski, A. Gobel, B. Hoffmann, D. Loffler, C. S. McArdell, T. A. Ternes, A. Thomsen and H. Siegrist (2006). "Biological degradation of pharmaceuticals in

municipal wastewater treatment: Proposing a classification scheme." Water Research **40**(8): 1686-1696.

Kalsch, W. (1999). "Biodegradation of the iodinated X-ray contrast media diatrizoate and iopromide." Science of the Total Environment **225**(1-2): 143-153.

Katayama, H., K. Yamaguchi, T. Kozuka, T. Takashima, P. Seez and K. Matsuura (1990). "Adverse reactions to ionic and nonionic contrast media. A report from the Japanese Committee on the Safety of Contrast Media." Radiology **175**(3): 621-628.

Kidd, K. A., P. J. Blanchfield, K. H. Mills, V. P. Palace, R. E. Evans, J. M. Lazorchak and R. W. Flick (2007). "Collapse of a fish population after exposure to a synthetic estrogen." Proceedings of the National Academy of Sciences **104**(21): 8897-8901.

Kormos, J. L., M. Schulz, H.-P. E. Kohler and T. A. Ternes (2010). "Biotransformation of Selected Iodinated X-ray Contrast Media and Characterization of Microbial Transformation Pathways." Environmental Science & Technology **44**(13): 4998-5007.

Kormos, J. L., M. Schulz and T. A. Ternes (2011). "Occurrence of Iodinated X-ray Contrast Media and Their Biotransformation Products in the Urban Water Cycle." Environmental Science & Technology **45**(20): 8723-8732.

Kreuzinger, N., M. Clara, B. Strenn and H. Kroiss (2004). "Relevance of the sludge retention time (SRT) as design criteria for wastewater treatment plants for the removal of endocrine disruptors and pharmaceuticals from wastewater." Water Science and Technology **50**(5): 149-156.

Kunkel, U. and M. Radke (2008). "Biodegradation of acidic pharmaceuticals in bed sediments: Insight from a laboratory experiment." Environmental Science & Technology **42**(19): 7273-7279.

Lam, M. W., C. J. Young, R. A. Brain, D. J. Johnson, M. A. Hanson, C. J. Wilson, S. M. Richards, K. R. Solomon and S. A. Mabury (2004). "Aquatic persistence of eight pharmaceuticals in a microcosm study." Environmental Toxicology and Chemistry **23**(6): 1431-1440.

Lenz, S. (1999). Hydrogeological Investigation along the Wadi al Kafrein and the Kafrein Reservoir, Jordan. Diploma Thesis, University of Karlsruhe (TH).

Lipp, P., H. J. Gross and A. Tiehm (2012). "Improved elimination of organic micropollutants by a process combination of membrane bioreactor (MBR) and powdered activated carbon (PAC)." Desalination and Water Treatment **42**(1-3): 65-72.

Llorens, E., V. Matamoros, V. Domingo, J. M. Bayona and J. Garcia (2009). "Water quality improvement in a full-scale tertiary constructed wetland: Effects on conventional and specific organic contaminants." Science of the Total Environment **407**(8): 2517-2524.

Loffler, D., J. Rombke, M. Meller and T. A. Ternes (2005). "Environmental fate of pharmaceuticals in water/sediment systems." Environmental Science & Technology **39**(14): 5209-5218.

Loos, R., R. Carvalho, D. C. António, S. Comero, G. Locoro, S. Tavazzi, B. Paracchini, M. Ghiani, T. Lettieri, L. Blaha, B. Jarosova, S. Voorspoels, K. Servaes, P. Haglund, J. Fick, R. H. Lindberg, D. Schwesig and B. M. Gawlik (2013). "EU-wide monitoring survey on emerging polar organic contaminants in wastewater treatment plant effluents." Water Research **47**(17): 6475-6487.

Loos, R., B. M. Gawlik, G. Locoro, E. Rimaviciute, S. Contini and G. Bidoglio (2009). "EU-wide survey of polar organic persistent pollutants in European river waters." Environmental Pollution **157**(2): 561-568.

Loos, R., G. Locoro, S. Comero, S. Contini, D. Schwesig, F. Werres, P. Balsaa, O. Gans, S. Weiss, L. Blaha, M. Bolchi and B. M. Gawlik (2010). "Pan-European survey on the

occurrence of selected polar organic persistent pollutants in ground water." Water Research **44**(14): 4115-4126.

Maeng, S. K., S. K. Sharma, C. D. T. Abel, A. Magic-Knezev and G. L. Amy (2011). "Role of biodegradation in the removal of pharmaceutically active compounds with different bulk organic matter characteristics through managed aquifer recharge: Batch and column studies." Water Research **45**(16): 4722-4736.

Margot, J., C. Kienle, A. Magnet, M. Weil, L. Rossi, L. F. de Alencastro, C. Abegglen, D. Thonney, N. Chèvre, M. Schärer and D. A. Barry (2013). "Treatment of micropollutants in municipal wastewater: Ozone or powdered activated carbon?" Science of the Total Environment **461–462**(0): 480-498.

Massmann, G., U. Dunnboer, T. Heberer and T. Taute (2008). "Behaviour and redox sensitivity of pharmaceutical residues during bank filtration - Investigation of residues of phenazone-type analgesics." Chemosphere **71**(8): 1476-1485.

Matamoros, V., C. Arias, H. Brix and J. M. Bayona (2009). "Preliminary screening of small-scale domestic wastewater treatment systems for removal of pharmaceutical and personal care products." Water Research **43**(1): 55-62.

Matamoros, V., A. Caselles-Osorio, J. Garcia and J. M. Bayona (2008). "Behaviour of pharmaceutical products and biodegradation intermediates in horizontal subsurface flow constructed wetland. A microcosm experiment." Science of the Total Environment **394**(1): 171-176.

Meredith-Williams, M., L. J. Carter, R. Fussell, D. Raffaelli, R. Ashauer and A. B. A. Boxall (2012). "Uptake and depuration of pharmaceuticals in aquatic invertebrates." Environmental Pollution **165**(0): 250-258.

Mersmann, P., T. Scheytt and T. Heberer (2002). "Säulenversuche zum Transportverhalten von Arzneimittelwirkstoffen in der wassergesättigten Zone." Acta Hydrochimica Et Hydrobiologica **30**(5-6): 275-284.

Milnes, E. and P. Perrochet (2006). "Direct simulation of solute recycling in irrigated areas." Advances in Water Resources **29**(8): 1140-1154.

MWI - Ministry of Water and Irrigation (2004). National Water Master Plan. Amman.

MWI - Ministry of Water and Irrigation (2010). Annual Report 2010. D. o. M. a. W. Awareness. Amman, Water Authority of Jordan - Jordan Valley Authority.

Muller, B., T. Scheytt and G. Grutzmacher (2013). "Transport of primidone, carbamazepine, and sulfamethoxazole in thermally treated sediments-laboratory column experiments." Journal of Soils and Sediments **13**(5): 953-965.

Naidoo, V., K. Wolter, R. Cuthbert and N. Duncan (2009). "Veterinary diclofenac threatens Africa's endangered vulture species." Regulatory Toxicology and Pharmacology **53**(3): 205-208.

Nakada, N., K. Komori, Y. Suzuki, C. Konishi, I. Houwa and H. Tanaka (2007). "Occurrence of 70 pharmaceutical and personal care products in Tone River basin in Japan." Water Science and Technology **56**(12): 133-140.

Nakamura, Y., H. Yamamoto, J. Sekizawa, T. Kondo, N. Hirai and N. Tatarazako (2008). "The effects of pH on fluoxetine in Japanese medaka (Oryzias latipes): Acute toxicity in fish larvae and bioaccumulation in juvenile fish." Chemosphere **70**(5): 865-873.

Oaks, J. L., M. Gilbert, M. Z. Virani, R. T. Watson, C. U. Meteyer, B. A. Rideout, H. L. Shivaprasad, S. Ahmed, M. J. Chaudhry, M. Arshad, S. Mahmood, A. Ali and A. A. Khan (2004). "Diclofenac residues as the cause of vulture population decline in Pakistan." Nature **427**(6975): 630-633.

Onesios, K. M. and E. J. Bouwer (2012). "Biological removal of pharmaceuticals and personal care products during laboratory soil aquifer treatment simulation with different primary substrate concentrations." Water Research **46**(7): 2365-2375.

Oppel, J., G. Broll, D. Loffler, M. Meller, J. Rombke and T. Ternes (2004). "Leaching behaviour of pharmaceuticals in soil-testing-systems: a part of an environmental risk assessment for groundwater protection." Science of the Total Environment **328**(1-3): 265-273.

Paterson, G. and C. D. Metcalfe (2008). "Uptake and depuration of the anti-depressant fluoxetine by the Japanese medaka (Oryzias latipes)." Chemosphere **74**(1): 125-130.

Patterson, B. M., M. Shackleton, A. J. Furness, J. Pearce, C. Descourvieres, K. L. Linge, F. Busetti and T. Spadek (2010). "Fate of nine recycled water trace organic contaminants and metal(loid)s during managed aquifer recharge into a anaerobic aquifer: Column studies." Water Research **44**(5): 1471-1481.

Perez, S. and D. Barcelo (2007). "Fate and occurrence of X-ray contrast media in the environment." Analytical and Bioanalytical Chemistry **387**(4): 1235-1246.

Preuß, G., U. Willme and N. Zullei-Seibert (2001). "Verhalten ausgewählter Arzneimittel bei der künstlichen Grundwasseranreicherung – Eliminierung und Effekte auf die mikrobielle Besiedlung." Acta Hydrochimica Et Hydrobiologica **29**(5): 269-277.

Putschew, A., S. Wischnack and M. Jekel (2000). "Occurrence of triiodinated X-ray contrast agents in the aquatic environment." Science of the Total Environment **255**(1-3): 129-134.

Rauch-Williams, T., C. Hoppe-Jones and J. E. Drewes (2010). "The role of organic matter in the removal of emerging trace organic chemicals during managed aquifer recharge." Water Research **44**(2): 449-460.

Sacher, F., M. Ehmann, S. Gabriel, C. Graf and H. J. Brauch (2008). "Pharmaceutical residues in the river Rhine - results of a one-decade monitoring programme." Journal of Environmental Monitoring **10**(5): 664-670.

Sacher, F., F. T. Lang, H. J. Brauch and I. Blankenhorn (2001). "Pharmaceuticals in groundwaters - Analytical methods and results of a monitoring program in Baden-Wurttemberg, Germany." Journal of Chromatography A **938**(1-2): 199-210.

Salameh, E. (2001). "The potential of groundwater artificial recharge in the Jordan Valley area / Jordan." Freiberger Forschungshefte **C494** (Selected Contributions to Applied Geology in the Jordan Rift Valley): 63-81.

Salameh, E. and H. El-Naser (2000). "Changes in the Dead Sea level and their impacts on the surrounding groundwater bodies." Acta Hydrochimica Et Hydrobiologica **28**(1): 24-33.

Schaffer, M., N. Boxberger, H. Börnick, T. Licha and E. Worch (2012). "Sorption influenced transport of ionizable pharmaceuticals onto a natural sandy aquifer sediment at different pH." Chemosphere **87**(5): 513-520.

Schaider, L. A., R. A. Rudel, J. M. Ackerman, S. C. Dunagan and J. G. Brody (2014). "Pharmaceuticals, perfluorosurfactants, and other organic wastewater compounds in public drinking water wells in a shallow sand and gravel aquifer." Science of the Total Environment **468–469**(0): 384-393.

Scheytt, T., P. Mersmann, M. Leidig, A. Pekdeger and T. Heberer (2004). "Transport of pharmaceutically active compounds in saturated laboratory columns." Ground Water **42**(5): 767-773.

Scheytt, T., P. Mersmann, R. Lindstadt and T. Heberer (2005). "Determination of sorption coefficients of pharmaceutically active substances carbamazepine, diclofenac, and ibuprofen, in sandy sediments." Chemosphere **60**(2): 245-253.

Scheytt, T. J. (2008). Pharmaceuticals in groundwater as indicators of land use practice and water management. Joint Meeting of the Geological Society of America, Soil Science Society of America, American Society of Agronomy, Crop Science Society

of America, Gulf Coast Association of Geological Societies with the Gulf Coast Section of SEPM.

Schulte-Oehlmann, U., J. Oehlmann and W. Püttmann (2007). "Humanpharmakawirkstoffe in der Umwelt: Einträge, Vorkommen und der Versuch einer Bestandsaufnahme." Umweltwissenschaften und Schadstoff-Forschung **19**(3): 168-179.

Siemens, J., G. Huschek, C. Siebe and M. Kaupenjohann (2008). "Concentrations and mobility of human pharmaceuticals in the world's largest wastewater irrigation system, Mexico City-Mezquital Valley." Water Research **42**(8-9): 2124-2134.

Stevens-Garmon, J., J. E. Drewes, S. J. Khan, J. A. McDonald and E. R. V. Dickenson (2011). "Sorption of emerging trace organic compounds onto wastewater sludge solids." Water Research **45**(11): 3417-3426.

Taggart, M. A., R. Cuthbert, D. Das, C. Sashikumar, D. J. Pain, R. E. Green, Y. Feltrer, S. Shultz, A. A. Cunningham and A. A. Meharg (2007). "Diclofenac disposition in Indian cow and goat with reference to Gyps vulture population declines." Environmental Pollution **147**(1): 60-65.

Teijon, G., L. Candela, K. Tamoh, A. Molina-Diaz and A. R. Fernandez-Alba (2010). "Occurrence of emerging contaminants, priority substances (2008/105/CE) and heavy metals in treated wastewater and groundwater at Depurbaix facility (Barcelona, Spain)." Science of the Total Environment **408**(17): 3584-3595.

Ternes, T. A. (1998). "Occurrence of drugs in German sewage treatment plants and rivers." Water Research **32**(11): 3245-3260.

Ternes, T. A., M. Bonerz, N. Herrmann, B. Teiser and H. R. Andersen (2007). "Irrigation of treated wastewater in Braunschweig, Germany: an option to remove pharmaceuticals and musk fragrances." Chemosphere **66**(5): 894-904.

Ternes, T. A. and R. Hirsch (2000). "Occurrence and behavior of X-ray contrast media in sewage facilities and the aquatic environment." Environmental Science & Technology **34**(13): 2741-2748.

Ternes, T. A., M. Meisenheimer, D. McDowell, F. Sacher, H. J. Brauch, B. H. Gulde, G. Preuss, U. Wilme and N. Z. Seibert (2002). "Removal of pharmaceuticals during drinking water treatment." Environmental Science & Technology **36**(17): 3855-3863.

Tiehm, A., N. Schmidt, P. Lipp, C. Zawadsky, A. Marei, N. Seder, M. Ghanem, S. Paris, M. Zemann and L. Wolf (2012). "Consideration of emerging pollutants in groundwater-based reuse concepts." Water Science and Technology **66**(6): 1270-1276.

Tiehm, A., N. Schmidt, M. Stieber, F. Sacher, L. Wolf and H. Hoetzl (2011). "Biodegradation of Pharmaceutical Compounds and their Occurrence in the Jordan Valley." Water Resources Management **25**(4): 1195-1203.

Toll, M. (2007). An integrated approach for the investigation of unconsolidated aquifers in brackish environment - A case study on the Jordanian side of the Lower Jordan Valley Dissertation, University of Göttingen.

Vieno, N., T. Tuhkanen and L. Kronberg (2007). "Elimination of pharmaceuticals in sewage treatment plants in Finland." Water Research **41**(5): 1001-1012.

Wolf, L., Pöschko, A., Zemann, M., Werz, H., Sawarieh, A., Seder, N., Stieber, M., Sacher, F., Tiehm, A. (2009). "Understanding occurrence of xenobiotics in closed river basins, examples from the Jordan Valley." International Conference on Xenobiotics in the Urban Water Cycle (XENOWAC).

Wolf, L., C. Zwiener and M. Zemann (2012). "Tracking artificial sweeteners and pharmaceuticals introduced into urban groundwater by leaking sewer networks." Science of the Total Environment **430**: 8-19.

Xu, J., L. Wu and A. C. Chang (2009). "Degradation and adsorption of selected pharmaceuticals and personal care products (PPCPs) in agricultural soils." Chemosphere **77**(10): 1299-1305.

Yu, L., G. Fink, T. Wintgens, T. Melin and T. A. Ternes (2009). "Sorption behavior of potential organic wastewater indicators with soils." Water Research **43**(4): 951-960.

Zorita, S., L. Mårtensson and L. Mathiasson (2009). "Occurrence and removal of pharmaceuticals in a municipal sewage treatment system in the south of Sweden." Science of the Total Environment **407**(8): 2760-2770.

Zwiener, C. and F. H. Frimmel (2003). "Short-term tests with a pilot sewage plant and biofilm reactors for the biological degradation of the pharmaceutical compounds clofibric acid, ibuprofen, and diclofenac." Science of the Total Environment **309**(1-3): 201-211.

SM 1: Concentrations of the five substances DIA, IPA, CBZ, IPR and IBU at specific locations from 2008 to 20112. GW: groundwater, SW: surface water, TWW: treated wastewater. Limit of detection (LOD) for pharmaceuticals in GW and SW: 20 ng/L, for ICM: 10 ng/L. LOD for TWW: 50 ng/L. Mean concentrations were calculated using 0.5*LOD for results <LOD.

Sampling sites	Water type	Diatrizoic acid				Iopamidol				Carbamazepine				Iopromide				Ibuprofen			
		2008 Apr [ng/L]	2011 Mar [ng/L]	2011 Sep [ng/L]	2012 Feb [ng/L]	2008 Apr [ng/L]	2011 Mar [ng/L]	2011 Sep [ng/L]	2012 Feb [ng/L]	2008 Apr [ng/L]	2011 Mar [ng/L]	2011 Sep [ng/L]	2012 Feb [ng/L]	2008 Apr [ng/L]	2011 Mar [ng/L]	2011 Sep [ng/L]	2012 Feb [ng/L]	2008 Apr [ng/L]	2011 Mar [ng/L]	2011 Sep [ng/L]	2012 Feb [ng/L]
JVN 1	GW	< LOD	22	13	18	< LOD	< LOD	910	39	< LOD	< LOD	500	< LOD	< LOD	< LOD	250	< LOD	< LOD	< LOD	< LOD	< LOD
JVN 2	GW	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	64	< LOD	< LOD	< LOD	64	< LOD	< LOD	< LOD	< LOD	11	< LOD	< LOD	< LOD	< LOD
Sp 027 -22	GW	250	290	190	490	38	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD
Sp 026 - 22	GW	370	350	160	380	< LOD	14	< LOD	730	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	54
AB 1110	GW	110	140	n.s.	n.s.	< LOD	< LOD	n.s.	n.s.	< LOD	< LOD	n.s.	n.s.	< LOD	< LOD	n.s.	n.s.	< LOD	< LOD	n.s.	n.s.
Shuna no.5	GW	66	330	100	580	44	< LOD	19	37	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	59
FU 192 DA 28	GW	130	11	17	37	< LOD	< LOD	16	87	< LOD	< LOD	< LOD	78	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD
We 1 - 28	GW	130	710	360	760	< LOD	23	25	13	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD
AN 1025	GW	88	n.s.	55	71	72	n.s.	1400	36,000	< LOD	n.s.	< LOD	83	< LOD	n.s.	< LOD	18	< LOD	n.s.	< LOD	58
AN 1026	GW	86	97	45	54	< LOD	4400	1300	33,000	34	< LOD	< LOD	70	70	96	< LOD	24	< LOD	< LOD	< LOD	54
JVN 3 -- FU 40 Da 13	GW	< LOD	< LOD	16	< LOD	< LOD	< LOD	59	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	14	< LOD	< LOD	< LOD	< LOD
JVN2 pool	SW	< LOD	n.s.	< LOD	140	< LOD	n.s.	990	1200	< LOD	n.s.	470	810	< LOD	n.s.	270	220	< LOD	n.s.	< LOD	74
T.O. 33 (KAC)	SW	< LOD	< LOD	< LOD	< LOD	< LOD	13	12	78	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD
T.O. 65 (KAC)	SW	< LOD	140	< LOD	< LOD	< LOD	18	12	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	< LOD	58
KTR	SW	< LOD	850	75	350	590	1000	1800	6000	760	2100	1500	1200	600	880	640	720	1400	< LOD	< LOD	85
IW 22 pool	SW	< LOD	290	< LOD	290	4200	970	2200	3500	1000	1900	1300	1700	1400	810	550	980	53	65	< LOD	130
KAC 28	SW	< LOD	< LOD	98	260	3800	57	2100	7800	990	76	1700	1400	1400	23	800	680	74	< LOD	< LOD	120
Shueib Reservoir	SW	< LOD	270	27	220	72	1600	700	1000	210	1200	410	340	< LOD	790	19	22	32	< LOD	< LOD	430
Kafrein Reservoir	SW	26	< LOD	100	120	< LOD	8000	78,000	17,000	88	170	470	73	74	4500	130	500	< LOD	< LOD	< LOD	250
Tal Mantah WWTP	TWW	< LOD	< LOD	< LOD	< LOD	33,000	36	1300	6600	10,000	17,000	4300	2800	< LOD	1700	51	< LOD	< LOD	95	280	340
Es Sir WWTP	TWW	< LOD	< LOD	< LOD	300	2300	160,000	680,000	600,000	1800	3700	3400	200	2200	280,000	2100	24,000	< LOD	130	180	380
As Salt WWTP	TWW	110	< LOD	< LOD	< LOD	1600	810	6900	13,000	< LOD	82	2800	< LOD	190	1900	120	260	750	< LOD	200	290
Fuheis WWTP	TWW	< LOD	120	< LOD	< LOD	2200	15,000	15,000	6600	3600	6900	7900	3800	860	< LOD	120	16	< LOD	< LOD	110	280
Statistical numbers																					
Mean GW	[ng/L]	113	196	96	240	18	447	380	6992												29
Mean SW	[ng/L]	8	224	40	174	1085	1665	10,727	4573									200	18		145
Mean TWW	[ng/L]	46	49	25	94	9775	43,962	175,800	156,550	3856	6921	4600	1706	819	70906	598	6075	195	61	193	323
Detections GW	[%]	73	80	90	90	27	30	80	70	9	0	20	30	9	10	10	40	0	0	0	40
Detections SW	[%]	13	57	50	75	50	100	100	88	63	71	75	75	50	71	75	75	50	14	0	88
Detections TWW	[%]	25	25	0	25	100	100	100	100	75	100	100	75	75	75	100	75	25	50	100	100
Median of positive detects in GW	[ng/L]	120	215	55	226	44	23	62	87	34		282	78	70	96	250	16				56
Median of positive detects in SW	[ng/L]	26	280	87	240	2195	970	1395	3500	760	1200	1300	1200	1000	810	550	680	64	65		125
Median of positive detects in TWW	[ng/L]	110	120		300	2250	7905	10,950	9800	3600	5300	3850	2800	860	1900	120	260	750	113	190	315
Ratio of Median SW/GW	[-]	0.2	1.3	1.6	1.1	50	42	23	40	22		5	15	14	8	2	43				2
Ratio of Median TWW/GW	[-]	0.92	0.56	0.00	1.33	51	344	178	113	106		14	36	12	20	0.48	16				6

SM 2: Accumulated numbers of different S-values for all individual sampling sites for diatrizoic acid (DIA) and iopamidol (IPA) and their corresponding significance level. Positive S-values indicate increasing trends, negative S-values indicate decreasing trends.

Significance level	S-value	Number of groundwater sites		Number of surface water sites	
[%]	**[-]**	**DIA**	**IPA**	**DIA**	**IPA**
4,20	6	0	0	0	1
	5	0	1	1	1
16,70	4	1	1	0	1
	3	2	2	3	0
37,50	2	1	0	2	2
	1	1	4	0	1
62,50	0	2	0	1	0
	-1	1	0	1	1
37,50	-2	0	0	0	0
	-3	0	1	0	0
	-4	1			
total S		8	16	14	19

3. Tracking Changing X-ray Contrast Media Application Practice to an Urban Influenced Karst Aquifer in the Wadi Shueib, Jordan

Reproduced from: Zemann M., Wolf L., Grimmeisen F., Tiehm A., Klinger J., Hötzl H., Goldscheider N. (2015): Tracking Changing X-ray Contrast Media Application to an Urban-influenced Karst Aquifer in the Wadi Shueib, Jordan. Environmental Pollution 198: 133-143, doi: 10.1016/j.envpol.2014.11.033. The final publication is available at www.sciencedirect.com.

Abstract: Sewage input into a karst aquifer via leaking sewers and cesspits was investigated over five years in an urbanized catchment. Of 66 samples, analyzed for 25 pharmaceuticals, 91 % indicated detectable concentrations. The former standard iodinated X-ray contrast medium (ICM) diatrizoic acid was detected most frequently. Remarkably, it was found more frequently in groundwater (79 %, median: 54 ng/l) than in wastewater (21 %, 120 ng/l), which is supposed to be the only source in this area. In contrast, iopamidol, a possible substitute, spread over the aquifer during the investigation period whereas concentrations were two orders of magnitude higher in wastewater than in groundwater. Knowledge about changing application of pharmaceuticals thus is essential to assess urban impacts on aquifers, especially when applying mass balances. Since correlated concentrations provide conclusive evidence that, for this catchment, nitrate in groundwater rather comes from urban than from rural sources, ICM are considered useful tracers.

3.1. Introduction

Despite the worldwide release and occurrence of pharmaceutical residues in the aquatic environment, little is still known about the long-term evaluation of concentrations, especially in groundwater. Many studies deal with concentrations in effluents from wastewater treatment plants (WWTP) (Ternes and Hirsch, 2000; Bueno et al., 2012; Loos et al., 2013; Kostich et al., 2014), surface water (Schwab et al., 2005; Sacher et al., 2008; Loos et al., 2009; Vulliet and Cren-Olivé, 2011), and groundwater (Sacher et al., 2001; Loos et al., 2010; Maeng et al., 2011; López-Serna et al., 2013), although most of them address single sampling campaigns. Time series data related to groundwater have been scarcely published (Wolf et al., 2012; Zemann et al., 2014) despite the fact that they can contribute significantly to a better understanding of substance behavior and long-term threads. As groundwater resources are often used for drinking water supply, special attention needs to be directed to exposed locations (Tiehm et al., 2011) such as karst aquifers and urban areas. Karst aquifers are known to be highly vulnerable because contaminants can easily enter the subsurface through cracks and swallow holes and are quickly transported via conduits (Ford and Williams, 2007). Urban areas, on the other hand, endanger the groundwater quality as they feature multiple pollution sources for the underlying aquifers. Consequently, a conglomerate of cities in karst areas has an increased potential for groundwater contamination.

Until today, only a few studies in karst areas have focused on pharmaceutical contaminations in groundwater, e.g. Einsiedl et al. (2010), Metcalfe et al. (2011); Reh et al. (2013), Katz et al. (2009) or Morasch (2013). However, there is vivid evidence of contaminant transport in karst, e.g. of nitrate (Huebsch et al., 2013), herbicides (Hillebrand et al., 2014), fecal bacteria (Pronk et al., 2006), pesticides (Mahler and Massei, 2007) or chloride used as a wastewater indicator (Schmidt et al., 2013).

Pharmaceuticals in urban groundwater primarily come from leaking sewers infiltrating wastewater into the subsoil (Osenbrück et al., 2007; Musolff et al., 2009). The urban impact on groundwater, regarding pharmaceuticals, was documented by various studies such as Wolf et al. (2012) or Reinstorf et al. (2008). Exfiltration rates calculated from pharmaceutical concentrations could be described as percentage of the dry weather flow. They show huge variations ranging from around 1 % in the cities of Linz (Fenz et al., 2005) or Tokyo (Kuroda et al., 2012), 9.9 to 13.0 % in Leipzig (Musolff et al., 2010), and between 6 and 50 % in Barcelona (Jurado et al., 2014). Other possible contamination sources might

come from infiltrating rivers fed with effluents from WWTP (López-Serna et al., 2013). The first possibility in particular introduces the highest concentrations of pharmaceuticals, comparable to inflows of WWTP. Pharmaceuticals may enter the entire urban sewage system, e.g. via the analgetics and anti-inflammatories used at home or as point sources from hospitals (Escher et al., 2011; Orias and Perrodin, 2013). Whereas, in Switzerland, 50 % of the X-ray contrast media (ICM) were found to have been administered in the investigated hospital, the respective remaining quantities were administered to out-patients. Concentrations in the hospital wastewater thus correlate with the daily consumption in the hospital reaching up to 580 g/d for iomeprol (IME) and 384 g/d for iopamidol (IPA) (Weissbrodt et al., 2009). A total contribution of 51 % analgetics and 49 % antibiotics from hospitals to the aggregate pharmaceutical inflow of the related WWTP was detected in Coimbra (Portugal). ICM reached 13 %, although only IME was investigated (Santos et al., 2013). Mean ICM effluent concentrations were determined at a Swiss hospital with 348 µg/l for diatrizoic acid (DIA), 2599 µg/l for IPA, and 439 µg/l for IME (Kovalova et al., 2012).

The prevalent occurrence of pharmaceuticals in Jordan water sources was first described by Tiehm et al. (2011) and Wolf et al. (2009). The first long-term assessment of trace organics in the Lower Jordan Valley indicated increasing groundwater pollution due to the intense reuse of treated wastewater for irrigation in agriculture. Concentrations and occurrences of the X-ray contrast media IPA and DIA were identified to show shifting prescription patterns in Jordan hospitals (Zemann et al., 2014). Wastewater concentrations measured upstream and downstream of two hospitals in Amman documented an increased input with concentrations between 3 to 7 µg/l of the analgetic diclofenac (DIC) and a single detection (26 µg/l) of the anti-inflammatory ibuprofen (IBU). However, they could not be related to influent concentrations at three local WWTP (Alahmad and Alawi, 2010).

The present study focuses on two wadis located around 40 km southeast of Amman, the capital of Jordan. The area features a high groundwater pollution risk as the vulnerable outcropping karst is covered by urban areas. Against this background, the following research issues were investigated:

- What substances occur in the urban karst aquifer of Wadi Shueib and Kafrein? What concentration levels do they reach?

- How can we make use of pharmaceutical occurrence and concentrations above the qualitative level?
- How can pharmaceuticals be used as tracers?
- Do temporal trends of X-ray contrast media (ICM) fit with recent findings of iopamidol (IPA) and diatrizoic acid (DIA) in the Jordan Valley, which were assumed to be due to prescription changes?

3.2. Materials and methods

3.2.1. Study area

Around 130,000 inhabitants live mainly in the upper part of Wadi Shueib in the cities Salt and Fuheis. The investigated area is rather steep with a dense drainage network. The Wadi Shueib course acts as a receiving stream for the entire area, and discharges into the Shueib reservoir at the southwestern outlet of the wadi towards the Jordan Valley (Werz, 2006). Two WWTPs in Fuheis and Salt release their effluents into this stream where they mix with the unused spring discharge. Within this study, the urban impact was investigated for five springs (Azzraqu, Baqquriah, Hazzir, Shoreia, and Farkha) and two wells (Yesidia, Um Attija). The main catchment of the Hazzir and Shoreia spring groundwater contribution zone lies directly within the urban area of Salt (Figure 17). The catchment zone for the Baqquriah spring is located downstream of the city, including parts of the upper aquifer discharged by the Hazzir, Shoreia, and Azzraqu springs (Figure 18). The Yesidia wells are located upstream of the city while the Um Attija well is located inside the city of Salt. Because the geology of the whole area is mainly characterized by outcropping karstified limestone (Figure 18), the groundwater in the area is highly vulnerable to anthropogenic hazards (Werz, 2006). Although an extensive sewer network exists inside the city, people still use septic tanks to avoid paying the costs of sewage connection. As costs for suction trucks are expensive as well, permeable bottoms and overflowing pits are common (Margane et al., 2010). In a survey performed in 2007, 58 % of the interviewees said they never emptied their cesspits (GTZ and WMIA, 2007). Therefore, large quantities of untreated wastewater extensively infiltrates into the subsoil. In 2009, the number of septic tanks in Salt was 3664, i.e. septic tanks were found in around 25 % of the households (Al-Kharabsheh et al., 2013). This is similar to estimations made for urban areas in northern Jordan (van Afferden et al., 2010).

Groundwater is mainly present in two aquifer systems (see Figure 18), Es Sir Limestone (A7) on top and the Hummar formation (A4) below. Both are assumed to be locally interconnected due to the tectonic conditions. All springs mentioned above drain from the As Sir limestone, whereas the Yesidia wells pump from the Hummar formation.

Being characterized by similar geologic conditions, Wadi Kafrein is located adjacent to the south of Wadi Shueib. A suburb of the capital Amman conveys its wastewater to the Es Sir WWTP, from where it is discharged into the wadi stream and later on stored in the Kafrein reservoir before being used for irrigation in the Jordan Valley. A medical center in the area discharges to this WWTP together with two pharmaceutical manufacturers (Alahmad and Alawi, 2010).

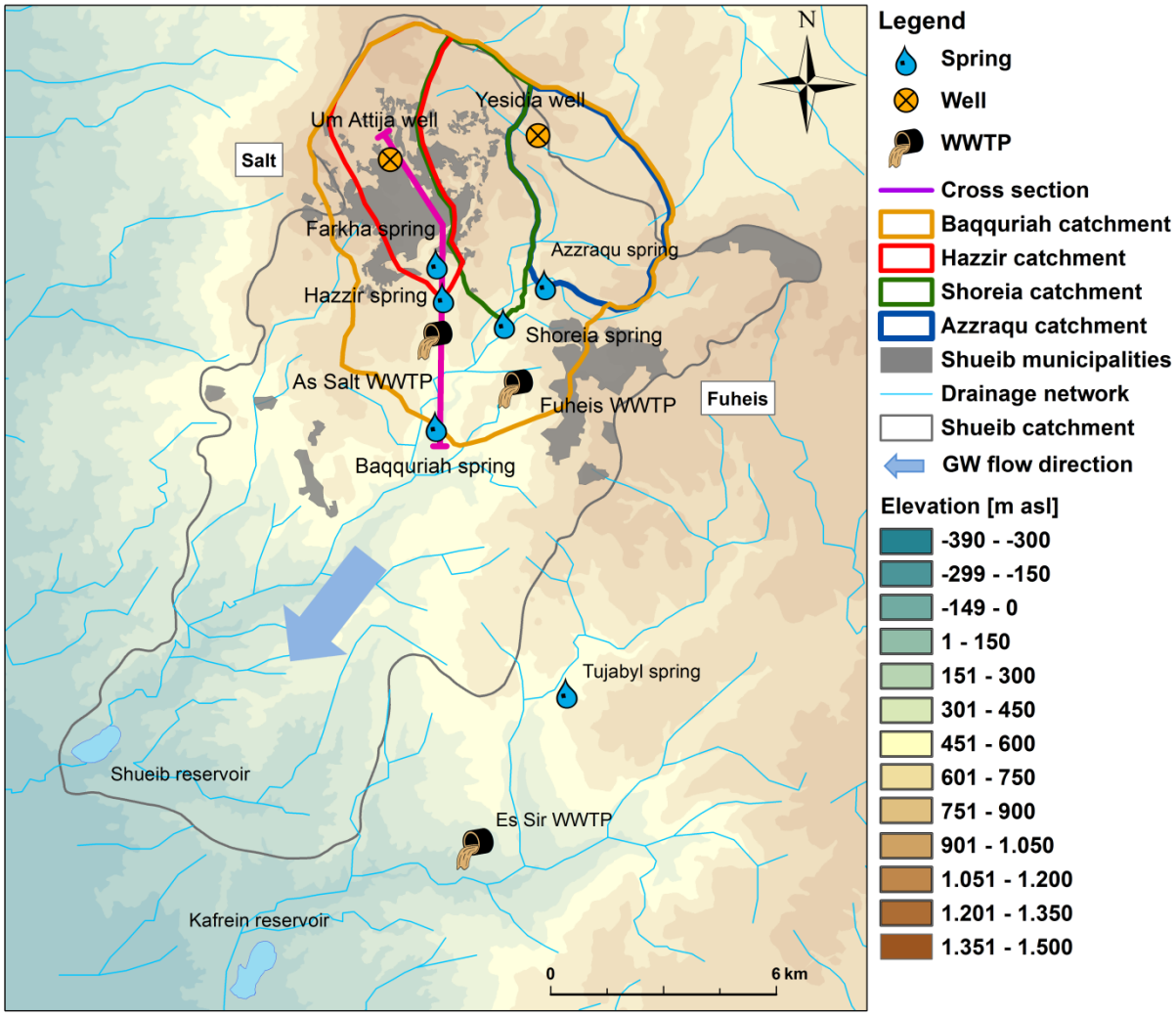

Figure 17: Sampling locations in Wadi Shueib, settlements, wastewater treatment plants (WWTP), and subsurface catchments of the main springs. Subsurface catchments were extracted from Margane et al. (2010).

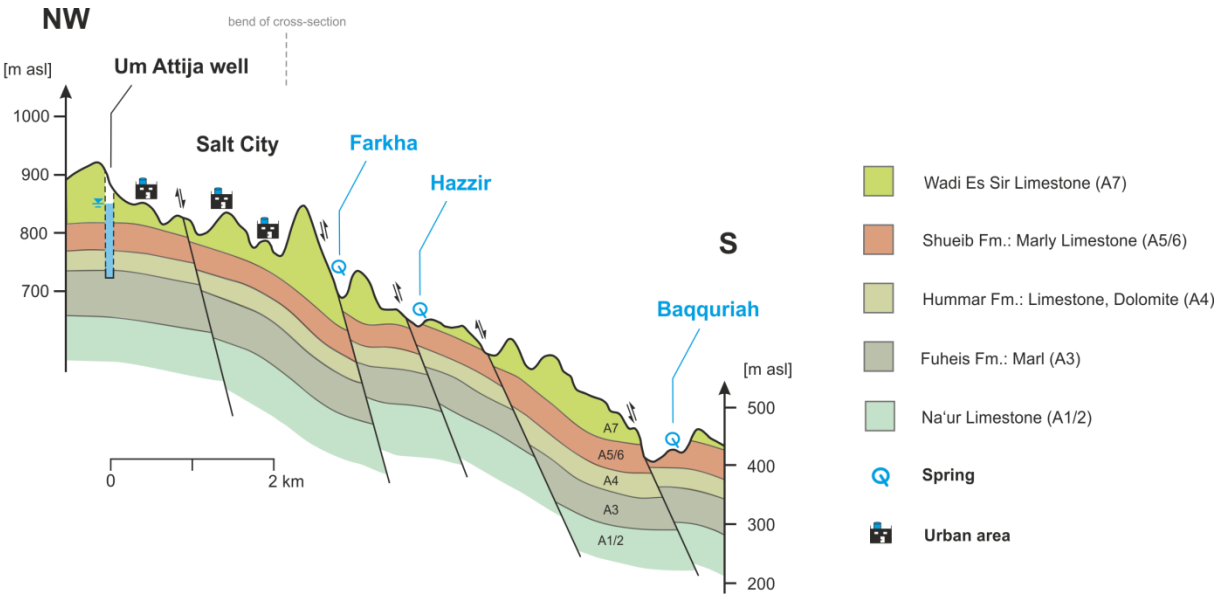

Figure 18: Schematic geological cross section (S- N – NW) through Wadi Shueib cutting two springs and one well[1].

3.2.2. Sampling campaigns and site selection

From 2008 to 2012, four water quality sampling campaigns were conducted in the study area. Some locations were already screened in 2007, including 5 samples from Kafrein reservoir taken within one week. In 2008, WWTPs and the reservoirs were sampled in April, while groundwater samples were taken in November. Minor parts of the dataset (some WWTPs and reservoirs) were already included in Zemann et al. (2014). In March 2012, six samples were taken at Hazzir spring during a 24-hour campaign and two others were taken in June 2013. All in all, 25 substances (Table 9), including 16 pharmaceuticals and 9 ICM, major ions, fecal bacteria, and physico-chemical parameters (EC), were monitored in the groundwater, surface water and treated wastewater with a total of 66 samples. The general sampling strategy was to completely assess all main water sources in the area, together with the effluents of the local WWTP as a possible contamination source. Sampling times were chosen to be before and at the end of the rainy season (November to March). No seasonal effects could be found within this study, however due to the limited samples, such effects cannot be excluded. Samples were taken from the WWTPs (Fuheis, As Salt, Es Sir), reservoirs (Shueib and Kafrein), and the main springs (Hazzir, Farkha, Azzraqu, Baqquriah, Shoreia, Tujabyl) and wells (Um Attija, Yesidia). Sampling locations are given in Figure 17.

[1] Due to the schematic character of the profile, the location of the wells with regards to the stratigraphy is not drawn at the exact position. A temporal sequence of normal faults and reverse faults should also not be drawn from the profile.

3.2.3. Sampling and analysis

At each location, grab samples were taken for pharmaceuticals and major cations and anions. In situ measurements of EC, redox, temperature, and oxygen were done using a WTW Multi 3430 device. Sampling preparation, storage, transport, and quality assessment at the laboratory were described in detail in Zemann et al. (2014). The pharmaceutical analysis method is described in Sacher et al. (2008), whereas the analyses of both studies were performed in the same laboratory using the same method. Samples for ion analysis were filtered with cellulose mixed ester (CME) filters (Roth: 0.45 µm). Cation samples where acidified with concentrated nitric acid (60 %) (1 ml per 100 ml sample) and analyzed by IC (DIONEX 1100) following DIN 38406. Anion analyses where conducted at the laboratories of the Water Authority of Jordan (WAJ) according to Eaton et al. (2005). The *E. coli* were identified by the most probable number (MPN) method using the Colilert system (IDEXX Laboratories Inc.).

Table 9: Screened substances sorted by generic product categories and acronyms used in the text.

Analgetics / Antiphlogistics	X-ray Contrast Media (ICM)	Lipid Lowering Agents
Pentoxifylline (PEN)	Diatrizoic acid (DIA)	Bezafibrate (BEZ)
Diclofenac (DIC)	Iodipamid (IDI)	Clofibric acid (CFA)
Ibuprofen (IBU)	Iohexol (IHE)	Etofibrate (ETO)
Indomethacin (IND)	Iomeprol (IME)	Fenofibrate (FFI)
Naproxen (NAP)	Iopamidol (IPA)	Fenofobric acid (FFA)
Phenacetin (PHE)	Iopromide (IPR)	Gemfibrozil (GEM)
Anti-inflammatory Drugs	Iotalamic acid (ITA)	**Antipsychotic Drugs**
Fenoprofen (FEN)	Ioxaglic acid (IXA)	Diazepam (DZP)
Ketoprofen (KET)	Ioxithalamic acid (IXI)	**Antiepileptic Drugs**
		Carbamazepine (CBZ)

3.3. Results

3.3.1. Pharmaceutical concentrations and detection rates

Six pharmaceuticals could be detected in the groundwater: The X-ray contrast media DIA (79 % detection frequency), IPA (51 %), and IHE (5 %) (Table 10). Besides, IBU was found in 14 % of the samples, CBZ in 13 %, and FFI in 6 %. Most substances showed lower detection frequencies in groundwater than in surface water and treated wastewater. Only DIA was found more frequently in groundwater than in treated wastewater. For most substances, e.g. ICM, detection frequencies in treated wastewater were slightly lower than

in surface water. Again, DIA showed the biggest discrepancy with 21 % in treated wastewater as compared to 85 % in surface water. BEN, IBU, and NAP showed decreasing frequencies from treated wastewater through to surface water to groundwater. FEN was only present in groundwater.

Table 10: Occurrence of pharmaceuticals and X-ray contrast media in three water types in the wadis Shueib and Kafrein. GW = groundwater, SW = surface water, TWW = treated wastewater. Total of samples = 66, total of sampling points = 14. Samples in GW = 39 for ICM and 32 for other pharmaceuticals, SW = 13, TWW = 14. LOD for ICM is 10 ng/l in GW and SW and 50 ng/l in TWW. LOD for pharmaceuticals is 20 ng/l in GW and SW and 50 ng/l in TWW. The bold substances were present in GW.

	GW	SW	TWW	GW	SW	TWW	GW	SW	TWW
	Detection frequency			Median of positives			Maximum values		
	[%]	[%]	[%]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]
Bezafibrate (BEZ)	0	15	64		26	270		89	480
Carbamazepine (CBZ)	13	100	86	80	240	3100	100	1200	7900
Clofibric acid (CFA)	0	0	7			150			150
Diazepam (DZP)	0	0	14			404			720
Diclofenac (DIC)	0	38	21		35	240		140	390
Etofibrate (ETO)	0	0	0						
Fenofibrate (FFI)	6	0	0	110			130		
Fenofibric acid (FFA)	0	0	7			160			160
Fenoprofen (FEN)	0	0	0						
Gemfibrozil (GEM)	0	100	71		150	1300		510	4800
Ibuprofen (IBU)	14	23	71	56	250	225	65	430	750
Indomethacin (IND)	0	0	0						
Ketoprofen (KET)	0	0	14			64			64
Naproxen (NAP)	0	8	21		69	95		69	240
Pentoxifyllin (PEN)	0	0	0						
Phenacetin (PHE)	0	0	0						
Diatrizoic acid (DIA)	79	85	21	54	110	120	220	270	300
Iodipamid (IDI)	0	0	0						
Iohexol (IHE)	5	77	71	19	5800	315	27	39,000	9000
Iomeprol (IME)	0	77	64		88	1400		5300	360,000
Iopamidol (IPA)	51	92	93	65	850	6900	1900	78,000	680,000
Iopromide (IPR)	0	92	79		995	860		4500	280,000
Iotalamic acid (ITA)	0	0	0						
Ioxaglic acid (IXA)	0	0	0						
Ioxithalamic acide (IXI)	0	0	0						

The term “pharmaceutical detection rates” is used in this study to describe the number of pharmaceutical substances detected at a sampling site within the set of 25 pharmaceutical substances screened in this study. This term was already introduced by Schaider et al. (2014). The different water types (SM 3) show a clear decreasing trend for this parameter

from treated wastewater through to surface water to groundwater for all campaigns, except for that of 2012. However, no temporal trends are visible from 2008 to 2012. Furthermore, until 2012, almost exclusively DIA and IPA were present in groundwater.

Most substances showed the highest concentrations in treated wastewater and the lowest in groundwater (Table 10). As an exception, DIA, NAP, IHE, and IPR featured higher median concentrations in surface water than in treated wastewater, but again had the lowest concentrations in groundwater (except for DIA). Median concentrations ranged between 19 and 110 ng/l in groundwater, 26 and 995 ng/l in surface water, and 64 and 6750 ng/l in treated wastewater. Maximum concentrations were found for IPA in all water types, where the maximum concentration in groundwater was 1900 ng/l at Tujabyl spring.

3.3.2. Diatrizoic acid

DIA was present in all groundwater samples of Baqquriah, Farkha, Hazzir, Shoreia, Tujabyl, and Um Attija from 2008 to 2012. It was never found at Azzraqu and Yesidia wells (compare SM 4), therefore showing a high, constant presence within this study. A spatio-temporal overview is given in Figure 19. Concentrations in groundwater ranged between 13 and 180 ng/l. Concentrations in surface water were between 26 and 270 ng/l, whereas it was only present twice in treated wastewater with 110 and 120 ng/l. Regarding Wadi Shueib, DIA showed constant occurrence at four springs and two wells, while it was present only once in treated wastewater. Mean and median values (see SM 4) for all water types range in the same order of magnitude, indicating no dilution effects from treated wastewater to groundwater or biodegradation processes. In none of the water types, concentrations reveal a distinct temporal trend.

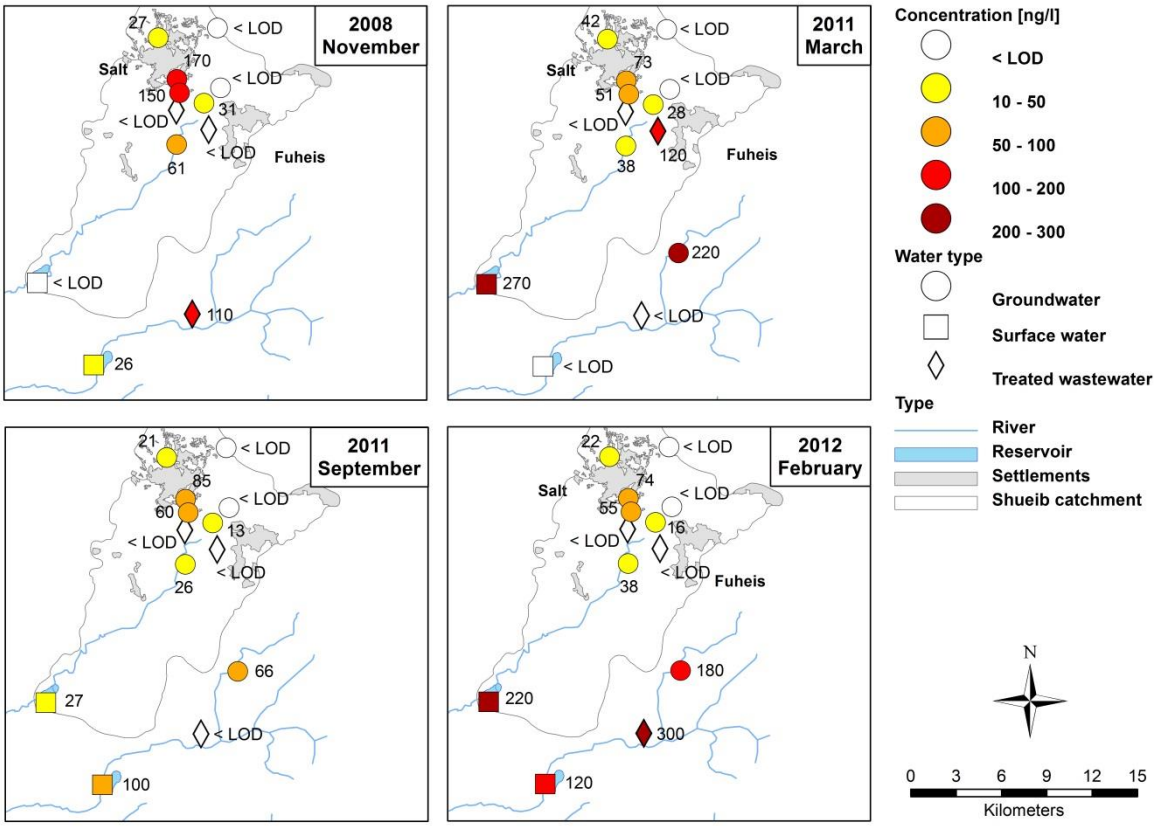

Figure 19: Diatrizoic acid concentrations in wells and springs around the city Salt between 2008 and 2012.

3.3.3. Iopamidol

Absolute numbers for IPA concentrations are given in the supplementary materials (SM 4) while their spatio-temporal occurrences are shown in Figure 20. In 2008, IPA was only found at Hazzir spring. Henceforth, detection rates were increasing gradually until 2012, when IPA was found in 75 % of the groundwater samples. Over the whole period, it was never found at Azzraqu spring and Yesidia well. Along with the detection rates, mean concentrations were gradually increasing as well. Except for Kafrein reservoir in 2008 and Es Sir WWTP in 2007, IPA was present in all samples of surface water and treated wastewater. Concentrations range from 11 to 1900 ng/l in groundwater, from 72 to 78,000 ng/l in surface water, and from 810 to 680,000 ng/l in treated wastewater. Therefore, concentrations of IPA decreased from treated wastewater through to surface water to groundwater.

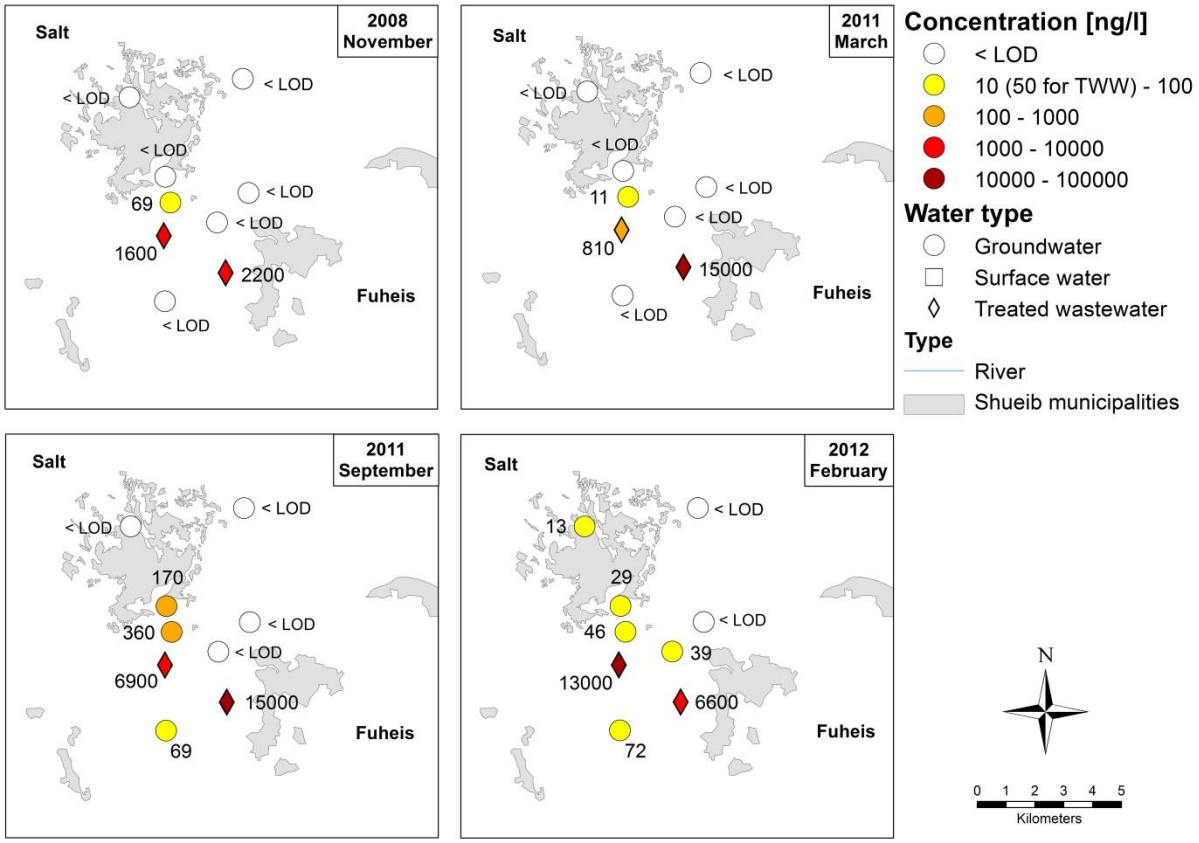

Figure 20: Iopamidol concentrations in wells and springs around the city Salt from 2008 to 2012.

3.3.4. Occurrence of other anthropogenic contaminants (fecal bacteria, nitrate)

Samples for fecal bacteria tests were always taken in parallel to the pharmaceutical samples. As some of the locations could not be considered during all campaigns, an overview of the dates and the sampled locations is given in Table 11. At each groundwater sampling site, *E. coli* was present at least once. While Um Attija and Yesidia showed only minor rates with 2 MPN/100 ml, Shoreia, Baqquriah, and Azzraqu featured rates between 18 and 72 MPN/100 ml. The highest *E. coli* rates were found at Hazzir and Tujabyl spring with a mean of 582 and 1260 MPN/100 ml, respectively. Nitrate concentration in the groundwater ranges from 25 to 64 mg/l in springs and from 1 to 38 mg/l in wells. Highest concentrations were detected at Hazzir spring, while the lowest were found at Yesidia well.

Table 11: *E. coli*, nitrate, and pharmaceutical detections (number of detected substances out of 25) at springs and wells in Wadi Shueib between 2008 and 2012. PHA = pharmaceutical, NOS = number of single substances, n.a. = not analyzed.

	February 2012			September 2011			March 2011			April/November 2008		
	PHA	***E. coli***	**Nitrate**	**PHA**	***E. coli***	**Nitrate**	**PHA**	***E. coli***	**Nitrate**	**PHA**	***E. coli***	**Nitrate**
	[NOS]	[MPN/ 100 ml]	[mg/l]	[NOS]	[MPN/ 100 ml]	[mg/l]	[NOS]	[MPN/ 100 ml]	[mg/l]	[NOS]	[MPN/ 100 ml]	[mg/l]
Azzraqu spring	1	17	27.0	0	33	25.1	0	< 1.8	27.3	0	< 1.8	n.a.
Baqquriah spring	3	5	43.3	2	170	36.1	2	33	36.9	1	79	n.a.
Farkha spring	3	920	57.1	2	23	64.1	1	6.8	61.6	1	49	n.a.
Hazzir spring	2	110	50.6	2	490	53.9	3	1600	54.5	2	130	n.a.
Shoreia spring	2	13	34.5	1	33	28.8	1	n.a.	30.1	1	8	n.a.
Tujabil spring	3	1600	49.9	1	920	56.6	2	n.a.	50.8	n.a.	n.a.	n.a.
Um Attija well	2	n.a.	38.0	1	< 1.8	37.0	1	n.a.	37.3	1	2	n.a.
Yesidia well	0	2	6.6	0	n.a.	21.0	0	n.a.	0.8	0	< 1.8	n.a.
Total detections	16 in 8	7 in 7		9 in 8	6 in 7		10 in 8	3 in 4		6 in 7	5 in 7	
Mean concentration [mg/l]			38.4			40.3			34.6			

3.3.5. Correlation of pharmaceuticals with nitrate and *E. coli*

The simultaneous occurrence of different pollution indicators at many sampling sites suggested that high numbers of pharmaceutical detections might correlate with high nitrate and *E. coli* values. The different indicators are plotted against each other (Figure 21). Correlation analyses (Spearman and Kendall) confirm the positive correlation of increasing pharmaceutical detection rates with both, increasing nitrate concentrations (Figure 22) and *E. coli* (Figure 23) numbers for a significance level of < 1 %.

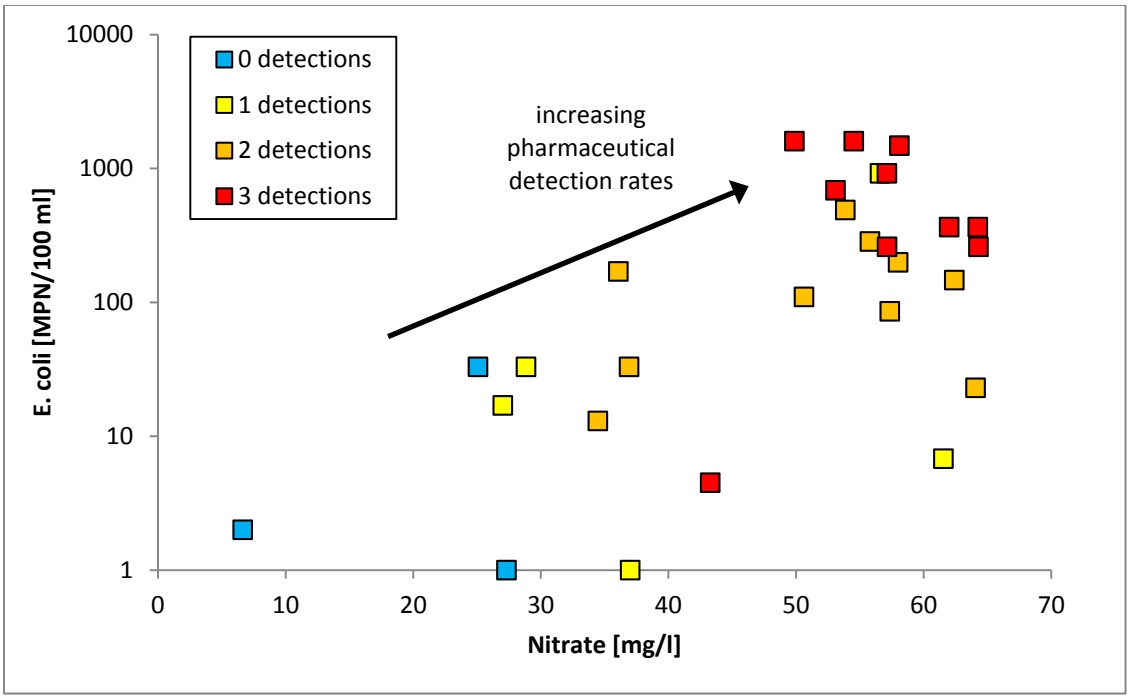

Figure 21: Correlation of *E. coli* and nitrate with the pharmaceutical detection rate in the groundwater of Wadi Shueib/Kafrein for samples taken between 2008 and 2012.

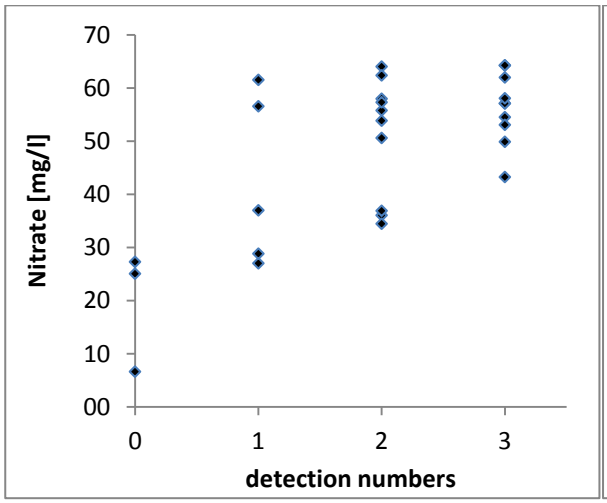

Figure 22: Pharmaceutical detection rates plotted against nitrate.

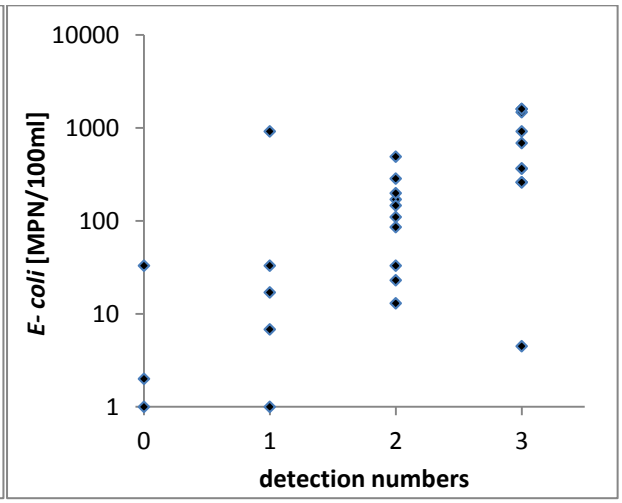

Figure 23: Pharmaceutical detection rates plotted against *E. coli.*

Due to the frequent detection of DIA, correlation analyses (as above plus Spearman) were also performed for nitrate and DIA (see Figure 24). DIA is strongly positively correlated with nitrate (significance level < 10^{-6}).

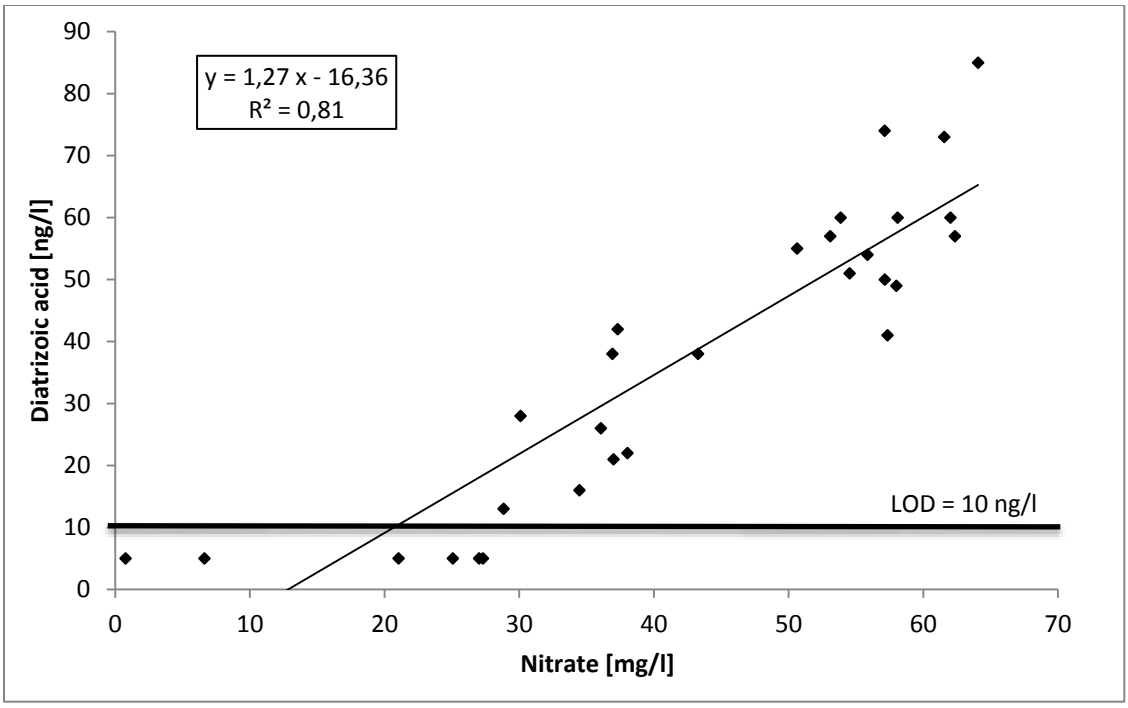

Figure 24: Correlation of nitrate and diatrizoic acid concentrations in the groundwater of Wadi Shueib and Kafrein for all samples taken between 2008 and 2012. Concentrations < LOD were calculated with 0.5*LOD. LOD for DIA = 10 ng/l.

3.4. Discussion

3.4.1. Pharmaceuticals in karst and urban aquifers

The following section tries to give an overview of published research related to pharmaceuticals in urban groundwater and in karst, indicating concentration ranges and substance spectrums. However, only few studies on pharmaceuticals in karst systems are available.

Concentrations reported in karst springs and groundwater usually range in the lower ng/l level. Springs at the Frankonian Alb (Germany) featured concentrations of IBU between 4 and 15 ng/l and DIC between 1 and 7 ng/l with an effluent from a wastewater treatment plant as the most likely source (Einsiedl et al., 2010). CBZ concentrations between 1.5 and 1.9 ng/l were detected in a large karstic basin (Florida/USA), coming from the land application of treated wastewater (Katz et al., 2009). Two karst springs in Switzerland showed different pharmaceuticals, amongst them DIC (0.7 to 1 ng/l), KET (4 to 8 ng/L), and NAP (4 ng/l) (Morasch, 2013). Results from a two year study in a German karst system underlying an urban area (Reh et al., 2013) are more similar to the results from Jordan documented in this study. Although many more substances were detected,

frequencies and concentrations were similar for CBZ (detection frequency: 12.9 %, median concentration: 38.4 ng/l), iohexol (4.3 %, 26.2 ng/l), and IBU (0.6 %, 23.0 g/l). A recently published study of the Lower Jordan Valley mainly showed similar results regarding concentration ranges and detected substances and frequencies (Zemann et al., 2014), e.g. DIA occurred in 79 % of all water samples, while IPA was present in 49 % and CBZ in 13 %. The differences in maximum concentrations or detection frequencies can be explained by the smaller investigation area, featuring a potentially more specific administration of single substances and different transport mechanisms.

Studies in urban areas typically show a broad spectrum of pharmaceuticals in groundwater (Ellis, 2006; Wolf et al., 2012; López-Serna et al., 2013). Therefore, the total of six substances found in groundwater within this study does not represent a vast urban impact. However, there could be different reasons (dilution, biodegradation, local description and application practice) for the absence of substances. CBZ could not be detected in any groundwater sample between 2008 and February 2012, despite its constant occurrence in the local WWTP (Table 10). During two short-term sampling campaigns at Hazzir spring in March 2012 and June 2013, CBZ could be detected occasionally with a maximum concentration of 100 ng/l. CBZ is known to be persistent against biodegradation processes and has often been suggested as a suitable anthropogenic tracer. Consequently, various reports exist on its frequent occurrence in urban groundwater, e.g. (Fenz et al., 2005; Osenbrück et al., 2007; Kuroda et al., 2012; Jurado et al., 2014). The overall detection frequency (13 %) is significantly lower than in other studies, for example the 33 % detected in Rastatt (Germany) (Wolf et al., 2012), 66 % in Linz (Austria) (Fenz et al., 2005) or the 96 % in Barcelona (Spain) (Jurado et al., 2014). The comparatively scarce occurrence of CBZ in the springs in this study might be attributed to the LOD, which was 10 ng/l for ICM but 25 ng/l for CBZ.

In contrast to CBZ, the X-ray contrast media DIA and IPA were found frequently within this study with concentrations in the range of 10 ng/l to a few hundred ng/l (compare Table 10 and SM 4). Wolf et al. (2012) detected DIA in groundwater (Rastatt, Germany) with 27 % frequency, IPA with 4 %, and average concentrations of DIA with 66 ng/l and IPA with 12 ng/l. IPA concentrations ranging from 6 to 272 ng/l were reported after different drinking water treatment processes in Berlin (Germany) (Kormos et al., 2011). Creeks in the Hessian Ried (Germany) receive relatively large amounts of treated

wastewater, which infiltrates into the aquifer. Concentrations in groundwater were determined for DIA (median: 30 ng/l) and IPA (160 ng/l). While the detection frequency for DIA (80 %) is similar to that found by this study, the one for IPA (75 %) is much higher (Ternes and Hirsch, 2000).

Compared to a European screening of 90 WWTPs (Loos et al., 2013), concentrations and detection frequencies in treated wastewater were similar for most pharmaceuticals, e.g. CBZ, NAP, BEZ, GEM, and IBU. While DIC and DIA frequencies were much smaller in this study, frequencies for all other detected ICM were much higher than the European average. Concentrations of ICM were also much higher than the European average, while DIA concentrations were much smaller.

3.4.2. Use and usefulness of pharmaceuticals for hydrogeological interpretation

Any detection of pharmaceuticals in groundwater indicates an anthropogenic influence. Pharmaceuticals are solely used in human and veterinary application and diagnostics. As there are no natural background concentrations, their occurrence in groundwater can be attributed to wastewater or manure. In urban areas, leaking sewers and cesspits were assumed to be the main pollution sources to the groundwater. Mass balance approaches are often uncertain due to missing information on amounts of leaking wastewater or applied pharmaceuticals. Therefore, their absolute input is difficult to measure but might be estimated indirectly. Within this study, input functions for the investigated substances (i.e. DIA and IPA) were as well not available. The monitoring of WWTP effluents contributes important information regarding the pharmaceutical substances applied in an urban catchment. Effluent concentrations might thus serve as a lower bound for released concentrations from a leaking sewer or cesspit into the subsoil, although leakage concentrations were expected to be higher.

Pharmaceuticals were rarely detected continuously (except for DIA and IPA) within this study. Mostly, they occurred scattered in the investigated area and at different times. An aim to utilize these single records, is the pharmaceutical detection rate (see chapter 3.3.1.). The number of all detected substances might serve as comparative pollution parameter. The correlation of increasing detection rates along with other increasing parameters in this study indicates the potential of these parameters for a semi-quantitative assessment. High detection rates indicate a high pollution degree. Persistent

substances like CBZ or some ICM can be detected much longer than fecal bacteria, which die off within weeks up to months (Auckenthaler, 2003). Therefore the pharmaceutical impact might be measurable much longer than such parameters. However, the detection rate depends on the number of investigated substances and their respective limits of detection (LOD). Results from different studies were therefore difficult to compare and conclusions might rather be of local relevance.

Comparing different pollution indicators at Wadi Shueib resulted in a spatial distribution of groundwater pollution within the context of the catchment. Springs and wells extracting groundwater upstream (Yesidia) or sideways (Azzraqu) of the city show typically low concentrations of nitrate and *E. coli* (Table 11) combined with no or few detections of pharmaceuticals (Yesidia and Azzraqu, each with only a single detection in 2012). Springs downstream of the city showed higher nitrate concentrations and *E. coli* numbers accompanied by multiple detections of pharmaceutical substances (see SM 3 and Table 11). The spatial distribution of DIA and IPA findings (Figure 19 and Figure 20) shows similar patterns. As Hazzir and Farkha show the highest values for each indicator, the distance to the urban area appears to have an influence as well. Both were located closest to the city, with big shares of their catchments inside the urban area (Figure 17). Shoreia and Baqquriah, located further downstream, subsequently showed lower concentrations of all indicators. These results were in accordance with findings in the city of Rastatt (Germany), where decreasing detection frequencies for ICM and their decreasing concentration could be correlated with the distance of the next upstream sewer. Carbamazepine and the artificial sweetener acesulfame occurred primarily in the city center (Wolf et al., 2012). Increasing concentrations could be found in the groundwater of Halle (Germany) along the underground passage of the city for most investigated xenobiotics (Reinstorf et al., 2008).

Correlation analysis was applied to identify possible causal connections between two pollution indicators. A significant correlation was found for DIA and nitrate in the entirety of all groundwater samples at Wadi Shueib. The reason for this could be similar sources or transport mechanisms. Nitrate might come both from wastewater or agricultural activities. On the other hand, DIA is mostly applied in hospitals end excreted there or at home. No reports on its usage in livestock breeding were documented for Jordan. Thus, for DIA, the source and transport mechanism via sewer leakages and infiltration (see

chapter 3.2.1.) into the unsaturated aquifer seems to be confined to urban areas. In consequence, the strong correlation between DIA and nitrate (Figure 24) suggests that the nitrate pollution at Wadi Shueib originates rather from urban than from rural sources. However, due to the limited amount of samples and sampling sites, other influences (e.g. fertilizer, precipitation) could not be excluded although their impact might be of minor relevance or seem unlikely.

3.4.3. Occurrence, fate, and possible pollution sources of DIA and IPA

Except for some scattered detections for other pharmaceuticals, DIA and IPA were the only substances occurring area-wide and with very high detection frequencies. DIA was present in all groundwater samples inside and downstream of Salt, but was only present in one out of nine treated wastewater samples in Wadi Shueib (see SM 4). Both substances were reported to behave very conservatively regarding biodegradation and sorption under different environmental conditions, e.g. in laboratory studies (Kalsch, 1999; Joss et al., 2006; Kormos et al., 2010; Hebig et al., 2014), during wastewater treatment (Ternes and Hirsch, 2000; Deblonde et al., 2011; Margot et al., 2013), and in groundwater (Kormos et al., 2011). Therefore, their environmental persistence suggests to enable long residence times in the groundwater and subsurface.

The distribution of concentrations of DIA differs from that found by other studies as they are in the same order of magnitude in all water types. This is in contrast to an assumed concentration decrease from source (wastewater) to sink (groundwater) which is mainly driven by dilution. In the urban water cycle of Berlin, this has already been described for primidone (Hass et al., 2012). Dilution factors from treated wastewater to groundwater between 100 and 1000 were determined for IBU and DIC in a karst aquifer (Einsiedl et al. 2010). This is in accordance with the IPA concentrations in Wadi Shueib which were decreasing from the WWTP to surface water to groundwater (SM 4). The detection frequencies in groundwater were increasing over time with constantly high effluent concentrations in all WWTPs.

The main sources for ICM are typically hospitals, with two being located in the study area, the Al Hussein Hospital, in Salt with 150 facilities, and the National Psychiatric Center in Fuheis. The radiology of Al Hussein Hospital is doing approx. 60 X-ray diagnostics per day (Storz, 2004). Around 50 % of a hospital`s patients were usually outpatients, which implies that ICM administered in diagnostics were excreted at home (Weissbrodt et al.,

2009). Furthermore, due to the widespread use of "perforated" septic tanks, a relevant share of ICM does not reach the WWTP but may infiltrate area-wide into the unsaturated zone. Leaky sewers might also contribute an important share. The unsaturated zone only exists in the upper limestone aquifer and varies between 50 and 75 m (Werz, 2006). Water and contamination transport in such aquifers are assumed as mixtures of highly mobile (conduit and fracture flow) and rather immobile (matrix flow) components which can be affected by exchange processes between them (Ford and Williams, 2007). This exchange was observed, e.g. during dye tracer transport in alpine karst systems (Goldscheider, 2005). Whereas fast components may react within hours to days after a precipitation event, the matrix flow is much slower with typical flow velocities for limestone and dolomite between 0.01 – 10 m/y (Freeze and Cherry, 1979). Other processes, like storage and slow release of water in immobile fluid regions (Goldscheider, 2008), are possible as well. A persistent contaminant such as DIA might therefore be omnipresent in the whole matrix after several decades of infiltration.

After the ban on DIA for intravascular application (DCGMA, 2000) and the local advice to change application from high osmolar to low osmolar ICM to avoid adverse effects (Jordan Food and Drug Administration, 2006) the fresh input of DIA today is assumed to be considerably reduced. Reports from shifting application practice in a Jordan hospital (Zemann et al., 2014) accompanied by low DIA concentrations found in wastewater support this as well. However, the residual DIA in the matrix could still leach through the unsaturated zone. There, it may be slowly drained by conduits and subsequently released at springs (Figure 19) with constant, but low concentrations. This is in accordance with the findings of this study, where DIA was found in the majority of well and spring samples, while it was hardly documentable in the WWTP effluents. The whole process is illustrated in Figure 25 with the situation for DIA being presented on the left. Minor amounts at the WWTP may result from an outstanding but reduced oral administration of DIA as ICM. A similar transport mechanism was already assumed for IBU and DIC. Both appeared in groundwater at the Franconian Alb 20 years after WWTP effluent infiltration was terminated (Einsiedl et al., 2010).

Simultaneously to the reduced DIA administration, IPA, a possible substitute for DIA, is detected with increasing frequency within the last 5 years (Figure 20) along with constant high loads in the local WWTPs which might indicate the increased use of IPA. The impact on the local aquifer system is assumed to take place at an earlier stage of spreading

(Figure 25, left) with the main transport occurring fast via the conduits. Direct responses of flow volumes or water quality to precipitation events, usually within days or few weeks, are documented for the springs in the area (Werz, 2006).

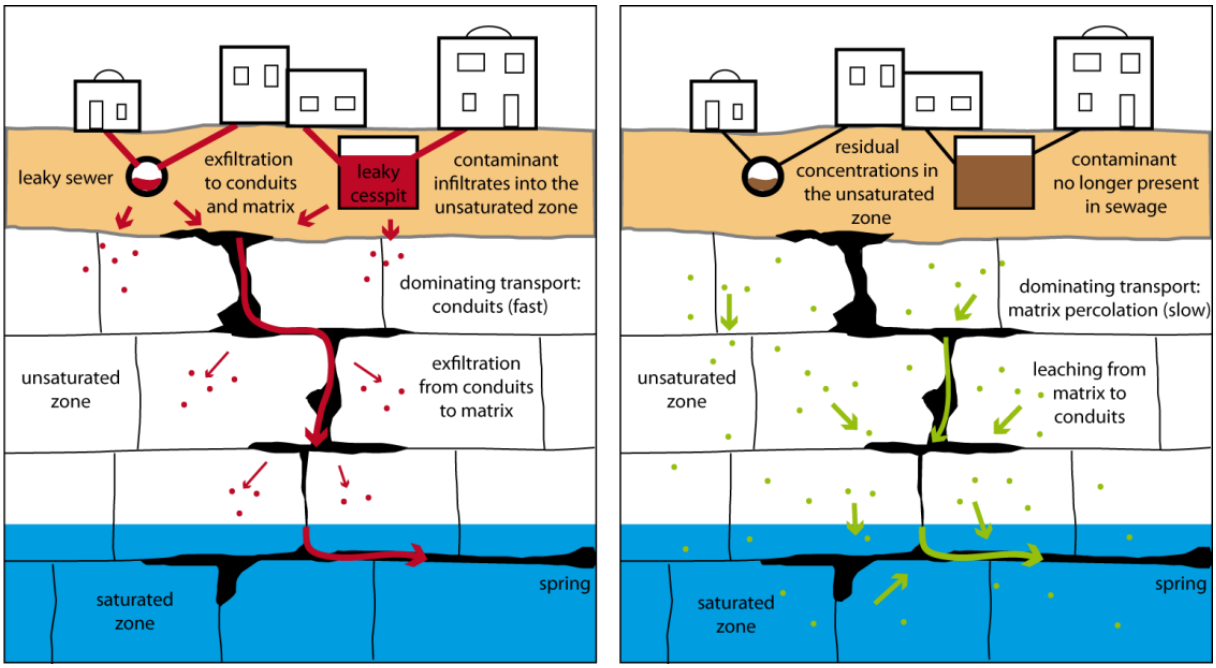

Figure 25: Conceptual model of two different pharmaceutical pathways in a karstified urban limestone aquifer with an extensive unsaturated zone. Left: Recent spreading of IPA (red). Right: Leaching of DIA (green) from the unsaturated zone.

This conceptual model is also in accordance with the significant correlation of nitrate and DIA (see chapter 3.3.5). Nitrate is also assumed to be ubiquitously distributed in the unsaturated zone due to prolonged input into the environment, e.g. by infiltrating wastewater. Nitrate pollution sources were thus likely to be related to the input of DIA. Due to the lacking use of DIA in agriculture, relevant shares of nitrate were assumed to come from urban sources. The rather new IPA displays poor correlation with nitrate (see SM 5), which might indicate a more recent entry. Similar approaches were used for the artificial sweetener acesulfame together with different pharmaceuticals to identify single WWTPs as groundwater contamination sources (Van Stempvoort et al., 2013). A similar behavior for DIA and IPA was already reported for water resources in the Lower Jordan Valley (Zemann et al., 2014).

3.5. Conclusions

Within this study, pharmaceuticals and X-ray contrast media were screened in the water resources of Wadi Shueib and Wadi Kafrein to assess the influence of urban pollution on the local karst aquifer. Overall, the following conclusions can be drawn:

- The occurrence of pharmaceuticals, especially X-ray contrast media, demonstrates the urban impact on the karst aquifer at Wadi Shueib and Kafrein where they were first of all infiltrated via leaking sewers and cesspits. Pharmaceuticals could be detected in 91 % of the 66 samples. Concentrations of the investigated substances were higher than in other karst studies whereas the number of detected substances was lower than in comparable urban studies.
- Diatrizoic acid showed continuous occurrence in groundwater during the 5-year period while only few detections were observed in the wastewater, which is supposed to be the only source in the catchment. Concentrations in groundwater and wastewater thus appeared in the same order of magnitude. Diatrizoic acid was a standard substance in X-ray diagnostics in the years before 2008, but its application has been reduced. Therefore, the observed distribution pattern can be explained by residual concentrations in the unsaturated zone and matrix of the limestone aquifer that are now leaching towards the groundwater and main conduit network.
- In contrast, the possible substitute iopamidol was increasingly detected over the 5-year investigation period. Concentrations in wastewater were two orders of magnitude higher than in groundwater.
- The statistically significant correlation of diatrizoic acid and nitrate concentrations points to sewage as the most likely source of nitrate in the investigated area as X-ray contrast media were not applied to livestock in Jordan agriculture. Therefore, diatrizoic acid can be considered as potential tracer for the urban impact on an aquifer.
- Combined assessment of the pharmaceutical detection rate, *E. coli* and nitrate in groundwater quantifies the anthropogenic influence on the karst aquifer in Wadi Shueib. Spatial occurrence of these pollution indicators can be linked to the hydrogeological setting (e.g. flow direction, aquifer system) and the distance to the city. Highest rates occurred downstream of the urban areas, while the upstream springs and wells were mostly unpolluted or showed less impact. The significant correlation of the three pollution parameter identifies the pharmaceutical detection rate as a suitable tool on a semi-quantitative level and therefore makes use of single scattered detections of different substances.

The results show that knowledge about changing application patterns of pharmaceuticals is essential to assess the urban impact on aquifers, i.e. when applying mass balance techniques. Future water quality assessments are highly recommended to verify the hypothesis of increasing iopamidol concentrations and the expected disappearance of diatrizoic acid within the coming years. Additionally, the effects of upcoming sewer rehabilitation in the karst springs should be monitored for nitrate and pharmaceuticals.

3.6. Acknowledgements

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3.7. References

Al-Kharabsheh, N. M., A. Al-Kharabsheh and O. M. Ghnaim (2013). "Effect of Septic Tanks and Agricultural Wastes on Springs` Water Quality Detoriation in Wadi Shuèib Catchment Area-Jordan." Jordan Journal of Agricultural Sciences **9**(No.1): 15.

Alahmad, W. R. and M. A. Alawi (2010). "Hplc/Uv/Fluorescence Detection of Several Pharmaceuticals in Sewage Treatment Plant Wastewaters of Jordan." Fresenius Environmental Bulletin **19**(5): 805-810.

Auckenthaler, A. (2003). Pathogene Mikroorganismen im Grund- und Trinkwasser : Transport, Nachweismethoden, Wassermanagement. Basel, Birkhäuser.

Bueno, M. J. M., M. J. Gomez, S. Herrera, M. D. Hernando, A. Agüera and A. R. Fernández-Alba (2012). "Occurrence and persistence of organic emerging contaminants and priority pollutants in five sewage treatment plants of Spain: Two years pilot survey monitoring." Environmental Pollution **164**(0): 267-273.

DCGMA, Drug Comission of the German Medical Association (2000). "Mitteilungen: Arzneimittelkommission der deutschen Ärzteschaft BfArM ordnet Widerruf von ionischen Röntgenkontrastmitteln (Amidotrizoesäure, Iotalaminsäure) zur intravasalen Anwendung an." Dtsch Arztebl International **97**(33): 2187.

Deblonde, T., C. Cossu-Leguille and P. Hartemann (2011). "Emerging pollutants in wastewater: A review of the literature." International Journal of Hygiene and Environmental Health **214**(6): 442-448.

Eaton, A. D., M. A. H. Franson, A. P. H. Association, A. W. W. Association and W. E. Federation (2005). Standard Methods for the Examination of Water & Wastewater, American Public Health Association.

Einsiedl, F., M. Radke and P. Maloszewski (2010). "Occurrence and transport of pharmaceuticals in a karst groundwater system affected by domestic wastewater treatment plants." Journal of Contaminant Hydrology **117**(1–4): 26-36.

Ellis, J. B. (2006). "Pharmaceutical and personal care products (PPCPs) in urban receiving waters." Environmental Pollution **144**(1): 184-189.

Escher, B. I., R. Baumgartner, M. Koller, K. Treyer, J. Lienert and C. S. McArdell (2011). "Environmental toxicology and risk assessment of pharmaceuticals from hospital wastewater." Water Research **45**(1): 75-92.

Fenz, R., A. P. Blaschke, M. Clara, H. Kroiss, D. Mascher and M. Zessner (2005). "Monitoring of carbamazepine concentrations in wastewater and groundwater to quantify sewer leakage." Water Science and Technology **52**(5): 205-213.

Ford, D. and P. W. Williams (2007). Karst hydrogeology and geomorphology. Chichester, Wiley.

Freeze, R. A. and J. A. Cherry (1979). Groundwater, Prentice-Hall.

Goldscheider, N. (2005). "Fold structure and underground drainage pattern in the alpine karst system Hochifen-Gottesacker." Eclogae Geologicae Helvetiae **98**(1): 1-17.

Goldscheider, N. (2008). "A new quantitative interpretation of the long-tail and plateau-like breakthrough curves from tracer tests in the artesian karst aquifer of Stuttgart, Germany." Hydrogeology Journal **16**(7): 1311-1317.

GTZ and WMIA (2007). Water awareness on Household and Community Level. Amman.

Hass, U., U. Duennbier and G. Massmann (2012). "Occurrence and distribution of psychoactive compounds and their metabolites in the urban water cycle of Berlin (Germany)." Water Research **46**(18): 6013-6022.

Hebig, K. H., K. Nödler, T. Licha and T. J. Scheytt (2014). "Impact of materials used in lab and field experiments on the recovery of organic micropollutants." Science of the Total Environment **473-474**: 125-131.

Hillebrand, O., K. Nödler, T. Geyer and T. Licha (2014). "Investigating the dynamics of two herbicides at a karst spring in Germany: Consequences for sustainable raw water management." Science of the Total Environment **482–483**(0): 193-200.

Huebsch, M., B. Horan, P. Blum, K. G. Richards, J. Grant and O. Fenton (2013). "Impact of agronomic practices of an intensive dairy farm on nitrogen concentrations in a karst aquifer in Ireland." Agriculture, Ecosystems & Environment **179**(0): 187-199.

Jordan Food and Drug Administration (2006). Jordan National Drug Formulary. Q. L. Nuseirat A., Ghanem M. http://www.jfda.jo/jndfbook/index.htm, Drug Directorate Rational Drug Use Department.

Joss, A., S. Zabczynski, A. Gobel, B. Hoffmann, D. Loffler, C. S. McArdell, T. A. Ternes, A. Thomsen and H. Siegrist (2006). "Biological degradation of pharmaceuticals in municipal wastewater treatment: Proposing a classification scheme." Water Research **40**(8): 1686-1696.

Jurado, A., R. López-Serna, E. Vázquez-Suné, J. Carrera, E. Pujades, M. Petrovic and D. Barceló (2014). "Occurrence of carbamazepine and five metabolites in an urban aquifer." Chemosphere(0).

Kalsch, W. (1999). "Biodegradation of the iodinated X-ray contrast media diatrizoate and iopromide." Science of the Total Environment **225**(1-2): 143-153.

Katz, B. G., D. W. Griffin and J. H. Davis (2009). "Groundwater quality impacts from the land application of treated municipal wastewater in a large karstic spring basin: Chemical and microbiological indicators." Science of the Total Environment **407**(8): 2872-2886.

Kormos, J. L., M. Schulz, H.-P. E. Kohler and T. A. Ternes (2010). "Biotransformation of Selected Iodinated X-ray Contrast Media and Characterization of Microbial Transformation Pathways." Environmental Science & Technology **44**(13): 4998-5007.

Kormos, J. L., M. Schulz and T. A. Ternes (2011). "Occurrence of Iodinated X-ray Contrast Media and Their Biotransformation Products in the Urban Water Cycle." Environmental Science & Technology **45**(20): 8723-8732.

Kostich, M. S., A. L. Batt and J. M. Lazorchak (2014). "Concentrations of prioritized pharmaceuticals in effluents from 50 large wastewater treatment plants in the US and implications for risk estimation." Environmental Pollution **184**(0): 354-359.

Kovalova, L., H. Siegrist, H. Singer, A. Wittmer and C. S. McArdell (2012). "Hospital Wastewater Treatment by Membrane Bioreactor: Performance and Efficiency for Organic Micropollutant Elimination." Environmental Science & Technology **46**(3): 1536-1545.

Kuroda, K., M. Murakami, K. Oguma, Y. Muramatsu, H. Takada and S. Takizawa (2012). "Assessment of groundwater pollution in Tokyo using PPCPs as sewage markers." Environmental Science and Technology **46**(3): 1455-1464.

Loos, R., R. Carvalho, D. C. António, S. Comero, G. Locoro, S. Tavazzi, B. Paracchini, M. Ghiani, T. Lettieri, L. Blaha, B. Jarosova, S. Voorspoels, K. Servaes, P. Haglund, J. Fick, R. H. Lindberg, D. Schwesig and B. M. Gawlik (2013). "EU-wide monitoring survey on emerging polar organic contaminants in wastewater treatment plant effluents." Water Research **47**(17): 6475-6487.

Loos, R., B. M. Gawlik, G. Locoro, E. Rimaviciute, S. Contini and G. Bidoglio (2009). "EU-wide survey of polar organic persistent pollutants in European river waters." Environmental Pollution **157**(2): 561-568.

Loos, R., G. Locoro, S. Comero, S. Contini, D. Schwesig, F. Werres, P. Balsaa, O. Gans, S. Weiss, L. Blaha, M. Bolchi and B. M. Gawlik (2010). "Pan-European survey on the

occurrence of selected polar organic persistent pollutants in ground water." Water Research **44**(14): 4115-4126.

López-Serna, R., A. Jurado, E. Vázquez-Suñé, J. Carrera, M. Petrović and D. Barceló (2013). "Occurrence of 95 pharmaceuticals and transformation products in urban groundwaters underlying the metropolis of Barcelona, Spain." Environmental Pollution **174**: 305-315.

Maeng, S. K., S. K. Sharma, K. Lekkerkerker-Teunissen and G. L. Amy (2011). "Occurrence and fate of bulk organic matter and pharmaceutically active compounds in managed aquifer recharge: A review." Water Research **45**(10): 3015-3033.

Mahler, B. and N. Massei (2007). "Anthropogenic contaminants as tracers in an urbanizing karst aquifer." Journal of Contaminant Hydrology **91**(1–2): 81-106.

Margane, A., A. Subah, I. Hamdan, Z. Hajali and T. Almomani (2010). Delineation of groundwater protection zones for the springs in Wadi Shuayb. Groundwater Resources Management - Technical Report. **No. 14:** 98.

Margot, J., C. Kienle, A. Magnet, M. Weil, L. Rossi, L. F. de Alencastro, C. Abegglen, D. Thonney, N. Chèvre, M. Schärer and D. A. Barry (2013). "Treatment of micropollutants in municipal wastewater: Ozone or powdered activated carbon?" Science of the Total Environment **461–462**(0): 480-498.

Metcalfe, C. D., P. A. Beddows, G. G. Bouchot, T. L. Metcalfe, H. Li and H. Van Lavieren (2011). "Contaminants in the coastal karst aquifer system along the Caribbean coast of the Yucatan Peninsula, Mexico." Environmental Pollution **159**(4): 991-997.

Morasch, B. (2013). "Occurrence and dynamics of micropollutants in a karst aquifer." Environmental Pollution **173**(0): 133-137.

Musolff, A., S. Leschik, M. Moder, G. Strauch, F. Reinstorf and M. Schirmer (2009). "Temporal and spatial patterns of micropollutants in urban receiving waters." Environmental Pollution **157**(11): 3069-3077.

Musolff, A., S. Leschik, F. Reinstorf, G. Strauch and M. Schirmer (2010). "Micropollutant Loads in the Urban Water Cycle." Environmental Science & Technology **44**(13): 4877-4883.

Orias, F. and Y. Perrodin (2013). "Characterisation of the ecotoxicity of hospital effluents: A review." Science of the Total Environment **454–455**(0): 250-276.

Osenbrück, K., H. R. Gläser, K. Knöller, S. M. Weise, M. Möder, R. Wennrich, M. Schirmer, F. Reinstorf, W. Busch and G. Strauch (2007). "Sources and transport of selected organic micropollutants in urban groundwater underlying the city of Halle (Saale), Germany." Water Research **41**(15): 3259-3270.

Pronk, M., N. Goldscheider and J. Zopfi (2006). "Dynamics and interaction of organic carbon, turbidity and bacteria in a karst aquifer system." Hydrogeology Journal **14**(4): 473-484.

Reh, R., T. Licha, T. Geyer, K. Nödler and M. Sauter (2013). "Occurrence and spatial distribution of organic micro-pollutants in a complex hydrogeological karst system during low flow and high flow periods, results of a two-year study." Science of the Total Environment **443**: 438-445.

Reinstorf, F., G. Strauch, K. Schirmer, H. R. Gläser, M. Möder, R. Wennrich, K. Osenbrück and M. Schirmer (2008). "Mass fluxes and spatial trends of xenobiotics in the waters of the city of Halle, Germany." Environmental Pollution **152**(2): 452-460.

Sacher, F., M. Ehmann, S. Gabriel, C. Graf and H. J. Brauch (2008). "Pharmaceutical residues in the river Rhine - results of a one-decade monitoring programme." Journal of Environmental Monitoring **10**(5): 664-670.

Sacher, F., F. T. Lang, H. J. Brauch and I. Blankenhorn (2001). "Pharmaceuticals in groundwaters - Analytical methods and results of a monitoring program in Baden-Wurttemberg, Germany." Journal of Chromatography A **938**(1-2): 199-210.

Santos, L. H. M. L. M., M. Gros, S. Rodriguez-Mozaz, C. Delerue-Matos, A. Pena, D. Barceló and M. C. B. S. M. Montenegro (2013). "Contribution of hospital effluents to the load of pharmaceuticals in urban wastewaters: Identification of ecologically relevant pharmaceuticals." Science of the Total Environment **461–462**(0): 302-316.

Schaider, L. A., R. A. Rudel, J. M. Ackerman, S. C. Dunagan and J. G. Brody (2014). "Pharmaceuticals, perfluorosurfactants, and other organic wastewater compounds in public drinking water wells in a shallow sand and gravel aquifer." Science of the Total Environment **468–469**(0): 384-393.

Schmidt, S., T. Geyer, A. Marei, J. Guttman and M. Sauter (2013). "Quantification of long-term wastewater impacts on karst groundwater resources in a semi-arid environment by chloride mass balance methods." Journal of Hydrology **502**(0): 177-190.

Schwab, B. W., E. P. Hayes, J. M. Fiori, F. J. Mastrocco, N. M. Roden, D. Cragin, R. D. Meyerhoff, V. J. D'Aco and P. D. Anderson (2005). "Human pharmaceuticals in US surface waters: A human health risk assessment." Regulatory Toxicology and Pharmacology **42**(3): 296-312.

Storz, R. (2004). GIS-based Groundwater Hazard Mapping of the Wadi Shu'eyb Catchment Area, Jordan. Diploma, Karlsruhe University.

Ternes, T. A. and R. Hirsch (2000). "Occurrence and behavior of X-ray contrast media in sewage facilities and the aquatic environment." Environmental Science & Technology **34**(13): 2741-2748.

Tiehm, A., N. Schmidt, M. Stieber, F. Sacher, L. Wolf and H. Hoetzl (2011). "Biodegradation of Pharmaceutical Compounds and their Occurrence in the Jordan Valley." Water Resources Management **25**(4): 1195-1203.

van Afferden, M., J. A. Cardona, K. Z. Rahman, R. Daoud, T. Headley, Z. Kilani, A. Subah and R. A. Mueller (2010). "A step towards decentralized wastewater management in the Lower Jordan Rift Valley." Water Science and Technology **61**(12): 3117-3128.

Van Stempvoort, D. R., J. W. Roy, J. Grabuski, S. J. Brown, G. Bickerton and E. Sverko (2013). "An artificial sweetener and pharmaceutical compounds as co-tracers of urban wastewater in groundwater." Science of the Total Environment **461-462**: 3480-3359.

Vulliet, E. and C. Cren-Olivé (2011). "Screening of pharmaceuticals and hormones at the regional scale, in surface and groundwaters intended to human consumption." Environmental Pollution **159**(10): 2929-2934.

Weissbrodt, D., L. Kovalova, C. Ort, V. Pazhepurackel, R. Moser, J. Hollender, H. Siegrist and C. S. McArdell (2009). "Mass flows of x-ray contrast media and cytostatics in hospital wastewater." Environmental Science and Technology **43**(13): 4810-4817.

Werz, H. (2006). The use of remote sensing imagery for groundwater risk intensity mapping in the Wadi Shueib, Jordan. Doktor, University of Karlsruhe.

Wolf, L., A. Pöschko, M. Zemann, H. Werz, A. Sawarieh, N. Seder, M. Stieber, F. Sacher and A. Tiehm (2009). "Understanding occurrence of xenobiotics in closed river basins, examples from the Jordan Valley." International Conference on Xenobiotics in the Urban Water Cycle (XENOWAC).

Wolf, L., C. Zwiener and M. Zemann (2012). "Tracking artificial sweeteners and pharmaceuticals introduced into urban groundwater by leaking sewer networks." Science of the Total Environment **430**: 8-19.

Zemann, M., L. Wolf, A. Pöschko, N. Schmidt, A. Sawarieh, N. Seder, A. Tiehm, H. Hötzl and N. Goldscheider (2014). "Sources and processes affecting the spatio-temporal distribution of pharmaceuticals and X-ray contrast media in the water resources of the Lower Jordan Valley, Jordan." Science of the Total Environment **488–489**(0): 100-114.

3.8. Supplementary materials (SM)

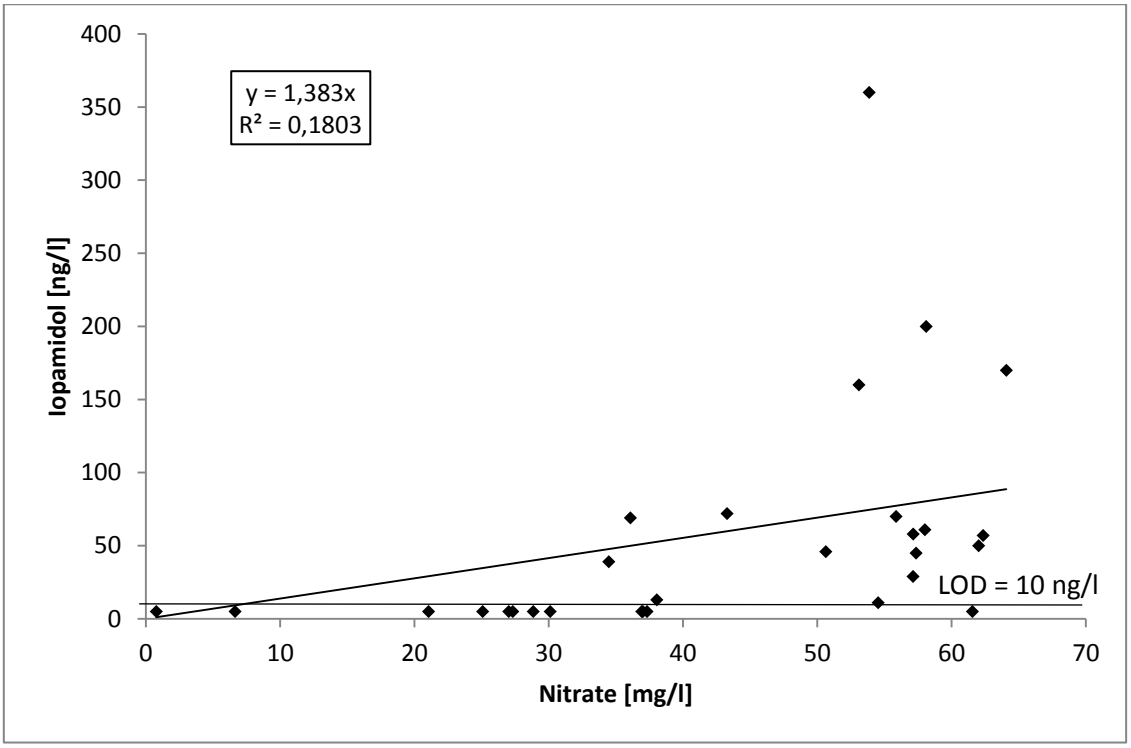
SM 3: Total detections in the different water types of Wadi Shueib and Kafrein. TWW = treated wastewater, GW = groundwater, SW = surface water. Numbers in SW and TWW were taken in April 2008. * = 2008

	number of samples	2008 Nov/Apr	2011 May	2011 Nov	2012 Feb
TWW	3	21	18	19	24
SW	2	12	14	11	17
GW	8 (7*)	6	10	9	18

SM 4: Concentrations for the different sampling campaigns for diatrizoic acid and iopamidol at each sampling site. LOD is 10 in GW and SW and 50 for TWW. Calculations of means and median values were done by calculating values > LOD as 0.5 * LOD. n.s. = no sample. * = mean values from a 24h sampling campaign. # = reservoir and WWTP samples were taken in April.

		Diatrizoic acid					Iopamidol				
Sampling date		Nov 07	Nov 08#	Mar 11	Sep 11	Feb 12	Nov 07	Nov 08	Mar 11	Sep 11	Feb 12
	Water type	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]	[ng/l]
Azzraqu spring	GW		< LOD	< LOD	< LOD	< LOD		< LOD	< LOD	< LOD	< LOD
Baqquriah spring	GW		61	38	26	38		< LOD	< LOD	69	72
Farkha spring	GW		170	73	85	74		< LOD	< LOD	170	29
Hazzir spring	GW		150	51	60	55		69	11	360	46
Shoreia spring	GW		31	28	13	16		< LOD	< LOD	< LOD	39
Um Attija well	GW		27	42	21	22		< LOD	< LOD	< LOD	13
Yesidia well	GW		< LOD	< LOD	< LOD	< LOD		< LOD	< LOD	< LOD	< LOD
Tujabyl spring	GW		n.s.	220	66	180		n.s.	180	< LOD	1900
Kafrein reservoir	SW	103*	26	< LOD	100	120	803*	< LOD	8000	78,000	17,000
Shueib reservoir	SW	< LOD	< LOD	270	27	220	1000	72	1600	700	1000
Es Sir WWTP effluent	TWW	< LOD	110	< LOD	< LOD	300	< LOD	2300	160,000	680,000	600,000
As Salt WWTP effluent	TWW	< LOD	< LOD	< LOD	< LOD	< LOD	4500	1600	810	6900	13,000
Fuheis WWTP effluent	TWW		< LOD	120	< LOD	< LOD		2200	15,000	15,000	6600
Detections GW	[%]		71	75	75	75		14	25	38	75
Mean GW	[ng/l]		64	58	35	49		14	28	78	264
Mean SW	[ng/l]		16	138	64	170		39	4800	39,350	9000
Mean TWW	[ng/l]		53	57	25	117		2033	58,603	233,967	206,533
Median GW	[ng/l]		61	46,5	43	46,5		69	95,5	170	42,5
Maximum GW	[ng/l]		170	220	85	180		69	180	360	1900
Maximum SW	[ng/l]		26	270	100	220		72	8000	78,000	17,000
Maximum TWW	[ng/l]		110	120		300		2300	160,000	680,000	600,000

SM 5: Correlation of nitrate and iopamidol for all groundwater samples in Wadi Shueib and Kafrein from 2008 to 2012. Values < LOD were calculated as 0.5 * LOD.



4. Accumulation of Pharmaceuticals in Groundwater under Arid Climate Conditions – Results from Unsaturated Column Experiments

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Abstract: Intense reuse of treated wastewater in agriculture is practiced all over the world, especially in arid and water-scarce regions. In doing so, pharmaceutical residues in the water are irrigated to the soil and subsequently can percolate into the local aquifers. Since evaporation rates in these areas are typically high, persistent substances might enrich in the groundwater recharge of closed catchments like the Jordan Valley. Against this background, unsaturated column tests were conducted to investigate the potential for evaporative accumulation of the two pharmaceuticals bezafibrate and carbamazepine under simulated arid climate conditions. Parallel tests were conducted with inhibited microbiological activity where both substances showed an increase in the effluent concentrations proportional to the evaporation loss of the inflow solution. The mean accumulation factors of the pharmaceuticals correspond to the evaporated water loss. The experiments indicate the accumulation potential for pharmaceuticals with high persistence against biodegradation. For the first time, the overall potential for evaporative enrichment could be demonstrated for pharmaceuticals. Under the given experimental conditions, the two investigated pharmaceuticals did not enrich faster than chloride, which might result in soil salting prior to reaching harmful pharmaceutical concentrations in soil water. The findings are relevant to future assessments of environmental impacts of persistent trace substances, which need to take into account that concentrations in the aquatic cycle might increase further due to evaporative enrichment.

4.1. Introduction

The intensive use of treated wastewater in agriculture bears the potential of introducing emerging pollutants into the groundwater. Although some substances are degraded while percolating through the unsaturated zone, many persistent pharmaceuticals were found in aquifers underlying agricultural areas irrigated with treated wastewater (Kinney et al., 2006; Siemens et al., 2008; Avisar et al., 2009; Grossberger et al., 2014). Laboratory results indicate the potential of pharmaceuticals of reaching the groundwater after irrigation with treated wastewater as well (Chefetz et al., 2008; Siemens et al., 2010). Recent monitoring results in the Lower Jordan Valley (LJV) showed higher pharmaceutical concentrations in groundwater than in the infiltrating surface water (Wolf et al., 2009). As irrigation is mostly applied in dry climates, such substances could be assumed to accumulate in the shallow groundwater due to high evaporation rates in semi-arid areas. For the LJV, the short-cycled agricultural use of mixtures of treated wastewater with the locally pumped groundwater, combined with the closed character of the catchment with the Dead Sea as final sink, supports an enrichment as well. Nevertheless, the issue of evaporative accumulation of pharmaceutical substances has not been addressed to date.

Pharmaceuticals in the aquatic environment are ubiquitous and their negative effects at environmental concentrations were already reported for e.g. fish (Kidd et al., 2007; Pomati et al., 2008; Brodin et al., 2013). However, after treatment, levels of residues in drinking water are very low and are considered unproblematic (Webb et al., 2003; Schwab et al., 2005; Houeto et al., 2012). Due to the assumed accumulation, environmental concentrations might increase to up to harmful levels and therefore entail increasing effort for drinking water treatment.

Against this background, the hypothesis of accumulating pharmaceutical concentrations over time was investigated following two approaches: Long-term field investigations and small-scale column studies at lab scale. While the field studies were insufficient to provide statistically valid evidence for the Lower Jordan Valley (Zemann et al., 2014), the studies showed that the tempo-spatial distributions of x-ray contrast media concentrations were most likely due to shifting medication and application patterns in this area (Zemann et al., 2015). However, the idea of accumulating persistent substances by evaporation processes, e.g. similar to salt enrichment during salting of soil, seems still realistic. The increase in pharmaceutical concentrations in wastewater-irrigated soils was already

investigated in China (Chen et al., 2011) and Tunisia (Fenet et al., 2012). Soils irrigated with treated wastewater in Colorado (USA) showed rising carbamazepine (CBZ) concentrations after several months of irrigation, while the soil organic matter (SOM) was assumed to be the controlling factor for pharmaceutical retention (Kinney et al., 2006). However, none of these studies considered evaporative processes.

The experiments of this study were conducted under most realistic natural conditions, i.e. by choosing temperatures, humidity and irrigation rates as they were measured in the LJV (Ministry of Water and Irrigation, 2004). In addition, this includes the use of real treated wastewater, the use of natural sand, and the consideration of the relevant processes, i.e. degradation and sorption under unsaturated flow conditions. Substances were selected according to the range of pharmaceuticals detected in previous studies in the LJV (Zemann et al., 2014). Out of this spectrum, pharmaceuticals with different persistence against biodegradation were selected. Those were the rather easily biodegradable lipid lowering agent bezafibrate (BEZ) and the antiepileptic CBZ. Their degradability (BEZ) and persistence (CBZ) was reported by different authors, e.g. (Maeng et al., 2011; Grossberger et al., 2014; Rühmland et al., 2015).

4.2. Methodology

4.2.1. Experimental setup and conceptual idea

The experiments were conducted in six stainless steel columns filled with prewashed quartzous sand and packed by a pounder. Relevant experimental characteristics are listed in Table 12. Each column had a percolation length of 50 cm and a diameter of 10 cm, with a total volume of 3695 cm^3. A ceramic filter plate at the outlet prevented leaching of sand and a potential clogging of the outflow pipe and led to uniform drainage at the column bottom. The grain size of the used sand was determined as medium-fine sand with shares of fine sand (57 %), medium sand (41 %), coarse sand (~1 %), and silt (~1 %), respectively. The hydraulic conductivity of $3*10^{-4}$ m/s was determined according to Hazen from the grain size distribution. Each column was spiked with four equally distributed soil moisture sensors (ECH2O EC-5) from UMS Co. as shown in the supplementary materials (SM 6, left).

Table 12: Physio-chemical soil parameters and experimental conditions for all columns.

Soil properties			Column conditions		
p_s	[g/cm]	2.65	L	[cm]	50
n	[%]	36	A	[cm^2]	73.9
k_f	[m/s]	$3*10^{-4}$	Q	[ml/day]	108
C_{org}	[wt.%]	0.13 - 0.23	T	[°C]	30
			Φ	[%]	45
			C_0	[µg/l]	20

p_s = Grain density

n = Porosity

k_f = Hydraulic conductivity

C_{org} = Carbon content

L = Length

A = Surface area

Q = Flow rate

T = Temperature

Φ = Humidity

C_0 = Spiked pharmaceutical concentration

All columns were operated under unsaturated conditions. The feeding solution was trickled onto the top of each column by four cannulas at a distance of 1 cm (see SM 6, left). The feeding solution was stored in a fridge and supplied to the columns by a peristaltic pump via stainless-steel pipes 0.5 mm in diameter (SM 6, bottom right). The specified pumping rate according to the hose diameter (0.51 mm) was 108 ml/day. This rate features a mean daily irrigation amount used by farmers in the LJV (Ministry of Water and Irrigation, 2004). The effluent of each column was pumped back to the fridge where it was stored as a collective sample in separated glass bottles. The columns were placed on a table in one row (SM 6, top right). Of the six columns, respectively three were operated under the same conditions to obtain replicate results. Column numbers I1, I2 and I3 (I = inhibited) were operated with a toxic feeding solution to inhibit microbiological activities during percolation. The columns operated under normal, uninhibited conditions were labeled U1, U2 and U3 (U = uninhibited). Columns U3 and I3 did not have internal monitoring facilities. The three inhibited and the three uninhibited columns were each

fed from the same storage bottle. Subsequently, the inflowing amount of each column was calculated from mean weight differences of the feeding solution bottles divided by three and related to the share of the effluent at each column.

To obtain semi-arid evaporation conditions, the whole experiment was built inside a climatic chamber (approx. 2.5 m*4 m) adjustable for temperature and humidity. The temperature was kept at 30°C and the humidity at 45 % during the whole experiment. The numbers were chosen according to mean values for the LJV (Ministry of Water and Irrigation, 2004). Beside the sampling and control times, the room was kept dark to avoid any photochemical degradation processes. During the whole experiment, the columns were placed on scales to monitor the water balance.

The feeding solutions consisted of treated wastewater from a local wastewater treatment plant (WWTP) in Karlsruhe Neureut and were spiked with the investigated pharmaceuticals and LiBr as conservative tracer. Pharmaceuticals were added to a concentration of 20 µg/l from an aqueous stock solution. Residual concentrations in the treated wastewater might even increase this concentration. The feed solution of three columns was acidified with NaN_3 (0.1 %) to inhibit microbiological degradation and growth. This feeding solution was mixed in 2.5 l volumes and refreshed once a week. Bezafibrate (CAS: 41859-67-0) and CBZ (298-46-4) were purchased from Sigma-Aldrich. The LiBr concentration was 20 mg/l. LiBr and NaN3 were used in pure quality supplied by Merck.

4.2.2. Experimental execution and monitoring

The experiment started on March 27th 2012 and ended on August 8th 2012, lasting a total duration of 134 days. For the first 53 days, the initially dry columns were operated with final effluent from the Karlsruhe Neureut WWTP which was refreshed regularly. During this adaption phase, the growth of a microbial community inside each column should be enabled. After this initial phase, the columns were fed with autoclaved treated wastewater taken during one single sampling. The autoclave treatment was done to avoid biodegradation during storage. Subsequently, nutrient and ion inflow concentrations were obtained constant during the whole experiment. The addition of pharmaceuticals started after 53 days. The initial phase lasted for 4.6 pore volumes (PVs) while the main experiment with the pharmaceutical accumulation lasted for 81 days (7.3 PVs).

Water samples were taken from the column outlet on a regular basis at short intervals (0.3, 1 and 3 days) at the beginning of the spiking and larger intervals (one week) after three weeks of spiking with pharmaceuticals. The surplus feeding solution was sampled almost every week. All effluent samples were weighed and measured for EC, pH, redox and dissolved oxygen. Afterwards, the inflow and all effluent samples of the uninhibited experiments were acidified with HNO_3. The inflow and effluent samples of the inhibited experiments were not acidified, as the NaN_3 should inhibit any microbiological degradation effects. All samples were stored cool and dark in glass bottles until analysis. Mass differences in the feeding solution bottles were weighed on a daily basis. Water samples were analyzed for pharmaceuticals, major ions including Li^+ and Br^-, and DOC/TOC.

The soil moisture was measured once a day. The scales were logged at 15 min intervals. At the end of the experiment, the columns were dismantled and segmented. The soil was analyzed for water content by drying at 105 °C and for soil organic matter (SOM) by ignition loss. Additionally, parts of the soil were dry frosted and eluted with methanol. The aliquot was analyzed for pharmaceuticals. However, residual pharmaceutical concentrations in the sediment remained negligible.

4.2.3. Analysis

Pharmaceutical analyses of CBZ and BEZ were carried out using an LC-ESI-MS/MS system from PE SCIEX (API 3000) with an HP 1100 pump. Chromatographic separation was performed with a Gemini C_{18} column (150 x 3 mm, 5 µm) from Phenomenex. 100 ml aliquots of each sample were enriched (enrichment factor 1:100) via solid-phase extraction cartridges HRX from Chromabond (6 ml, 200 mg) from Macherey-Nagel. CBZ d10 was spiked prior to enrichment as a surrogate standard for loss correction during SPE. Calibration was done externally in pure water. Limits of quantification (LOQ) were searched experimentally with a criterion of a standard deviation of 10 % and found between 100 to 150 ng/L before enrichment.

Samples for ion analysis were filtered with cellulose mixed ester (CME) filters (Roth: 0.45 µm). Cation samples where acidified with concentrated nitric acid (60 %) (1 ml per 100 ml sample) and analyzed by IC (DIONEX 1100) following DIN 38406. The measurement error is in the range of +/- 3 %.

4.3. Results and discussion

4.3.1. Water, chloride mass balances, and soil humidity

To illustrate the general evaporation concept of the experiments, water balances were calculated in the first step. As shown in Figure 26a and Figure 26b, the outflow volume per day is always lower than the inflowing volume. This effect can be attributed solely to evaporation. The mean water mass difference for all uninhibited columns was -18.3 % and -17.6 % for all inhibited columns (see Table 13). Water mass balances were calculated from the total amount of feeding solution and the total effluent of each column plus the remaining pore water in the columns after the experiment. This was necessary, as all columns were dry at the beginning of the experiment. This was done in the same way for chloride and bromide and the pharmaceuticals. The results are shown in SM 7, with feeding rates of approx. 120 ml/day and corresponding outflow rates of 100 ml/day. Especially the inflow rates were rather stable, therefore being a good indicator for constant flow conditions through the columns during the whole experiment.

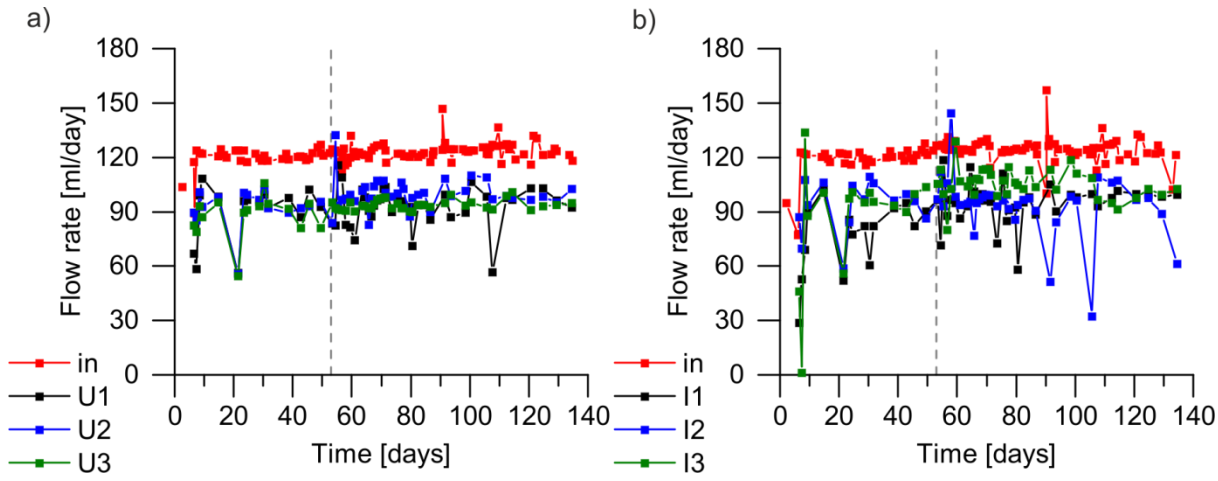

Figure 26, a-b: Inflow and effluent rates of the uninhibited (a) and inhibited (b) columns. Inflow was calculated as mean value for all columns. The grey line marks the beginning of pharmaceutical addition.

The soil moisture profiles remained almost stable for all columns after several days of operation (Figure 27). The moisture content decreased from the top (10 cm) of the column to the middle (20 cm to 30 cm) from where it increased again towards the bottom of the columns (40 cm). The moisture at a depth of 40 cm was varying over time within a small range which was accompanied by changes in the column weight in a range of around 60 g. This led to the conclusion that a specific minimum water table was necessary inside each column before the pressure head was high enough to release the water at the bottom.

During the first week after pharmaceutical spiking with the short scheduled sampling, only two out of three samples could be taken due to the small effluent amount.

The chloride mass balance was calculated for the whole experiment. The mass balance errors are given in SM 7. Balances include the remaining chloride inside the columns at the end of the experiment. The balance discrepancy ranges between -4.1 % to 1.7 % for the uninhibited and 2.5 % to 8.8 % for the inhibited columns. These values were mostly within the range of the analysis errors and therefore, indicating no substantial mass losses during the experiment.

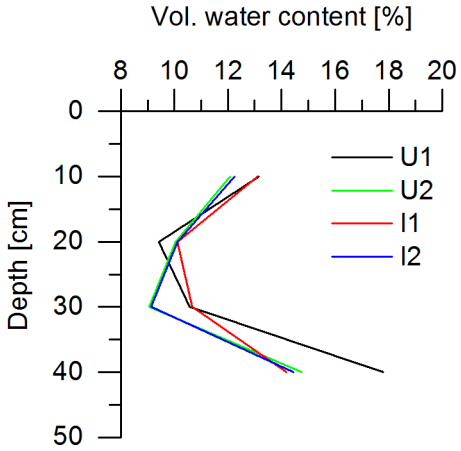

Figure 27: Mean measured humidity profiles for U1, U2, I1 and I2 from May 19th 2012 until August 8th 2012.

4.3.2. Conservative transport

Travel times through the columns were determined using the breakthrough of the conservative tracer bromide. Bromide was spiked to the feeding solution parallel to the pharmaceuticals after 53 days (Figure 28a and Figure 28b). The higher inflow concentrations in the inhibited columns are due to residual bromide in the NaN_3. The mean bromide travel velocity was calculated from the time when 50 % of the outflow concentration was met. The time of this inflection point represents the mean travel velocity (Boulding and Ginn, 2003). The bromide travel time for each column was derived after curve fitting via Origin™ from the Boltzmann function. The calculations were given in the supplementary materials (SM 8). The mean travel time for bromide over all columns was 3.7 days (3.3 to 4.4 days). This results in an overall mean travel velocity of 0.135 m/day. The mean travel time for the uninhibited columns was 4.0 days while the mean travel time for the inhibited columns was 3.4 days. One explanation of this faster travel time within the inhibited columns would be due to the missing microorganisms, including the missing sorption effects of the associated biofilms (Wunder et al., 2011).

However, the overall SOM within the presented study was rather small with values between 0.13 to 0.23 wt.%. Therefore, sorption seems to be of minor relevance for these experiments. This was confirmed by the results from the eluted soil samples which were taken after the end of the column experiment where the soil samples showed only negligible concentrations of BEZ and CBZ. According to the travel times of BEZ and CBZ which were each contrarily in the inhibited and the uninhibited columns (see SM 9), this variation might possibly result from the temporal resolution of the sampling events.

Mean travel times for CBZ were calculated as 4.7 days which is around 1 day slower than the bromide travel time. Mean BEZ travel time was calculated as 4.2 days. This is in accordance with contemporary literature, where the smaller bromide ion is expected to be more mobile and travel faster than larger molecules (Ptak et al., 2004). The travel times were also used to calculate retardation factors for CBZ and BEZ which were 1.26 and 1.14, respectively. The observed travel times of BEZ and CBZ were in the same range as the small sorption and retardation rates of other lab-scale transport studies:

BEZ retardation was determined in an irrigation setup on unsaturated columns with 1.46 (Siemens et al., 2010). Biodegradation batch test on BEZ in river sediment found sorption as well of minor importance (Kunkel and Radke, 2008). Ternes et al. (2007) predicted poor sorption rates to soil (K_{OC}) from K_{OW} values for BEZ and CBZ. CBZ showed no sorption during unsaturated column experiments (Patterson et al., 2011) and as well negligible sorption to sterilized sediment in batch test (Martínez-Hernández et al., 2014). Only small retardation factors of 1.8 could be detected during the percolation through sand-filled unsaturated columns (Scheytt et al., 2006). Similar CBZ retardation rates between 1.06 and 1.37 were calculated from saturated column studies with thermal treated sediment (Muller et al., 2013). Theoretical retardation was determined between 2.1 to 3.1 in soils irrigated with treated wastewater (Durán-Álvarez et al., 2012). Saturated-column studies with river sediment showed higher retardation between 5.3 at pH 4 and 3.6 at pH 8 indicating only a weak dependency of the pH (Schaffer et al., 2012). CBZ retardation was reported to increase with increasing amounts of SOM (Drillia et al., 2005; Chefetz et al., 2008; Yu et al., 2009). This was as well shown for CBZ and BEZ in sterilized soil experiments (Revitt et al., 2014).

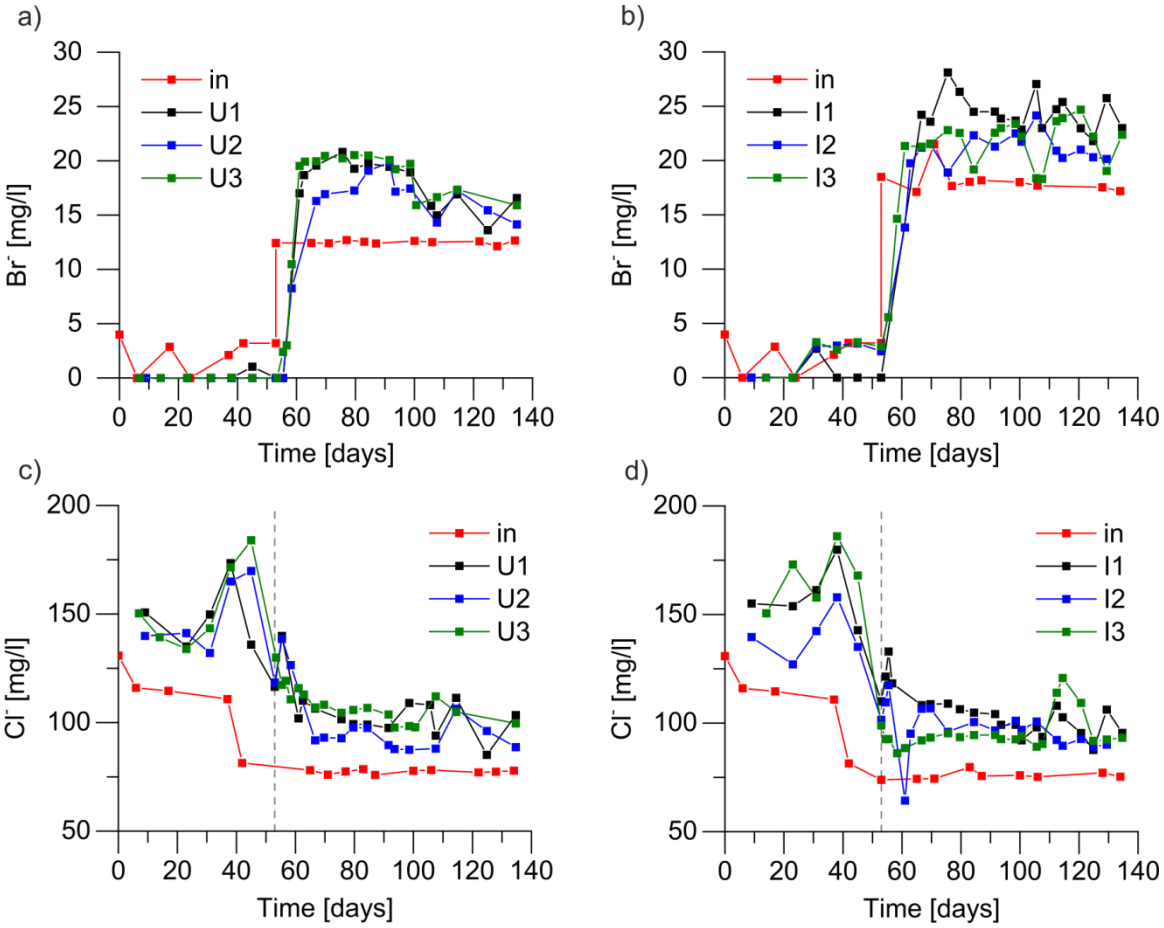

Figure 28, a-d: Inflow and outflow concentrations of the uninhibited (a) and inhibited (b) bromide columns and of the uninhibited (c) and inhibited (d) chloride columns. The grey line marks the beginning of pharmaceutical addition.

Chloride was already present in the used treated wastewater (Figure 28c and Figure 28d). The chloride effluent concentrations were always higher than in the inflow for all columns. This shows that the effect of evaporation was working for conservative species during the whole experiment under both, inhibited and uninhibited conditions. The same effect was expected for the pharmaceuticals as well. Due to a rain event shortly before the final abstraction at the WWTP Neureut, the chloride concentrations in the autoclaved water (starting from day 53) were smaller than the preceding concentrations during the adaption phase (first 53 days). This can be seen by a drop of the red inflow line in Figure 28c and Figure 28d where the concentrations decreased to a lower level. The effluent concentrations follow this trend and stabilized again with a shift of approximately 10 days. Because of this concentration decrease and the subsequent stabilization period, the following accumulation considerations were only calculated for the last 69 days of the experiment (May 31th 2012 to August 8th, 2012). Focusing on this 69 day period ensured constant and uniform flow and transport conditions 12 days after the onset of the pharmaceutical spiking.

4.3.3. Biodegradation

Due to separation of the experiments into a sterile (inhibited) and a non-sterile (uninhibited) environment, biodegradation and elimination effects could be observed. The uninhibited columns showed almost complete degradation of BEZ after a short initial concentration increase of up to 7 µg/l in all three columns (Figure 29a). This short peak is attributed to the adaption of the microbiology to the new introduced carbon source. A second peak which started after around 100 days is possibly a result of a one-day overheating inside the chamber with temperatures climbing up to 50 degrees for some hours. As a result, the microbiological activity might be either decreased or even stopped. The high temperature event matches as well with travel velocity as the outflow concentrations rise between 3 to 8 days after the heating event. A higher resolution was not available as the sampling interval was seven days. Desorption of adsorbed BEZ seems unlikely as this peak did neither show in the inhibited columns of BEZ (Figure 29b) nor for the CBZ concentrations.

The inhibited-columns (Figure 29b) effluent showed concentrations in the range of the inflow or above during the whole experiment. As biodegradation effects are of negligible relevance here, this difference can solely be attributed to accumulation.

Half-lives for BEZ and CBZ were calculated assuming first-order degradation kinetics and by using the mean inflow and effluent concentration of the uninhibited columns together with the mean travel times. Concentrations were corrected by the evaporation rates (Table 13). Sorption was reflected as well by considering the mean mass balance of the inhibited columns (see Table 13), as they should not feature any degradation and therefore the difference in their mass balance must be either attributed to measurement errors or to sorption. While sorption could not be proven by the elution test it was neglected for half-life calculation. BEZ showed a mean half-life of 1.3 days while the mean half- life for CBZ was 9.7 days.

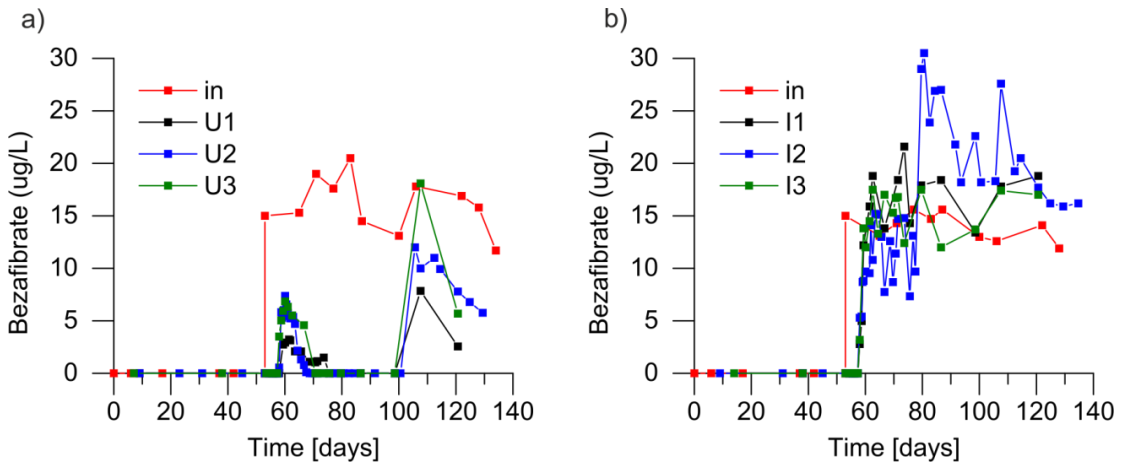

Figure 29, a-b: Inflow and outflow concentrations of bezafibrate of the uninhibited (a) and inhibited (b) columns.

Half-lifes of BEZ were in accordance with literature values as shown below. For better understanding, some half-lives needed to be translated from the original results by first-order kinetics:

BEZ half-lives were determined in bed sediments between 4.3 to 8.4 days (Kunkel and Radke, 2008). Aerobic soil aquifer treatment column experiments on lab scale operated with treated wastewater showed BEZ half-lives < 1.5 days (Schmidt, 2014). BEZ elimination in aerobic sand columns showed half-lives between 0.7 to 1.2 days (Maeng et al., 2011). BEZ was investigated in a setup irrigating natural soil with spiked treated wastewater in Israel. BEZ half-lives were determined in microcosm studies between 0.5 days to 1.2 days (Grossberger et al., 2014).

In contrast to BEZ, CBZ (Figure 30a and Figure 30b) did show much smaller mass balance differences between the inhibited and the uninhibited setup (Table 13). However, the mean CBZ half-life of 9.7 days is inconsistent to other soil column experiments where biodegradation of CBZ is mostly reported to be negligible (Scheytt et al., 2006; Monteiro and Boxall, 2009; Rauch-Williams et al., 2010; Maeng et al., 2011; Grossberger et al., 2014). Calculated half-lives were reported from microcosm studies with 69.7 to 92.6 days (Lam et al., 2004), from water/sediment batch studies with 328 days (Loffler et al., 2005), and from anaerobic column studies simulating managed aquifer recharge with > 100 days (Patterson et al., 2010). One reason for the inconsistent CBZ results might be the measurement error of the pharmaceutical analyses of our experiment. Besides, the high CBZ elimination rates might also be explained by the higher temperature of the column experiments. Comparing the typical laboratory temperature of 20°C or even colder field

conditions with 35°C[2] in the climatic chamber, microbiological activities are most likely increased.

4.3.4. Pharmaceutical accumulation

Considering conditions of constant in- and outflow to the columns, negligible sorption, and inhibited biodegradation, pharmaceutical concentrations should reach higher concentrations in the effluent than in the feeding water due to the evaporative water loss. For uninhibited conditions, at least biodegradation needs to be considered which might lead to lower effluent concentrations compared to the effluent concentrations of the inhibited columns.

For BEZ (Figure 29b) and CBZ (Figure 30b), the columns partly show the expected accumulation within the single experiments. For all columns, the inflowing concentration (red line) starts to increase at the spiking date and establishes a constant inflow level while the effluent concentrations increase with approx. 5 days delay. Especially for column I2 (blue line), both substances show a clear accumulation with higher effluent than influent concentrations after ~80 days. The other columns only show a slight accumulation. The CBZ concentration in column I1 is even slightly reduced compared to its inflow. Due to these variations and the replicate experimental setup, the following accumulation calculations were done as arithmetic mean over the inhibited and the uninhibited columns, respectively.

While there are many ways to decrease the concentrations (sorption, degradation, precipitation) of pharmaceuticals within the experimental setup, only evaporative processes can increase concentrations in a closed experimental environment. Therefore, the positive results of BEZ and CBZ under inhibited conditions give strong evidence of an evaporative pharmaceutical accumulation.

While BEZ showed, the expected reduction in the effluent concentration due to biodegradation (Figure 29a), CBZ (Figure 30a) concentrations in column U2 (blue line) showed accumulation even under uninhibited conditions. The results for CBZ from this uninhibited column indicate the potential for accumulation under conditions with active

[2] The temperature during the experiment was not 35°C but 30° as it is mentioned multiple times in the publication.

microbiological degradation processes. Thus accumulation of CBZ may occur also under real world conditions in the field.

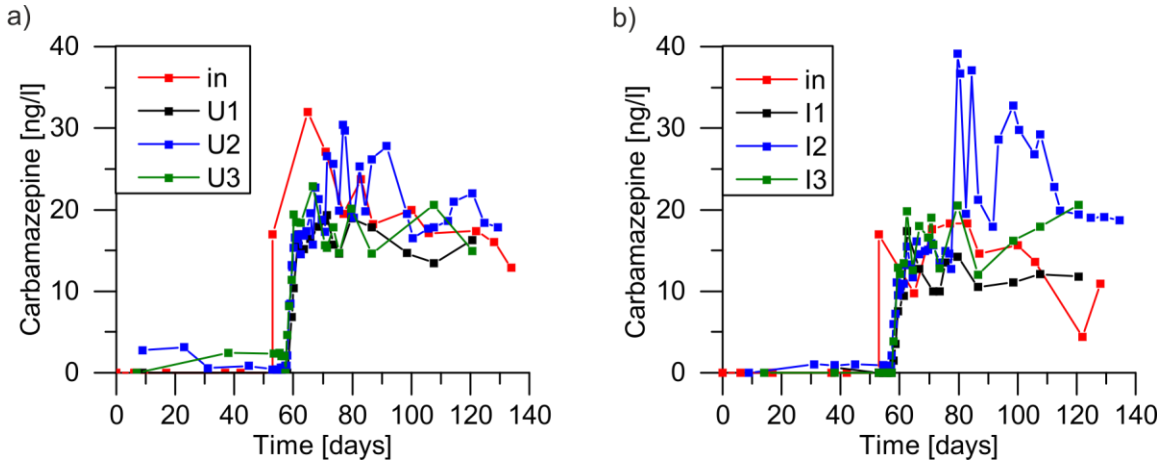

Figure 30, a-b: Inflow and outflow of the CBZ concentration of the uninhibited (a) and the inhibited (b) columns[3].

For a better comparison of evaporation rates and accumulation rates, accumulation factors were calculated for each column by dividing the mean outflow concentration by the mean inflow concentration. This was only done over the period of the last 69 days, assuming stable flow and accumulation conditions. A weighted mean was not calculated due to some data gaps in the analysis results. For the water flow, the value was calculated by diverting the mean inflow rate by the mean outflow rate. Final factors as given in Table 13 were calculated as mean from the three replicates of the uninhibited and inhibited columns, respectively. The flow rate thereby shows mean evaporation factors between 1.21 and 1.22 for water. Mean chloride factors were in the same range with 1.28 to 1.29 while mean bromide accumulation factors were slightly higher ranging from 1.24 to 1.42. Considering bromide as conservative tracer, this range is assumed to be the maximum accumulation for persistent substances within this experiment under the given conditions. Mean factors of the inhibited columns for e.g. BEZ and CAR were in similar ranges than the evaporative water loss, therefore indicating that no relevant sinks had been neglected. Factors smaller one, for CBZ and especially BEZ in the uninhibited columns, indicate the amount of biodegradation during the percolation. One single result from CBZ in column U2 (Figure 30a) even indicated the potential of accumulation under uninhibited conditions with an accumulation factor of 1.03.

[3] The concentrations scale of carbamazepine in all experiments of this publication was not ng/l but µg/l. All discussions and results are based on the actual concentration scale.

Table 13: Accumulation factors and mass balance as mean of the each three inhibited and uninhibited replicate columns calculated for the final 69 days of the experiment. Numbers in brackets give the standard deviation (n=3).

	Uninhibited (U1, U2, U3)		Inhibited (I1, I2, I3)	
	Mean concentration accumulation factor c_{out}/c_{in} [-]	Mean mass balance m_{out}-m_{in} [%]	Mean concentration accumulation factor c_{out}/c_{in} [-]	Mean mass balance m_{out}-m_{in} [%]
Chloride	1.28 (± 0.06)	3.4	1.29 (± 0.03)	2.6
Bromide	1.42 (± 0.07)	-1.5	1.24 (± 0.08)	-9.4
Lithium	1.28 (±0.06)	-17.4	1.16 (± 0.04)	-6.2
Bezafibrate	0.14 (± 0.04)	-83.3	1.20 (± 0.07)	-6.5
Carbamazepine	0.86 (± 0.12)	-31.6	1.20 (± 0.29)	-6.2
	Mean evaporation factor Flow $rate_{in}$/Flow $rate_{out}$ [-]		Mean evaporation factor Flow $rate_{in}$/Flow $rate_{out}$ [-]	
Water	1.22 (± 0.03)	-18.3	1.21 (± 0.04)	-17.6

4.3.5. Relevance and transfer of the results with regards to risk assessment for BEZ and CBZ

The presented accumulation experiment was setup close to conditions observed in the LJV. Therefore the relevance of the results and their transferability to other field conditions needs to be discussed. The experimental accumulation factors describe a setup with very uniform climate, irrigation and contamination conditions. Comparing this to the real conditions in the LJV, temperatures and contaminant concentrations might shift in daily patterns including shifting evaporation rates and contamination peaks and lows. On the other hand, irrigation is usually not applied continuously but during intense times with lower evaporation (e.g. morning and evening). Therefore, real flow rates will be much higher with subsequently lower evaporation shares. Plant uptake and photo-degradation effects might reduce the pharmaceutical concentrations additionally. Taking

all this into account, the obtained accumulation factors can be seen as an orientation to upper limits and worst case scenarios.

Environmental concentrations were reported by Zemann et al. (2014) with mean CBZ concentrations of 74 ng/l and a maximum detection of 500 ng/l in groundwater of the LJV. Concerning a risk assessment, a "predicted no-effect concentration" (PNEC) consumend via drinking water for humans for CBZ was reported with 232 µg/l by Cunningham et al. (2010) and with 7.86 µg/l by Khan et al. (2015). A drinking water guideline value for CBZ was derived with 1 µg/l (Schriks et al., 2010). Kümmerer (2008) suggested a provisional drinking water limit of 50µg/l. Consequently, reported concentrations in groundwater of 211 ng/l detected in Tunesia (Fenet et al., 2012) and the 500 ng/l reported from Jordan (Zemann et al., 2014) were therefore uncritical.

Corresponding with its degradability, BEZ was only detected in surface water of the LJV with 89 ng/l (Zemann et al., 2014). Discussing the risk assessment of BEZ, no mutagen or genotoxic effects could be found for BEZ and its photoproducts (Isidori et al., 2007). A PNEC for humans via drinking water was assessed with 39.3 µg/l (Khan et al., 2015). The Australian drinking water guideline calculated a threshold for drinking water for a therapeutic dose of BEZ with 300 µg/l (EPHC-NHMRC-NRMMC, 2008). Kümmerer (2008) suggested a provisional drinking water limit of 35 µg/l. Consequently, the environmental risk of BEZ needs to be considered despite its good degradability and the published guideline values which suggest that there is no acute concern regarding the concentrations in the LJV.

Given that groundwater concentrations already reach PNEC of e.g. CBZ, it must be mentioned, that due to treatment and purification processes prior to consumption of ground- and surface water, concentrations in drinking water are usually very low and considered unproblematic (Webb et al., 2003; Schwab et al., 2005; Houeto et al., 2012).

Considering closed cycles, where the local groundwater is used again for irrigation, a long-term accumulation of persistent substances needs to be taken into account. However, in such closed-loop situations, also salt will enrich in the irrigation water, possibly causing soil salinization prior to hazardous pharmaceutical concentration levels.

4.3.6. Inconsistencies within experiments and results

Despite a regular refreshment of the feeding solution, the inflow concentrations of many substances showed variations of up to ~20 %. In the outflow concentrations, these variations cannot always be distinguished from the natural variations which result from preferential flow paths in unsaturated soil. Such variations were observed in numerous unsaturated percolation experiments documented in the literature (Lewis and Sjöstrom, 2010). To address this issue, calculations in the paper rely on mean values. Some data gaps (missing samples) needed to be filled as well, which was mostly done using average values of the previous and the following samples. For the bromide inflow concentration, the first sample at the beginning of spiking is missing, as the feeding solution ran dry for the uninhibited columns due to a technical problem. This value therefore was taken from the inhibited feeding solution, which was based on the same wastewater.

Variations of the experimental setup, i.e. reduction of the flow rate would possibly have fostered the validity of the results. Though, due to possible interpretation of the results for the LJV, concentrations, flow rates and evaporation rates were not elevated artificially.

4.4. Conclusions

To investigate the hypothesis of evaporative accumulation of pharmaceuticals in groundwater under arid climate conditions, unsaturated-column studies were performed in a climatic chamber. The experiments were designed analogous to the use of treated wastewater in irrigated agriculture such as the Jordan Valley. The analysis focuses on two pharmaceuticals, the more easily biodegradable bezafibrate and the rather persistent carbamazepine. Experiments were conducted under close-to-natural conditions. A control was run with a toxic feeding solution to inhibit microbiological activities. For each environmental condition, three parallel columns were operated as replicate with similar flow conditions. Based on the results, the following conclusions can be drawn:

- Evaporative enrichment of pharmaceuticals must be taken into account under arid conditions. In the experiment, columns with inhibited microbiological activity showed higher concentrations of the pharmaceuticals carbamazepine and bezafibrate in the outflow water compared to the inflowing water.
- Biodegradation processes during groundwater recharge might effectively prevent concentrations from becoming higher than in the irrigation water for most pharmaceuticals. For columns with uninhibited microbiology, outflow

concentrations remained lower (bezafibrate) or equal (carbamazepine) compared to the inflow concentrations.

- The observed microbiological degradation of carbamazepine is higher at 30°C than in comparable studies from the literature at 20°C. Carbamazepine showed losses of approx. 30 % along the 50 cm soil passage with mean travel times of 4.7 days.
- For persistent pharmaceuticals and micro-pollutants, enrichment can occur at the same rate as the well-documented enrichment of salts. For columns with inhibited microbiology, both bezafibrate and carbamazepine showed enrichment similar to that of chloride.

Consequently, the observed concentrations provide conclusive evidence that a sufficiently persistent pharmaceutical with negligible biodegradation can accumulate in soil water if irrigated under arid conditions. The hypothesis of evaporative enrichment in arid environments therefore can be accepted for the given experiment conditions. Further research is required with similar methods to investigate the accumulation risk of other persistent substances such as e.g. artificial sweeteners or iodinated X-ray contrast media.

4.5. Acknowledgements

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4.6. References

Chávez, A., C. Maya, R. Gibson and B. Jiménez (2011). "The removal of microorganisms and organic micropollutants from wastewater during infiltration to aquifers after irrigation of farmland in the Tula Valley, Mexico." Environmental Pollution **159**(5): 1354-1362.

Chefetz, B., T. Mualem and J. Ben-Ari (2008). "Sorption and mobility of pharmaceutical compounds in soil irrigated with reclaimed wastewater." Chemosphere **73**(8): 1335-1343.

Chen, F., G. G. Ying, L. X. Kong, L. Wang, J. L. Zhao, L. J. Zhou and L. J. Zhang (2011). "Distribution and accumulation of endocrine-disrupting chemicals and pharmaceuticals in wastewater irrigated soils in Hebei, China." Environmental Pollution **159**(6): 1490-1498.

Cunningham, V. L., C. Perino, V. J. D'Aco, A. Hartmann and R. Bechter (2010). "Human health risk assessment of carbamazepine in surface waters of North America and Europe." Regulatory Toxicology and Pharmacology **56**(3): 343-351.

EPHC-NHMRC-NRMMC (2008). Australian Guidelines for Water Recycling - Augmentation of Drinking Water Supplies 22. Environment Protection and Heritage Council (EPHC), National Health and Medical Research Council (NHMRC) and Natural Resource Management Ministerial Council (NRMMC).

Drillia, P., K. Stamatelatou and G. Lyberatos (2005). "Fate and mobility of pharmaceuticals in solid matrices." Chemosphere **60**(8): 1034-1044.

Fenet, H., O. Mathieu, O. Mahjoub, Z. Li, D. Hillaire-Buys, C. Casellas and E. Gomez (2012). "Carbamazepine, carbamazepine epoxide and dihydroxycarbamazepine sorption to soil and occurrence in a wastewater reuse site in Tunisia." Chemosphere **88**(1): 49-54.

Ferrari, B., N. Paxeus, R. Lo Giudice, A. Pollio and J. Garric (2003). "Ecotoxicological impact of pharmaceuticals found in treated wastewaters: study of carbamazepine, clofibric acid, and diclofenac." Ecotoxicol Environ Saf **55**(3): 359-370.

Grossberger, A., Y. Hadar, T. Borch and B. Chefetz (2014). "Biodegradability of pharmaceutical compounds in agricultural soils irrigated with treated wastewater." Environmental Pollution **185**: 168-177.

Han, G. H., H. G. Hur and S. D. Kim (2006). "Ecotoxicological risk of pharmaceuticals from wastewater treatment plants in Korea: Occurrence and toxicity to Daphnia magna." Environmental Toxicology and Chemistry **25**(1): 265-271.

Harada, A., K. Komori, N. Nakada, K. Kitamura and Y. Suzuki (2008). Biological effects of PPCPs on aquatic lives and evaluation of river waters affected by different wastewater treatment levels. Water Science and Technology. **58:** 1541-1546.

Hua, J. M., P. L. An, J. Winter and C. Gallert (2003). "Elimination of COD, microorganisms and pharmaceuticals from sewage by trickling through sandy soil below leaking sewers." Water Research **37**(18): 4395-4404.

Isidori, M., A. Nardelli, L. Pascarella, M. Rubino and A. Parrella (2007). "Toxic and genotoxic impact of fibrates and their photoproducts on non-target organisms." Environment International **33**(5): 635-641.

Kinney, C. A., E. T. Furlong, S. L. Werner and J. D. Cahill (2006). "Presence and distribution of wastewater-derived pharmaceuticals in soil irrigated with reclaimed water." Environmental Toxicology and Chemistry **25**(2): 317-326.

Kunkel, U. and M. Radke (2008). "Biodegradation of acidic pharmaceuticals in bed sediments: Insight from a laboratory experiment." Environmental Science & Technology **42**(19): 7273-7279.

Lewis, J. and J. Sjöstrom (2010). "Optimizing the experimental design of soil columns in saturated and unsaturated transport experiments." Journal of Contaminant Hydrology **115**(1–4): 1-13.

Maeng, S. K., S. K. Sharma, C. D. T. Abel, A. Magic-Knezev and G. L. Amy (2011). "Role of biodegradation in the removal of pharmaceutically active compounds with different bulk organic matter characteristics through managed aquifer recharge: Batch and column studies." Water Research **45**(16): 4722-4736.

Martínez-Hernández, V., R. Meffe, S. Herrera, E. Arranz and I. de Bustamante (2014). "Sorption/desorption of non-hydrophobic and ionisable pharmaceutical and personal care products from reclaimed water onto/from a natural sediment." Science of the Total Environment **472**(0): 273-281.

Muller, B., T. Scheytt and G. Grutzmacher (2013). "Transport of primidone, carbamazepine, and sulfamethoxazole in thermally treated sediments-laboratory column experiments." Journal of Soils and Sediments **13**(5): 953-965.

Patterson, B. M., M. Shackleton, A. J. Furness, E. Bekele, J. Pearce, K. L. Linge, F. Busetti, T. Spadek and S. Toze (2011). "Behaviour and fate of nine recycled water trace organics during managed aquifer recharge in an aerobic aquifer." Journal of Contaminant Hydrology **122**(1-4): 53-62.

Patterson, B. M., M. Shackleton, A. J. Furness, J. Pearce, C. Descourvieres, K. L. Linge, F. Busetti and T. Spadek (2010). "Fate of nine recycled water trace organic contaminants and metal(loid)s during managed aquifer recharge into a anaerobic aquifer: Column studies." Water Research **44**(5): 1471-1481.

Ptak, T., M. Piepenbrink and E. Martac (2004). "Tracer tests for the investigation of heterogeneous porous media and stochastic modelling of flow and transport - A review of some recent developments." Journal of Hydrology **294**(1-3): 122-163.

Quinn, B., F. Gagné and C. Blaise (2008). "The effects of pharmaceuticals on the regeneration of the cnidarian, Hydra attenuata." Science of the Total Environment **402**(1): 62-69.

Rauch-Williams, T., C. Hoppe-Jones and J. E. Drewes (2010). "The role of organic matter in the removal of emerging trace organic chemicals during managed aquifer recharge." Water Research **44**(2): 449-460.

Revitt, D. M., T. Balogh and H. Jones (2014). "Sorption behaviours and transport potentials for selected pharmaceuticals and triclosan in two sterilised soils." Journal of Soils and Sediments **15**(3): 594-606.

Schaffer, M., N. Boxberger, H. Börnick, T. Licha and E. Worch (2012). "Sorption influenced transport of ionizable pharmaceuticals onto a natural sandy aquifer sediment at different pH." Chemosphere **87**(5): 513-520.

Scheytt, T. J., P. Mersmann and T. Heberer (2006). "Mobility of pharmaceuticals carbamazepine, diclofenac, ibuprofen, and propyphenazone in miscible-displacement experiments." Journal of Contaminant Hydrology **83**(1-2): 53-69.

Schmidt, N. (2014). Elimination anthropogener organischer Spurenstoffe im Zuge der Grundwasseranreicherung, TZW: DVGW-Technologiezentrum Wasser.

Schriks, M., M. B. Heringa, M. M. E. van der Kooi, P. de Voogt and A. P. van Wezel (2010). "Toxicological relevance of emerging contaminants for drinking water quality." Water Research **44**(2): 461-476.

Siemens, J., G. Huschek, C. Siebe and M. Kaupenjohann (2008). "Concentrations and mobility of human pharmaceuticals in the world's largest wastewater irrigation system, Mexico City-Mezquital Valley." Water Research **42**(8-9): 2124-2134.

Siemens, J., G. Huschek, G. Walshe, C. Siebe, R. Kasteel, S. Wulf, J. Clemens and M. Kaupenjohann (2010). "Transport of pharmaceuticals in columns of a wastewater-irrigated Mexican clay soil." Journal of Environmental Quality **39**(4): 1201-1210.

Ternes, T. A., M. Bonerz, N. Herrmann, B. Teiser and H. R. Andersen (2007). "Irrigation of treated wastewater in Braunschweig, Germany: An option to remove pharmaceuticals and musk fragrances." Chemosphere **66**(5): 894-904.

Wolf, L., A. Pöschko, M. Zemann, H. Werz, A. Sawarieh, N. Seder, M. Stieber, F. Sacher and A. Tiehm (2009). "Understanding occurrence of xenobiotics in closed river basins, examples from the Jordan Valley." International Conference on Xenobiotics in the Urban Water Cycle (XENOWAC).

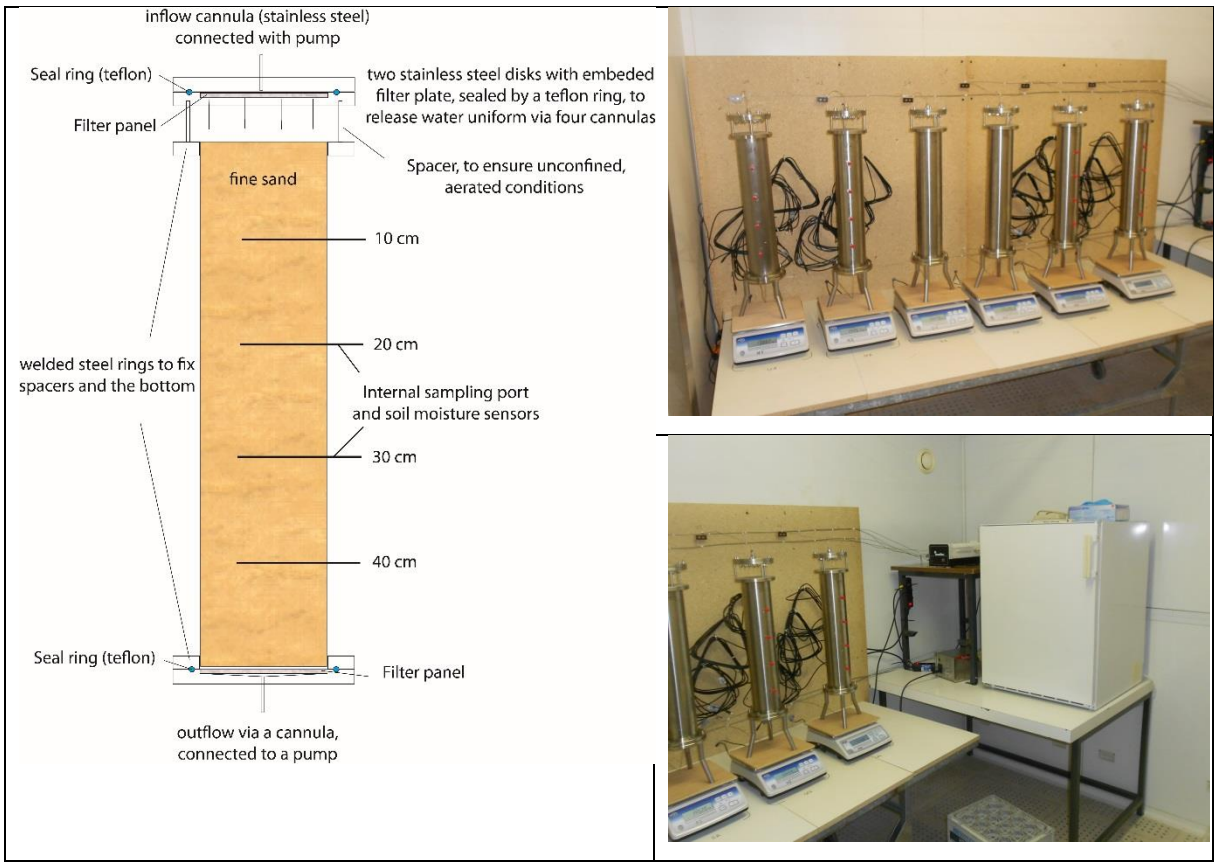
Yu, L., G. Fink, T. Wintgens, T. Melin and T. A. Ternes (2009). "Sorption behavior of potential organic wastewater indicators with soils." Water Research **43**(4): 951-960.

Zemann, M., L. Wolf, F. Grimmeisen, A. Tiehm, J. Klinger, H. Hotzl and N. Goldscheider (2015). "Tracking changing X-ray contrast media application to an urban-influenced karst aquifer in the Wadi Shueib, Jordan." Environmental Pollution **198**: 133-143.

Zemann, M., L. Wolf, A. Pöschko, N. Schmidt, A. Sawarieh, N. Seder, A. Tiehm, H. Hötzl and N. Goldscheider (2014). "Sources and processes affecting the spatio-temporal distribution of pharmaceuticals and X-ray contrast media in the water resources of the Lower Jordan Valley, Jordan." Science of the Total Environment **488–489**(0): 100-114.

4.7. Supplementary materials (SM)

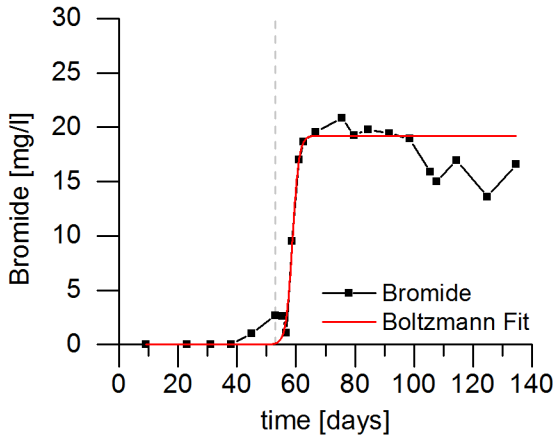
SM 6: Schematic column setup (left). Columns arrangement on scales named U1, I1, U3, I3, U2 and I2 from left to right (right top). Storage fridge and peristaltic pumps (right bottom).



SM 7: Chloride, bromide and water mass balances and the mean flow rate calculated over the whole experiment time.

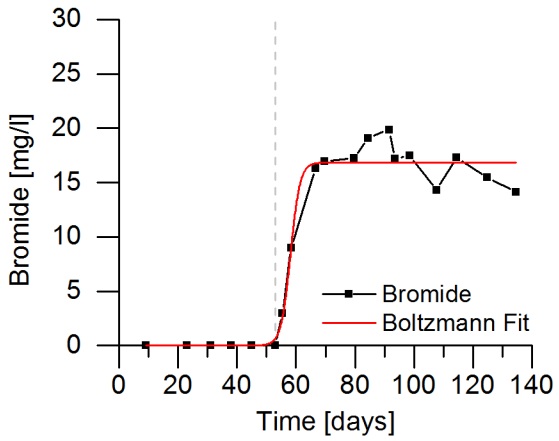
Column number	Mass balance			Mean inflow	Mean outflow
	H_2O [%]	Cl^- [%]	Br^- [%]	H_2O [ml/day]	H_2O [ml/day]
U1	-18.3	-1.6	5.6	122.9	98.7
U2	-18.3	-4.1	-1.0	122.9	103.9
U3	-18.3	1.7	12.8	122.9	98.7
I1	-17.6	5.1	7.9	121.8	97.4
I2	-17.6	2.5	-3.1	121.8	98.2
I3	-17.6	8.6	2.8	121.8	121.8

SM8-1: Bromide concentrations and derived Boltzmann fit for column U1 (uninhibited).



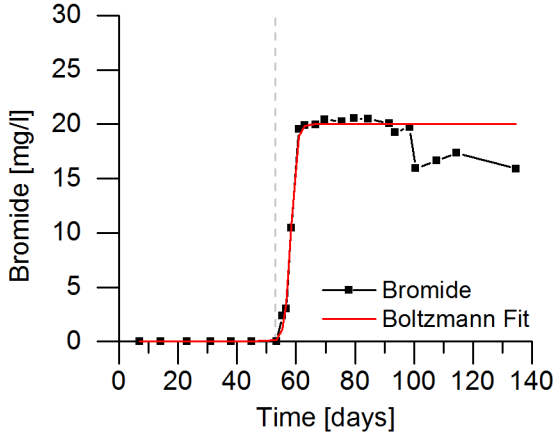
	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	4,42554		
4	Adj. R-Square	0,93713		
5			Value	Standard Error
6	Bromide	A1	0	0
7		A2	19,214	0
8		x0	58,78319	0,40801
9		dx	1,09909	0,35143
10		span	19,214	0
11		EC50	3,38234E25	1,38004E25

SM8-2: Bromide concentrations and derived Boltzmann fit for column U2 (uninhibited).



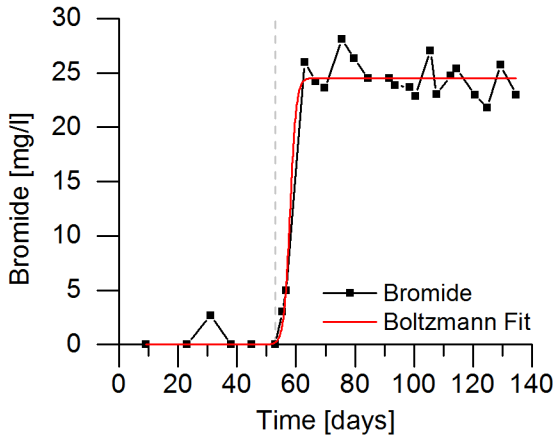
	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	1,95897		
4	Adj. R-Square	0,97044		
5			Value	Standard Error
6	Bromide	A1	0	0
7		A2	16,84484	0,42355
8		x0	58,12413	0,50713
9		dx	1,6058	0,55241
10		span	16,84484	0,42355
11		EC50	1,7498E25	8,87369E24

SM8-3: Bromide concentrations and derived Boltzmann fit for column U3 (uninhibited).



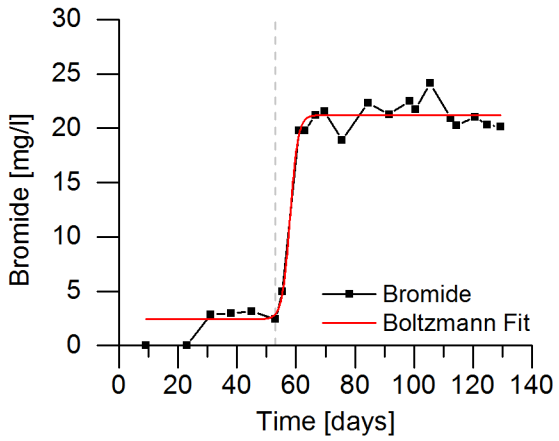
	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	2,52452		
4	Adj. R-Square	0,96954		
5			Value	Standard Error
6	Bromide	A1	0	0
7		A2	20,009	0
8		x0	58,23902	0,27438
9		dx	0,99019	0,25895
10		span	20,009	0
11		EC50	1,96284E25	5,38563E24

SM8-4: Bromide concentrations and derived Boltzmann fit for column I1 (inhibited).



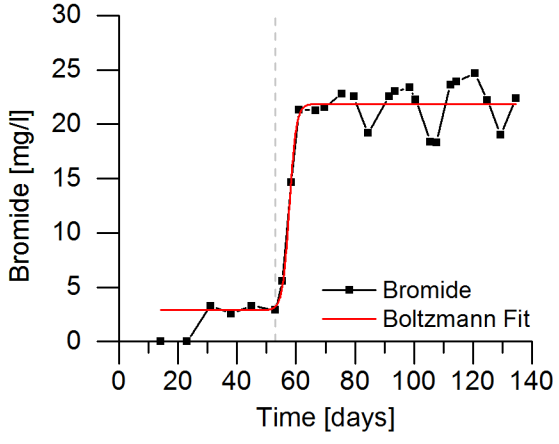
	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	2,40664		
4	Adj. R-Square	0,98028		
5			Value	Standard Error
6	Bromide	A1	0	0
7		A2	24,52792	0,36913
8		x0	57,96622	0,99802
9		dx	1,04007	0,65819
10		span	24,52792	0,36913
11		EC50	1,4942E25	1,49124E25

SM8-5: Bromide concentrations and derived Boltzmann fit for column I2 (inhibited).



	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	1,80649		
4	Adj. R-Square	0,97786		
5			Value	Standard Error
6	Bromide	A1	2,43	0
7		A2	21,19714	0,36893
8		x0	57,96636	0,69107
9		dx	1,35201	0,3216
10		span	18,76714	0,36893
11		EC50	1,49442E25	1,03275E25

SM8-6: Bromide concentrations and derived Boltzmann fit for column I3 (inhibited).



	A1(Y)	A2(Y)	A3(Y)	A4(Y)
1	Model	Boltzmann		
2	Equation	y = A2 + (A1-A2)/(1 + exp((x-x0)/dx))		
3	Reduced Chi-Sqr	3,44027		
4	Adj. R-Square	0,95619		
5			Value	Standard Error
6	Bromide	A1	2,88	0
7		A2	21,85001	0,44506
8		x0	57,72388	0,44948
9		dx	1,17292	0,39445
10		span	18,97001	0,44506
11		EC50	1,17263E25	5,27072E24

SM 9: Travel times of bromide, BEZ and CBZ for all columns.

	Bromide travel time	CBZ travel time	BEZ travel time
	[days]	[days]	[days]
U1	4.4	5.2	4.5
U2	3.7	5.0	3.8
U3	3.8	4.6	4.1
I1	3.5	4.7	4.6
I2	3.5	4.2	4.1
I3	3.3	4.5	4.2
Mean	3.7	4.7	4.2
Mean uninhibited	4.0	4.9	4.1
Mean inhibited	3.5	4.5	4.3

5. Conclusions and Perspectives

The accumulation potential of different pharmaceuticals under arid climate conditions was investigated within the framework of a multilateral water management project (BMBF-SMART) for the area of the Lower Jordan Valley. The two step approach included long term field investigations combined with unsaturated column percolation tests under laboratory conditions.

The investigated substances were chosen on the basis of a previously conducted screening campaign in 2007 which showed higher concentrations of the X-ray contrast media diatrizoic acid in groundwater in comparison to the related surface water and treated wastewater inflow and therefore indicated a possible accumulation. The local setup provides favorable conditions for accumulation as an intensive reuse of treated wastewater in irrigation is combined with the usage of locally pumped groundwater, high temperatures and evaporation rates, almost no precipitation and the closed nature of the Dead Sea basin. Samples were taken from groundwater, surface water and treated wastewater in three areas of the valley (north, middle, south) and from two Wadis (Shueib, Kafrein) of the eastern escarpment over a period of five years (2008 – 2012). These study areas were chosen in order to represent aquifer systems with expositions to different regimes of anthropogenic pollution. The shallow alluvial aquifers of the Jordan Valley on the one hand, where irrigation water includes large volumes of treated wastewater, and on the other hand the limestone aquifer of the steep declining wadis, where urban pollution sources impact the karstic environment.

The results of the sampling campaigns show first and foremost the prevailing occurrence of multiple pharmaceuticals in groundwater, surface water and treated wastewater over the whole investigated area. This indicates that the local water sources are strongly affected by anthropogenic influences. In particular, the two X-ray contrast media diatrizoic acid (79 % detection rate) and iopamidol (~50 %) were present in groundwater samples of the whole area. Other substances like carbamazepine, ibuprofen or fenofibrate were detected in groundwater samples as well but with much smaller detection rates (< 14 %).

For the Lower Jordan Valley, the pharmaceutical detections can mainly be attributed to the intensive reuse of treated wastewater for irrigation in agriculture. Concentrations and

detection rates increase from north to south along the flow of the King Abdullah Canal, thereby reflecting the environmental impact of Amman`s biggest wastewater treatment plant As Samra. The detections in groundwater of Wadi Shueib and Wadi Kafrein originate most likely from leaking sewers or cesspits and illegal effluent disposal.

The data evaluation regarding the supposed accumulation of pharmaceuticals revealed particular trends for the two X-ray contrast diatrizoic acid and iopamidol in comparison to other substances. Almost all detected substances showed decreasing concentrations along their transport from wastewater sources to surface water and finally to groundwater with a roughly 1 log reduction from one compartment to the next. This is mostly due to dilution but might as well be attributed to microbiological degradation or sorption effects. The sole exception was diatrizoic acid which showed similar concentrations in all water types while it was mostly detected in groundwater and beside was almost always absent in treated wastewater. This would undergird the accumulation hypothesis. However, this observation was made for both investigated areas, the Lower Jordan Valley and the Wadis. While evaporation plays an important role in the alluvial aquifers of the Lower Jordan Valley, the limestone aquifers in Wadi Shueib and Kafrein feature an unsaturated zone of 50 to 70 m and hence should not be subjected to evaporative processes. The tempo-spatial occurrence of iopamidol showed increasing detection rates over the five year investigation period which might as well be interpreted as an indicator for slowly increasing concentrations caused by evaporation. However this trend could again be observed for all investigated areas, including Wadi Shueib. Improved detection limits over time could be ruled out after consulting the analyzing laboratory.

In respect to concentration and the tempo-spatial evolution in groundwater, surface water and treated wastewater, the non-ionic X-ray contrast media iopamidol and the ionic diatrizoic acid showed a clear different environmental behavior. Possible reasons might be caused either by a more recent introduction of the non-ionic iopamidol or by its lower persistence. As both substances were reported with a similar environmental persistence the findings points to the more recent introduction of iopamidol. This goes along with reported changes in X-ray contrast media prescription practice for Jordan which the author therefore suggests as reason for the observed patterns. Information obtained from different hospitals as well as literature indicates that the ionic contrast media diatrizoic acid was banned for intravascular application between 2000 and 2006 due to adverse health effects. The non-ionic iopamidol as one possible substitute in radiology diagnostics

was afterwards used predominantly. Those observations fit to the tempo-spatial evolution of the two substances in the study area that showed a constant occurrence of diatrizoic acid combined with a spreading of iopamidol. The observed diatrizoic acid therefore should be described as residual concentration resulting from its high environmental persistence. Within the alluvial aquifer of the Lower Jordan Valley, such residual concentrations might easily be explained by the combination of slow groundwater movement and the constant reuse use of locally pumped groundwater for irrigation. The karstic aquifers in Wadi Shueib and Wadi Kafrein however would release this pollutant much faster via the springs. A possible explanation for the observed concentrations at the springs are residual concentrations which are still present in the huge unsaturated zone (50 - 75 m) and the matrix of the limestone aquifers after years of exfiltration from leaking sewers and cesspits. Nowadays, they are leaching towards the groundwater and the main conduit network.

Summarizing this discussion, an accumulation of pharmaceuticals could neither be deduced from the available dataset nor could the assumption be declined. Instead persistent pharmaceuticals particularly showed their potential as suitable tracers to backtrack and interpret anthropogenic impacts or pollutant fluxes. Another assessment evaluating the tracer potential of pharmaceuticals was performed within this study including additional anthropogenic wastewater indicators like *E. coli* and nitrate. For Wadi Shueib and Kafrein, the correlation of diatrizoic acid and nitrate concentrations showed a significant, almost linear, correlation of both substances in all groundwater resources throughout the investigated period. This leads to the conclusion that nitrate concentrations result from the same source as the pharmaceuticals. Since X-ray contrast media were not applied to livestock in Jordan, the infiltration of slurry can be neglected which makes infiltrating sewage the most likely source. Diatrizoic acid therefore was utilized successfully as an anthropogenic tracer.

In order to quantify the anthropogenic influence to the karst aquifer, a combined assessment of the "pharmaceutical detection rate", which describes the number of detected substances at one sampling point, together with *E. coli* and nitrate concentrations, was performed. For the same area, the spatial occurrence of those pollution indicators could be linked to the hydrological setting (flow direction) and the distance to the city Salt. The highest rates of all three indicators could be observed downstream of the urban areas, while the upstream wells and springs were unpolluted

most of the time. The significant correlation of all three indicators recommends the pharmaceutical detection rate as a suitable tool for pollution indication on a semi-quantitative level as it makes use of single scattered detections of different substances. Restrictions must be made with respect to the number of analyzed pharmaceuticals, as the detection rate only reflects the investigated substances which always might only be a small share of the unknown total number. However, the higher the number of analyzed substances, the higher is the significance of the pharmaceutical detection rate.

While pharmaceuticals showed their potential as suitable tracers on various levels within this study, they unfortunately offer a restricted applicability. High costs for analysis are only one reason. Many substances are subject to sorption or degradation processes and thereby enhance the natural attenuation process which is highly appreciated in reducing xenobiotics or other anthropogenic trace substances as a pretreatment step before water abstraction. Distinct substances like some persistent X-ray contrast media might feature the potential as ideal tracers and can add valuable information if their presence in groundwater is evaluated and interpreted in an adequate way. However, concerning a sustainable groundwater management it is unwanted to introduce such substances artificially into drinking water aquifers.

As the accumulation of pharmaceutical substances could not be validated in the field during the five year observation period of this study, the question remains whether persistent pharmaceuticals can accumulate under “suitable” conditions. Processes like soil salting as result of inadequate irrigation in arid areas seems very comparable to the assumed enrichment of persistent pharmaceuticals in the Lower Jordan Valley. In addition to the described field investigations, the hypothesis of evaporative pharmaceutical accumulation under arid climate conditions was subsequently addressed in laboratory experiments based on unsaturated columns which were trickled with treated wastewater in a climatic chamber.

The experimental setup was designed as analogy to the use of treated wastewater in irrigated agriculture such as found in the Lower Jordan Valley. This was done to evaluate the risk of the long term application of treated wastewater to the area. In order to create near to natural conditions, the columns were operated with treated wastewater, adapted flow rates, temperature and humidity according to measured data from the Lower Jordan Valley. The analysis focused on two pharmaceuticals, the more easily biodegradable lipid

lowering agent bezafibrate and the rather persistent antiepileptic carbamazepine. By operating different columns with active and inhibited microbiological environment, the most important aspects of pharmaceutical transport (sorption, degradation) were investigated along with their evaporative enrichment. For each environmental condition, three parallel columns were operated as replicates with similar flow conditions.

The column experiments showed an accumulation of both pharmaceuticals under inhibited microbiological conditions with higher concentrations of bezafibrate and carbamazepine in the outflow, compared to the inflow of the columns. The hypothesis of evaporative enrichment of pharmaceuticals under the given conditions can therefore be accepted and is furthermore a process which must be taken into account under arid conditions. The observed accumulation occurred at the same rate for bezafibrate and carbamazepine and was similar to the enrichment of chloride and in the ratio of the evaporated water. For persistent pharmaceuticals and micro-pollutants, enrichment therefore can occur at the same rate as the well documented enrichment of salts.

Biodegradation processes during groundwater recharge might effectively prevent concentrations from becoming higher than the irrigation water for most pharmaceuticals. This was observed in the experiments, where columns with uninhibited microbiology showed lower (bezafibrate) or equal (carbamazepine) outflow concentrations compared to the inflow concentration. The observed biodegradation of carbamazepine (half-life: 9.7 days) is higher at 30°C than in comparable studies from literature under 20°C. This might be attributed to a more active microbiological community at higher temperatures. Carbamazepine showed losses of approx. 30 % along the 50 cm soil passage with a mean travel time of 4.7 days. Bezafibrate showed a biodegradation (half-life: 1.3 days) similar to results reported from comparable studies. The loss along the column was around 80 % with a mean travel time of 4.2 days. Retardation rates were determined as well with 1.26 for carbamazepine and 1.14 for bezafibrate.

One single column showed enrichment of carbamazepine for the uninhibited microbiological environment, therefore indicating that an accumulation in the field might be possible as well. However, such enrichment might only take place under idealized conditions where other persistent substances like salts would enrich in parallel.

Regarding a risk assessment, the pharmaceutical concentration levels found in the Jordan Valley were far below the levels of e.g. chloride or other salts. They therefore should be

given a higher priority as effects like soil salting, inedibility of drinking water or harmful effects on plant growth will occur prior to toxic pharmaceutical levels. Even though the experiments where conducted under near to natural conditions, they may still overestimate the real conditions in the Lower Jordan Valley (temperature changes, intermitted irrigation etc.). Against this background, combining the pharmaceutical concentrations documented for the Lower Jordan Valley and their temporal fate together with the laboratory result, no risk is arising regarding harmful pharmaceutical levels for humans. From this follows, that, from a pharmaceutical point of view, the use of treated wastewater in irrigation can be continued without objection for the foreseeable future. However, the use of treated wastewater in agricultural irrigation should be monitored carefully, as other wastewater borne ingredients like endocrine disruptors or heavy metals bear their own risk potential.

Within the scope of this work, only four sampling campaigns were conducted. While the results are valuable to characterize the pharmaceutical concentrations and the anthropogenic impact to the groundwater resources in the Lower Jordan Valley, especially the question of pharmaceutical accumulation in the field still needs to be evaluated on a much wider data base in order to gain statistical evidence. Hence, further monitoring campaigns are highly recommended. As shown within this work, pharmaceutical residues offer multiple evaluation possibilities with regard to anthropogenic contamination due to their unique origin. Other markers for sewage-born anthropogenic impacts may be cheaper to analyze, but mostly the combination of different substances gives distinct evidence of a contamination source. For example coliform bacteria might as well originate from livestock farming or nitrate and salts may bear geogenic backgrounds.

In the future, other matters related to pharmaceuticals might become of interest, like the upcoming issue of antibiotic residuals in wastewater treatment plant effluents. Antibiotic resistances are a well-known problem in medicine and spread from hospital wastewater to treatment plants and further to the aquatic environment where they could already be detected in agricultural soils. Residual endocrine disrupters from birth control pills are under suspicion to be responsible for mutations observed in male fish for years. The release of pharmaceuticals into the environment is recently observed more critically and restrictions in effluent concentrations are already being discussed. As a consequence, decision makers are starting to include these substances in their environmental

monitoring concepts. Regarding the release of pharmaceuticals into the environment, the EU started in 2013 to add three substances (2 hormones and diclofenac) to the watch list of their water framework directive. Substances on this list are monitored and their impact on the aquatic environment is documented. Three antibiotics were added in 2015 and further measures are delayed until 2017. Australia implemented thresholds for pharmaceuticals in groundwater recharge and drinking water augmentation. The public awareness is nowadays increasing towards all kinds of contaminants, including pharmaceuticals. As of late, environmental politics are going towards precautionary principles like the restricted usage of pesticides and pharmaceuticals. Such principles might eventually substitute the end-of-pipe treatment that is actually practiced in wastewater treatment and might even lead towards a more sustainable management of pharmaceutical substances in the future.

Declaration of authorship

Chapter 1.6

Citation:

Zemann M. (2008): Artificial recharge tests at an infiltration basin in the Wadi Kafrein, Jordan. Diploma thesis at the University of Karlsruhe (TH), Institute for Applied Geology (AGK), 126 p., unpublished.

Zemann, M., Wolf, L., Hassan, J., Schmidt, N., Zawadsky, C., Tiehm, A., Abassi, B. (2013): Assessment of groundwater risk by treated wastewater. Deliverable No. 503 within the IWRM SMART-Project, XX p., available at http://www.iwrm-smart2.org.

Declaration of authorship:

The description of the geology and hydrogeology of the study site was adapted according to text fragments which were originally published in a project report (Deliverable No. 503) for the SMART project and the diploma thesis of Moritz Zemann. The diploma thesis was solely written by Moritz Zemann. The complete deliverable 503 was written by several authors whereas the used text passages were solely written by Moritz Zemann.

Chapter 2:

Citation:

Zemann, M., Wolf, L., Pöschko, A., Schmidt, N., Sawarieh, A., Seder, N., Tiehm, A., Hötzl, H., Goldscheider, N., 2014: Sources and processes affecting the spatiotemporal distribution of pharmaceuticals and X-ray contrast media in the water resources of the Lower Jordan Valley, Jordan. Science of the Total Environment 488-489, 100-114.

Declaration of authorship:

The sampling concept was designed together with Leif Wolf. Site location in the Jordan Valley was according to former works of Antje Pöschko. Moritz Zemann conducted the four monitoring campaigns. Organizational help was provided from Ali Sawarieh and

Nayef Seder, regarding language problems and the selection of the sampling locations. All samples were taken by Moritz Zemann who as well conducted the in situ measurements. Help was provided for analyses of anion and coliforms which were conducted at the WAJ laboratories of the MWI in Jordan. Cation analyses were conducted at the AGW laboratories. The complete pharmaceutical analyses were conducted from the Water Technology Centre (TZW) in Karlsruhe. Statistical analyses and data evaluation was solely conducted by Moritz Zemann. Small parts of the dataset (spring data from 2008) were adopted from the diploma thesis of Antje Pöschko. GIS data as background for the maps were provided from the MWI. The results were discussed with Leif Wolf. Moritz Zemann wrote almost the complete manuscript. Parts of chapter 2.4.6. (Shifting application patterns) were written by Leif Wolf. Figure 13 was provided by Antje Pöschko. The final manuscript was reviewed by all authors.

Chapter 3:

Citation:

Zemann, M., L. Wolf, F. Grimmeisen, A. Tiehm, J. Klinger, H. Hötzl and N. Goldscheider (2015): Tracking changing X-ray contrast media application to an urban-influenced karst aquifer in the Wadi Shueib, Jordan. Environmental Pollution **198**: 133-143.

Declaration of authorship:

Analyses and field work were conducted in the same way as for paper 1. In addition, 2 single sampling events at Hazzir spring were conducted from Felix Grimmeisen and analyzed at the same facilities as above. Statistical analyses and data evaluation was solely conducted by Moritz Zemann. GIS data as background for the maps were provided from the Ministry of Water and Irrigation (MWI) in Jordan. The results and the conceptual model were mainly discussed with Leif Wolf. The chapter on geology was discussed with Felix Grimmeisen. Moritz Zemann wrote the complete manuscript. Only Figure 18 was contributed from Felix Grimmeisen. The final manuscript was reviewed by all authors.

Chapter 4:

Citation:

Zemann M., Majewsky M., Wolf L (2016): Accumulation of Pharmaceuticals in Groundwater under Arid Climate Conditions – Results from Unsaturated Column Experiments. Chemosphere **154**, pp 463-471.

Declaration of authorship:

Moritz Zemann designed the experimental setup together with Leif Wolf. The columns were constructed from the AGW workshop. Moritz Zemann conducted the laboratory experiments. Help was provided for sampling and maintenance of the experiment by a student co-worker. Ion analyses were conducted from Moritz Zemann under the guidance of AGW laboratory staff. Pharmaceutical analyses were conducted at the EBI laboratories. Pycnometer tests were conducted from AGW laboratory staff. Ignition loss, dry weight and elution of the soil were conducted by Moritz Zemann. Data evaluation was conducted by Moritz Zemann. The results were discussed with Marius Majewski and Leif Wolf. The manuscript was almost completely written by Moritz Zemann. Only parts of chapter 4.2.3 were contributed by Marius Majewski. The final manuscript was reviewed by all authors.