Integrated Evaluation of Wastewater Irrigation for Sustainable Agriculture and Groundwater Development

PhD in Integrated Management of Water, Soil and Waste

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Technische Universität Dresden
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Dissertation

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Place:	Date:
Dresden, Germany	18th January 2019

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Acknowledgements

The PhD research work would not be possible without the support of my eminent supervisors Dr. Stephan Hülsmann (UNU-FLORES, Germany) and Prof. Dr. Rudolf Liedl (TU Dresden, Germany). Their tremendous support, critical advice and constructive criticism motivated me throughout the PhD research period. Our research meetings definitely helped me to prepare the publications for high impact factor scientific journals. I am highly thankful to Dr. Priyanie Amerasinghe (IWMI, Sri Lanka) for her unconditional support in terms of data and scientific thinking during the PhD research. I am also thankful to Dr. Sahebrao Sonkamble (CSIR-NGRI, India) and Ms. Karla Locher (UNU-FLORES) for our fruitful scientific discussions. I would like to thank Dr. Narakula Srinivasa Rao (CSIR-NGRI) for his assistance in multi-way modelling. Part of the research data used in the thesis work is collected under the project "Enhancement of natural water systems and treatment methods for safe and sustainable water supply in India - Saph Pani" funded by the European Commission within the Seventh Framework Programme (grant agreement number 282911). I am thankful to UNU-FLORES and TU Dresden Graduate Academy for their funding support.

I would like to express my gratitude to Mr. Mohamad Haroun, Mr. Arne-Hendrik Klass and Ms. Eunjoo Lee for updating and installing the required software at UNU-FLORES. I am very thankful to Dr. Reza Ardakanian, Mr. Benjamin Zhu, Mr. Rafael Hernandez, Ms. Julie Coulombe, Ms. Luisa Arndt, Mr. Mohamad Haroun and Mr. Bastian Nebel for all the logistical support at UNU-FLORES. I am also thankful to Dr. Mathew Kurian and Prof. Dr. Hiroshan Hettiarachchi for the motivation and great discussions. I am thankful to all my colleagues at UNU-FLORES, with whom I shared culture, food and great scientific discussions in the international work environment. I am also highly indebted to my friends Thuy, Ramakrishna, Solomon, Agossou and Patra for the financial support during the PhD research period. I am especially thankful to my PhD colleagues and friends, with whom I cherished great memories in the last four years.

I am grateful for the endless support by my parents Annapurna and Nageswararao, my aunt Sadasiva Lakshmi and my brothers Harish and Vishnu. I am thankful to all the friends who helped me and took good care of me in Dresden. Finally, I express my gratitude to all friends and family members for being wonderful all along.

General Abstract

Many agricultural landscapes in India are irrigated with wastewater, and it is a common livelihood practice particularly in urban and peri-urban areas. Farmers around urban agglomerations continuously depend on the wastewater released from nearby urban centres. While providing opportunities with respect to water and nutrient supply, irrigating with wastewater has adverse environmental impacts, particularly on the local aquifer systems. Therefore, addressing the wastewater irrigation influence on local aquifer systems is crucial for sustainable groundwater management. The present research demonstrates the impacts of wastewater irrigation, seasonality and spatio-temporal variations in the groundwater quality and its geochemical evolution and mixing processes in different land use and crop settings. The doctoral research aims at understanding the aquifer heterogeneity, land use conditions, groundwater dynamics and contaminant fate and transport in the long-term wastewater irrigation system to develop sustainable and suitable groundwater management strategies. The selected study watershed is located on the banks of Musi River in a peri-urban catchment of the Musi River basin in India. Statistical techniques, land use change modelling and solute flow and transport modelling tools are employed to identify and quantify the linkages between contaminants, agricultural use and environmental variables, particularly those characterizing the groundwater qualities. The research results suggest that concentrations of the major ionic substances increase after the monsoon season, especially in wastewater irrigated areas and the major polluted groundwaters to come from the wastewater irrigated parts of the watershed. Clusters of chemical variables identified indicate that groundwater pollution is highly impacted by mineral interactions and long-term wastewater irrigation. The groundwater geochemistry of the watershed is largely controlled by long-term wastewater irrigation, local rainfall patterns and water-rock interactions. The detected land use changes in the watershed indicate that, as a consequence of urban pressures, agricultural landscapes are being converted into built-up areas and, at the same time, former barren land is converted to agricultural plots. The mapped land use data are used in modelling the aquifer conditions and to observe the groundwater dynamics in the peri-urban environment. The study results provide the basis for sustainable agriculture and groundwater development using the efficient scenarios identified for wastewater irrigation management. The resulting strategies for integrated management of water and waste will contribute to the water security and achieve the respective Sustainable Development Goals (SDGs 2, 3, 6, 11 and 15).

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List of Symbols and Abbreviations

%	Percentage
~	Approximately
>	Greater than
°	Degrees
°C	Degree Centigrade
±	Plus or Minus
mm	Millimetres
m	Meters
km	Kilometres
km^2	Square kilometres
m/s	Meter per second
m/day	Meter per day
m^2/s	Meter square per second
mm/yr	Millimetre per year
m^3	Cubic meters
MCM	Million Cubic Meters
MGD	Million Gallons per Day
kg/ha	Kilogram per hectare
amsl	Above mean sea level
bgl	Below ground level
µS/cm	Micro Siemens per Centimetre
meq/L	Milliequivalents per Litre
mg/L	Milligrams per Litre
Δi	Ionic Delta, difference between measured and conservative mixing
aParagrass	Actual water abstractions for paragrass crop
approx.	Approximately
aRice	Actual water abstractions for Rice crop
aVegetables	Actual water abstractions for vegetables
Cl_s	Chloride concentration for groundwater sample
Cl_f	Chloride concentration for freshwater sample
Cl_{ww}	Chloride concentration for wastewater sample
F_{ww}	Wastewater fraction
ET	Evapotranspiration
R^2	Coefficient of determination
i_s	Ionic concentration of the groundwater sample
i_m	Ionic mixing concentration
i_{ww}	Ionic concentrations of wastewater

i_f	Ionic concentrations of freshwater
T	Transmissivity
K	Hydraulic conductivity (chapter 5)
K	Solubility product (chapter 3)
K1/Ks1	Groundwater sample location in the watershed
Log	Logarithmic
PET	Potential evapotranspiration
rParagrass	Crop water requirements for paragrass crop
rRice	Crop water requirements for Rice crop
rVegetables	Crop water requirements for vegetables
vs	Versus
W_{11}, W_{12}	Scenarios for management strategies
$AgNO_3$	Silver nitrate
Ca	Calcium
Cl	Chloride
DO	Dissolved Oxygen
EC	Electrical Conductivity
EDTA	Ethylenediaminetetraacetic acid
E. coli	Escherichia coli
F	Fluoride
H_2SO_4	Sulfuric acid
HCO_3	Bicarbonate
K	Potassium
Mg	Magnesium
Na	Sodium
NO_3	Nitrate
PO_4	Phosphate
SO_4	Sulphate
TDS	Total Dissolved Solids
$CaCO_3$	Calcite mineral composition
CaF_2	Fluorite mineral composition
$CaMg(CO_3)_2$	Dolomite mineral composition
$CaSO_4.2H_2O$	Gypsum mineral composition
NaCl	Halite mineral composition
APHA	American Public Health Association
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
BAU	Business as Usual
BIS	Bureau of Indian Standards
CAI	Chloroalkaline Index

CPCB	Central Pollution Control Board
DEM	Digital Elevation Model
FA	Factor Analysis
FAO	Food and Agricultural Organization
GIS	Geographical Information System
GPS	Global Positioning System
GW	Groundwater
HCA	Hierarchical Cluster Analysis
HDPE	High-Density Polyethylene
IAP	Ion Activity Product
KML	Keyhole Markup Language
KSMWS	Kachiwani Singaram Micro-Watershed
Landsat	Land Remote Sensing Satellite (System)
LCM	Land Change Modeler
MAR	Managed Aquifer Recharge
Max	Maximum
Min	Minimum
MODFLOW	Modular Three-Dimensional Finite-Difference Groundwater Flow
MODIS	Moderate Resolution Imaging Spectroradiometer
MLP	Multilayer Perceptron
MT3DMS	Modular Transport Three Dimensional Multi-Species
PALSAR	Phased Array type L-band Synthetic Aperture Radar
PARAFAC	Parallel Factor Analysis
RMS	Root Mean Square
SAT	Soil Aquifer Treatment
SAR	Sodium Absorption Ratio
SD	Standard Deviation
SDGs	Sustainable Development Goals
SEE	Standard Error of the Estimate
SI	Saturation Index
SimWeight	Logistical regression and simulation weighting
SOM	Self-Organizing Maps
UN	United Nations
UPA	Urban and Peri-urban Agriculture
UTM	Universal Transverse Mercator
WGS	World Geodetic System
WHO	World Health Organization
WW	Wastewater

1 General Introduction and Study Background

1.1 General Overview

Water demand is increasing at a higher rate than the population growth (Oki and Kanae, 2006) and there is a critical need to reappraise perspectives on urban water and wastewater management. Globally, around 20 million hectares of agricultural area is under wastewater irrigation (Hamilton et al., 2007). In the developing world, especially in South Asia wastewater irrigation is a common livelihood practice. Farmers around the urban agglomerations continuously depend on the wastewater released from nearby urban centres (Qadir et al., 2010), which is untreated or partially treated or diluted wastewater and also sometimes it is a mixture of industrial wastewater (Amerasinghe et al., 2009). The scant data on wastewater generation, treatment and reuse in the developing world makes it difficult to understand the fluxes and processes in peri-urban systems (Minhas and Samra, 2004). There is a need to take advantage of emerging scientific methods for evaluating the wastewater reuse options for sustainable crop production and groundwater management (Minhas and Samra, 2004; Qadir et al., 2010). Considering the interlinkages of resources – and looking at them in a balanced way – is an approach increasingly referred to as the Nexus Approach (Hettiarachchi and Ardakanian, 2016a), suited to overcome the vicious cycle of resources overexploitation, degradation, and pollution (Lal, 2015; Schwärzel et al., 2016). In recent years, the nexus approach is increasingly getting attention for assessing and managing the environmental resources in an integrated manner. Especially resources management in wastewater irrigation systems is a good example for water-soil-waste nexus approach (Figure S1.1). Good practice examples of wastewater irrigation for intensification of agricultural production have been documented (Hettiarachchi and Ardakanian, 2016b), but so far little attention has been paid to understand the potential impacts on groundwater.

In many parts of the world, there has been recognition of benefits associated with wastewater for irrigation (Haruvy, 2006; Levine and Asano, 2004). Farmers depend on wastewater for irrigation because of its nutrient benefits or they have less water resource competition with other sectors (e.g. industry and urban). WHO (2006) developed wastewater safe use guidelines to minimize health risks, but the environmental risks and concentration pathways in wastewater irrigated systems are often unclear. Irrespective of these environmental effects farmers are choosing suitable crops based on their economic viability (Biggs and Jiang, 2009; McCartney et al., 2009), typically receiving little or no support for selecting the most suitable crops for wastewater irrigation. They are often completely unaware of further environmental or health problems involved (Drechsel et al., 2010). When economic constraints limit the treatment of wastewater, a sustainable crop choice or improved wastewater quality should be selected to limit health problems.

The transport of chemicals and toxic elements in the groundwater system is poorly understood in wastewater irrigation systems. This type of irrigation systems has adverse environmental impacts on local aquifer systems in the long run (Rattan et al., 2005). Several authors reported that the influence of wastewater irrigation has adverse impacts on groundwater and changes the chemical characteristics of the aquifer (Candela et al., 2007; Gallegos et al., 1999; Kass et al., 2005; Rattan et al., 2005). In developing countries, treating the wastewater to quality requirements for irrigation is nearly impossible because of social and economic conditions. Developing sustainable management strategies based on local conditions for groundwater development is clearly an option to conserve the environment.

1.2 Statement of the Research Problem

In general, environmental problems associated with the wastewater irrigated systems in South Asia and Africa are very similar because of their inadequate infrastructure for wastewater treatment. Using the untreated wastewater for irrigation provides opportunities for farmers in peri-urban areas to benefit from water and nutrient resources. At the same time, this practice increases the health and environmental concerns. The Musi River case study represents one such wastewater system located in Southern India. A number of research studies on the Musi River basin have dealt with wastewater and health risks (Ensink et al., 2008), hydrology and water quality (Amerasinghe et al., 2015; Mahesh et al., 2015b; Perrin et al., 2010), salinity implications (Biggs and Jiang, 2009; McCartney et al., 2009), livelihood opportunities and socio-economics of the wastewater irrigation (Buechler and Devi, 2003; Mahesh et al., 2015b). However, investigations related to the impact of wastewater irrigation on groundwater and the natural processes that attenuate contamination in the peri-urban areas of the Musi River basin have not been undertaken. Also, the transport of chemicals and toxic elements in irrigation water-groundwater continuum is poorly understood in these wastewater irrigation systems.

Wastewater used for irrigation may not have the same quality over the years in the Musi River basin, it changed with the Hyderabad city dynamics and may be influenced by several factors including industrial pollutant loads, treatment level of the urban treatment plants, climate, rainfall, stormwater runoff etc. over various time spans. The rate of wastewater application for plant growth depends on crop choice and agronomic factors. These factors have a definite impact on the groundwater quality over different seasons in a hydrological year. Earlier studies examined how wastewater irrigation of certain food and fodder crops such as paddy rice, paragrass and vegetables alters the local aquifer properties and impacts the groundwater quality (Biggs and Jiang, 2009; Rattan et al., 2005). The long-term wastewater irrigation makes local aquifers unsuitable for domestic consumption in these peri-urban systems. Understanding and predicting the contaminant fate and solute transport in wastewater irrigated systems of irrigation water-groundwater continuum can improve the management of the peri-urban environment.

When considering the sustainability and environmental conservation for groundwater development, a series of research questions need to be answered for assessing the wastewater irrigation systems influence on local aquifers in the Musi River basin, especially in the peri-urban systems of Hyderabad city.

1. What are the seasonal dynamics and spatio-temporal variability of groundwater quality influenced by wastewater irrigation?
2. What are the geochemical and mixing processes that control the groundwater chemistry in wastewater irrigated areas?
3. What are the land use shifts and agricultural cropping area dynamics in the peri-urban environment influenced by wastewater irrigation?
4. How does the intensive land use shifts in the peri-urban system influence the groundwater recharge dynamics?
5. What are the fate and transport of contaminants in the local aquifer with the wastewater influence and how to control the pollution and manage the groundwater resources sustainably?

In order to understand these hydrogeological and contaminant processes in the wastewater irrigation systems, the proposed research aims at evaluating the wastewater irrigation impacts on groundwater quality with the change in wastewater quality, choice of crop and seasonality.

1.3 Objectives of the Research

The overarching goal of the PhD research is to develop sustainable irrigation water management strategies which would ultimately enhance groundwater quality in the wastewater irrigated peri-urban agricultural systems. To assess groundwater quality influences and generate different sustainable and suitable scenarios for groundwater development (based on statistical analysis, land use change analysis and groundwater modelling), the following sub-objectives shall be addressed:

1. To evaluate the spatio-temporal variability and chemical parameter distribution in groundwater with the wastewater irrigation influence.
2. To evaluate the hydrogeochemical and groundwater mixing processes influenced by wastewater irrigation.
3. To observe the temporal trends of micro-level land use changes and evaluate the irrigated areas and cropping patterns in the watershed.
4. To elucidate the groundwater recharge dynamics in relation to the change in land use patterns and irrigation practice.
5. To assess the contaminant fate and transport processes in the local aquifer under different wastewater application rates and qualities used for irrigation and to develop potential agricultural interventions and groundwater development strategies under irrigation management.

1.4 Situation at Hyderabad City and Musi River

The present case study, Hyderabad city and its associated wastewater agriculture conditions are very similar to many other urban agglomerations situation across South Asia. Due to the continuously increasing urban water demands of Hyderabad city, more than 90% of the water supply is supplemented from other river basins into the Musi River basin (Figure 1.1).

With the increase in water use, wastewater generated from the city has also increased largely over the years. These large volumes of wastewater, both untreated and partially treated from the city of Hyderabad are discharged into the Musi River and utilised by people who live downstream of the Hyderabad city (Figure 1.1). At present, around 500 MCM per year of partially treated or untreated wastewater is being discharged into the Musi River from Hyderabad city (Mahesh et al., 2015). With the increase in wastewater availability in the Musi River, the agriculture area has vastly increased over the years.

Wastewater irrigation became a livelihood practice due to the year-round water availability in the Musi River (Amerasinghe et al., 2009). Agriculture production takes place along the banks of the river and water is channelled via irrigation canals or lifted via irrigation pumps (Figure 1.2). The concentrations of chemical compounds within wastewater used for irrigation have adverse impacts on soil salinization (Biggs and Jiang, 2009) and also lead to groundwater contamination with salinity, nitrates and microbes (Amerasinghe et al., 2009). Even though groundwater is contaminated in these wastewater irrigated systems, local people around these systems are still dependent on the groundwater for domestic purposes, which increases health risks (Buechler et al., 2002; Drechsel et al., 2010).

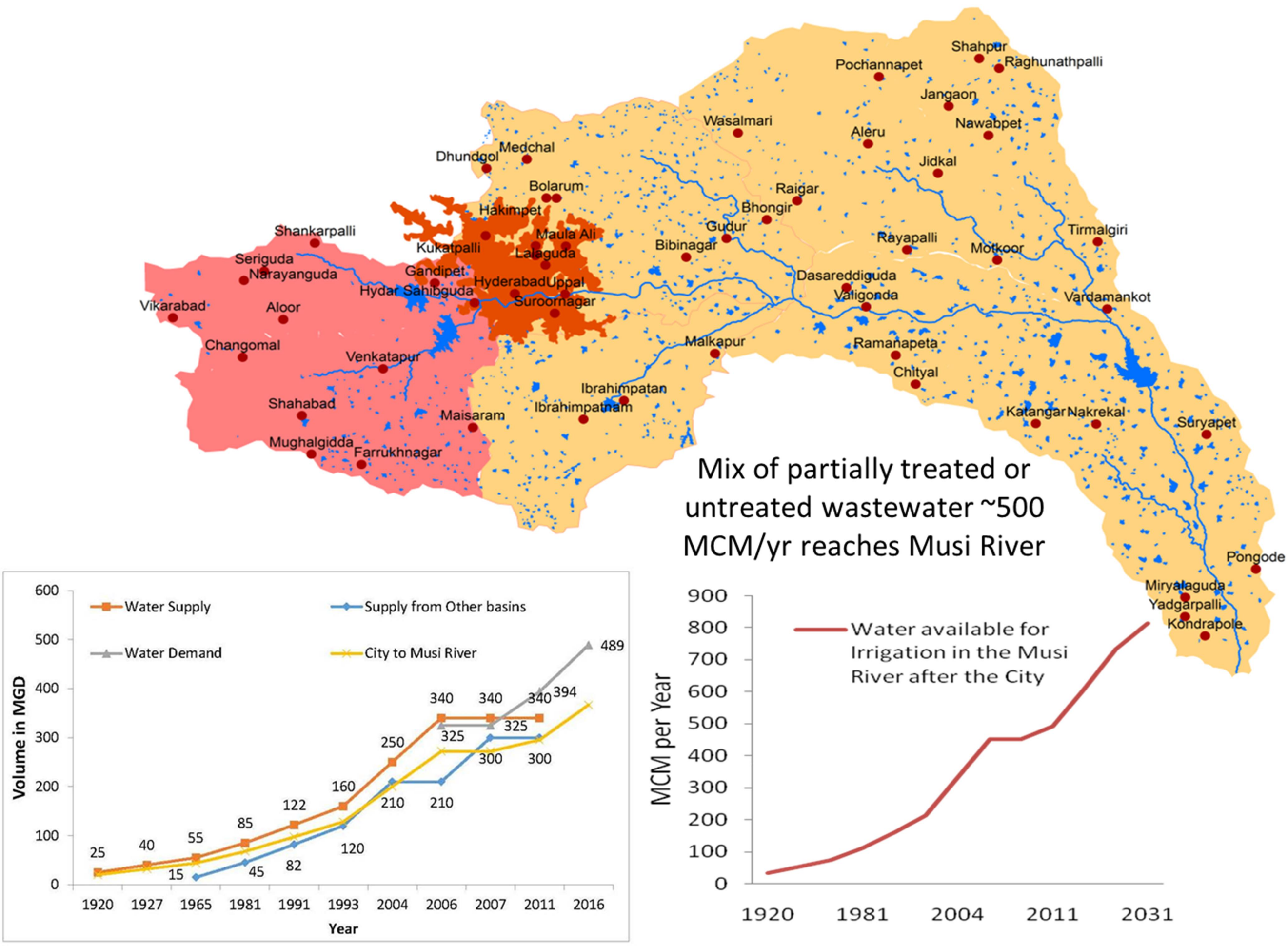

Figure 1.1 Overview of the water flows in the Musi River basin. Magenta - freshwater area; Brick red - Hyderabad Urban Agglomeration; Yellow – influenced by urban wastewater with predominant agriculture; Blue polygons are water reservoirs. MCM – million cubic meters; MGD – million gallons per day. The graphs show Hyderabad city's water demand, total supply, supply from other river basins and city wastewater into Musi River and also year-wise wastewater available for irrigation (more than 90% of city's water supply come from other basins into Musi River basin); modified from Amerasinghe et al., 2015 and Mahesh et al., 2015.

To deal with the pollution, we have to manage groundwater resources in a holistic manner in the wastewater irrigated systems. An innovative research approach is needed for a detailed understanding of the pollutant linkages and interactions within the system, which will further help to develop sustainable management options for groundwater development.

Figure 1.2 Wastewater flows and irrigation infrastructure in the peri-urban node. From top left to right: Wastewater canal refill; Irrigated paddy rice using wastewater from the canal; Lift irrigation pumps from canal to upstream irrigated area; Wastewater attributed flows into the Musi River after the city. Modified from Mahesh et al., 2015a.

1.5 Study Area Description

Wastewater irrigated agriculture is a significant resource for many inhabitants in the semi-arid peri-urban environment of Hyderabad urban agglomeration The present delineated study area is a peri-urban micro-watershed located in and around the village of Kachiwani Singaram (KSMWS) of the Musi River basin, next to the Hyderabad city and comprises an area of 2.74 km^2 (Figure 1.3). The location map and land use classification of the study area with agriculture and peri-urban built-up areas illustrated in Figure 1.3 (DEM, Figure S1.2).

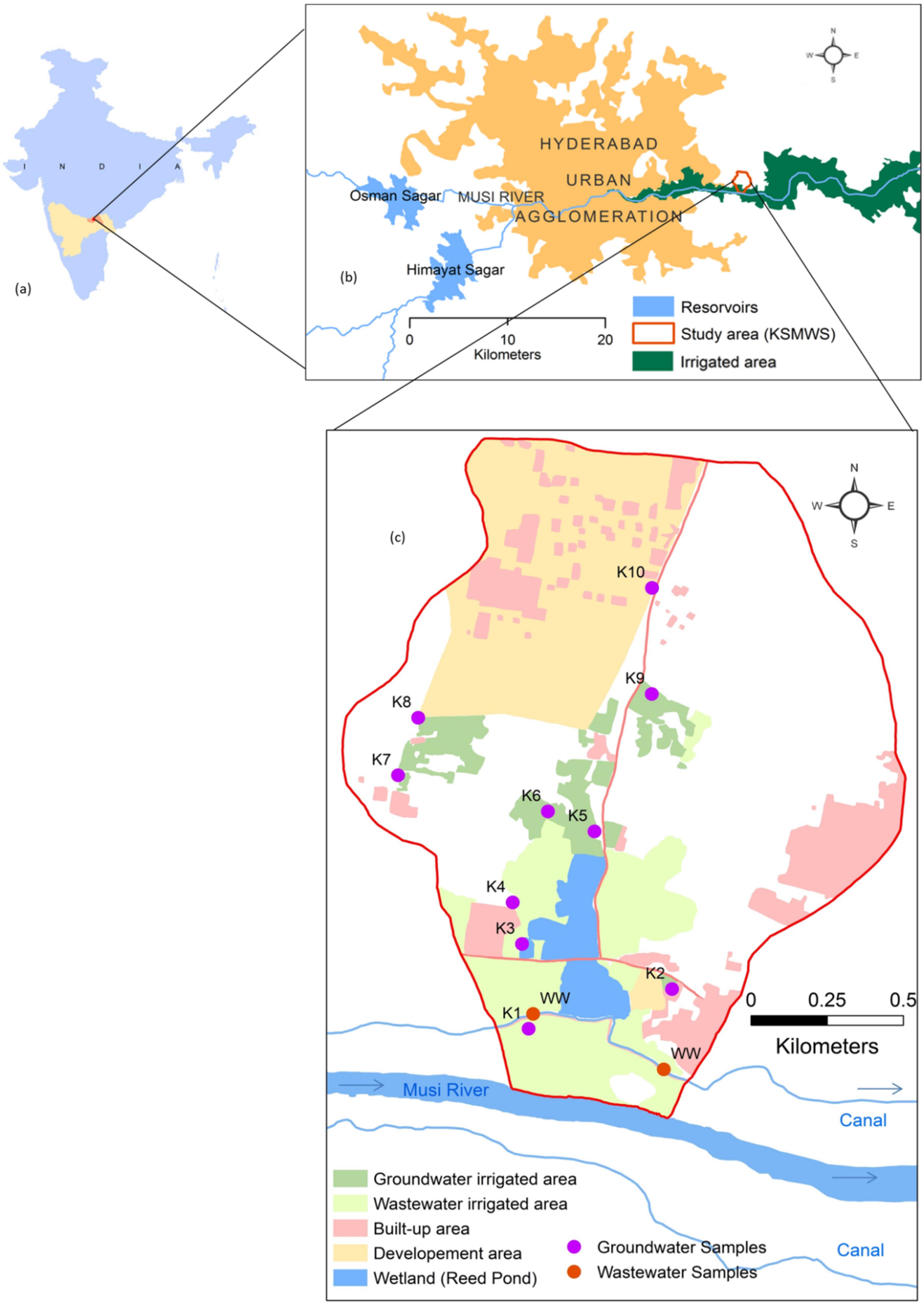

Figure 1.3 Location map: study area (a) in India, (b) next to Hyderabad urban agglomeration, (c) watershed with sampling locations (groundwater and wastewater) and background land use information (groundwater and wastewater irrigated areas with peri-urban built-up areas); modified from Jampani et al., 2018.

1.5.1 Climate and hydrological conditions

Musi River is a tributary of the Krishna River, and it constitutes approximately 4% of the total area of Krishna River basin in Southern India. The river passes through the Hyderabad city, as the city developed around the Musi River.

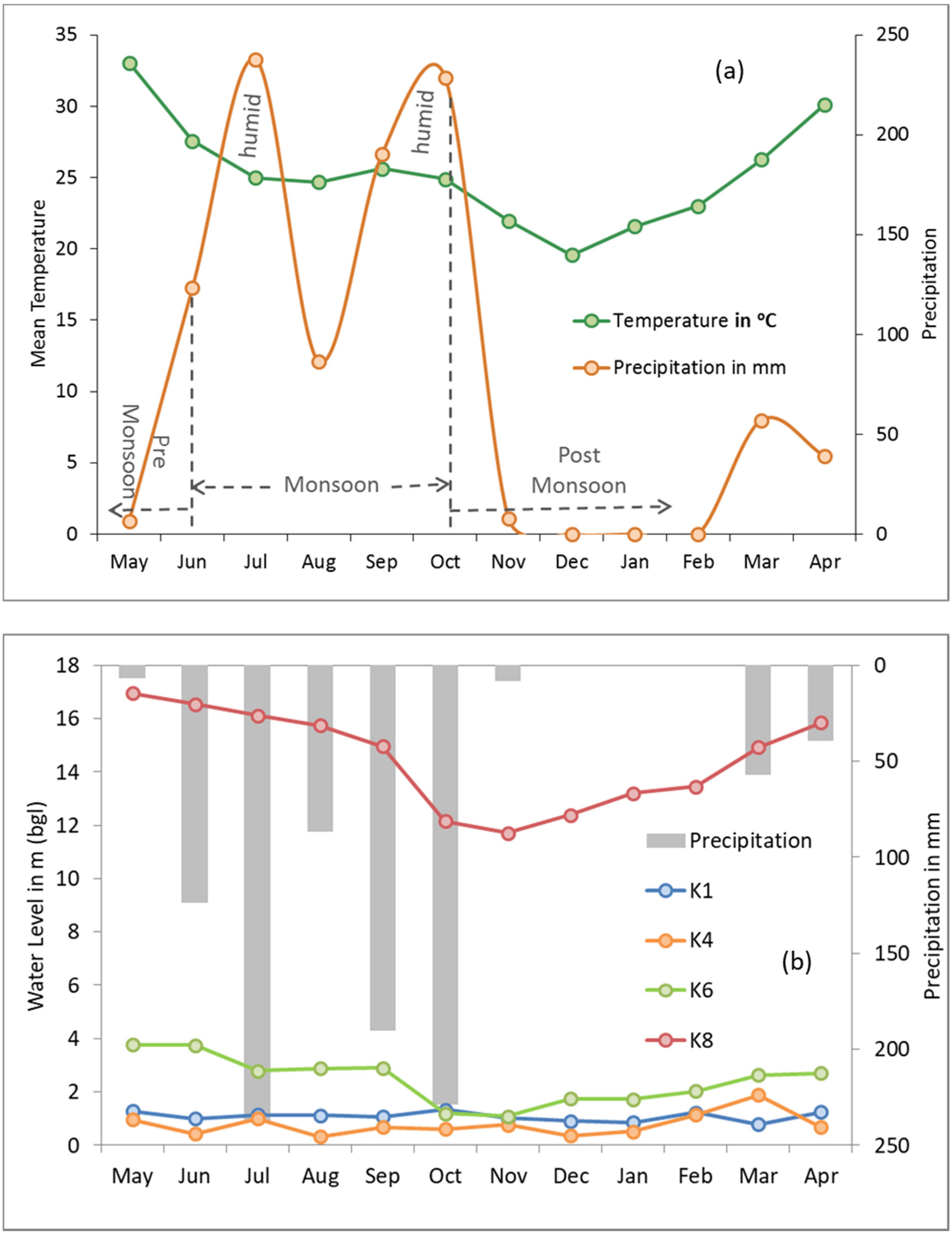

Figure 1.4 Seasonal development of precipitation and temperature (hydrological year: 2013 to 2014) (a) ombrothermic diagram and (b) groundwater level variation in the piezometers of the watershed with respect to precipitation.

Before entering the urban area, freshwater amounts were withdrawn from the Musi River into Hussain Sagar and Osman Sagar reservoirs (Figure 1.3b), which are the main drinking water sources for the Hyderabad city. After the freshwater withdrawal, there is a negligible amount of water flow in the Musi River (almost runs dry except monsoon season), which makes the river flow ephemeral. On its way through the city, partially treated (~40%) and untreated (~60%) wastewater flows from the Hyderabad city is making up the Musi River flow (Mahesh et al., 2015b), which peri-urban farmers withdraw for irrigation. Currently, an estimated 1.2 million m^3/day of wastewater (mix of domestic and industrial) is funnelled into the Musi river (Amerasinghe et al., 2015).

Musi River, which passes through the Hyderabad city is unique and characterizes the river basin into three large segments. In the upstream, it is a source of drinking water supply and at the Hyderabad city collects the wastewater discharges, and in the downstream, the collected wastewater is used for agriculture (Figure 1.1 and Figure S1.3). In the peri-urban systems of Hyderabad city, built-up areas such as villages and irrigated areas (wastewater and groundwater) co-exist together. The study area has a semi-arid climate with average annual rainfall of 750 mm and the rain events occur in the monsoon season between June to October (Figure 1.4). The mean annual temperature is about 27 °C, although during the summer months the maximum temperature can reach up to 47 °C.

1.5.2 Land use and hydrogeology of the study watershed

Upper parts of the watershed consist of barren land, peri-urban development and built-up areas. Overall, the study watershed is a characteristic of multi-functional land use system, a mixture of agriculture, wetland, small-scale industry, barren and built-up areas, which is representative for several peri-urban areas in India, Asia and elsewhere (Binns et al., 2003; Ensink et al., 2002; Graefe et al., 2008; Keraita et al., 2003; Khai et al., 2007; Murray and Ray, 2010). The watershed is located on orthogneissic granite basement with quartz and dolerite intrusions and granite as the main lithology (Dewandel et al., 2008). The unconfined crystalline hard rock aquifer system of the watershed includes a distinctive weathering profile with a saprolite layer of 10 to 15 m thickness above a fissured layer of 15 to 20 m thickness, which is underlain by a fresh unfissured granitic basement (Mahesh et al., 2015b). The saprolite layer is the main groundwater storage zone because of its low permeability and high porosity. The fissured zone consists of a horizontal fissuring network, which provides main transmissive functions of the aquifer (Figure S1.4). Aquifer material is made up of the fractured and weathered layer with fresh crystalline bedrock in the basement (Amerasinghe et al., 2015; Mahesh et al., 2015b). With the year-round cultivation, large return flows are generated from irrigated agricultural fields and contribute to groundwater recharge in the watershed (Dewandel et al., 2008; Mahesh et al., 2015b; Perrin et al., 2010).

1.5.3 Irrigation conditions

The partially treated or untreated wastewater is funnelled from the Hyderabad urban agglomeration into the Musi River and serves as the potential irrigation supply source for the downstream peri-urban farmers. Wastewater available in the Musi River diverted to a series of irrigation canals for agriculture in the downstream of Hyderabad city (Ensink et al., 2009). Land use patterns of peri-urban Hyderabad are complex and mostly influenced by urban drivers, livelihood practices, wastewater availability and socio-economic conditions (Amerasinghe et al., 2012; Mahesh et al., 2015b). The current experimental watershed shows the unique characteristics of typical watersheds in the wastewater irrigated systems of the developing world (Perrin et al., 2010). The major crop types irrigated in the Musi River system are paragrass or fodder grass (*Urochloa mutica L.*), which is a perennial crop throughout the year, paddy rice (*Oryza sativa L.*), which is irrigated approximately 300 days a year and vegetables, which are also irrigated throughout the year. Cultivation of these crops provides economic benefits to the local farmers. Upper parts of the watershed irrigated with groundwater and the lower parts of the agricultural areas irrigated with wastewater from the canal. Nearly 74% of the cultivable land is irrigated with wastewater lifted from the Musi River, and the rest is irrigated with groundwater (Mahesh et al., 2015b). Shallow groundwater is pumped for irrigation in areas where canal water is not accessible, or according to farmers, too polluted for certain crops, especially paddy rice (Starkl et al., 2015).

1.6 Research Flow and Chapter Descriptions

To achieve the overarching goal of the PhD research work, the following chapters are arranged according to the numbered research questions and objectives mentioned in the above sections (1.3 and 1.5).

Objective 1: Chapter 2 analyzes the seasonality, chemical characterization and spatio-temporal distribution of the groundwater resources in the watershed to understand the groundwater conditions in the aquifer influenced by wastewater irrigation.

Objective 2: Chapter 3 analyzes the hydrogeochemical signatures and groundwater mixing processes in the watershed to understand the amount of long-term wastewater irrigation contribution to the aquifer geochemistry.

Objective 3: Chapter 4 analyzes the temporal shifts and micro-level land use changes in the watershed to assess the peri-urban agriculture dynamics influenced by urban pressures and wastewater availability.

Objective 4: Chapter 5 analyzes the groundwater recharge dynamics based on spatio-temporal land use change patterns in the peri-urban environment to assess the aquifer recharge conditions influenced by intensive wastewater irrigation.

Objective 5: Chapter 6 analyzes the groundwater contaminant transport processes in the watershed and generate efficient scenarios to produce sustainable and suitable groundwater development strategies under irrigation water management.

Chapter 7 explains the overarching conclusions of the PhD research with policy recommendations based on the research analysis results from Chapters 2 to 6. Chapter 7 is followed by the corresponding references of the thesis and supplementary material with extra figures and tables of the current research work.

2 Spatio-temporal Distribution and Chemical Characterization of Groundwater Quality

This chapter is a modified version of the published paper: *Mahesh Jampani, Stephan Huelsmann, Rudolf Liedl, Sahebrao Sonkamble, Shakeel Ahmed, Priyanie Amerasinghe (2018). 'Spatio-temporal Distribution and Chemical Characterization of Groundwater Quality of a Wastewater Irrigated System: A Case Study'. Science of the Total Environment (Elsevier). Vol. 636. pp. 1089-1098.*

2.1 Introduction

Many people around the world depend on wastewater for irrigation, as it has become a common livelihood practice especially among poor urban and peri-urban dwellers (Raschid-Sally and Jayakody, 2008). In the developing world, the water used for irrigation is mostly withdrawn from rivers and lakes, which consists to a large extent of wastewater that has been either partially treated or remained untreated (Hamilton et al., 2007; Qadir et al., 2010). The scarcity of alternative or cost-effective water resources pushes farmers to completely depend on wastewater for irrigation (Kurian et al., 2013). On the one hand, non- or insufficiently treated wastewater for irrigation provides opportunities for local farmers in peri-urban areas to profit from a continuous water supply and rich nutrient resources (Buechler et al., 2002). At the same time, this type of irrigation practice increases health and environmental concerns (Amerasinghe et al., 2009). Even though the World Health Organization (WHO, 2006) has developed guidelines and safe practices for using wastewater for irrigation, they are often not followed or unknown to farmers (Amerasinghe et al., 2013a; Qadir et al., 2010).

The use of wastewater for increasing soil productivity and enhancing water use efficiency has been identified as a suitable measure towards sustainable resources management strategies (Kurian and Ardakanian, 2015). Understanding the impacts of wastewater irrigation on groundwater would be a critical step for re-thinking and designing sustainable wastewater reuse within agroecological systems via suitable combinations of crops, crop performance assessment, integration of informal wastewater schemes into partnerships etc. (Weckenbrock and Alabaster, 2015).

Several authors have reported that with the increase in the use of wastewater for irrigation, local groundwater aquifers become highly susceptible to contamination with heavy metals, nutrients and pathogens (Candela et al., 2007; Gallegos et al., 1999). When the soils accumulate salts during long-term irrigation, these salts may eventually be flushed into the local aquifers, which may increase the groundwater salinization. Dominant contaminants in the groundwater of wastewater irrigated systems include organic and inorganic chemical compounds (nitrates, phosphates, chlorides, sulphates, organochloro pesticides etc.), heavy metals (cadmium, chromium etc.), helminths (*Ascaris spp*, etc.), protozoan parasites (*Giardia* spp.) and microbial contaminants (*E. coli* and other coliform bacteria, etc.) (Amerasinghe et al., 2009; Gallegos et al., 1999). The long-term wastewater irrigation makes local aquifers unsuitable for domestic consumption, especially in the peri-urban systems (Drechsel et al., 2010). Understanding the chemical constituent dynamics in these irrigation water-groundwater systems can help to improve the peri-urban environments, especially for agriculture, which is very common in developing countries.

Ecological and environmental problems associated with wastewater irrigated systems in Asia and Africa are very similar, because of inadequate infrastructure for necessary wastewater treatment (Amerasinghe et al., 2013). Musi River in Southern India illustrates one such wastewater irrigation system. In the last decade, a number of studies have been carried out to understand the wastewater influenced Musi River agricultural system. Biggs and Jiang (2009) found that the high chemical concentrations of wastewater used for irrigation have adverse impacts on soil salinity. Other studies investigated the impact of hydrological processes on water quality (Amerasinghe et al., 2015; Mahesh et al., 2015b; Perrin et al., 2010), contaminations with nitrates and microbes (Amerasinghe et al., 2009), health risks associated with wastewater irrigation (Ensink et al., 2008; Srinivasan and Reddy, 2009), food crops and associated human health risks (Chary et al., 2008), salinity implications (Biggs and Jiang, 2009; McCartney et al., 2009), livelihood opportunities, and socio-economics of wastewater irrigation (Buechler et al., 2002; Mahesh et al., 2015b; Starkl et al., 2015). However, investigations related to the impact of wastewater irrigation on groundwater and the distribution of chemical variables impacting the seasonality in peri-urban areas of the Musi River basin have not been carried out.

To understand the dynamics and interrelations of complex environmental systems, multivariate methods and self-organizing maps (SOM) have proved to be effective techniques for analyzing large data sets (Astel and Małek, 2008; Tsakovski et al., 2010). In particular, a combination of statistical approaches was shown to provide a better understanding of driving forces and/or decisive interrelations (Singh et al., 2004). The multi-way modelling is considered to be an efficient and robust method (Bro, 2006; Smilde et al., 2005), and if combined with multivariate analysis, yields more coherent results (Villez et al., 2008). To our knowledge, the combination of multivariate analyses, multi-way models and SOMs was not used in research studies for understanding and evaluating groundwater quality dynamics influenced by wastewater irrigation. Combining these methods for analyzing chemical data sets, the current study aims to characterize the spatio-temporal variability of chemical variables defining groundwater quality and evaluate the impact of wastewater irrigation. This chapter results will benefit the researchers and water resources managers to understand the groundwater pollution dynamics with respect to the quality of wastewater applied for irrigation and seasonal effects arising from precipitation-irrigation interactions.

2.2 Materials and Methods

2.2.1 Sampling and Analytical Procedures

Water samples from the canal (wastewater) and groundwater samples from ten bore wells were collected every month for one hydrological year from May 2013 to April 2014 (twelve months) (Figure 1.3). Samples were collected in one litre high-density polyethylene plastic bottles, which were pre-cleaned with double distilled water and then three times with sampling water. To increase the accuracy of analysis, replicates (total 144) were collected from each sampling station in all the sampling months. Samples were labelled and transported to the laboratory on the same day and preserved at 4 ^{o}C in the laboratory for chemical analysis. In situ measurements such as pH, electrical conductivity (EC) and dissolved oxygen (DO) were measured in the field using the HQ40D portable multi meter (HACH, USA) during the sample collection. Analytical procedures for chemical analysis were performed using methods recommended by APHA (2005) in the Telangana state groundwater quality laboratory, Hyderabad. Prior to the sample analysis in the laboratory collected samples were filtered using Whatman filter paper. The volumetric titration method was applied for measuring bicarbonate (HCO_3) with standard sulfuric acid (H_2SO_4), chloride (Cl) with standard silver nitrate ($AgNO_3$), magnesium (Mg) and calcium (Ca) with standard Ethylenediaminetetraacetic acid (EDTA). Fluoride (F), phosphate (PO_4) and nitrate (NO_3) were determined using atomic absorption spectrophotometer; total dissolved solids (TDS) using gravimetric method; sulphate (SO_4) using turbidity meter; sodium (Na) and potassium (K) using flame photometer. The ionic balance error between total cations (Ca, Mg, Na and K) and total anions (HCO_3, Cl, SO_4, NO_3 and F) was within the acceptable range (± 5 %). All the measured variables were expressed in mg/L except pH (-) and EC (µS/cm).

2.2.2 Multivariate Statistical Analysis

Factor analysis (FA) and hierarchical cluster analysis (HCA) are the most commonly used and sophisticated multivariate statistical techniques for analyzing hydrochemical and geochemical datasets (Bayo and López-Castellanos, 2016; Dhakate et al., 2013; Magesh et al., 2017). Complex structures within the chemical datasets could be explained by reducing them to few significant factors without losing any information (Papatheodorou et al., 2007; Singh et al., 2004). Also, various pollutant impacts on surface water and groundwater have been studied widely with the factor, discriminant and cluster multivariate statistical techniques (Kumar et al., 2017; Singh et al., 2004).

Factor analysis is a well-recognized data reduction method which uses extraction of the eigenvalues and eigenvectors from the correlation matrix. In the analysis, factors are extracted by using the principal component method and the interpretation is based on rotated factors and loadings (Bayo and López-Castellanos, 2016; Dhakate et al., 2013). The data matrix is reduced to a minimum number of component loadings, which explains the majority of the variance. Normally, the first component loading explains the majority of the variance compared to subsequent components. Factor scores were estimated by using the Anderson-Rubin method, which ensures estimated factors and scores to be orthogonal. Factor scores with values >1 indicate samples reflecting major transformations of groundwater chemistry due to either anthropogenic or geogenic influence with the largest factor score explaining higher concentrations. To elucidate the spatial dynamics of groundwater quality in the study area, factor analysis was performed for each month separately, meaning for twelve spatially-explicit groundwater chemistry data sets to provide clear spatial variations in the individual two-way data.

Cluster analysis was performed to identify the significant chemical variable clusters that dominate the groundwater chemistry. The hierarchical clustering method joins the variables of similar data observations into one group, followed by the next most similar observations into another group (Papatheodorou et al., 2007). A dendrogram can be constructed to illustrate different similarity levels. Several authors (Chabukdhara et al., 2017; Dhakate et al., 2013) suggested that Euclidean distance as a distance (or dissimilarity) measure and Ward's method as a linkage rule produce the most distinctive clusters between the variables. To identify distinct and dominant chemical variables in the groundwaters of the studied watershed, cluster analysis was performed to the one hydrological year groundwater chemical database. Factor analysis and cluster analysis were performed in this chapter using SPSS software.

2.2.3 Multi-way Modelling

Even though factor analysis proved to be a powerful data reduction method, employed here to study spatial variations, it only analyses two-way data sets, i.e., samples x chemical variables or samples x seasons. To overcome this, multi-way models were developed, able to handle more complex data sets with multiple dimensions (Bro, 2006). For our purposes, parallel factor analysis (PARAFAC) was found appropriate as it allows analyzing the three-way dimensions in the database. In particular, PARAFAC segregates information within the data corresponding to the three variable modes: sampling sites (spatial), physicochemical variables (hydrochemical) and sampling campaigns (temporal) (Astel and Małek, 2008). The normal probability distribution map of the groundwater samples for PARAFAC model is presented in Figure S2.1. The corresponding three-way data matrix input consists of ten groundwater samples vs fourteen chemical variables vs twelve months, i.e., 1680 data points in total. PARAFAC was carried out by using the N-way Toolbox for MATLAB developed by (Bro, 1997). The detailed model structure is explained by (Bro, 1997) and (Rutledge and Bouveresse, 2007).

2.2.4 Self-Organizing Maps

Self-organizing maps (SOMs) provide visible data clusters for uncomplicated classification of the results and are well known in environmental and chemosphere studies (Astel et al., 2007). SOM is a neural-network model proposed by (Kohonen et al., 1996) and can learn and organize information without being given the dependent output values for the input network pattern (Kohonen et al., 1996; Mukherjee, 1997). SOM is noise tolerant and organizes data by adjusting the synaptic weights, which refer to the connection strength between the network nodes. SOM consists of neurons organized on a regular grid, and the number of neurons may vary from a few up to several thousand (Astel et al., 2007), connected to other adjacent neurons similar to the human brain based on neighbourhood relations. SOMs are applied to the data set, which contains 1680 results, to further understand the clustering of chemical variables. SOM allows comparative visualization of clustering of variables and helps to identify chemical variables which do not belong to any organized group. SOM presents the spatial distribution of similarity between the negatively and positively correlated variables (Astel et al., 2007). A Kohonen map instructs the structure of the variables (here it is groundwater major ions) with similar data observations. Training of the network was performed for the whole one-year groundwater chemistry database to optimize the weights. For the groundwater quality data, five by five visual cluster maps were produced for all the variables using 25 neural networks in the process. The neural network cluster toolbox of MATLAB was used to produce the chemical variable self-organizing maps.

2.3 Results and Discussion

Descriptive statistics (minimum, maximum, mean, standard deviation, and skewness) for physicochemical variables and major ions for one hydrological year are presented in Table 2.1 and show wide ranges of concentrations, partly reaching or even exceeding maximum permissible values according to the Bureau of Indian Standards (BIS) drinking water specifications. Groundwater pH values range from neutral to alkaline (6.87 to 8.31). Electrical conductivity varies between 673 and 3470 µS/cm. NO_3 and F concentrations in the groundwater samples are frequently exceeding the maximum permissible BIS limits.

Table 2.1 Descriptive statistics of the groundwater quality data*

	Units	Minimum	Maximum	Mean	Standard deviation	Skewness	Bureau of Indian Standards†	
							Desirable / Acceptable	Maximum permissible
pH		6.87	8.31	7.67	0.29	-0.07	6.5-8.5	-
EC	µS/cm	673	3470	1719	522	0.68	-	-
TDS	mg/L	430	2221	1100	334	0.68	500	2000
HCO_3	mg/L	76	606	330	93	-0.43	300	600
Cl	mg/L	10	650	250	117	0.74	250	1000
F	mg/L	0.49	2.29	1.14	0.33	0.94	1.0	1.5
NO_3	mg/L	5.18	189	35	23	2.88	-	45
SO_4	mg/L	40	430	132	77	1.70	200	400
Na	mg/L	35	379	164	73	0.58	-	-
K	mg/L	1.06	35	5.99	6.71	2.36	-	-
Ca	mg/L	48	248	111	39	0.80	75	200
Mg	mg/L	10	151	55	26	0.87	30	100
PO_4	mg/L	0.03	6.84	0.78	1.28	2.51	-	-
DO	mg/L	0.48	11.03	4.77	1.89	0.51	-	-

*Based on 120 groundwater samples in total (sampling hydrological year: 2013-2014); †Limits mentioned for human consumption (https://bis.gov.in/)

The relative dominance of anions and cations in the entire database is found to be Cl > SO_4 > HCO_3 > NO_3 and Na > Ca > Mg > K, respectively. The order of dominance varies in the monthly results depending on the wastewater application rates and monsoon rainfall. The standard skewness is considered to be extreme if it is outside the range ±2. For the data presented in Table 2.1, only the nutrients NO_3, K and PO_4 were right skewed with positive values and considered to be extreme.

2.3.1 Spatio-temporal distribution

The groundwater in the watershed was continuously evolving and changing its chemical quality depending on the change in wastewater quality applied for irrigation and monsoon rainfall during the study period.

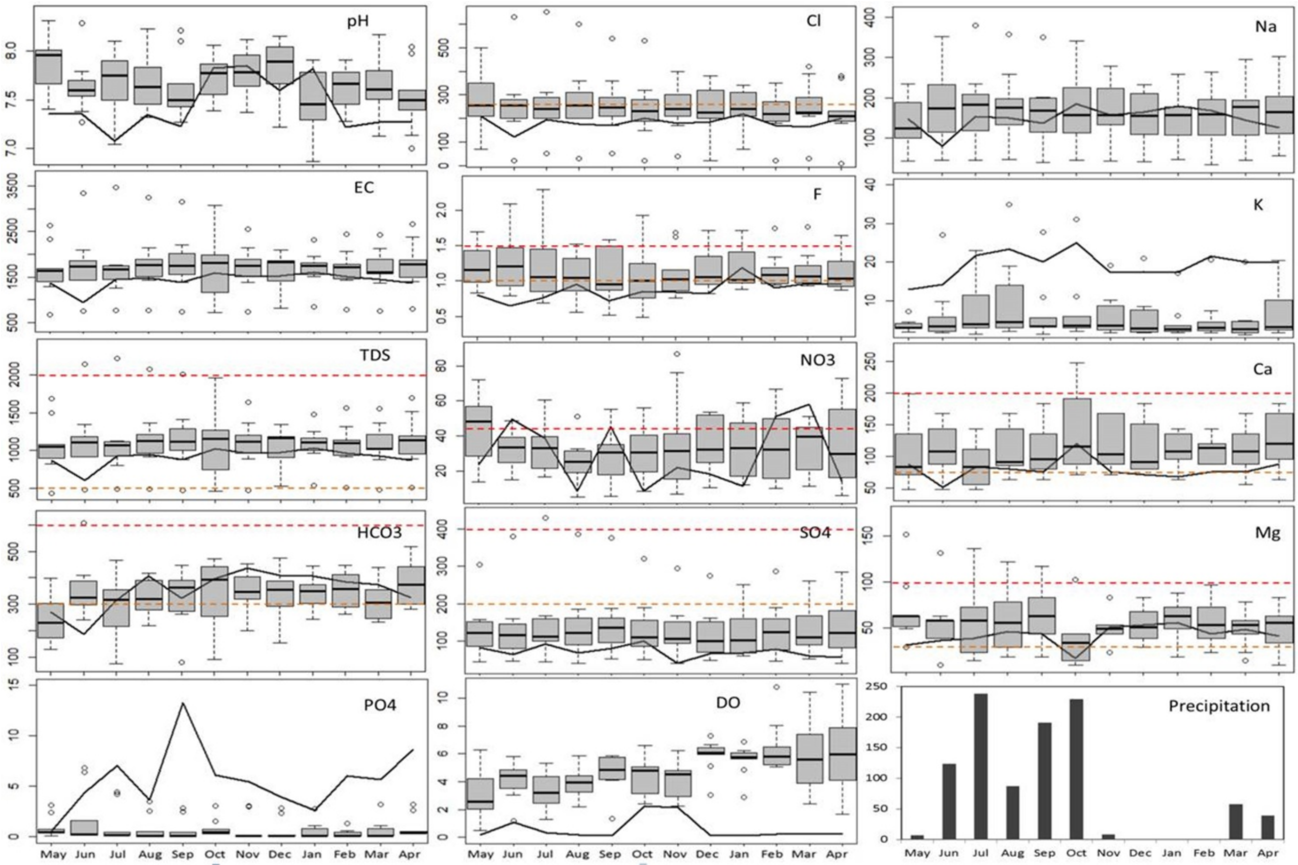

Figure 2.1 Seasonal changes in water quality variables from May 2013 to April 2014: Box plots display values from groundwater samples (median, 25 and 75% quartiles (boxes), 10% and 90% percentiles (whiskers) and outliers (dots)). The solid lines represent the quality of wastewater used for irrigation (mean value of two wastewater samples) in the study area. The orange dotted lines represent the acceptable and the red dotted lines represent maximum permissible water quality standards (BIS) and parameters with no relevant standards don't have these lines. The last graph represents the precipitation data in mm. All chemical variables are expressed in mg/L except for pH (-) and EC (µS/cm).

Seasonal variations of the groundwater quality data of various chemical variables were analyzed using box plots given in Figure 2.1 from May 2013 to April 2014. The pH values were highly variable in a manner similar to precipitation and EC, TDS, Cl, and SO_4 show a similar pattern. Wastewater samples have high K and PO_4 contents, whereas respective concentrations in groundwater are low, maybe due to soil nutrient intake and plant uptake. The DO values of wastewater are very low and ecological functions will not be sustained in these chemical environments. DO values of the groundwater are gradually increasing in the drier months due to the delayed recharge from monsoon rainfall in the previous months, whereas the reverse is true for the wet months due to increased interaction with irrigated wastewater. The high concentrations of NO_3 values in groundwater are significantly impacted by the wastewater irrigation in the watershed and often exceed the maximum permissible limits for drinking water. Fluoride concentrations in the groundwater are also often exceeding the maximum permissible limits and reach a maximum of 2.29 mg/L during the study period. The quality of the groundwater changes over the hydrological year with respect to the land use and choice of water used for irrigation. The ionic composition of groundwater samples is compared using a Piper plot (Figure 2.2a).

Groundwater quality in the upper part of the watershed shows the characteristics of calcium-bicarbonate waters, whereas groundwater samples from wastewater irrigated areas show the characteristics of mixed and calcium-sulphate waters. The analysis of hydrochemical facies suggests that cation exchange alters the interaction with carbonate minerals and calcium dominance. Irrigation suitability in terms of salinity can be deduced from a Wilcox diagram (Figure 2.2b). In general, if the water quality of a particular water sample falls in high to very high sodium or salinity hazard, then that particular water sample is not suitable for irrigation. Groundwater samples from the upper peri-urban area (Ks10) of the watershed express medium salinity hazard and low sodium hazard. Samples from groundwater irrigated area (Ks5-Ks9) indicate a high salinity hazard and low sodium hazard, whereas samples from wastewater irrigated area (Ks1-Ks4) indicate high to very high salinity hazard and low to medium sodium hazard. It seems that the groundwater from the watershed is unsuitable for irrigation with the current high salinity levels (Figure 2.2b). Local farmers have already reported a decrease in yields of paddy rice with the increase in salinity levels. As a consequence, many farmers have switched to paragrass cultivation, which is salinity tolerant and has currently become the dominant crop in the watershed (Biggs and Jiang, 2009; Mahesh et al., 2015b).

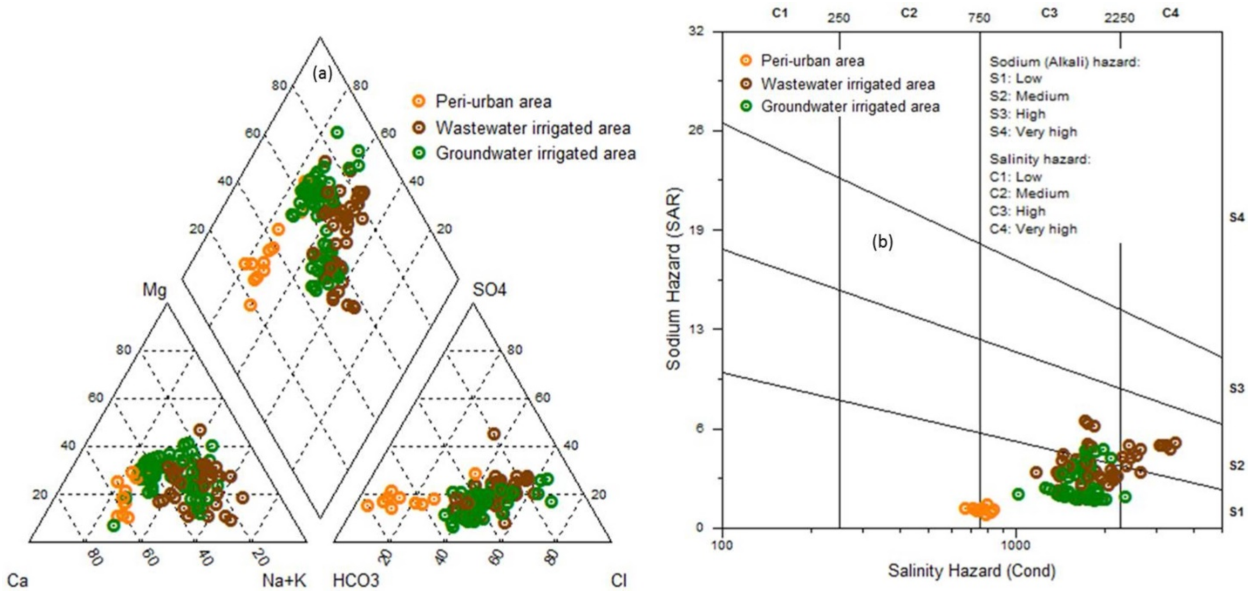

Figure 2.2 (a) Piper plot, (b) Wilcox diagram (SAR - Sodium Absorption Ratio, Cond – electrical conductivity). Water types given in orange, brown and green colours represent the three land use types of peri-urban, wastewater irrigated and groundwater irrigated areas, respectively.

2.3.1.1 Factor Analysis

Factor analysis was used to describe monthly spatial distribution and dominant chemical variables in a particular month. To observe overall seasonal variations in the groundwater is important, but at the same time understanding the monthly dominant chemical fluctuations is also salient. Principal component extraction method was used to perform factor analysis for the individual monthly groundwater quality database, to understand the spatio-temporal evolution. Factor analysis results of the first factor component score are shown in Figure 2.3 and Figure S2.3. Total explained variance is lowest in December (27.5%) and highest in August with 46.2%. Three to five components explain the total variance depending on the month (Table S2.1). In the majority of the months, the first component is dominated by the variables EC, TDS, Cl, SO_4, Na, and Mg, which reflect the anthropogenic influences of the wastewater on groundwater quality and at the same time monsoon rainfall impacts. The second component largely contains information about HCO_3, F, and NO_3, which majorly expresses the geogenic influences with a mixture of wastewater impacts. Dominant variables of factor 1 and their corresponding scores indicate that the groundwater quality in the upstream part of the watershed is rather unchanged over the months compared to the lower parts of the watershed, where wastewater irrigation is heavily practiced.

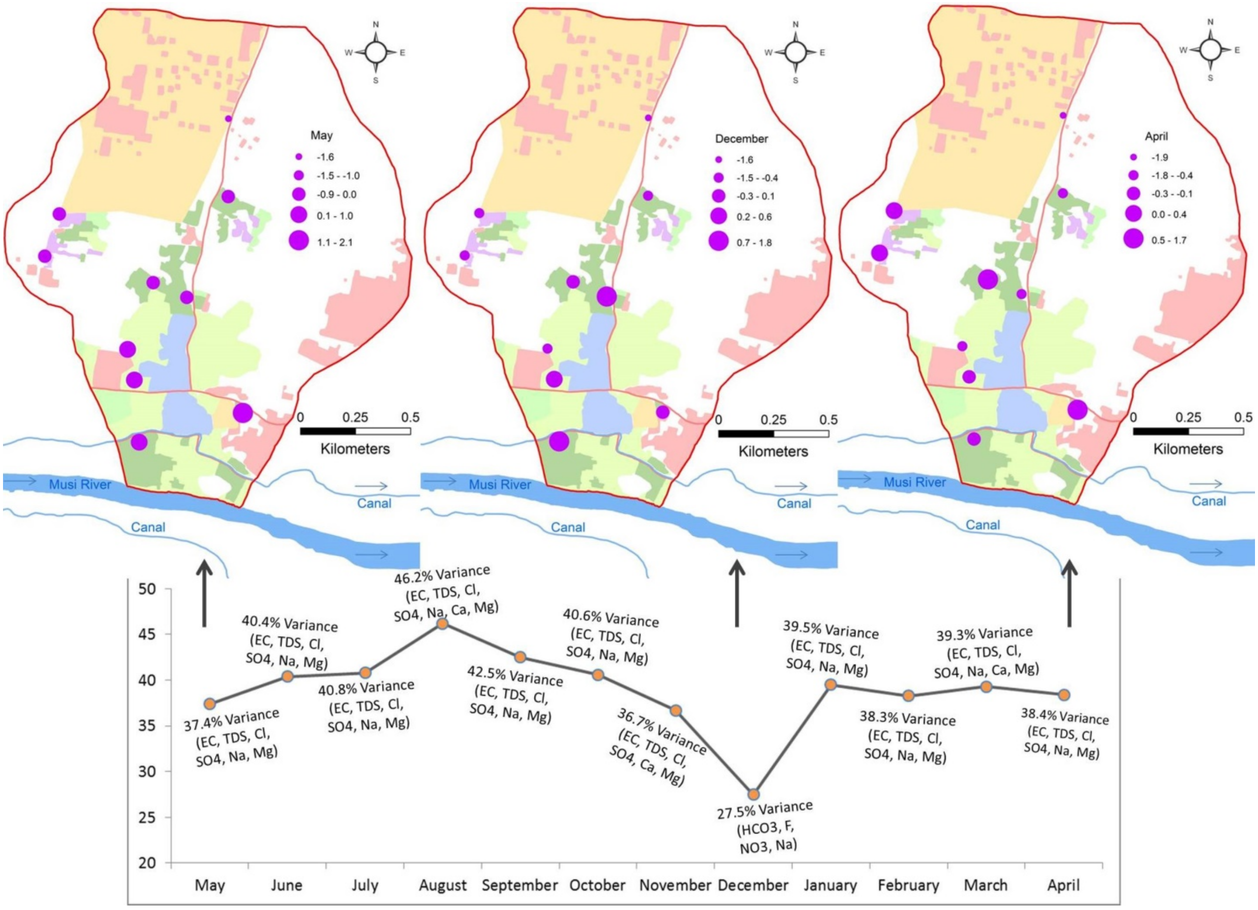

Figure 2.3 Factor Analysis (FA) results: Upper panel shows spatial variation of the first component factor scores of groundwater samples during the starting and ending months, May and April of the monitored hydrological year and December, when maximum variation for the year is observed. The smaller size of the colour symbol represents the minimum factor score, and larger size represents the factor score with maximum variation of the groundwater chemical concentrations in the watershed. The lower panel shows the temporal variation of explained variance of the first component and dominant variables.

The groundwater sampling location that is heavily influenced by wastewater irrigation is located south-east of the watershed (Ks2), where peak seasonal variations and higher salinity levels are observed throughout the year (Figure 2.1 and Figure 2.3). Factor score variation (lower part of Figure 2.3) indicates that the monsoon (June-October) recharge has a significant impact on the watershed groundwater quality. The drop in explained variance in December indicates a shift in dominant variables from EC, TDS, Cl, SO_4, Na and Mg to HCO_3, NO_3, Na and F. This indicates that poor quality of the water used for irrigation has increased concentrations of Cl and SO_4 in the groundwater and induces the higher salinity levels in the local aquifer. Because of the water-rock interactions with granite-gneiss, the major rock type in the study area, the fluoride content fluctuates over the months and is associated with precipitation, which can be explained by fluoride containing minerals such as fluorite and biotite (Brindha et al., 2011). NO_3 content levels in the groundwater can be explained by the long-term wastewater irrigation, with nitrates being excessively leached into local groundwater aquifers. Increase in salinity levels in the groundwater after the monsoon is most likely due to flushing of the soils by flood irrigation and intensive rainfall events.

2.3.1.2 Parallel Factor Analysis

PARAFAC is used for three-way analysis to evaluate the seasonal variations with spatial dynamics. Two modes, seasonal and sample mode of the total groundwater quality are displayed in Figure 2.4. Both precipitation and wastewater application rates for irrigation have important roles in changing the local groundwater quality.

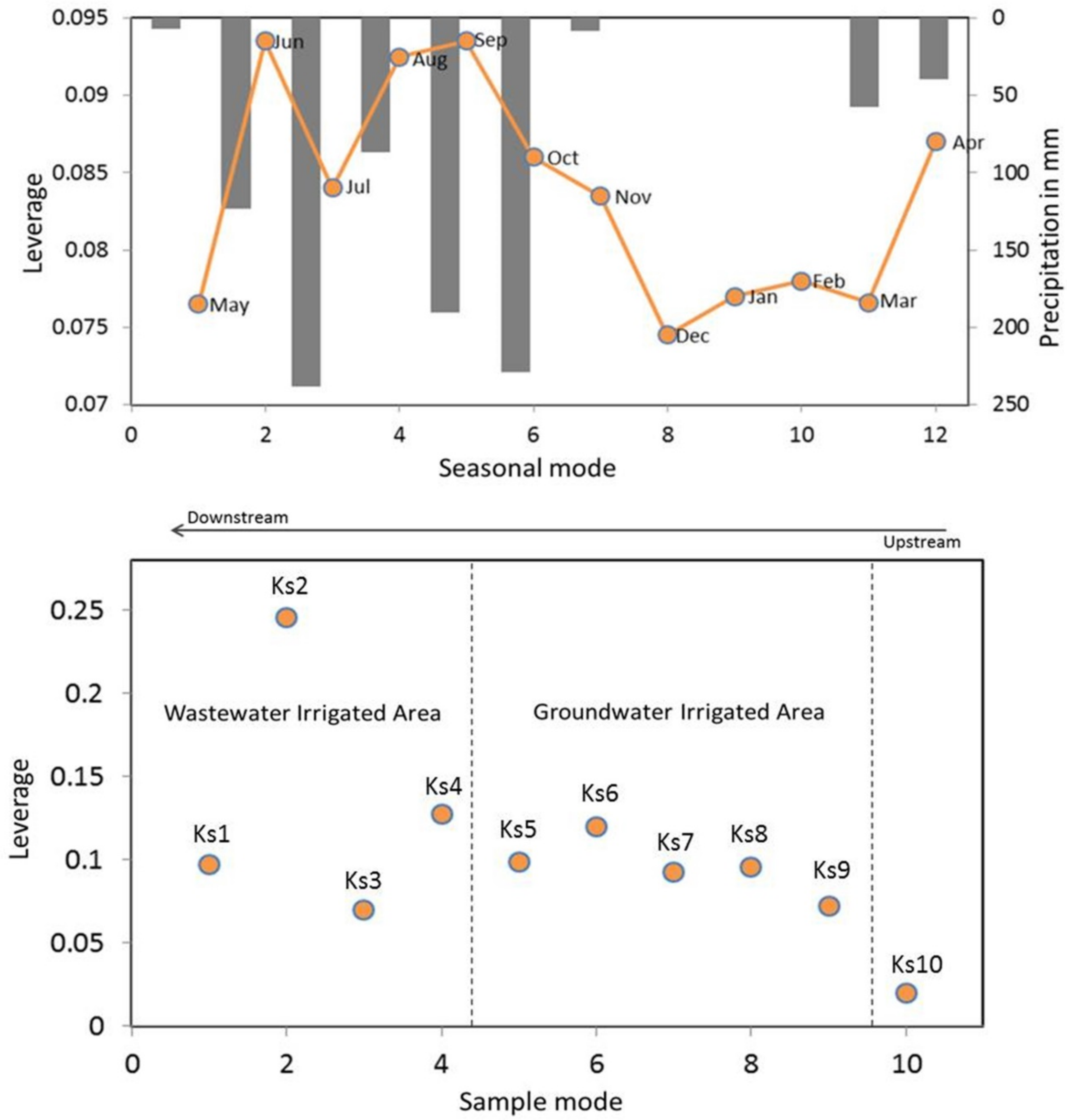

Figure 2.4 PARAFAC: Seasonal mode and sample mode of the groundwater quality in the watershed. The grey bars in the seasonal mode depict the precipitation in mm over the year.

Periods of high precipitation correspond to high values of groundwater chemical concentrations observed in the following months with a recharge delay rate of one to two months. After the first monsoon showers in June, the groundwater quality changes, and thereafter, continuous monsoon showers from July to October have a significant influence on the variance drop from December to March of the following year. The results of the PARAFAC sample mode explain the variation of the groundwater quality along the watershed over a year. It clearly shows an inclining trend of groundwater quality from upstream to downstream of the watershed, close to Musi River in the wastewater irrigated areas. Sample Ks2, which is in close proximity to both wastewater irrigated area and peri-urban built-up area (Figure 1.3) appears to be associated with higher chemical concentrations in the groundwater. Groundwater samples in the wastewater irrigated area are highly influenced by wastewater application rates on the agricultural fields. And also the return flows of the wastewater irrigated system have a significant influence on the local aquifer. In the built-up area of the upper parts of the watershed, groundwaters are less influenced by wastewater and mostly influenced by precipitation. In regular watersheds, the contaminants in the groundwater get diluted after the monsoon rainfall, whereas in the current long-term wastewater irrigated study context, the reverse is true. It appears that accumulated contaminants in the soil media are flushed out to the groundwater aquifers after the first monsoon and that the unsaturated zone is acting as a chemical buffer.

2.3.2 Chemical variable characterization

HCA results in five groups of linked chemical variables. The initial cluster analysis yields only two major cluster groups: cluster I with EC and TDS and all remaining chemical variables in a group that was labelled as cluster II (Figure 2.5a, left panel). To depict a more detailed distribution of organized groups, chemical variables in cluster II are re-analyzed. In doing so, cluster II is divided into four more clusters: cluster III with F, PO_4, pH, DO, K, NO_3 and Mg, cluster IV with SO_4, Na and Ca, cluster V with HCO_3, and cluster VI with Cl (Figure 2.5a, right panel). Cluster analysis gave a glimpse of dominant chemical variables, but SOM explains more perspicuous chemical variables in the data set.

The resulting SOM visualization maps indicate a diverging variable cluster grouping compared to HCA (Figure 2.5b; U-matrix presented in Figure S2.2). HCA and SOM results agree on most clustering results (see Table S2.2), except the distribution of Cl. This chemical variable forms an individual cluster in cluster analysis, while its SOM structure is very similar to EC and TDS. Therefore, Cl was assigned to the EC and TDS cluster. Further, it is found that SO_4, Na and Ca are negatively correlated to each other, but show a similar distribution. In the same way, all the nutrients NO_3, PO_4, and K are also negatively correlated with each other but express similar distributions. F and Mg are contained in one group, which is not identified in the cluster analysis. pH and DO show similar visualization structures, but they are negatively correlated with each other. HCO_3 is the only chemical variable expressing a unique structure, thus falling into a group by itself. The EC, TDS, Cl group is the dominant cluster and is highly influenced by wastewater irrigation flows in the watershed. High salinity content in the groundwater is associated with high chloride concentrations in the irrigation water. Salinity levels continue to increase all over the year in the local groundwater because soils do not retain or absorb chloride which leaches directly into the local aquifers.

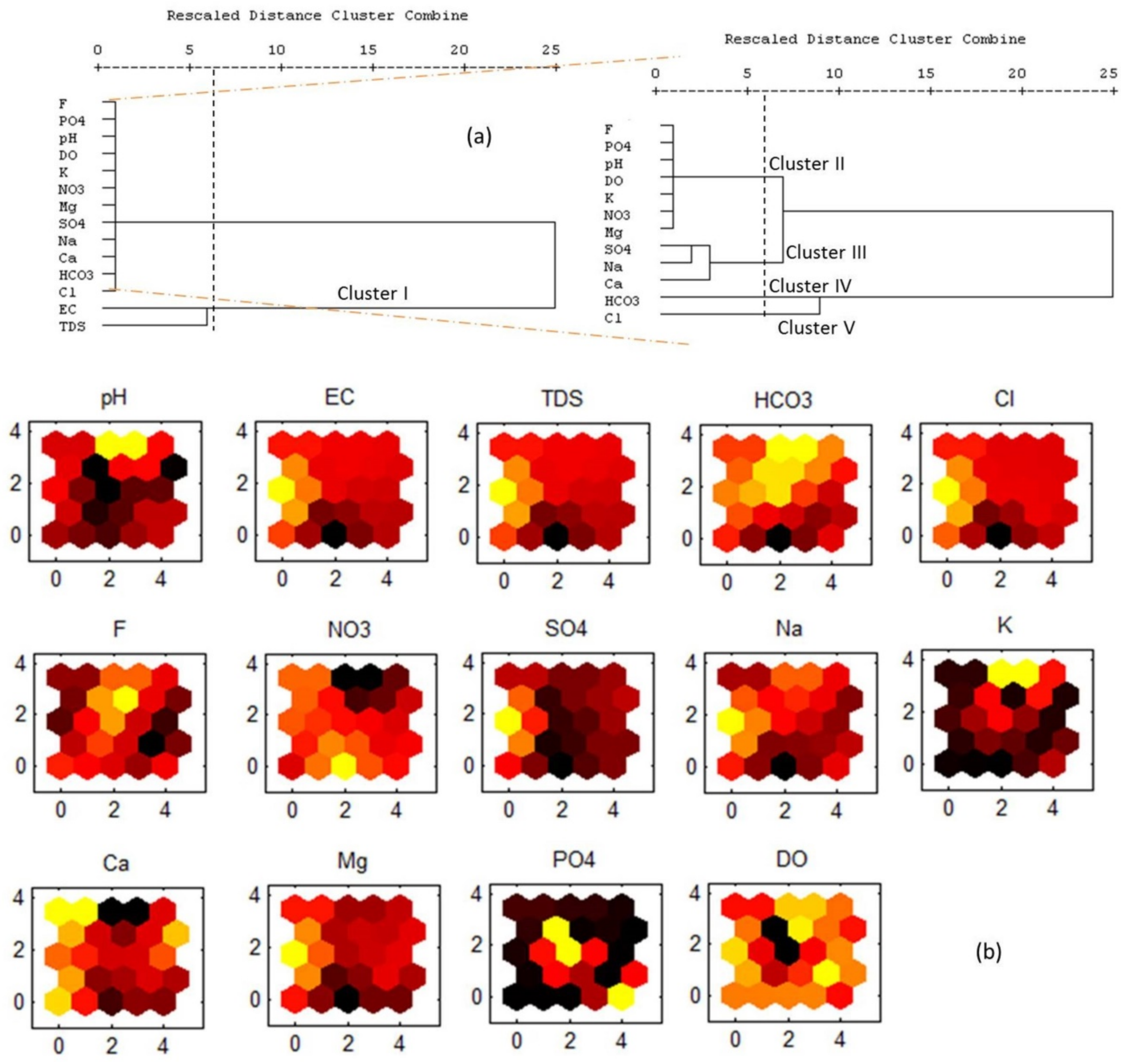

Figure 2.5 (a) Dendrogram of hierarchical cluster analysis, (b) SOM planes and visualization of the distribution of groundwater quality variables mapped with the 5x5 neural network.

The SO_4, Na and Ca cluster explains the anthropogenic impacts of wastewater as well as cation exchanges with the rainfall events. The nutrient cluster explains that, with the soil absorption and plant uptake, groundwater has less K and PO_4, while the nitrates reach the groundwater more freely, at least in some months. Nutrient content in the irrigation water highly fluctuated from month to month because of intricate urban wastewater flows. Nutrient uptake is considered to be high for paragrass compared to other crops such as vegetables and paddy rice, which should explain why nutrient concentrations in the groundwaters were at a minimum. Approximately an amount of 200 to 400 kg/ha of nitrates is taken up per year by grass crops (Candela et al., 2007). The F, Mg cluster is influenced by geogenic impacts and the F levels increase after the monsoon rainfall (Figure 2.1), which is a clear indication of water-rock interactions. The fluoride and magnesium contents might originate from fluorite and dolomite minerals, respectively, due to the interaction of water with granitic rocks (Reddy et al., 2010). The pH, DO cluster is highly influenced by wastewater irrigation and at the same time after the monsoon DO values of the groundwater increase with the dilution because of long-term wastewater irrigation practices. HCO_3 is itself a single cluster, indicating that only the groundwaters in the built-up area in upper parts of the watershed (Ks10, Figure 1.3) are rich in bicarbonates (Figure 2.2). The SOM groups represent the chemical variables which are of high influence in the groundwater chemical composition due to wastewater irrigation. The combined analysis of multivariate modelling and SOM suggests that wastewater pollution heavily influences high concentrations of Cl, SO_4 and low concentration of DO. As such, the groundwater in the watershed is not suitable for human consumption, evidently caused by the continuous irrigation of land with wastewater.

2.3.3 Implications of wastewater irrigation

Wastewater from the Hyderabad city turned out to be a potential resource for irrigation and provides nutrients and organic matter for agriculture, and at the same time, groundwater contamination is a potential problem after long-term wastewater irrigation in the Musi River irrigated plain. Often the sewage from the urban agglomerations in the developing world is mixed with industrial contaminants, which also holds for Hyderabad city. Based on wastewater quality, treatment should be performed in a target-oriented manner for reuse in agriculture, depending on the type and amount of pollutants in the wastewater on one hand and the requirements of the grown crops as well as the capacities of the system to attenuate and degrade remaining pollutants on the other hand. In this context adopting natural treatment systems for pollution abatement, such as constructed wetlands, is a viable option that decision-makers need to consider and such systems have been already proposed for Musi River (Sonkamble et al., 2018). Additionally, there were no guidelines to control the chemical loadings and protect groundwater quality, a regulatory mechanism should be implemented to control the dominant chemical loading rates in wastewater. With the farmers´ heavy dependence on the wastewater for agriculture in peri-urban areas (Amerasinghe et al., 2009), potential toxic contaminants such as heavy metals and microbial pathogens reaching the groundwater aquifers continue to increase, which needs more attention in future studies because the local aquifers in peri-urban areas are a potential water supply source for drinking and domestic purposes. With wastewater being increasingly accepted and used as a source of water and nutrients for agricultural usage (Hettiarachchi and Ardakanian, 2016b), local water resources departments need to be conscious in regular monitoring of groundwater in the wastewater irrigation areas to evaluate the potential threats to human health.

2.4 Conclusions

The current study presents a combination of multivariate analysis, multi-way modelling and self-organizing maps to assess the groundwater quality in a watershed where wastewater irrigation has been a practice for over 40 years. This chapter concludes that the continuous wastewater irrigation resulted in increased groundwater salinity. Groundwater quality is not suitable for irrigation and the farmers' decision to move to crops that are salinity tolerant is thus understandable. Farmers' decisions were made on practical experiences, like low crop yields, especially for paddy rice. From multi-way modelling and SOM, it is evident that the monsoon rainfall from June to October impacted the groundwater quality during the period from October to January of the following year. Wastewater irrigation leads to an increasing trend in the concentration of chemical variables because of clogged chemicals in the soils being flushed to the groundwater aquifer. In the groundwater, Cl, SO_4 and DO are greatly influenced by wastewater irrigation. Nutrient concentrations (NO_3, PO_4, and K) are high in wastewater and low in groundwater, which leads to the conclusion that nutrient uptake by crops from wastewater and/or immobilization in the soil was sufficiently high to prevent nutrient enrichment of groundwater. To ensure that nutrient enrichment will not become problematic in the future, one needs to evaluate the optimum requirement of wastewater to irrigate crops, where farmers practice flood irrigation. Local water resources management authorities should be mindful about the growing problem of groundwater pollution and take steps to treat the wastewater that is used for irrigation or develop efficient groundwater management strategies to manage the pollution.

3 Hydrogeochemical and Mixing Processes Controlling Groundwater Chemistry

This chapter is a modified version of the published paper: *Mahesh Jampani, Rudolf Liedl, Stephan Hülsmann, Sahebrao Sonkamble, Priyanie Amerasinghe (2020). 'Hydrogeochemical and mixing processes controlling groundwater chemistry in a wastewater irrigated agricultural system of India'. Chemosphere (Elsevier). Vol. 239. 124741.*

3.1 Introduction

Aquifer systems around the world are increasingly influenced by anthropogenic activities, which can alter the hydrochemical processes that control the groundwater geochemistry in natural settings (Devic et al., 2014; Ramyapriya and Elango, 2018). With rapid urbanization and development related human interventions, anthropogenic influences are often greater in the urban and peri-urban settings when compared to rural settings. Various factors influence the quality of groundwater in urban and peri-urban areas, including domestic and industrial wastes, excessive groundwater pumping, sewage from urban systems, wastewater irrigation etc. (Kurian et al., 2013; Kurian and McCarney, 2010; Mahesh et al., 2015b; Padgham et al., 2015; Parkinson and Tayler, 2003; Tiwari et al., 2014). Often it is hard to determine the influencing processes of groundwater quality in these systems due to intermixing complexity of anthropogenic and geogenic factors, which occur concurrently (Huang et al., 2013; Shi et al., 2018; Zhai et al., 2017).

Around the world, various studies have been carried out to understand the complex processes of aquifer geochemistry and provided critical knowledge on groundwater evolution with the change in environmental and social conditions (Argamasilla et al., 2017; Choi et al., 2014; Dehbandi et al., 2018; Liu et al., 2017; Rao et al., 2012). Popular applied methods include basic water quality plots such as Gibbs diagram, Piper plot, Wilcox diagram, Chadha diagram and Durov plots that provide the understanding of groundwater chemical processes (Argamasilla et al., 2017; Chadha, 1999; Gibbs, 1970; Jampani et al., 2018; Lu et al., 2015; Mushtaq et al., 2018; Sonkamble et al., 2012; Zhu et al., 2017). Further, chemical mass balance, ionic delta analysis and saturation indices provide the detailed knowledge of water mixing and mineralogical processes of a particular aquifer system (Al-qudah et al., 2017; Liu et al., 2017; Najib et al., 2016; Olmez et al., 1994; Slama and Bouhlila, 2017; Zhao et al., 2017).

Rapid groundwater salinization is one of the severe problems faced by many aquifers around the world, especially in water-logged, coastal and wastewater irrigated areas (Biggs and Jiang, 2009; Datta and Jong, 2002; Gil-Márquez et al., 2017; McCartney et al., 2009). Over the last few decades, wastewater irrigation has increasingly been practiced by farmers in various parts of the world due to limited availability of freshwater and nutrient benefits in the wastewater (Qadir et al., 2010; Raschid-Sally and Jayakody, 2008). Particularly in the peri-urban settings around the world, reuse of wastewater has become a common practice for agriculture and horticulture by farmers and urban dwellers (Amerasinghe et al., 2012; Bedbabis et al., 2015). When farmers and urban dwellers use wastewater that is untreated, it will have detrimental impacts on the local ecological system (Alghobar and Suresha, 2016; Becerra-Castro et al., 2015). Poor agricultural management practices in wastewater irrigation systems also have adverse impacts on local aquifers (Candela et al., 2007; Kass et al., 2005). Monitoring of groundwater quality and analyzing geochemical processes is largely lacking in many parts of the developing world in these wastewater irrigation systems, where peri-urban dwellers depend on the local aquifers for drinking and domestic water supply. To mitigate the groundwater pollution, understanding the factors and processes that control the aquifer geochemistry is critical for sustainable groundwater management in wastewater irrigated systems.

One such study area that is heavily influenced by wastewater is the Musi River basin in southern India. The wastewater flows for the Musi River comes from a major urban agglomeration of India, Hyderabad, and has been studied with regard to socio-economic and environmental impacts and processes (Amerasinghe et al., 2015, 2009; Biggs and Jiang, 2009; Buechler et al., 2002; Ensink et al., 2009; Mahesh et al., 2015b; Sonkamble et al., 2018). Further, Jampani et al., (2018) analysed the groundwater conditions impacted by wastewater irrigation with respect to seasonality, spatio-temporal patterns and chemical characterization.

This chapter evaluates groundwater quality and their controlling hydrogeochemical and mixing processes in a micro-watershed of the Musi River basin, where wastewater irrigation has been a common practice for decades. The main aim of the current chapter is to characterize the hydrogeochemical facies of groundwater, distinguish the geochemical signatures, and evaluate the chemical balance of different ions, mineral phases and source appropriation of groundwater. Results should enable a clear indication which water quality variables of the groundwater are influenced by wastewater irrigation to enable a clear strategy for irrigation and wastewater management. It envisages that this chapter will provide an improved understanding of the geochemical processes that control groundwater chemistry in wastewater irrigated systems and will help authorities to develop practices that will minimise aquifer contamination.

3.2 Materials and Methods

3.2.1 Water Sampling and Analytical Procedures

Water quality in the watershed was analyzed by sampling the irrigation source wastewater from the canal and groundwater from ten bore wells collected on a monthly basis for one hydrological year (May 2013 to April 2014). Out of the ten bore wells, four had piezometers installed, in which the groundwater levels were continuously monitored (Figure 1.4b) using automatic water level recorders. The samples were analyzed for pH, electrical conductivity (EC), total dissolved solids (TDS), major anions (HCO_3, Cl, SO_4, and F), major cations (Na, K, Ca and Mg) and nutrients (NO_3 and PO_4). Water samples were collected every month with replicates in one litre pre-cleaned high-density polyethylene (HDPE) plastic bottles. The collected water samples were labelled, preserved and transported to the laboratory on the same day. Water quality analytical procedures for chemical analysis were performed using the methods recommended by (APHA, 2005). Bicarbonate (HCO_3), chloride (Cl), calcium (Ca) and magnesium (Mg) were measured using volumetric titration method; sulphate (SO_4) using turbidity meter; electrical conductivity (EC) using conductivity meter; total dissolved solids (TDS) using gravimetric method; sodium (Na) and potassium (K) using flame photometer; fluoride (F), nitrate (NO_3) and phosphate (PO_4) using atomic absorption spectrophotometer. The charge balance error achieved for measured cations and anions of the data was within the acceptable range (± 5 %). All the measured chemical parameters were expressed in mg/L except EC (µS/cm) and pH (-).

The factors influencing the groundwater chemistry can be demonstrated by Chadha's diagram. The graphical representation of Chadha's diagram (Chadha, 1999) is the difference between alkaline earths (Ca+Mg) and alkaline metals (Na+K) plotted against the difference between weak acids (HCO3 or CO3+HCO3) and strong acids (Cl+SO4). Typically the concentrations used for Chadha's diagram is presented in milliequivalents. The measured water samples will fall under four categories of water types: Ca-Mg-HCO3 (recharge waters with temporary hardness), Ca-Mg-Cl (reverse ion-exchange waters with permanent hardness), Na-Cl (saline waters) and Na-HCO3 (bases ion-exchange waters).

Gibbs plots are boomerang structured diagrams used to determine the functional sources or mechanisms that control the groundwater chemistry. A typical Gibbs diagram is a weight ratio of anions Cl/(Cl+HCO3) or cations Na/(Na+Ca) plotted against total dissolved solids (TDS). The two plots proposed by Gibbs (1970) demonstrate the influence on groundwater by climate conditions, lithology of the study area and local rainfall, which are mentioned in the Gibbs plot as evaporation/crystallization, rock weathering and precipitation, respectively.

3.2.2 Chloroalkaline Indices (CAI) and Saturation Index (SI)

The evaluation of base ion exchange between the groundwater system and its host environment is explained by chloroalkaline indices (CAI). The two distinct equations of CAI-I and CAI-II can be written as:

$$CAI - I = \frac{Cl - (Na + K)}{Cl}$$

Eq. 1

$$CAI - II = \frac{Cl - (Na + K)}{(SO_4 + HCO_3 + NO_3)}$$

Eq. 2

When the CAI values are positive, then it means that there is direct base ion exchange of Na and K in the water with the Ca and Mg in rocks. If the values are negative, then the exchange is reverse and indirect (Kumar et al., 2018; Rao et al., 2015).

A saturation index informs about the degree of interaction between water and minerals. Saturation indices of the groundwater samples were determined by using PHREEQC model to evaluate the mineral saturation conditions with the influence of long-term wastewater irrigation and monsoon rainfall. In general, the saturation stage between mineral and solution is indicated by Saturation Index (SI). It can be expressed as (Appelo and Postma, 2004);

$$SI = \log\left(\frac{IAP}{K}\right)$$

Eq. 3

where K is the solubility product and IAP is the ion activity product. In general, if SI values of a particular mineral component are negative, they indicate groundwater undersaturation or mineral dissolution. If the SI values are positive, then it means groundwater is being supersaturated or mineral precipitation takes place. Supersaturation of a mineral in the groundwater also occurs because of many other factors including temperature rise, evaporation, dissolution etc. If the SI value is zero, then it means groundwater is in equilibrium with the dissolved aquifer minerals (Aghazadeh and Mogaddam, 2011; Appelo and Postma, 2004; Sridharan and Nathan, 2018).

3.2.3 Wastewater Fraction and Ionic Deltas

Wastewater fraction in the groundwater samples is evaluated by using modified seawater fraction equation (Appelo and Postma, 2004), where chloride is used as a conventional tracer for estimation.

$$F_{ww} = \frac{Cl_s - Cl_f}{Cl_{ww} - Cl_f}$$

Eq. 4

where Cl_s, Cl_f and Cl_{ww} are the concentrations of chloride for groundwater sample, freshwater and wastewater respectively.

Wastewater fraction is used to calculate the concentration of each ion from the conservative mixing of wastewater and fresh groundwater. The mixing concentrations for each ion can be written as:

$$i_m = (F_{ww} * i_{ww}) + \left((1 - F_{ww}) * i_f\right)$$ Eq. 5

where i_{ww} and i_f are ionic concentrations of wastewater and freshwater respectively, and F_{ww} is the calculated wastewater fraction from the previous equation.

The ionic reactions or ionic delta (Δ) for each ionic concentration is the difference between the conservative mixing of the sample and measured sample of the groundwater. The chemical reaction equation can be written as:

$$\Delta i = i_s - i_m$$

Eq. 6

where i_s and i_m are the ionic concentrations of the groundwater sample and calculated ionic mixture from equation 5. The potential chemical reactions and geochemical processes in the aquifer are quantified using the ionic deltas. If the ionic delta is positive, that means groundwater is getting enriched with that particular ion (i) and a negative value indicates depletion of the particular ionic concentration compared to the theoretical mixing (Appelo and Postma, 2004; Najib et al., 2016).

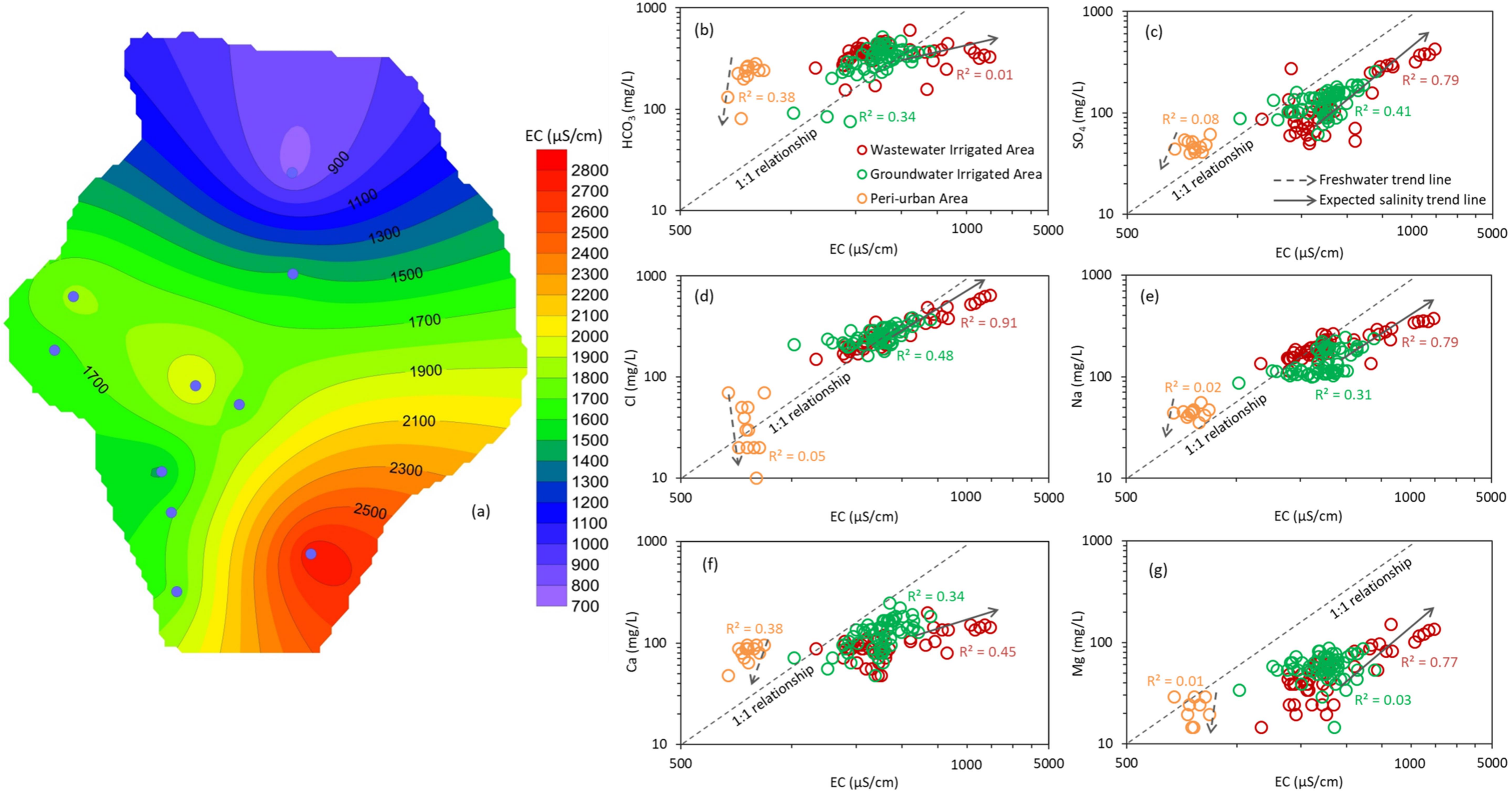

Figure 3.1 Groundwater quality variations of major ions: (a) average groundwater electrical conductivity of one hydrological year; scatter plots of the relationship between Log EC vs (b) Log HCO_3, (c) Log SO_4, (d) Log Cl, (e) Log Na, (f) Log Ca, and (g) Log Mg. The arrows indicate freshwater trend line and observed salinity trend line with respect to the plotted chemical parameter.

3.3 Results and Discussion

The pH values of the groundwater samples in the study area range from 6.8 to 8.3, which indicate neutral to alkaline characteristics. Detailed descriptive statistics of the groundwater samples in wastewater irrigated, groundwater irrigated and upstream peri-urban areas of the watershed with source wastewater used for irrigation are presented in Table 3.1. The electrical conductivity of the groundwater samples is highly variable in the wastewater irrigated area ranging from 1166 to 3470 µS/cm, whereas low variability observed in the upstream peri-urban area of the watershed (673 to 841 µS/cm). Cl is recorded as the dominant parameter observed in the groundwater samples all over the year.

Table 3.1 Descriptive statistics of groundwater and wastewater quality data of the watershed*

Parameter	Wastewater Irrigated Area[1]			Groundwater Irrigated Area[2]			Peri-urban Area[3]			Wastewater (Canal)[4]		
	Min	Max	SD	Min	Max	SD	Min	Max	SD	Min	Max	SD
pH	6.87	8.29	0.32	7.05	8.31	0.23	7.0	8.1	0.36	7.08	7.85	0.27
EC	1166	3470	581	1015	2380	246	673	841	46	939	1611	175
TDS	746	2220	372	650	1523	158	431	538	29	601	1031	112
HCO_3	155	606	84	76	518	88	81	281	59	189	435	71
Cl	150	650	133	163	390	51	10	70	20	120	220	26
SO_4	50	430	106	62	255	36	40	62	6.5	41	101	16
F	0.75	2.29	0.37	0.49	1.77	0.32	0.87	1.4	0.15	0.65	1.19	0.14
NO_3	5.18	189	28.5	5.78	76	13.7	12.02	106	25	8.4	58.3	18.3
PO_4	0.03	6.84	1.68	0.03	2.4	0.36	0.03	0.41	0.14	0.51	13.28	3.22
Na	116	379	66	87	248	44	35	56	5	81	186	27
K	1.06	35	8.63	1.3	7.2	0.97	1.65	3	0.37	12.9	25	3.6
Ca	48	199	30	48	248	42	48	96	14.5	52	120	16
Mg	15	151	33	15	88	15	10	29	7.8	17	56	11

*All the values mentioned are in mg/L except for pH (-) and EC (µS/cm); [1]well numbers: Ks1 to ks4, [2]well numbers: Ks5 to Ks9, [3]well numbers: Ks10 and [4]sampling points: WW in Figure 1.3.

The average EC contour map of the watershed (Figure 3.1a) depicts the lower parts of the watershed with high values, where intensive wastewater irrigation is being practiced and subsequently the next higher values in groundwater irrigated area and lower values observed in the upstream peri-urban area. The relationship between EC and other major ionic concentration is also examined to check the wastewater mixing conditions with fresh groundwater (Figure 3.1; Figure S3.1). The relationships between EC with HCO_3, SO_4, Cl, Na, Ca and Mg ions clearly show that the groundwater in the upstream peri-urban area can be characterized as freshwater and is not yet influenced by wastewater. In the EC vs Cl and EC vs Na graphs, it is evident that the sample pattern of groundwater and wastewater irrigated areas are very similar (Figure 3.1). There was a probable mixing of groundwaters from groundwater irrigated area with the return flows from the wastewater irrigated area. The relationship between EC with other major elements shows poor correlation for the samples from the upstream peri-urban area except for Ca and HCO_3, which are moderately correlated (R^2 = 0.38). The relationship for the samples from groundwater irrigated area explains moderate correlation except for Mg, which is of poor correlation (R^2 = 0.03). In case of wastewater irrigated area, the relationship between EC and other major ionic concentrations are strongly correlated except for HCO_3 with poor correlation (R^2 = 0.01) and Ca moderately correlated (R^2 = 0.45), which is almost the reverse case for the upstream peri-urban area.

3.3.1 Hydrochemical characteristics

Plotting Ca+Mg vs HCO_3+SO_4 shows that the groundwater samples from wastewater and groundwater irrigated area are skewed to one side of the 1:1 equiline. This indicates they are influenced by the monsoon rains and silicate weathering (Figure 3.2a). All the samples from the upstream peri-urban area signify the carbonate dissolution. The groundwater samples under silicate weathering are indicative of mineral surface composite activated for dissolution.

Major water type and dominant ionic concentrations of the groundwater is explained in simpler terms by Chadha's diagram (Figure 3.2b). More than 91% of the groundwater samples from groundwater irrigated area are of Ca-Mg-Cl water type, ~7% of Na-Cl water type and only ~2% of Ca-Mg-HCO3 water type, which illustrates the groundwater salinity with permanent hardness (Chadha, 1999). Samples from wastewater irrigated area are a mixture of Na-Cl and Ca-Mg-Cl water types, which represents saline waters. The majority of the samples from the upstream peri-urban area is of Ca-Mg-HCO3 water type, which depicts temporary hardness, resulting from carbonate dissolution.

CAI-I and CAI-II show positive values for the majority of the groundwaters from groundwater irrigated area, which indicates a direct ion exchange between the Na and K in the waters and Ca and Mg in the host rock environment (Figure 3.2c and Figure 3.2d). The samples from wastewater irrigated area and upstream peri-urban area resulted in positive and negative values, which indicates the impact of wastewater and monsoon rainfall, respectively. The return flows from wastewater irrigation play a dominant role for the direct and indirect base ion exchange processes in the groundwater environment.

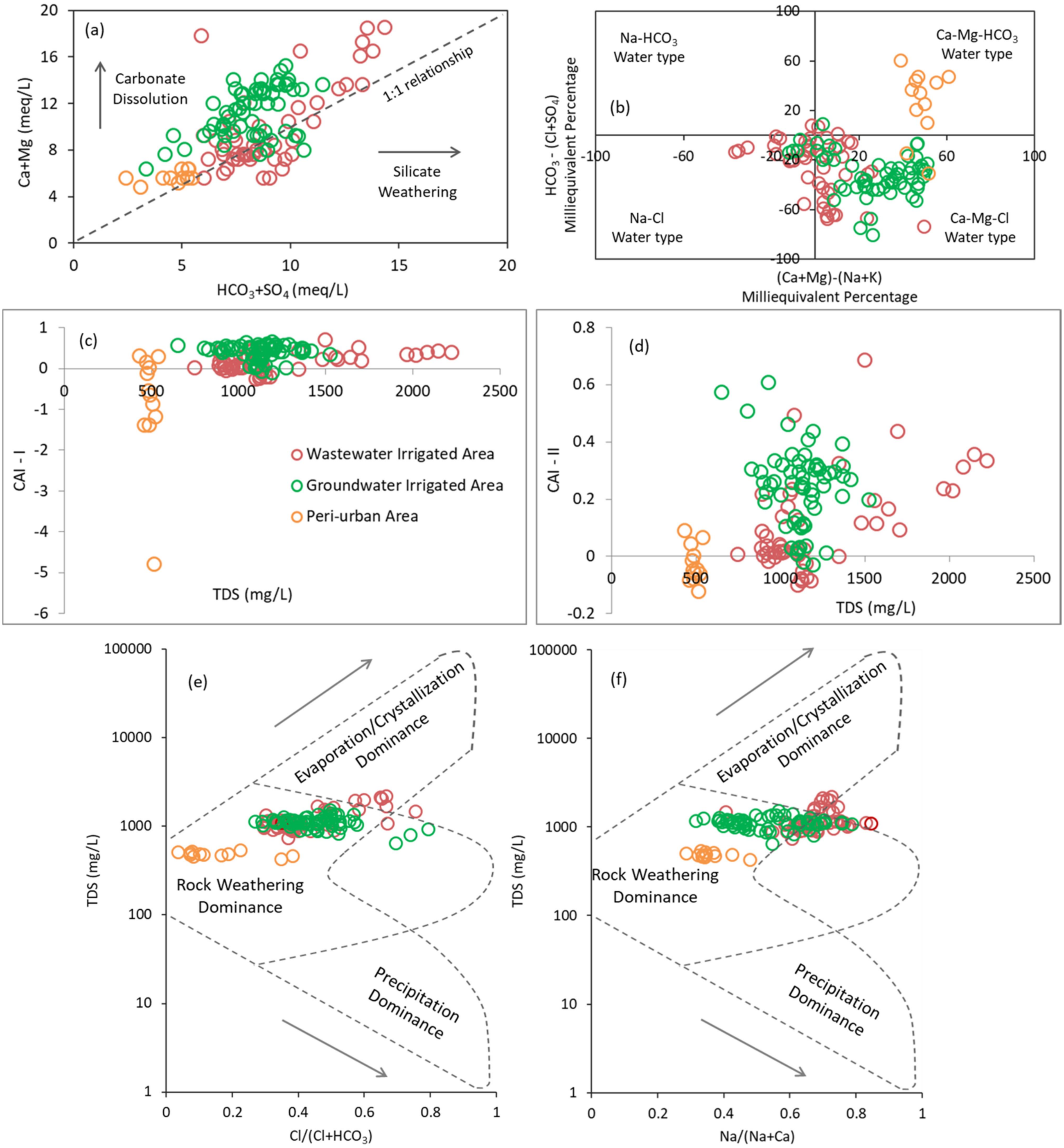

Figure 3.2 Hydrochemical variations: (a) scatter plot of HCO_3+SO_4 vs Ca+Mg, (b) Chadha's classification diagram explaining groundwater type, (c) TDS vs chloroalkaline index I (CAI-

I), (d) TDS vs chloroalkaline index II (CAI-II), (e) Gibbs diagram of Cl/(Cl+HCO_3) vs Log TDS, and (f) Gibbs diagram of Na/(Na+Ca) vs Log TDS.

The mechanisms that control the groundwater chemistry based on the ratio of major cations and anions with TDS are illustrated by Gibbs diagram. The demonstrated three major dominant mechanisms from Gibbs boomerang diagram are a) rock weathering dominance, b) evaporation/crystallization dominance and c) precipitation dominance (Gibbs, 1970). The groundwater chemistry of the watershed reveals that the upstream peri-urban area shows the dominance of chemical weathering and the wastewater and groundwater irrigated areas influenced by the combination of evaporation and chemical weathering (Figure 3.2e and Figure 3.2f). The samples under chemical weathering dominance in the study area are explained by the geochemical processes such as ion exchange, oxidation-reduction and precipitation-dissolution. The samples under evaporation dominance are explained by climate control and precipitated components of wastewater and groundwater deposited as evaporites, which further percolate into the saturated zone and increase the groundwater salinity (Biggs and Jiang, 2009; Jampani et al., 2018; Sridharan and Nathan, 2018).

3.3.2 Determination of Mineral Saturation Indices (SI)

SI values are calculated for minerals halite (NaCl), gypsum ($CaSO_4.2H_2O$), calcite ($CaCO_3$), dolomite ($CaMg(CO_3)_2$) and fluorite (CaF_2) and plotted against their dominant ionic concentrations Na+Cl, Ca+SO_4, Ca+HCO_3, Ca+Mg and F respectively (Figure 3.3). When it comes to mineral saturation, all the groundwater samples show either mineral precipitation or dissolution irrespective of the irrigation or peri-urban area. Attained results suggest that the calculated SI values are greater than zero for calcite (Figure 3.3c) and dolomite minerals (Figure 3.3d), thus groundwater in the watershed is oversaturated for calcite and dolomite, and indicating mineral precipitation. The SI values obtained for halite, gypsum and fluorite minerals are less than zero, and thus indicating groundwater is undersaturated and suggesting dissolution of these minerals (Table S3.1). Even though fluorite is being the dominant mineral in the granitic hard rock aquifer system of the Musi River basin (Reddy et al., 2010), the scatter plot of SI of calcite vs SI of fluorite (Figure 3.3f) indicates calcite oversaturation and at the current stage fluorite mineral is in equilibrium condition or undersaturation in the study watershed. The correlation between fluorite mineral and fluoride is positive and the overall likelihood of fluorite mineral dissolution is controlled by surface reaction, which is a chemical process at the mineral surface.

The role of halite mineral is an important process contributing to the groundwater salinization in the study watershed. The possibility of halite mineral dissolution is controlled by diffusion control reaction, which is a physical process of dissolved components diffusing into the subsurface and then to the groundwater interface.

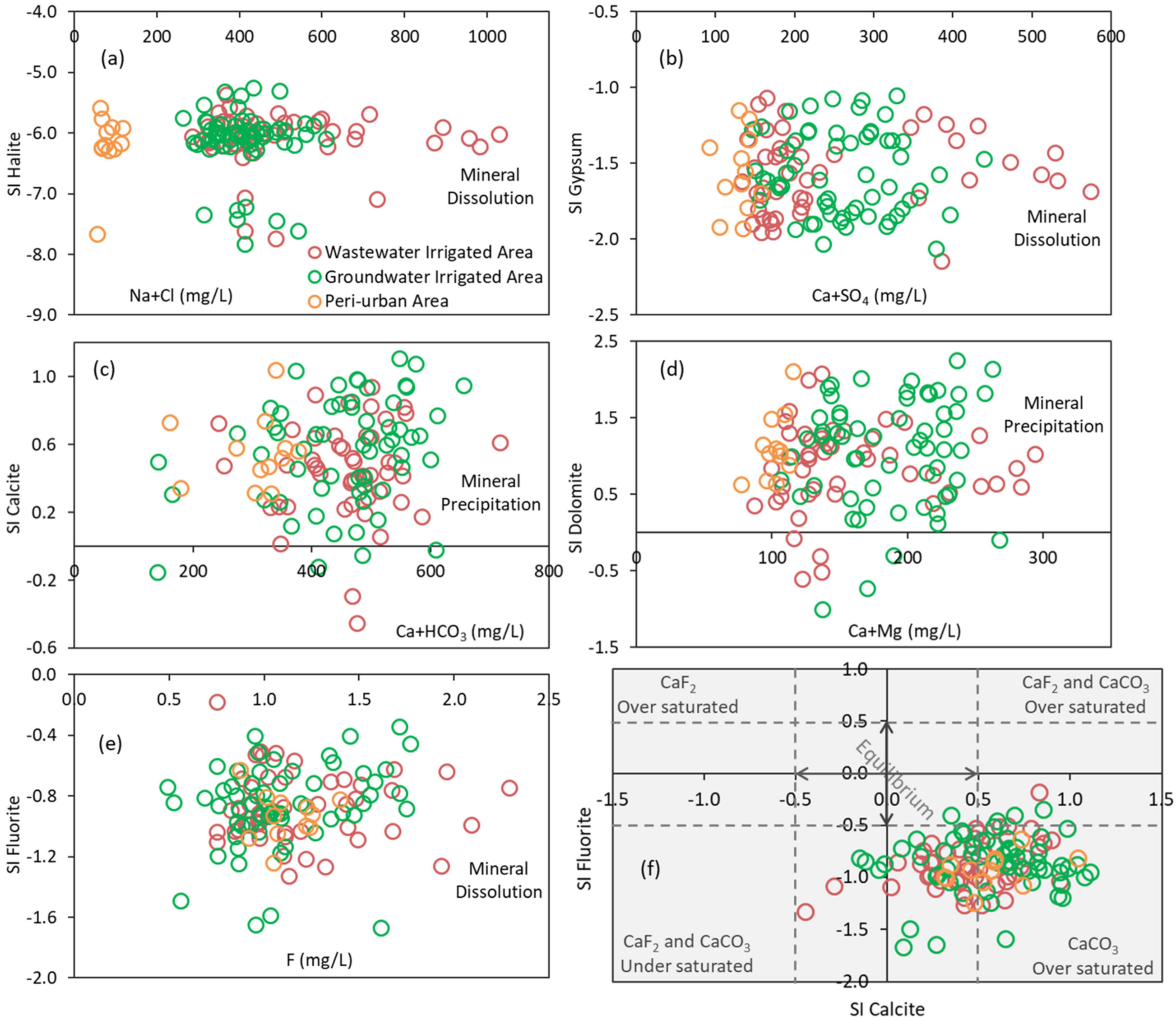

Figure 3.3 Scatter plots of saturation indices of minerals with dominant ions: (a) Na+Cl vs SI of Halite, (b) Ca+SO_4 vs SI of Gypsum, (c) Ca+HCO_3 vs SI of Calcite, (d) Ca+Mg vs SI of Dolomite, (e) F vs SI of Fluorite, and (f) SI of Calcite vs SI of Fluorite.

Gypsum mineral dissolution is explained by the positive relationship between Ca+SO_4 and gypsum, where irrigation with wastewater allows the mineral formation on the surface as a consequence of the evaporation process. Gypsum and halite are to be considered as main sources for accumulated salts over the last four decades in the soils, which are further leached into the groundwater table by monsoon rainfall and continuous irrigation. Carbonate weathering is the major process that explains the supersaturation of groundwater samples and excess of Ca and Mg ions in the water are from carbonate minerals. The mineral precipitation of carbonate and dolomite can also interact with other ions at the surface. Mineral precipitation is controlled by ionic aggregation and specific bonding reactions of calcite and dolomite at the surface. Groundwater samples with higher salinity values are oversaturated with calcite and dolomite minerals (Najib et al., 2016).

3.3.3 Ionic Reactions and Wastewater fraction

In general, chloride does not participate in geochemical reactions or water-rock interactions, thus it is often used as a tracer for salinity, especially for coastal aquifers to identify the degree of saltwater intrusion (Slama and Bouhlila, 2017).

Table 3.2 Descriptive statistics of calculated wastewater fraction and the groundwater quality data of the watershed*

	Wastewater Irrigated Area			Groundwater Irrigated Area			Peri-urban Area		
	Min	Max	SD	Min	Max	SD	Min	Max	SD
F_{WW}	0.67	3.05	0.63	0.73	1.81	0.24	0.0	0.29	0.1
ΔHCO_3	-13.65	1.69	4.16	-7.82	2.19	2.14	-1.1	3.28	1.38
ΔSO_4	-1.59	4.25	1.59	-0.83	2.3	0.61	-0.28	0.23	0.15
ΔNO_3	-1.94	1.15	0.51	-1.16	0.21	0.27	-0.21	1.48	0.43
ΔCa	-6.78	0.51	1.65	-3.99	6.21	2.01	-1.03	2.39	0.92
ΔMg	-3.81	2.75	1.25	-3.59	1.93	1.21	-0.34	1.4	0.62
ΔNa	-10.56	3.41	2.62	-6.49	2.05	2.04	-1.48	0.92	0.69
ΔK	-1.71	0.17	0.5	-1.06	0.41	0.15	-0.17	0.002	0.05

*All the values mentioned are in meq/L except for F_{WW}.

Calculated wastewater fraction in the groundwater samples of the wastewater irrigated area is ranging from 0.67 to 3.05 and the F_{ww} values for groundwater irrigated area are ranging from 0.73 to 1.81 and the lowest F_{ww} values observed in the upstream peri-urban area, ranging from 0 to 0.29 (Table 3.2 and Table S3.2). F_{ww} values more than 1.0 means that the observed chloride values in groundwater are higher than the observed values in wastewater (Table 3.1), which means excess chloride in the groundwater is coming from accumulated salts in the soil with long-term wastewater irrigation. Samples from groundwater irrigated area with high wastewater fraction contents are due to the mixing of fresh groundwater with wastewater return flows from wastewater irrigated area. The ionic deltas (ΔHCO_3, ΔSO_4, ΔNO_3, ΔNa, ΔCa and ΔMg) are plotted against wastewater fraction, which reveals the variations in the groundwater with respect to the hydrological and geochemical controls (Figure 3.4). The ΔHCO_3 values of the groundwater decrease with respect to increase in wastewater fraction (Figure 3.4a).

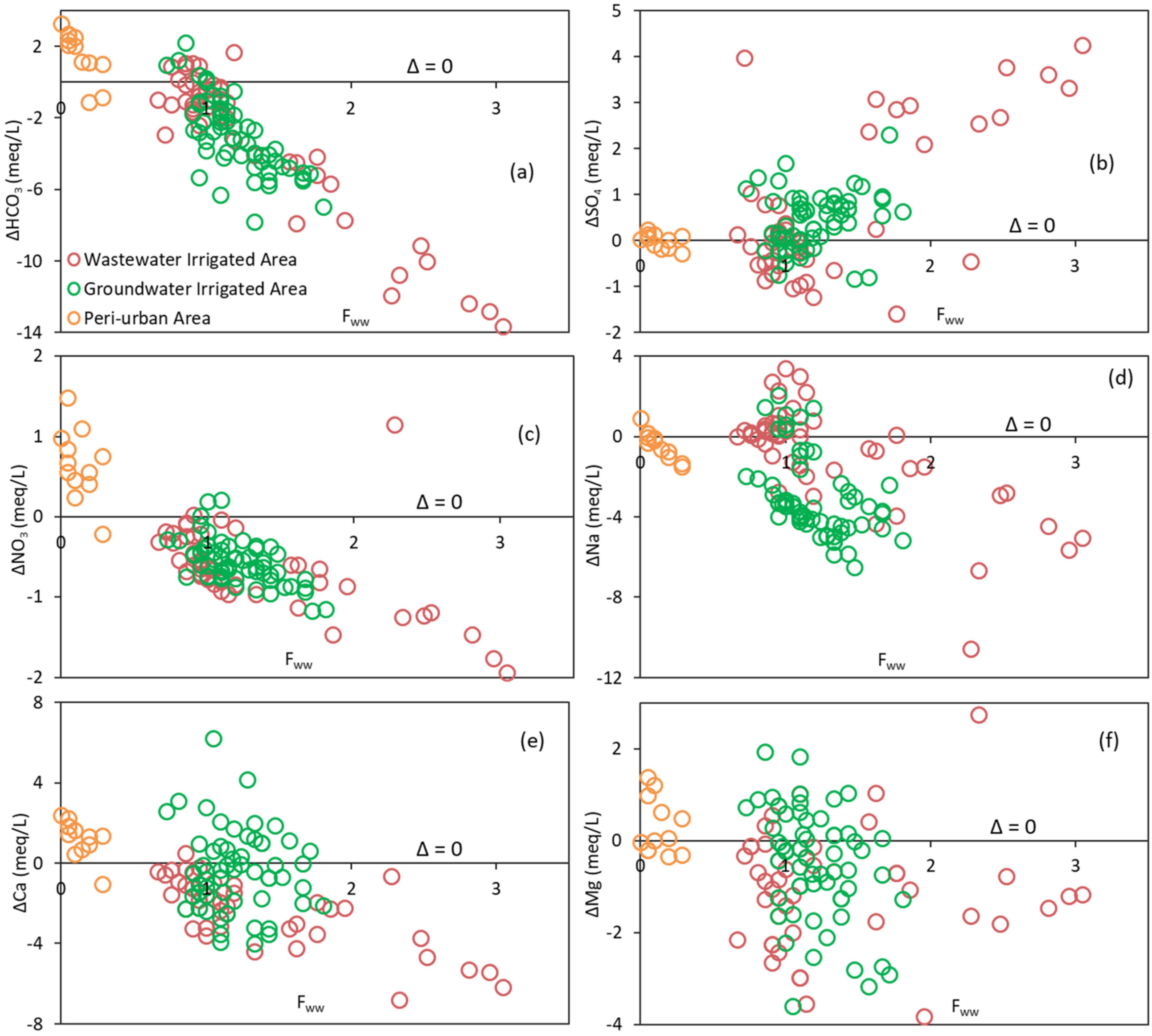

Figure 3.4 Wastewater fraction of the groundwater samples plotted against analyzed ionic deltas: (a) F_{ww} vs ΔHCO_3, (b) F_{ww} vs ΔSO_4, (c) F_{ww} vs ΔNO_3, (d) F_{ww} vs ΔNa, (e) F_{ww} vs ΔCa, (f) F_{ww} vs ΔMg

The ΔSO_4 values for the upstream peri-urban area are almost zero ($\Delta \approx 0$), which explains no influence of sulphate on groundwater enrichment. The ΔSO_4 values increase with the high wastewater fractions; this is due to wastewater fraction samples having higher sulphate concentrations. Groundwater samples in the wastewater and groundwater irrigated areas are enriched by sulphate concentrations from wastewater except in the late monsoon season, which shows sulphate reduction due to rainfall dilution. The ΔNO_3 values indicate that the upstream peri-urban groundwater is enriched by nitrate, whereas negative values of the ΔNO_3 in the wastewater and groundwater irrigated areas indicate a reduction of nitrate relative to the theoretical mixture. Even though higher wastewater fraction values are observed for nitrate, decreasing ΔNO_3 signifies the uptake of the nitrate content by plants in both irrigated areas. The negative ΔNa values of the upstream peri-urban area indicate sodium depletion relative to the theoretical mixture except in the monsoon season. The ΔNa values increase with the increase in wastewater fraction up to 1.2 and then decreases with increase in wastewater fraction. This shows ΔNa is sensitive to hydrological control such as monsoon rainfall. ΔCa and ΔMg values for upstream peri-urban area indicate that the groundwater is enriched by Ca and Mg ions. The ionic deltas of Ca and Mg ions are depleting relative to the theoretical mixture in the wastewater irrigated area, whereas samples from groundwater irrigated area represent a mixture of ionic enrichment and ionic depletion with the influence of several other factors including bases ion exchange, soil enrichment, wastewater irrigation return flows and monsoon rainfall.

3.4 Summary and Conclusions

The current study presents a hydrogeochemical evaluation and groundwater mixing processes of long-term wastewater irrigation influenced peri-urban watershed of the Musi River basin in India. Analytical results of water quality suggest that the groundwater quality deterioration in the downstream of the watershed is attributed to the wastewater irrigation practices. The groundwater qualities in the middle and upstream of the watershed are also affected due to the mixing of return flows from wastewater irrigated area. High salinity in the local aquifer is observed downstream of the watershed and can be attributed to the wastewater used for irrigation purposes. The samples from downstream and middle of the watershed are characterized by fresh to saline water with Ca-Mg-Cl and Na-Cl hydrochemical facies, respectively, whereas upstream groundwaters are of Ca-Mg-HCO_3 water type. Direct and indirect ion exchange processes explain the groundwater salinity in the watershed and the aquifer geochemistry in the watershed is controlled by evaporation dominance and water-rock interactions.

Analysis of saturation indices elucidates that the halite and gypsum mineral dissolution and calcite and dolomite mineral precipitation are the dominant mineralization processes. Sulphate enrichment in the groundwater is responsible for the mineral dissolution of gypsum. The mineralization processes in the watershed are heavily influenced by irrigation practices and seasonal hydrological factors such as evaporation and monsoon rainfall. Ionic delta analysis of the groundwater signifies that cation exchange and carbonate weathering are the dominant phenomena. The enrichment or reduction of cations in the groundwater is influenced by wastewater irrigation and Na is found to be exchanged with Ca and Mg in the local aquifer of the watershed. The rise in sulphate ionic concentration is associated with the contribution of wastewater fraction in the groundwater. Enrichment of the nitrate in the upstream peri-urban area is because of nutrient and wastewater application rates in the watershed, whereas the reduction in ionic delta concentration of nitrate in the wastewater and groundwater irrigated areas is associated with the nutrient plant uptake by crops. Overall, even though wastewater irrigation is a livelihood opportunity, local planners and decision-makers should be mindful of aquifer pollution in the peri-urban systems. For guiding water resource planners towards better groundwater management in the wastewater irrigated systems, this chapter clearly indicates that wastewater is applied in high quantities and this practice needs to be changed, implying efficient irrigation with less water and/or partial wastewater treatment.

4 Multi-functionality and Land Use Dynamics in the Peri-urban Environment

4.1 Introduction

Peri-urban ecosystems are complex landscapes continuously influenced by diverse human-induced changes. Peri-urban spaces are multifunctional land use systems forming a mosaic of built-up and agricultural areas (Padgham et al., 2015). Urban or peri-urban system sustainability and land use changes are often looked at separately, which can lead to equivocal results and fallacious conclusions (Seto et al., 2012). Food security discussions nowadays focus on sustainable approaches to feed the increasing global population, but as emphasized by Seto and Ramankutty, 2016, more attention needs to be paid to the fact that this population growth will be majorly associated with the urban centres. Urbanization and food security are strongly interlinked, and the much-needed focus should concentrate on urban and peri-urban agriculture to supplement the food supply to the urban agglomerations. Globally, it is estimated that around 456 Mha of total croplands are under urban and peri-urban agriculture cultivation (Thebo et al., 2014), and this could be further enhanced if efficient planning is practiced. These systems provide merits and showcase functions of multifunctional land use systems which require integrated management approaches for achieving the social welfare and environmental benefits (Zhang and Schwärzel, 2017).

Urban and peri-urban agriculture is influenced by direct and indirect drivers, including the availability of water, and small-scale industries, land costs and land availability (Kurian et al., 2013; Kurian and McCarney, 2010). These drivers have a significant influence on the local or regional land use change dynamics. Mapping the associated dynamics with water bodies and agriculture farms in urban and peri-urban systems using remote sensing based approaches can provide insightful details to the local policymakers, but often without ancillary data, remotely mapped locations can provide misinformation of the land use systems (Brown and McCarty, 2017). Improved accuracy in mapping the urban and peri-urban spaces helps local dwellers for improved land use management and for policymakers to develop effective policies. In the past two decades, technological advancements of remote sensing field have resulted in high resolution satellite products that have helped to capture the multi-functionality of these transformative spaces. Google Earth's free and open source data provide the composite of several satellite datasets with a high spatial resolution up to 1 m for urban areas and selected test sites (Gorelick et al., 2017).

Google Earth, due to its user friendly accessibility, free access and high spatial resolution imagery, has increasingly been used in a scientific context for understanding social, earth system and environmental processes with anthropogenic impacts (Pulighe et al., 2016). Several studies explored the options to apply Google Earth in earth system science (Fisher et al., 2012; Yamagishi et al., 2010), urban agriculture (Taylor and Lovell, 2012), land use change and landscape processes (Huang et al., 2017; Hu et al., 2013), epidemiology (Chang et al., 2009), climate adaptation (Stocker et al., 2012), habitat quality (Benham et al., 2011), forestry (Dorais and Cardille, 2011; Ploton et al., 2012; Singh et al., 2015), biological and ecological applications (Al-Abdulrazzak and Pauly, 2014). Various case studies around the world utilized Google Earth as auxiliary support for training, classifying and validating land use/cover maps (Gbanie et al., 2018; Hu et al., 2013) and also as ground truth data to validate the Landsat, MODIS and PALSAR derived datasets (Dong et al., 2012; Dorais and Cardille, 2011; Gumma et al., 2017). Irrespective of the high spatial resolution imagery of Google Earth, it was used as a primary data source for land use/land cover mapping only in few studies. Spatial and temporal dynamics of land use change can be assessed/mapped at different levels and scales using Google Earth, but the temporal resolution can be a challenge due to missing or unsuitable images in a time series.

The current study explores the micro-level land use change dynamics using Google Earth data in the peri-urban environment of Hyderabad city in India, which has been the focus of several earlier studies. Gumma et al. (2011, 2017) analyzed the urban and peri-urban agriculture dynamics and their impacts associated with the urban area of Hyderabad, in which regional level land use changes were well illustrated by using Landsat and MODIS imagery. For the city or regional level planners, this type of land use mapping can be quite helpful, whereas, for local planning, mapping with higher spatial and temporal resolution is required (Seto et al., 2012). Similar to the current study area, in many parts of the developing world, peri-urban agriculture is dependent on wastewater from urban centres for irrigation (Amerasinghe et al., 2012, 2013; Mahesh et al., 2015b). In peri-urban Hyderabad, several authors investigated the wastewater irrigation opportunities (Buechler et al., 2002; Mahesh et al., 2015b), food, water, health, energy nexus (Miller-Robbie et al., 2017), health risks (Amerasinghe et al., 2009), impacts on the local aquifer (Jampani et al., 2018) and salinity implications (Biggs and Jiang, 2009) among others. There is, however, still limited understanding of the land use change in peri-urban systems with wastewater irrigation influence.

The current study aims to evaluate the micro-level land use change dynamics of a peri-urban landscape system influenced by wastewater irrigation. Micro-level changes are analyzed including land use type, crop type and irrigation water type used for agriculture. Understanding the temporal evolution of land use changes with a spatially explicit approach can help farmers and planners to better manage the natural resources in an integrated manner that are useful for food production, particularly for the peri-urban areas in developing countries where wastewater irrigation is a common practice.

4.2 Materials and Methods

4.2.1 Google Earth data and processing

The background images data from Google Earth are collected for a 16 year period from 2000 to 2015. A total of 34 images are available from Google Earth database for almost every year except for the years 2002, 2004 and 2007. Out of this available imagery on Google Earth, highest resolution (up to 1 m) images were selected for land use change analysis based on the quality of the image (clear visibility of land features and without cloud cover) for that particular year. To avoid any seasonal discrepancy, only selected the images that are available in the months of April and May, where the images are available for a maximum number of years. A total of nine years (2000, 2003, 2006, 2009, 2010, 2011, 2013, 2014 and 2015) were considered for land use analysis over the 16-year period. The date and year of the Google Earth images used for the land use analysis are mentioned in Table S4.1. Land use features were manually digitized with visual interpretation of the images to assess the land use changes in the watershed. Using Google Earth, one can easily digitize point, line and polygon objects (vectorial elements) representing relevant land use features. These objects are finally saved as Keyhole Markup Language (KML) files (Frankl et al., 2013).

For accuracy, ground truthing was performed with periodic field visits, Global Positioning System (GPS) point data collection and farmers survey (Table S4.2). The field data collection for ground truthing was carried out from 2010 onwards (every year between April to May for each agricultural plot and land use type in the watershed to identify the cropping patterns) to check the accuracy of the maps. The field observations include land use type, crop practice and irrigated water type used for agriculture of a particular plot of the watershed. The farmers in the watershed were interviewed in order to assess the past land use change patterns from 2000 to 2009 (Mahesh et al., 2015a, 2015b), where ground truthing data is not available. The questionnaire for farmers includes questions related to practiced crop type and source of irrigation water used for agriculture for every year in the last decade (Mahesh et al., 2015b; Starkl et al., 2015). Farmer's survey data is used for validation because farmers in the watershed area are practicing agriculture since a very long time (five decades), they were basically aware of the historical land use types in the previous years. Farmers´ responses and ground truth information were assigned to the particular agricultural plot of the Google Earth image. With the thorough mapping and the image validation process, detailed land use maps are generated for nine years between the years 2000 and 2015.

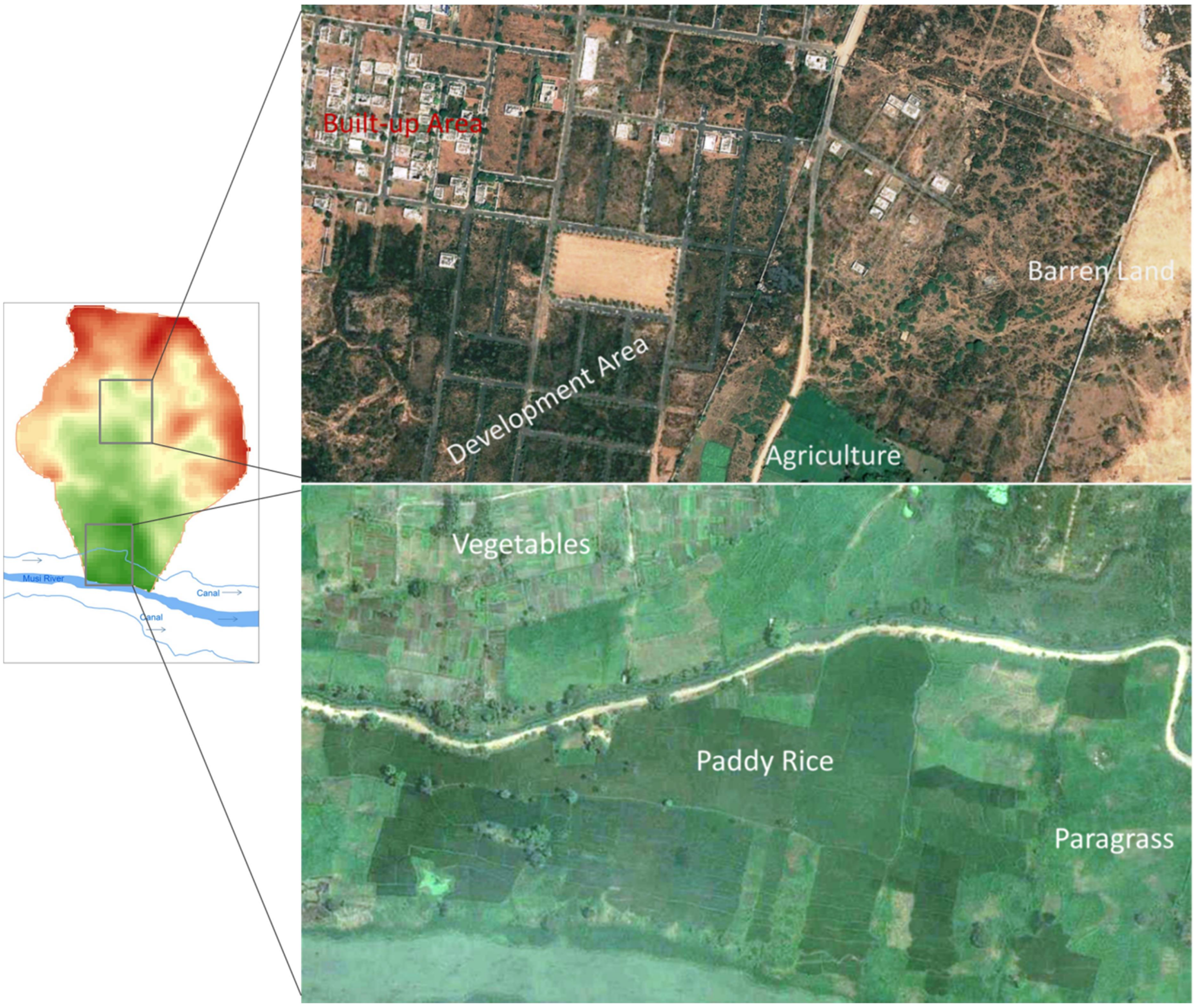

Figure 4.1 Identification of different land use classes and crop classifications from Google Earth images for spatio-temporal mapping of the watershed.

Land use and crop types were assigned based on Google Earth images following standardized procedures from satellite image interpretation (i.e. shape, shadow, tone/color, texture, pattern, height/depth, site/situation/association). The image interpretation was exported as KML format polygons or polylines to ArcGIS 10.5.1 for further analysis. To avoid any spatial distortion of the exported Google earth images to ArcGIS, the images are georeferenced using GPS control points to the Universal Transverse Mercator (UTM) 44^{O} N World

Geodetic System (WGS) 1984 coordinate system. At a later stage, these images were orthogonally rectified in ArcGIS. While mapping the images, a smaller scale (1:500) is used to delineate different land use classes and crop types. Even though zoom in and zoom out options are used to check out the neighbourhoods of the image, still a constant scale is maintained in the entire mapping process. Based on the image classification, different land use types: agricultural, built-up and development areas were mapped (Table 4.1 and Figure 4.1, upper panel). The unused land was assigned to barren land, defined as either empty land or with occasional grass patches depending on the season. In general, mapping different crop types using any available satellite imagery can be complex, but by using Google Earth high resolution imagery, we can easily identify and classify the crop types (Figure 4.1, lower panel). It is easy to identify the crops like paragrass (light green), paddy rice (dark green) and vegetables (small plots with mixed green and brown patches), whereas mapping any other crops requires the support of ground truth data in the study area context. As vegetables is a perennial crop in the watershed area, the brown color patches in the vegetable plots represent the crop cutting for those particular smaller plots in the imagery time (Figure 4.1). Wetlands with reed pond or elephant grass area texture are also easy to identify in the Google Earth images. In general, over the years, the road networks remained constant in the watershed except for a new road, which was built alongside the canal in 2008 especially for crop produce transport and to gain easy access to the agricultural plots. All these micro-level changes over the 16 year period were mapped manually to create high accuracy maps for the peri-urban watershed.

Table 4.1 Classification scheme of the land use changes in the micro-watershed

Dynamics Category	Land Use Class Name	Description of the Land Use Class
Major land use	Built-up area	Existing housing and industrial infrastructure
	Development area	Planned plots for built-up area development
	Agriculture	Croplands irrigated with GW and WW
	Barren Land	Empty lands with occasional shrubs
	Wetland	Natural reed pond with elephant grass
Crop	Paragrass	Fodder crop and grows all around the year
	Paddy Rice	Food crop and grows ~300 days in the year
	Vegetables	Food crop and grows all around the year
	Other Crops	Commercial crops such as chilli and cotton
Irrigation System	GW Irrigated	Agricultural area under groundwater irrigation
	WW Irrigated	Agricultural area under wastewater irrigation

Built-up, agriculture and development areas are classified under major land use dynamics Land use changes of the major crops in the watershed are classified under crop dynamics. Changes of the groundwater and wastewater irrigated areas are classified under irrigation system dynamics (Table 4.1). Groundwater irrigation and wastewater irrigation areas are mapped using field observations, farmers survey and attributes assigned to the mapped crop areas in the Google Earth images.

4.2.2 Landscape change modelling

Landscape change modelling requires different sources of information in addition to land use maps of different years. Information such as Digital Elevation Model (DEM), soil type, slope, geology, accessibility to the roads and accessibility to the canal (wastewater) are integrated as supporting input data. Land use change modelling typically includes the following procedure: (1) land use change analysis between the given years, (2) determining the potential driving forces of the land use change, (3) modelling transition potential under different transitions or change influences, (4) prediction of future land use based on the given transitions, and (5) validation of the model for given year with observed data (Ibrahim et al., 2016).

The Land Change Modeler (LCM) of Terrset IDRISI software (https://clarklabs.org/) was utilized for analyzing the transitions between 2000 and 2010 and for future land use change predictions for the year 2030. The two years 2000 and 2010 have the same specifications, such as colours, legends and spatial characteristics to observe and understand the nature of changes in the peri-urban micro-watershed. Before exporting the maps from GIS to land use change modelling platform, the mapping results are converted into raster format. These mapped changes were utilized to identify prevalent transitions and further generate the probability matrix and transition potential maps. The modelled transition maps between different land use categories are further used as inputs for land use change modelling prediction. Transition potential maps are critical inputs for land use change model prediction. Predictive variables such as DEM, slope, soil type, geology, wastewater (canal) accessibility and access to the roads are used in the modelling process.

Built-up and development areas are merged into the built-up area for transition modelling and model prediction as both represent the peri-urban built-up area. There are multiple models available for transition modelling: multilayer perceptron (MLP) neural network, logistical regression and simulation weighting (SimWeight). Here MLP neural network model is used as it is a feedforward neural network and better suited for smaller study areas model prediction compared to other land use change models (Mas et al., 2014). The predictive power of modelled transition from 2000 to 2010 is used for predicting the land use map of 2015. Further, the original mapped data of 2015 using Google Earth is compared with the predicted map for validation. Model validation of observed versus predicted is calculated using percent error estimation for each land use type (Ibrahim et al., 2016). Following the successful validation of the land use map of 2015, the model transitions of 2000 to 2010 are used to predict the land use change scenario for the year 2030.

4.3 Results and Discussion

4.3.1 Land use change analysis

The land use change analysis based on Google Earth data for the years 2000 to 2016 resulted in identifying three types of key dynamic transformations in the context of the peri-urban watershed: major land use, crop and irrigation dynamics. In total eight major classes of land use, namely, built-up area, development area, roads, wetlands, paddy rice, paragrass, other crops (commercial crops such as chilli and cotton) and vegetables are mapped using Google Earth images (Figure 4.2).

4.3.1.1 Major land use dynamics

Built-up and development areas are continuously increasing in the watershed, and the observed peak in built-up and development areas in 2003 coincided with the expansion of the greater Hyderabad city boundary, which raised the land prices and resulted in the conversion of agricultural landscapes into built-up areas (Gumma et al., 2011). The agricultural area almost remains constant over the 16-year period (Figure 4.2). From the examined mapping details between the years 2000 and 2015, it was clear that barren lands are converted into new built-up and agricultural areas, compensating the losses to the built-up area (Figure 4.3). The northern regions of the watershed were being developed faster than the other regions, being closer to a major road network. The major and moderate land use shifts were observed in the years 2003, 2006, 2010 and 2014. It was uncertain to track the reasons behind the shifts between built-up and agriculture land use, therefore further looked at the details of crop and irrigation system dynamics.

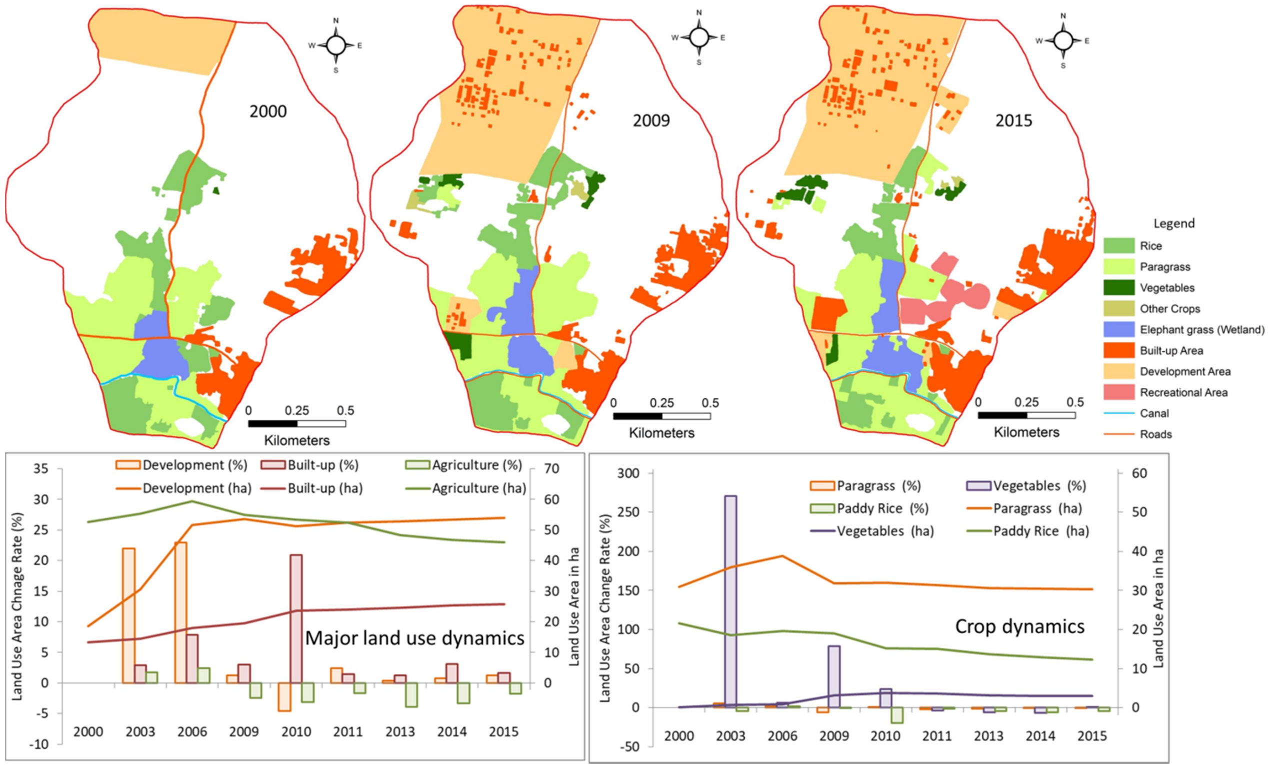

Figure 4.2 Land use change dynamics in the study micro-watershed. Upper panel is showing the spatial dynamics of the land use change for the years 2000, 2009 and 2015. The white part of the watershed is empty land or barren land. The lower panel is showing temporal dynamics in different settings: Major land use (three main use types) and crop (three main crop systems).

4.3.1.2 Crop dynamics

Within the watershed, crop dynamics are significant, mainly between the crops paddy rice, paragrass and vegetables. The temporal change patterns of paragrass and paddy rice are similar but in the opposite direction, where paragrass patches increased relative to paddy cultivation. The decrease in paddy rice cultivation is associated with the salinity implications associated with long term irrigation with wastewater, which resulted in low crop yields (Biggs and Jiang, 2009; Mahesh et al., 2015b). Increasing paragrass cultivation is associated with the demand for paragrass by the local dairy industry in the area, which supplies the milk to Hyderabad city (Buechler et al., 2002). Other factors that may be influencing the paragrass cultivation are high crop yields at shorter time intervals that result in quick profits, lower transportation costs and less labour requirement (Buechler et al., 2002). Thus, it can be expected that paragrass cultivation may increase in future also due to quick profits of the crop and less attention required by the farmer.

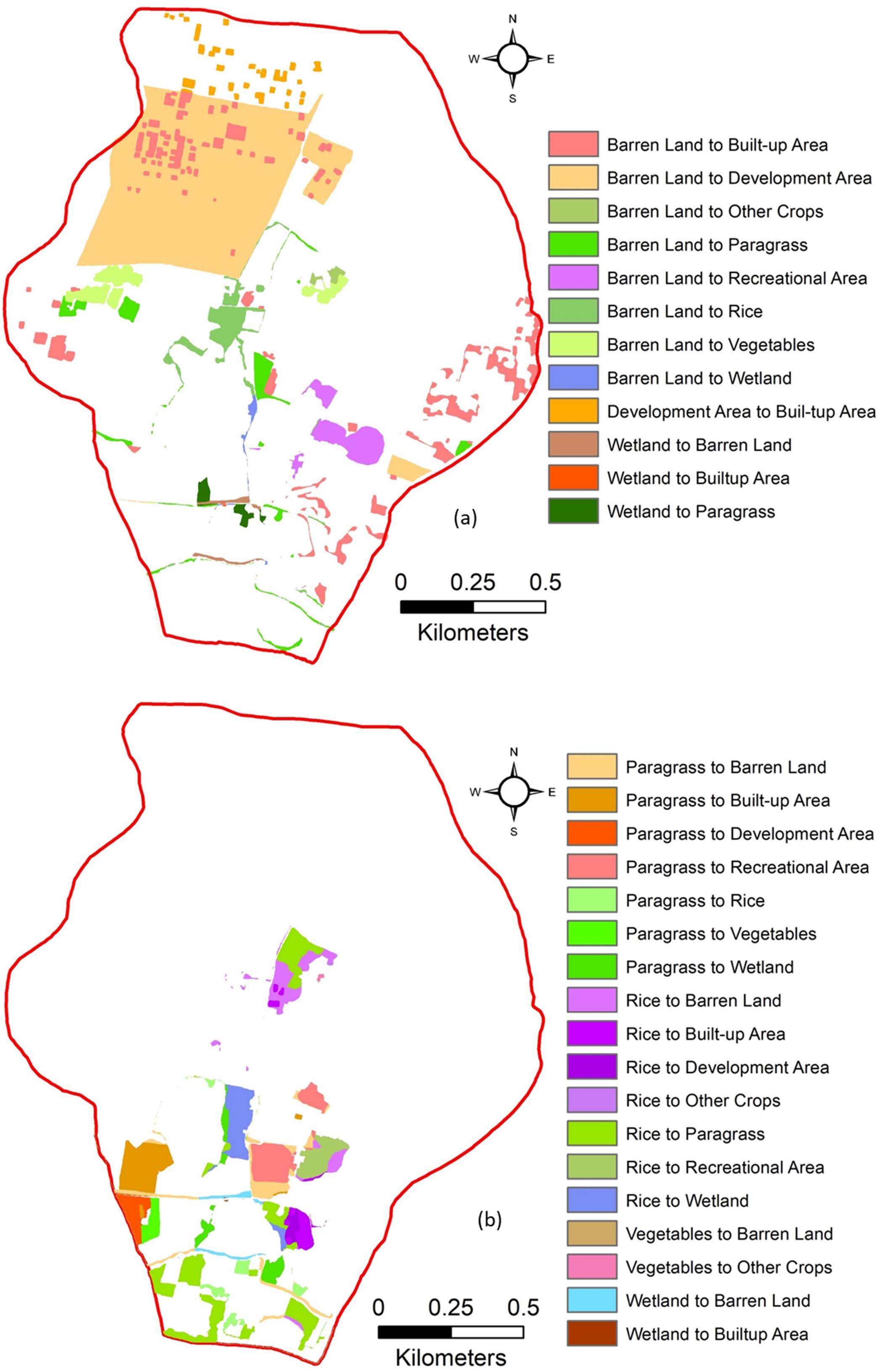

Figure 4.3 Observed land use changes between the years 2000 and 2015, (a) and (b). The land use classes are shown here irrespective of the change detection observed between the classes. For example, there were no changes observed between the built-up area to barren land and barren land to the wetland.

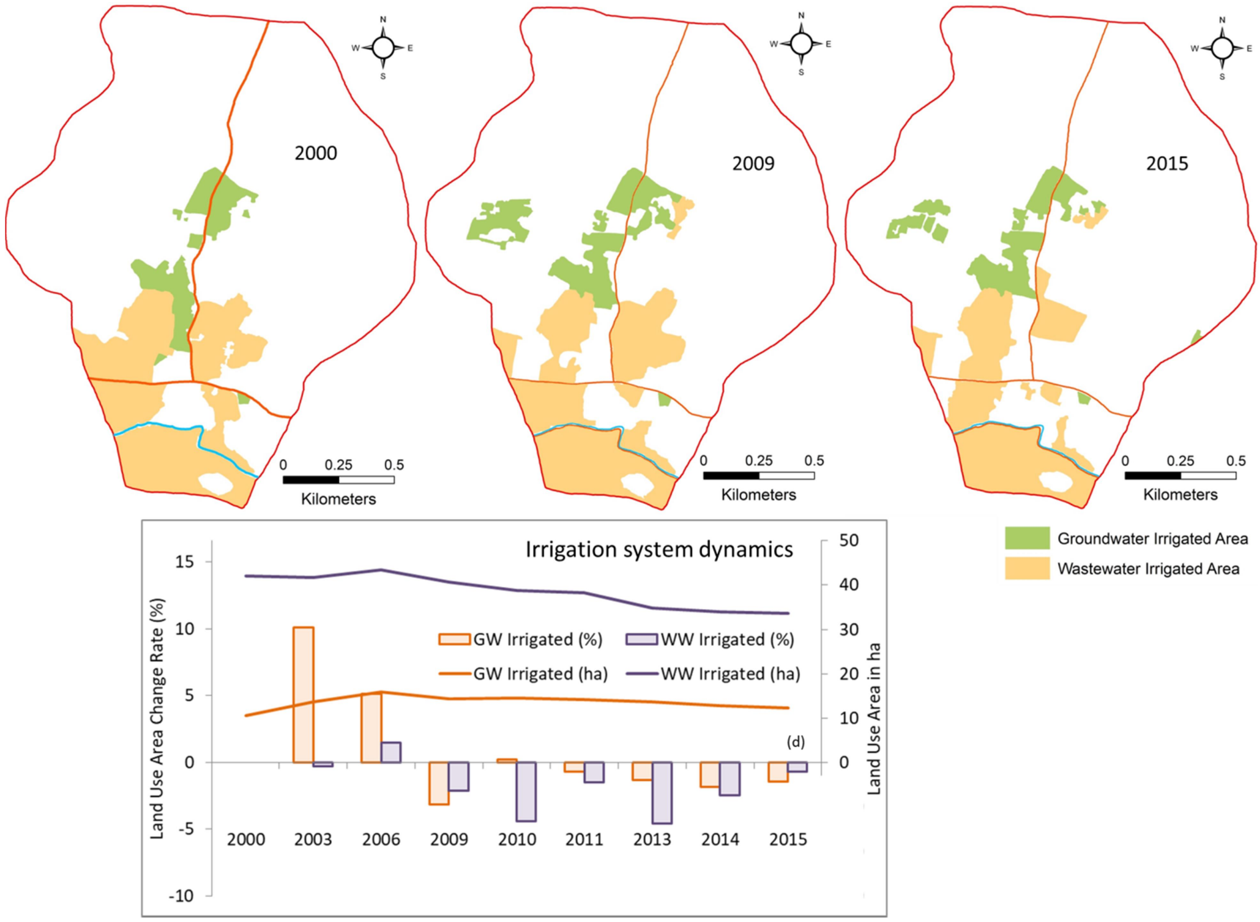

Figure 4.4 Irrigation system dynamics in the study micro-watershed. Upper panel is showing the spatial dynamics of the irrigation type land use change for the years 2000 and 2010. The lower panel is showing temporal dynamics of irrigation system (groundwater, GW vs wastewater, WW).

Vegetable cultivation in the area started around 2006 (Figure 4.2, lower panel) to supply the food for urban markets, which yielded high profit margins for local farmers (Mahesh et al., 2015b). Even though vegetables have a high profit and market value, decrease in the vegetable growing area after the year 2012 is associated with labour costs, continuous attention required for the crop cultivation and negative perception of using wastewater for agriculture by urban dwellers (Amerasinghe et al., 2013). The risks of vegetable production with wastewater irrigation, especially leafy vegetables can be mitigated by practicing safe reuse approaches for irrigation (Amerasinghe et al., 2009; Nabulo et al., 2012; WHO, 2006).

4.3.1.3 Irrigation system dynamics

Wastewater availability for irrigation has a significant influence on agricultural land use shifts in the watershed. Two irrigation systems, wastewater and groundwater, exist in the watershed and both irrigation systems have unique characteristics. Wastewater irrigation has been practiced in the area since the last 40 years and groundwater irrigation has been practised in the watershed for more than 100 years. Both types of irrigated areas are almost constant over the years with minor shifts in the cropping area (Figure 4.4). After 2003, there is a drop in wastewater irrigated area, which tends to imply the same shift in paragrass area from crop dynamics graph (Figure 4.2, lower panel). Over the years of observation, only one shift is observed from agriculture to built-up area, which is from paragrass to built-up area (Figure 4.3). It is assumed that this shift may be associated with the increasing land costs, but it is difficult to track the original reasons behind the minor land use shifts because of the complexity of social dynamics and demography of the peri-urban systems.

4.3.2 Landscape modelling

Resulting from the spatio-temporal mapping of land use and land cover, seven significant transitions were considered based on observed major land use shifts: paddy rice to paragrass, barren land to paragrass, barren land to paddy rice, barren land to vegetables, barren land to built-up area, paragrass to built-up area and paragrass to vegetables (Figure 4.5). Accuracy assessment is provided for the case of the land use change modelling process due to the fact that we performed an exhaustive mapping of all the study area. Transitions from barren land to other land use types are inevitable with urbanization, which demands the need for urban infrastructure and dwelling places. Other transitions including paddy rice to paragrass, which is a farmers´ choice because of increasing soil salinity (Jampani et al., 2018), and paragrass to vegetables or built-up area are also plausible choices. These seven transition submodels were used as input for the land use change prediction model, and all the individual transition submodels achieved a minimum accuracy of 70% with the MLP neural network approach.

Predicted land use results for the year 2015 were validated with the observed land use results. The estimated error for model validation for each land use class is below seven percent. It is expected that the dominant land use classes are expressing higher errors compared to other land use classes (Han et al., 2015) (Figure 4.6). The barren land and built-up areas are showing the highest errors with 6.11% and 4.65%, respectively, followed by paragrass and vegetables cropping areas showing errors of 1.11% and 1.01%, respectively. After the validation of the model, the land use mapping results were predicted for the year 2030, which illustrates an increase in built-up areas (Figure 4.7). Overall, the transition potential map explains that there is a huge potential for increasing both agriculture and built-up areas in the future. However, it seems that the land use prediction tends to result in increased built-up areas, which is a combination of both built-up and development areas. Between the years 2000 and 2015, the built-up and development areas together grew by 150%, whereas agricultural areas decreased by 12%.

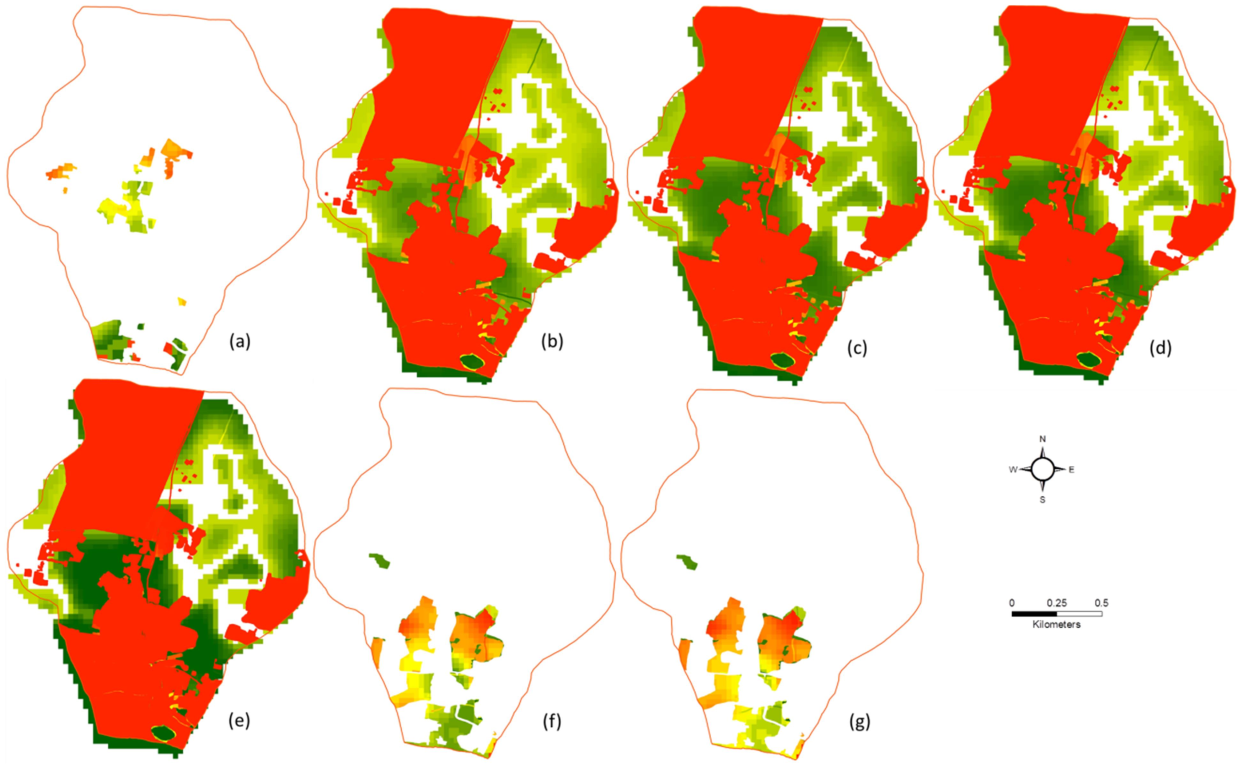

Figure 4.5 Mapped possible transitions between different crops and land use classes for future landscape change. Transitions maps showing; (a) Paddy Rice to Paragrass, (b) Barren land to Paragrass, (c) Barren land to Paddy Rice, (d) Barren land to Vegetables, (e) Barren land to Built-up area, (f) Paragrass to Built-up area, (g) Paragrass to vegetables. The colour scale range for the maps shown is from green to yellow to red, where green is a possible transition, yellow means transition may be possible or not possible, and red means either transition impossible or existing land use. White areas represent barren land or other land use types, which did not participate in the model transition.

If this growth rate of the built-up area continues, the predicted results would indeed hold true in the future. These changes are taken as the basis for the model predictions. It is clear that built-up areas are encroaching on agricultural areas, and there is a moderate chance that some of these irrigated areas can be also turned into built-up spaces for the peri-urban population. These changes are prompted by less attention to agricultural spaces and high land prices as progression of urbanization might be the reasons behind the big shift to built-up areas. The urbanization process can either push people to the peri-urban spaces or migrants who come looking for jobs can also occupy peri-urban spaces. This shift can be a threat to peri-urban agriculture in the future as it is a livelihood practice for the farmers in the region. Local governments are looking at policy options to improve and support the urban farms and peri-urban agriculture but institutional support for implementation is not in place. With the quantity of wastewater and groundwater that is available, peri-urban agriculture in the region can be easily sustained and can contribute to food security of the Hyderabad urban agglomeration.

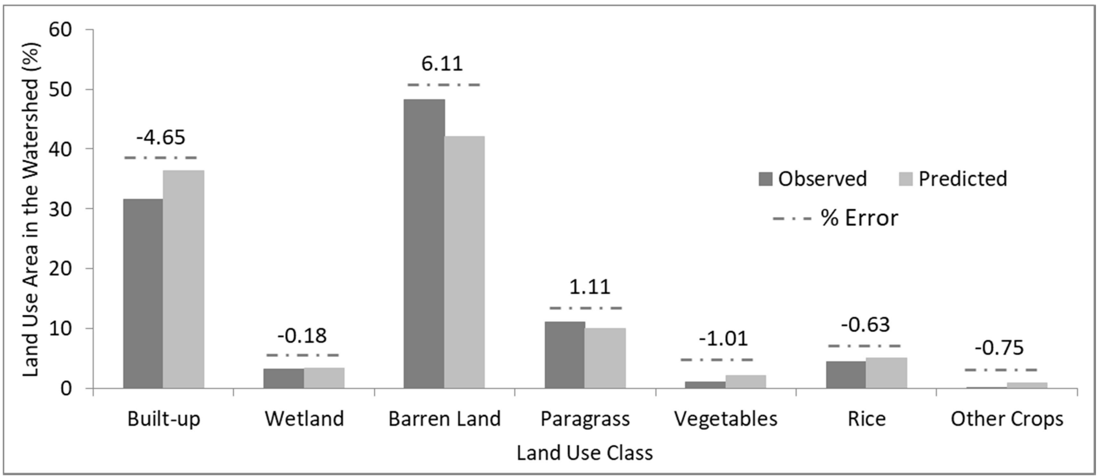

Figure 4.6 Estimated percent error for model validation of the observed land use from Google Earth vs predicted results from land change modeler for the year 2015.

4.3.3 Limitations of Google Earth data

Google Earth has some limitations with respect to large scale applications because of its inconsistent quality of images and intermittent data availability (Pulighe et al., 2016; Yu and Gong, 2012). Coincidentally for the current study area, high quality images were available with Google Earth for observing micro-level land use changes in the peri-urban context. Even though Google Earth provides high resolution images especially in the urban and peri-urban areas, the reliability of these images raises questions due to horizontal accuracy and limited background information or metadata released by Google (Pulighe et al., 2016). The current study utilized the local farmers' knowledge and thorough ground truthing of the watershed over the years to overcome this situation, which might not be possible in every case for mapping micro-level changes as it requires a lot of field work and person hours for validation.

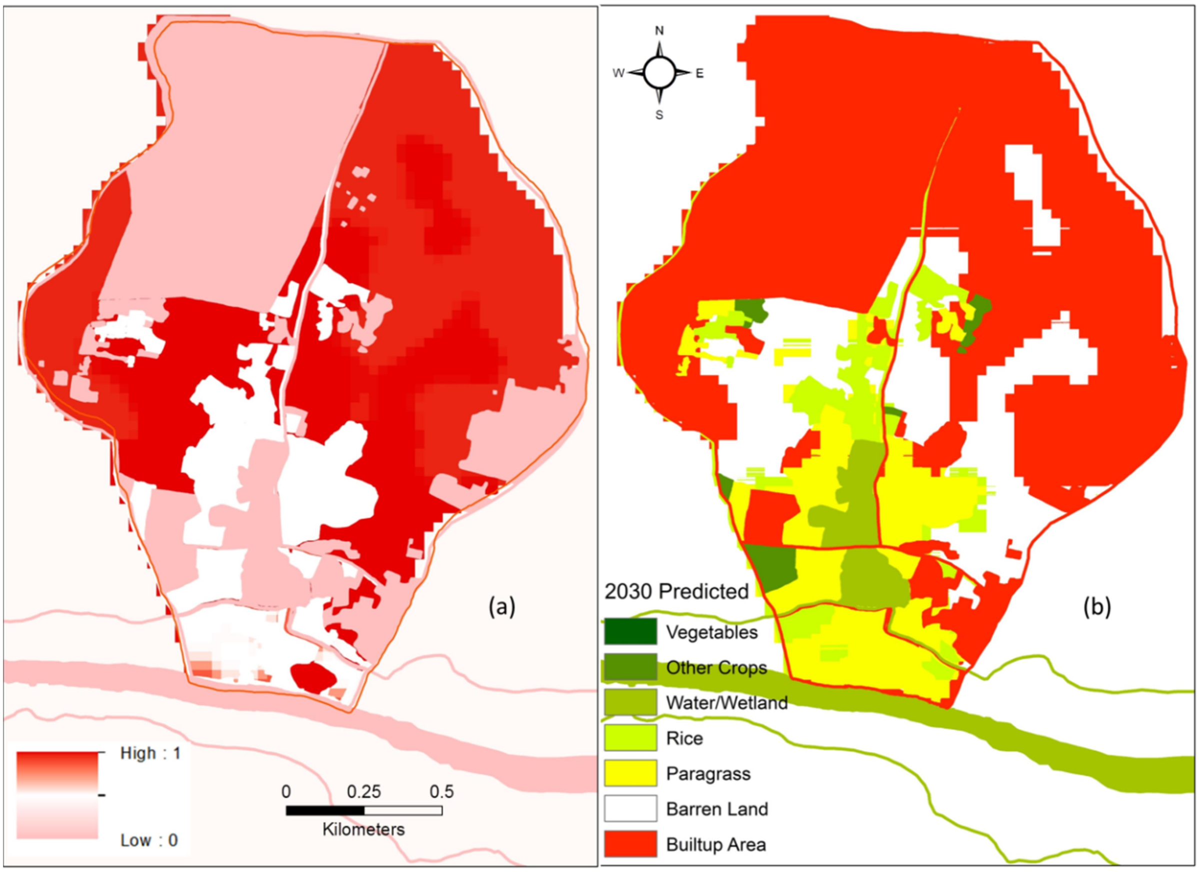

Figure 4.7 Landscape modelling (a) transition potential and (b) predicted land use for the year 2030. High (1) transition potential refers to possible for land use change and low (0) means transition of the land cover not possible.

4.3.4 Implications of peri-urban agriculture

This chapter elucidated the shifts in land use and their interrelations with wastewater irrigation. As the peri-urban spaces are the most dynamic compared to urban or rural areas, future research should focus on sustainable land use planning by understanding the social and economic factors. Even though vacant lands in the peri-urban systems offer ample opportunities for agricultural activities, these opportunities are not visible as expected because of various factors including land costs, farmers disinterest, water accessibility, the conundrum with wastewater (Amerasinghe et al., 2012, 2013) etc. The health risks associated to food production with wastewater appear to be manageable in principle (Starkl et al., 2015, 2013), but the issue of social acceptance remains. The second issue with wastewater irrigation is salinization, resulting in reduced soil fertility and productivity and environmental health. The observed shift from paddy rice to paragrass can be associated with this factor. This particular issue is quite site-specific, e.g. depending on soil characteristics and irrigation history. Whether the predicted shift to paragrass indeed would materialize depends on the actual salinization trajectory on single plots and many other, mostly socio-economic factors, which are not reflected in the land-use change model.

From the existing research, it was not clear what type of land use policy measures should be required to protect the peri-urban agriculture. Using sustainable irrigation practices for peri-urban agriculture may also improve the environmental conditions of urban agglomerations (Goldstein et al., 2016). As the cities are the biggest consumers, shifting food production closer to the places of high demand (e.g. urban and peri-urban farms) can aid in reducing greenhouse gas emissions and mitigating climate change (Eigenbrod and Gruda, 2015). The planning process in the urban and peri-urban areas should include agriculture as the priority in the vacant or abandoned places with the supporting policies to increase environmental sustainability (Oda et al., 2018). Developing integrated management plans for urban and agricultural systems in the nexus dimension considering reuse of wastewater, cost effectiveness and stakeholder interests can be a viable option for food production and also for land and water resources management (Amerasinghe et al., 2016; Hülsmann and Ardakanian, 2018; Kurian, 2017; McClintock et al., 2016; Miller-Robbie et al., 2017).

4.4 Conclusions

In this chapter, an attempt has been made to analyze the peri-urban land use system dynamics using high spatial resolution imagery of Google Earth and land use change modelling. The observed micro-level land use changes in the peri-urban watershed illustrate that there is an unintended competition between the built-up areas and agricultural areas. This chapter concludes that with the increasing urban pressures, new built-up areas are encroaching on the agricultural landscapes. On the other hand, new agricultural plots and built-up areas are being widely developed in the peri-urban barren lands. Availability of wastewater plays a critical role in continuing the agricultural activities despite the urban pressures. The crops produced find their way to the urban markets to feed the urban dwellers. No major changes are observed in the area under groundwater irrigation, but there is a shift in the choice of crops depending on the urban demands. Paragrass is the major crop in the watershed, and it is extensively irrigated by wastewater flowing into the Musi River from the Hyderabad city. Even though vegetables fetch a high price, fewer areas are being cultivated, due to negative impacts of wastewater irrigation. In fact, negative perception does not affect the paragrass cultivation as it is a direct feeder to the dairy industry, which further supplies the milk produced to the urban markets. Decreasing paddy rice cultivation over the years is linked with the salinity of the soils and low crop yields due to long-term wastewater irrigation.

The predicted land use change modelling for the year 2030 suggests that there is a greater chance of barren lands being converted to the development and built-up areas. Even though agricultural areas exist in the future, built-up areas will likely be dominant because of the increasing land prices. The developed maps with micro-level changes can be useful for decision makers and farmers to decide on the future practices of peri-urban agriculture. These results are site-specific depending on factors topography, proximity to the city, history of wastewater irrigation etc. and assuming similar developments in other peri-urban settings could be easily misleading. Other peri-urban systems can replicate this research methodology to observe the intense changes for controlled peri-urban planning. Local decision makers should take necessary actions to preserve the peri-urban agriculture as it is directly aiding the food production for increasing urban population and supports the livelihood of peri-urban farmers.

5 Land Use Change Impacts on Groundwater Recharge Dynamics

5.1 Introduction

In many parts of the world, groundwater is the main resource for food production and domestic and drinking water supply (Grönwall, 2016; Jankowfsky et al., 2014; Minnig et al., 2018). Increasing population trends certainly put pressure on the aquifers, particularly in urban and peri-urban areas. Anthropogenic activities and rapid land use changes in urban and peri-urban areas often influence the local groundwater recharge dynamics (Kadyampakeni et al., 2017; Sekhar et al., 2017). Construction of impervious surfaces such as buildings, roads, industrial settings etc. decreases the infiltration capacity of the surface and further influences the depleting trends in groundwater levels (Bonneau et al., 2018), while pervious surfaces such as parks, home gardens, urban and peri-urban agriculture etc. increases the recharge capacity (Packialakshmi et al., 2011). Human well-being in urban and peri-urban systems is greatly influenced by groundwater recharge, which is a hydrological input to the complex socio-ecological system (Han et al., 2017).

Urban and peri-urban agriculture (UPA) is a very important land use activity, and often its existence depends on various factors including land prices, labour availability, market prices of agricultural goods, positive persistency of urban dwellers etc. (Minhas and Samra, 2004; Thebo et al., 2014). Even though many factors influence the UPA including the fierce competition from built-up areas, it continues to exist in every urban agglomeration of the world because of the demand from urban markets, economic benefits and livelihood opportunities etc. (Amerasinghe et al., 2013; Saldías et al., 2016). In many parts of the developing world, these peri-urban agricultural areas are irrigated with wastewater or urban sewage (Qadir et al., 2010). In the quest for sustaining livelihoods, farmers are dependent on the reuse of these urban wastewaters for agriculture. Farmer’s choice of untreated or partially treated wastewater for irrigation is often perceived negative due to environmental or human health impacts (Bradford et al., 2003; Deng et al., 2018; Drechsel et al., 2010; Khalid et al., 2017). Safe use of wastewater for agriculture and using optimal water requirements for the crops is a viable option for sustainable food production in the urban and peri-urban areas, but often farmers are not aware of sustainable practices (Amerasinghe et al., 2013; Hettiarachchi and Ardakanian, 2016b; Qadir et al., 2010; WHO, 2006).

In the urban and peri-urban centres, it is often reported that the groundwater levels are declining at alarming rates (Minnig et al., 2018) and recharge rates per year are not able to cope with the levels of excessive and uncontrolled pumping rates (Barron et al., 2013). Governments are making efforts to increase the greener spaces in the urban and peri-urban areas for socio-ecological benefits and to control the groundwater depletion (Foster and Gun, 2016; Grönwall, 2016; Reddy, 2012). Several research studies provided evidence that the increase in pervious and greener surfaces can increase the recharge rates (Al-qudah et al., 2017; Barron et al., 2013; Han et al., 2017; Minnig et al., 2018; Turkeltaub et al., 2015). In urban agglomerations, the amount of recharge to the local aquifer is highly dependent on land use shifts (Minnig et al., 2018; Schirmer et al., 2013). In general, changes in the land use of rural areas over the years are often minimal, whereas, in the urban and peri-urban landscapes, these land use changes can be intense. These intense shifts in land use certainly influence the groundwater levels and conditions (Han et al., 2017; Schirmer et al., 2013). In recent research (Barron et al., 2013; Benz et al., 2016; Grönwall, 2016; Minnig et al., 2018; Packialakshmi et al., 2011), there were some efforts to understand the urban pressures on the hydrogeological conditions. Even though advanced groundwater modelling approaches are developed since the 1960s, it is still very complex to understand the land use change impacts on urban hydrogeology (Han et al., 2017; Schirmer et al., 2013). This is due to the fact that besides land use change various further factors need to be considered, e.g., wastewater irrigation influences and hydrogeological conditions in the urbanization context.

To understand the groundwater recharge dynamics in a wastewater irrigated peri-urban agriculture system of the Musi River basin in Southern India, the current chapter aims at assessing the spatio-temporal land use change impacts on the local aquifer in Kachiwani Singaram Micro-Watershed (KSMWS). Many case studies around the world and in the Musi River basin area are reported the impacts of wastewater irrigation on groundwater, in terms of contamination (Candela et al., 2007; Fridrich et al., 2014; Gallegos et al., 1999; Jampani et al., 2018; Perrin et al., 2010). But there were no studies aiming at understanding the wastewater irrigation impacts and its contribution to groundwater recharge considering temporal changes in the peri-urban landscape. This chapter provides insightful knowledge to the urban and peri-urban dwellers, local farmers and decision makers, how the intense changes in the built-up and agricultural areas are influencing the groundwater recharge conditions.

5.2 Materials and Methods

Groundwater observation wells were monitored for water level data from 2010 to 2015. The groundwater heads data collection from observation wells varies depending on the year, with respect to the accessibility to the well and possibility of monitoring the water level. In the watershed, farmer wells (fw) are tubed wells used for agriculture and open wells (ow) are also used for irrigation purposes (Figure 5.1). Dug wells (dw) are used for domestic purposes in the peri-urban area and installed piezometers (pi) are only used for monitoring purposes.

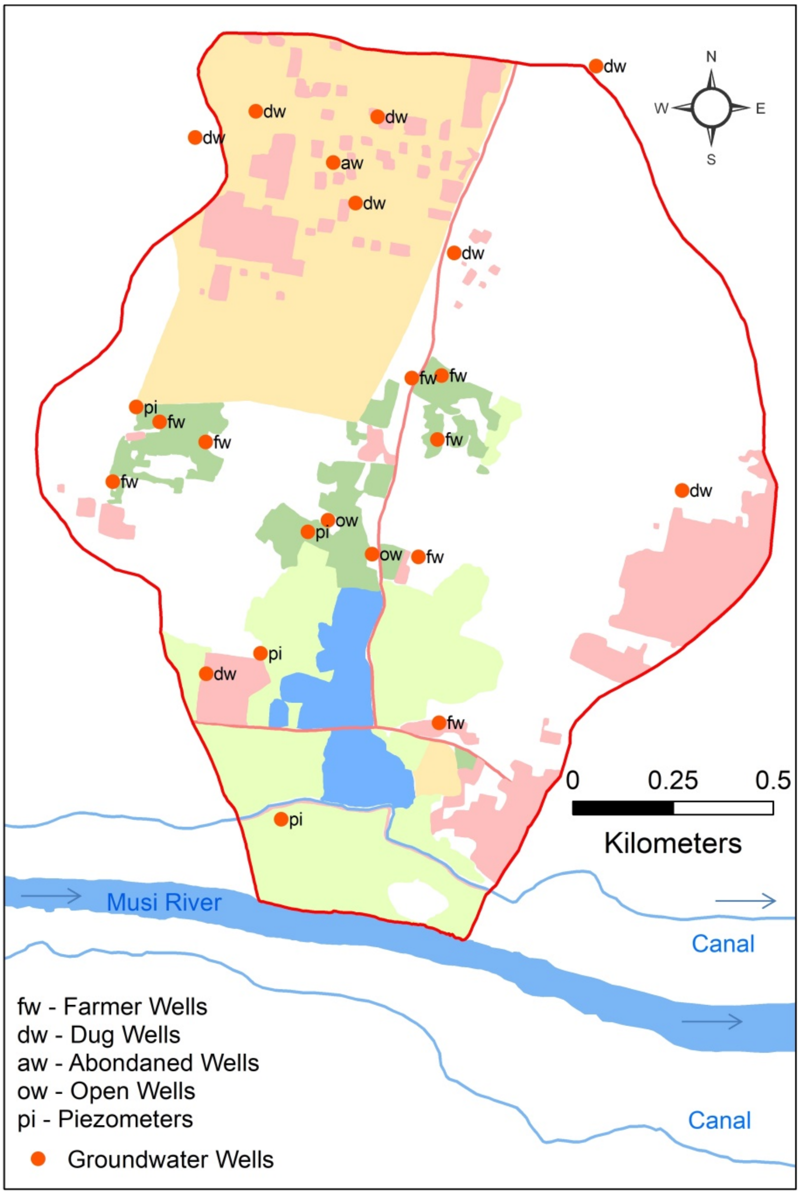

Figure 5.1 Location of observation wells monitored for groundwater levels between the years 2010 and 2015. Water level data of all the wells mentioned in the map is not continuously available during the monitoring period except for wells with installed piezometers (pi).

Four piezometers were installed and the corresponding water levels were monitored continuously from the year 2010 to 2015. In support of the piezometric data, water levels collected from other farmer bore wells or domestic bore wells are also used for model calibration (Figure 5.1; Figure S5.1; Figure 1.4b).

5.2.1 Water abstractions and crop water requirements

The abstraction of wastewater from the canal or groundwater from bore wells by farmers depends on the electricity availability for pumping. The electric power supply for irrigation water pumping in the region is based on the political promises of free electricity to farmers (Reddy, 2012). Information related to the pump capacity, duration of the pumping and pumping timings were obtained from farmers. Pumping rates by farmers to the respective agricultural fields (irrigation supply to per hectare area) in the watershed is assumed constant for all the years as their irrigation and pumping time practices did not change over the years. Based on the pumping conditions, discharge rates for each agricultural field were calculated in the watershed. These calculated discharge rates are mentioned as actual water abstractions by farmers (aCrop). To differentiate the water abstraction and the actual requirement for the crops, water requirements for each crop is calculated using CropWAT 8.0 (Smith, 1992).

Crop water requirement (rCrop) calculations need rainfall, wind speed, humidity, temperature, soil type, rooting depth, crop type and its growth stage information. The closest meteorological station is 6 km away from the study watershed, from where hydrometeorological data was collected to feed into the CropWAT model. The soil type in the study area is of loamy sand. The three principal crops of the study watershed paragrass, paddy rice and vegetables are chosen for crop water requirement calculations. For each crop, rooting depth and growth stages early, mature and late days were defined in the model. Paragrass is a typical perennial crop with 45 days growth stage and the rooting depth is up to 1.5 m. Paddy rice is cropped two times per year (May to September and November to March) and the rooting depth is up to 0.9 m. Leafy vegetables is also a perennial crop with 60 days growth stage and the rooting depth is up to 0.5 m.

5.2.2 Land use change and groundwater model setup

The groundwater flow model based on MODFLOW is constructed for the study watershed Kachiwani Singaram by synthesizing the information of land use, hydrological, climatic and hydrogeological parameters using Visual MODFLOW Flex 5.1. MODFLOW is a well versed and widely used finite volume groundwater model, which can simulate the transient groundwater conditions with single or more aquifer layers in different geological backgrounds (Harbaugh et al., 2000). The groundwater model grid size for the study watershed is divided into 50 rows by 50 columns. The smaller grid size for modelling is considered to assign the detailed land use based aquifer parameters and boundary conditions.

Detailed analysis of the historical (2000 to 2015) land use change in the peri-urban watershed is explained in Chapter 4 of the thesis. The present chapter is built up on the results of land use data generated from Chapter 4. In total, nine individual groundwater models were set up based on the historical land use data for the years 2000, 2003, 2006, 2009, 2010, 2011, 2013, 2014 and 2015. For all the modelled years, assumed model parameters and boundary conditions are almost similar except the spatial assigning of the conditions, which are changed based on the land use change of that particular year. For each land use change monitored year, an individual three dimensional finite difference groundwater flow model is constructed.

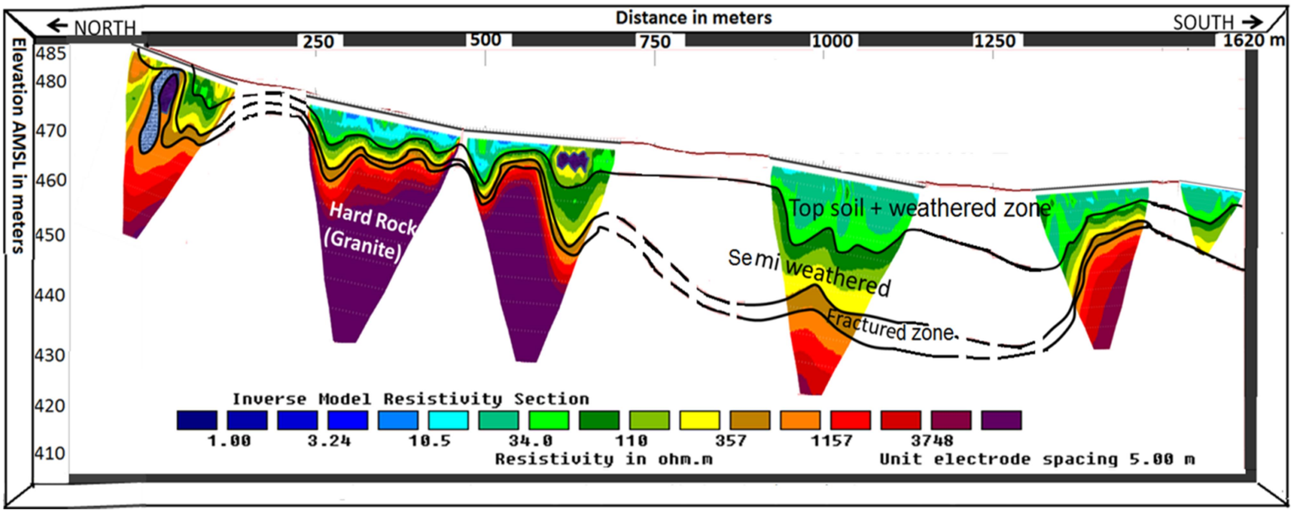

Figure 5.2 Hydrogeophysical profile of the study watershed with three subsurface layers considered for the groundwater flow modelling are depicted. The cross-section of the watershed is modified from Sonkamble et al., 2019.

A three layered aquifer model is considered based on the hydrogeophysical scanning of the study watershed (Figure 5.2). The first layer is with top soil and weathered zone (3 to 12 m thickness), the second layer is semi weathered (3 to 20 m thickness), the third layer is fractured zone (2 to 5 m thickness) and in the basement of the third layer is hard rock (granite) and all the three layers are hydraulically connected (Dewandel et al., 2008; Sonkamble et al., 2019). Musi River is with huge water flows at the bottom of the watershed and certainly, these river flows impact the groundwater conditions in the downstream of the watershed. At the end of the watershed, cells are assigned to the river conditions and simulated using MODFLOW river package. Surface water bodies are additional water supply to the recharging system in the watershed and canal flowing through the watershed is simulated using MODFLOW drain package. The recharge delay is insignificant and thus no major shifts in hydrogeological parameters are considered during the modelling period (Dhakate et al., 2012; Surinaidu et al., 2016).

5.2.3 Flow modelling and recharge calculations

Elevation data for the model top surface is assigned using Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) 30 m data. For conceptualizing the local groundwater flow system, boundary conditions of the model are critical. To understand the stream-aquifer interactions, the river and canal boundary conditions were assigned using the river and drain packages, respectively, and the corresponding width of the river and canal information is obtained from Google Earth images. Pumping tests were carried out in the four installed piezometers of the watershed to determine the aquifer parameters such as transmissivity (T, m^2/s) and hydraulic conductivity (K, m/s) (Perrin et al., 2010). The same values of hydraulic conductivity are assigned for different months in the model. The transmissivity and conductivity values of the upper layer in the watershed are varying from 86 to 293 m^2/day and 1.4 to 5.3 m/day, respectively (Figure 5.3).

Hydraulic conductivity values are assigned based on the land use conditions and their spatial distribution (Figure 5.3). Piezometer litholog information is also used as model input for layer selection, layer elevation and aquifer parameter distribution. The number of measured groundwater heads is varying depending on the model year (Figure 5.1; Table 5.1; Figure S5.1) and they are assigned accordingly as the observed heads in the model for calibration. Evapotranspiration (ET) and corresponding potential evapotranspiration (PET) values are calculated using climatic information fed to the CropWAT 8.0 model. Constant head conditions are assigned at the top and bottom of the watershed to control the inflow and outflow conditions of the model. Recharge boundary conditions are assigned based on the land use of the particular modelled year. Recharge conditions of the groundwater regime in the watershed are impacted by the irrigation return flows from the wastewater and groundwater irrigation, monsoon rainfall and seepage from the surface water bodies. Groundwater recharge and return flows to the aquifer are estimated by multiplying the geographical area of the land use, groundwater level change from pre-monsoon to postmonsoon and the specific yield. Estimated recharge of the watershed assigned to the recharge package is ~70 mm/yr (with variability 2 to 5 mm/yr for different years) and groundwater pumping considered in the well package is also ~70 mm/year (with variations 1 to 6 mm/yr). Groundwater pumping is mainly done by farmers in the middle part of the watershed to grow paddy rice and also by peri-urban dwellers in the upstream part for domestic consumption.

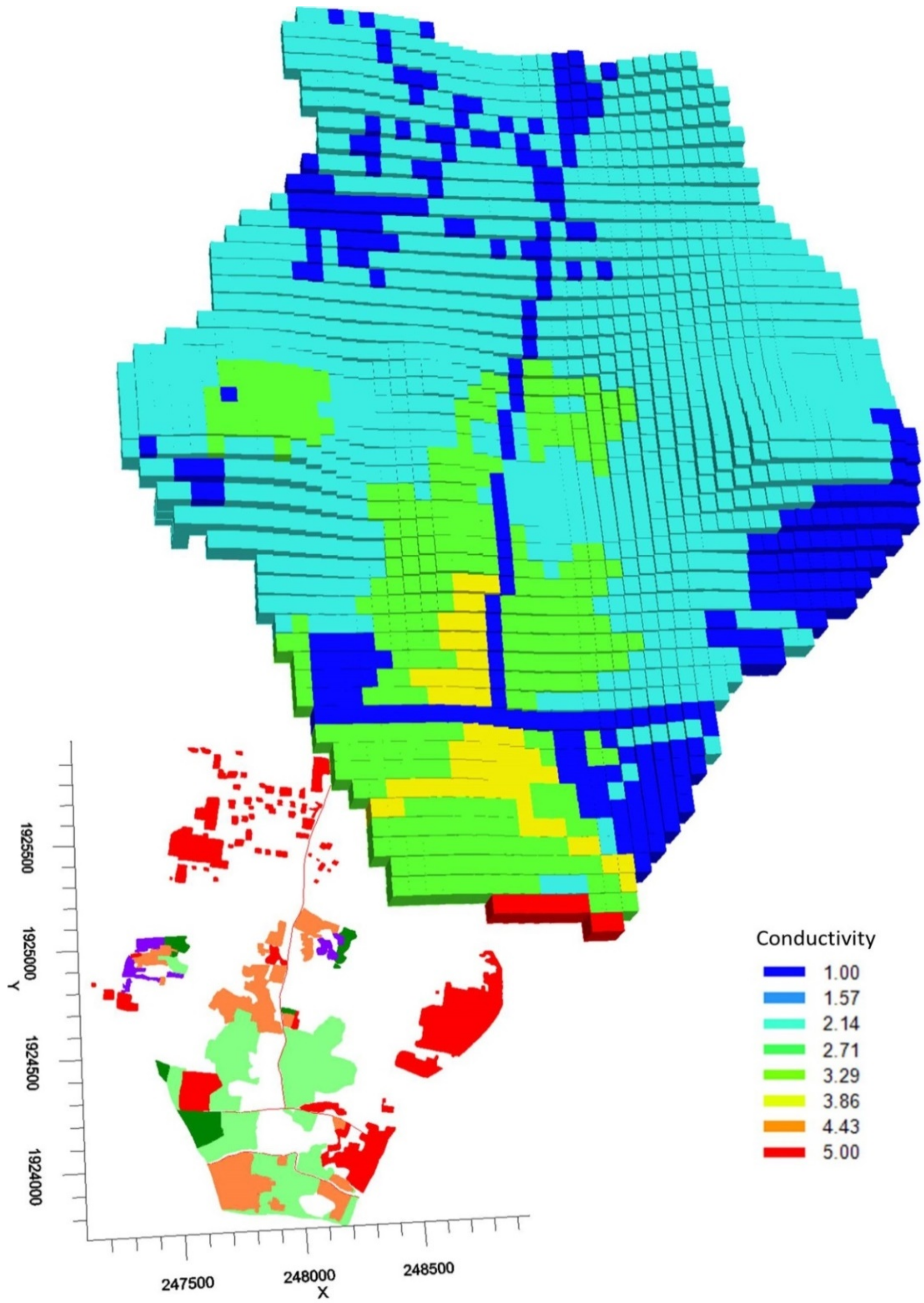

Figure 5.3 Spatial distribution of hydraulic conductivity values (in m/day) of the upper layer in the flow model based on the land use of the study watershed (example presented for the year 2010).

5.3 Results and Discussion

5.3.1 Calculated water abstractions vs crop water requirements

The water abstractions by farmers for the paragrass irrigation (aParagrass) in the watershed increased from 588,615 m^3 in the year 2000 to 701,998 m^3 in 2015, in line with the increase in land use area for the paragrass crop over the years.

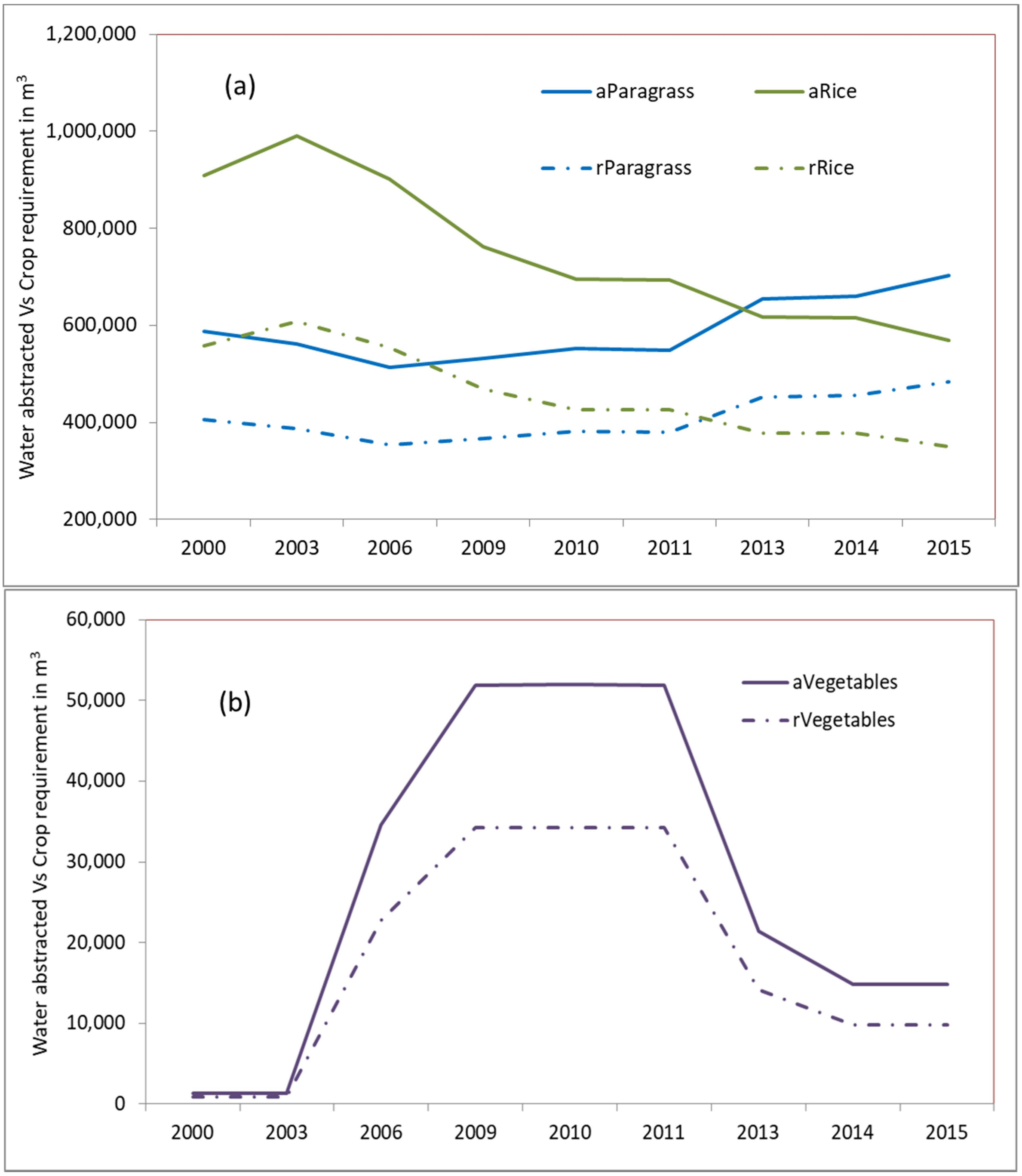

Figure 5.4 Calculated actual water abstraction vs the crop water requirements in the study watershed for (a) paragrass, paddy rice and (b) vegetable crops.

The calculated crop water requirements (rParagrass) illustrate that the farmers are applying 33% (approx.) excessive water than the required amounts for the paragrass cropping (Figure 5.4a). The paddy rice crop requires 10 to 20% more water for irrigation compared to paragrass (Biggs and Jiang, 2009). The actual water abstractions for paddy rice (aRice) in the study watershed amounted to 909,652 m^3 in the year 2000 and 568,761 m^3 in 2015. The drastic decrease in water abstraction (~47%) for paddy rice in the watershed is attributed to the decrease in land used for paddy rice cultivation. The calculated crop water requirements (rRice) illustrate that farmers apply 36% (approx.) more water than the required water for the crops (Figure 5.4a).

Vegetables consume less water compared to the paragrass and paddy rice. Until the year 2003, there is a very limited area under vegetable cultivation, after that, the increase in land use area for vegetable cropping changed the water abstraction patterns for vegetable production (Figure 5.4b). The actual water abstractions for vegetables (aVegetables) in the watershed increased from 900 m^3 in the year 2000 to 34,290 m^3 in 2010 and further, decreased to 14,870 m^3 in the year 2015. Intense changes in land use patterns certainly influenced the water abstraction patterns in the watershed. The calculated crop water requirements illustrate that farmers apply 34% (approx.) more than the actual water required for the vegetable production. The majority of the water abstraction for paragrass cultivation comes from the wastewater canal; the water for paddy rice cultivation comes from irrigation groundwater wells. At the current stage, almost equal amounts of wastewater and groundwater are abstracted for vegetable production in the watershed. These excessive water application rates for peri-urban agriculture in the watershed are certainly contributing to the groundwater recharge of the local aquifer.

5.3.2 Groundwater flow model calibration

The groundwater flow model is calibrated using trial and error method by adjusting parameters, hydraulic conductivity and recharge rates. During the calibration, monitored groundwater heads of the particular modelling year are used (Table 5.1). The flow model in all the years is highly sensitive to hydraulic conductivity and recharge rates. After increasing recharge rates at the bottom of the watershed, in the river alluvium area, a reasonable match between observed and computed heads is achieved. To reduce uncertainties in the period 2000 to 2009 as there was no monitoring of groundwater levels, the actual water abstractions for assigning recharge influxes and groundwater levels from the year 2010 are used as proxy data for model calibration. The corresponding models for each year are stand-alone and based on land use conditions. Creating groundwater flow models based on the changes in land use can increase the accuracy of predicting groundwater flow conditions in the watershed. With this kind of multi-model approach, we expect to reduce the uncertainties in estimating the aquifer parameters and boundary conditions. The observed values of the standard error of the estimate, root mean square error, normalized RMS and correlation coefficient for each modelled year achieved a good match between the observed versus the computed groundwater heads (Table 5.1).

Table 5.1 Year wise (2000 to 2015) groundwater flow model results of observed vs computed groundwater heads

Year	Number of data points	Standard error of the estimate (SEE) (m)	Root mean square error (RMS) (%)	Normalized RMS (%)	Correlation coefficient
2000	14*	0.24	8.89	4.77	0.87
2003	14*	0.25	8.78	4.65	0.88
2006	14*	0.24	8.75	4.53	0.89
2009	14*	0.23	7.92	4.32	0.91
2010	14	0.12	6.32	3.83	0.92
2011	18	0.09	7.39	3.99	0.92
2013	9	0.15	6.68	3.91	0.91
2014	8	0.17	5.45	3.28	0.92
2015	6	0.14	4.98	2.95	0.93

*For model calibration observed data points of the year 2010 were considered for proxy estimation of groundwater heads from 2000 to 2009.

5.3.3 Land use change impacts on groundwater recharge

Over the years, groundwater recharge processes in the watershed are influenced by highly heterogeneous spatial land use patterns (Figure 5.1 and Figure 5.5). Overall, the groundwater levels are influenced by the increase in built-up and agricultural areas (Figure S5.1). Estimated natural recharge from modelling in the watershed over the years stands at ~75 mm/yr in 2000 to ~68 mm/yr in 2015. Groundwater outflows from the watershed contribute to the base flow of the Musi River and at a rate of around ~134 mm/yr, which varies between 130 to 138 mm/yr for different years. Even though there are shifts in land use, the groundwater levels almost remain constant over the years. It is attributed to the increase in return flows from irrigated area pockets with the shifts in the agricultural area and at the same time decrease in recharge with increased impervious areas. The built-up areas are continuously increasing in the upstream of the watershed, where low hydraulic conductivity values are observed (Figure 5.3) and leads to a decrease in recharge to the groundwater.

5.3.3.1 Crop wise recharge estimations

Peri-urban agriculture is one of the main factors for sustaining groundwater levels in the watershed. The increased impervious surfaces are balanced by the groundwater return flows that are recharging local aquifers from irrigated areas. Paragrass is one of the main crops in the watershed, from where a major amount of irrigation return flows are contributing to the groundwater recharge.

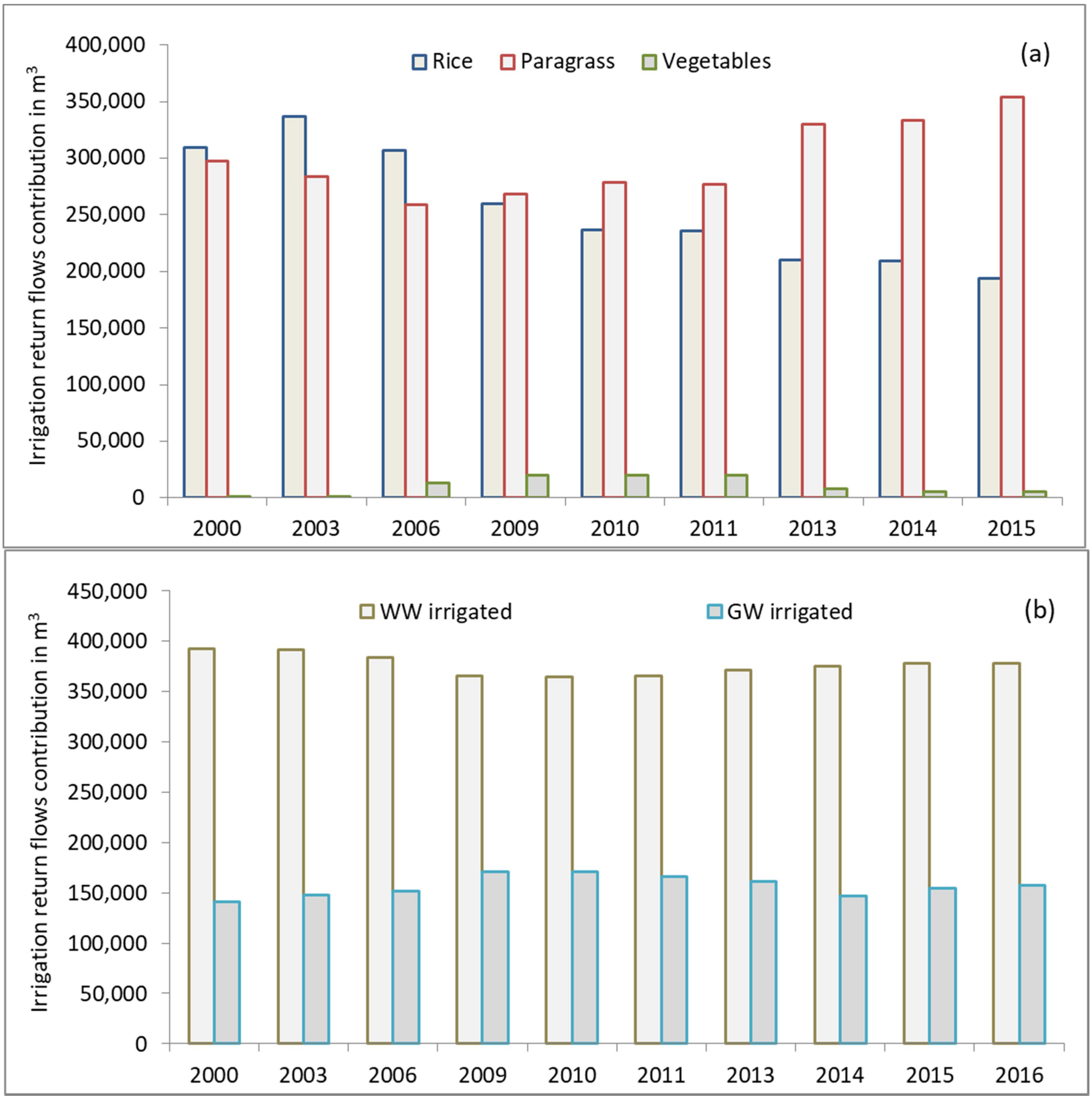

Figure 5.5 Crop and irrigated area wise irrigation return flows contribution to the local aquifer in the study watershed.

Resulting from the flow model estimations, total contributed irrigation return flows from areas with paragrass crop increased from 297,135 m^3 in the year 2000 to 354,372 m^3 in 2015 (Figure 5.5). Direct factors such as increase in area under paragrass cultivation and amount of water application rates to the crop and indirect factors such as less attention required for the crop, low labor costs and increased profit margin compared to other crops (Buechler et al., 2002; Mahesh et al., 2015b; Starkl et al., 2015) have certainly influenced the increase in return flows from the paragrass irrigated area to the groundwater recharge.

At the same time, return flows from paddy rice irrigated area decreased substantially from 309,539 m^3 in the year 2000 to 193,539 m^3 in 2015. Direct factors such as decrease in area under paddy rice cultivation and water application rates and indirect factors such as increased salinity of the irrigation waters, low crop yields, decreased profit margins (Amerasinghe et al., 2009; Biggs and Jiang, 2009) have influenced the decrease in return flows from paddy rice irrigated area to the groundwater recharge. Irrigation water application rates for leafy vegetables are less compared to other crops and thus the contribution of irrigation return flows to the groundwater recharge is low compared to other crops. Leafy vegetables contributed return flows of 525 m^3 in the year 2000 and increased their contribution to 19,998 m^3 in 2010 and further decreased to 5721 m^3 in 2015 (Figure 5.5). This change in return flows pattern is heavily influenced by land use shifts with respect to vegetable cultivation in the watershed. Direct factors such as less irrigated area in the watershed and fewer water requirements and indirect factors such as competition from other land use classes and needed continuous attention to the crop (Amerasinghe et al., 2015, 2009; Mahesh et al., 2015b) have influenced the return flow contribution from vegetables irrigated area to the groundwater recharge.

5.3.3.2 Wastewater irrigation influence

The local aquifer in the study watershed is heavily influenced by the wastewater irrigation and it is one of the main contribution sources to the groundwater recharge. Approximately 45% of the recharge in the watershed is contributed from irrigation return flows from the wastewater irrigated area. The return flows from wastewater irrigated area contributed 392, 417 m^3 in the year 2000 to 379,294 m^3 in 2015 to groundwater recharge. Over the years, the groundwater recharge contribution from wastewater irrigated area almost remained the same with 2 to 5% variation. Paragrass is the main crop in the wastewater irrigated area, which amounts to ~74% of the area under wastewater irrigation and also ~18% of the groundwater irrigated area is under paragrass cultivation in recent years (2013 to 2015). Over the last fifteen years, groundwater levels also substantially increased an average of 0.2 to 0.5 m in the downstream of the watershed, where wastewater irrigation is practiced. Farmers tend to practice flood irrigation with wastewater and apply excessive amounts of water for irrigation in the study area. If the current trend continues, groundwater levels may tend to raise more in the downstream of the watershed. The substantial amount of return flows (~98 mm/yr) from the wastewater irrigated area also contributes to the base flow of the Musi River at the watershed outlet.

5.3.4 Implications of peri-urban agriculture on groundwater recharge

Due to lack of pervious surfaces, groundwater levels are declining at an alarming rate in and around urban agglomerations in many parts of the world (Han et al., 2017; Minnig et al., 2018; Schirmer et al., 2013). Urban and peri-urban agriculture areas are pervious surfaces surrounded by impervious concrete structures and contribute to the food production and groundwater recharge. The current study provides integrated insights into the peri-urban agriculture, land use change, wastewater irrigation impacts on groundwater recharge. Even though with the increase in built-up areas expected to have an impact on the local aquifer, including the agriculture in the peri-urban environment certainly balances the system to sustain the groundwater levels. It is evident that wastewater availability from the Hyderabad city in the study area for irrigation has certainly improved the recharge conditions of the local aquifer. But at the same time, it is apparent that uncontrolled wastewater irrigation can cause soil and groundwater pollution (Candela et al., 2007; Fridrich et al., 2014; Gallegos et al., 1999; Jampani et al., 2018).

As the paragrass is the main crop in the peri-urban system influenced by wastewater irrigation, future research should focus on assessing the detailed water flow processes with respect to paragrass or fodder grass. The impact of paragrass cultivation on the vadose zone and further its link to the aquifer system should be studied at the field scale. There are some uncertainties in the current study, which includes unavailability of the historical groundwater head data and difficulties to validate the model as the model for each year is stand-alone. Future research should focus on curbing these limitations by integrating soil water data, multi-model assessment and data assimilation methods to generate more accurate land use based groundwater flow models. Also, long term monitoring of the infiltration capacities and groundwater levels can help in refining the groundwater flow models with the change in land use patterns, allowing better assessments in the future.

5.4 Conclusions

Overall changes in groundwater dynamics over the years are majorly affected by the changes in land use. But the groundwater levels in the watershed are almost the same over the years, only the contribution source to groundwater recharge has changed. Natural recharge from the watershed over the years stands at 75 mm/yr in 2000 to 68 mm/yr in 2015. The groundwater recharge conditions in the watershed are heavily influenced by wastewater irrigation. Even though only 14% of the land use in the watershed is wastewater irrigated area but it contributes approximately 45% of the groundwater recharge in the watershed through irrigation return flows. Paragrass is the major crop in the watershed and also in the wastewater irrigated area and the paragrass irrigated areas contribute to the majority of the groundwater recharge from the agricultural area. Future groundwater recharge in the watershed will be influenced by increasing impervious surfaces and pursuing peri-urban agriculture can improve the recharge flux in the watershed. As the peri-urban systems are in similar nature of multi-functional land use systems, the decision makers should plan the environmental resources management in water-soil-water nexus dimension for efficient and sustainable groundwater management. The study recommends local policy makers and farmers to use wastewater for irrigation as per optimal water requirements by crops and also employing the natural treatment systems such as managed aquifer recharge (MAR) and soil aquifer treatment (SAT) in the peri-urban systems can improve the groundwater recharge conditions and at the same time mitigate aquifer pollution.

6 Sustainable Groundwater Management Strategies

6.1 Introduction

Wastewater application for irrigation using safe practices has been recognized as one of the sustainable measures for environmental resources management (Hamilton et al., 2007; Qadir et al., 2010). Economic benefits associated with wastewater reuse are widely recognized in recent years (Kurian et al., 2013; Minhas and Samra, 2004). In many parts of the developed world, secondary treated wastewater is being used for industrial and horticulture purposes (Maass and Grundmann, 2016). Whereas in the developing world, due to insufficient infrastructure and lack of financial resources for wastewater treatment, partially treated or untreated wastewater is used for agricultural purposes (Amerasinghe et al., 2013; Mahesh et al., 2015b; Parkinson and Tayler, 2003). Long-term use of untreated wastewater for irrigation has detrimental impacts on the crop production and at the same time on groundwater resources (Bedbabis et al., 2015; Candela et al., 2007). Bad irrigation management practices by farmers have serious implications on soil and groundwater pollution (Qadir et al., 2010).

Insufficient policies, lacking support and advice to farmers for sustainable crop production, unsafe irrigation practices in the urban and peri-urban wastewater irrigation systems leading to poor resources management and groundwater contamination. Many of the local urban and peri-urban dwellers depend on the local aquifers for drinking and domestic purposes, which in case of wastewater contamination, will have adverse impacts on human health (Drechsel et al., 2010). In recent years, several researches (K’oreje et al., 2016; Papaioannou et al., 2016, 2016; Rizzo et al., 2013; Singh et al., 2010; Woldetsadik et al., 2017) have reported the risk of various toxic contaminants from urban wastewaters (heavy metals, pharmaceuticals, antimicrobial resistant bacteria etc.) entering into the natural environments even after partial treatment. Once the contaminants from wastewater came in contact with the groundwater resources, it is tough, costly and time consuming process to remediate the aquifers (Candela et al., 2007; Fridrich et al., 2014; Kass et al., 2005).

In terms of pollution, groundwater salinity is one of the major problems faced by aquifers around the world (Wang and Jiao, 2012; Wichelns and Qadir, 2015; Zhao et al., 2017). Especially in wastewater irrigation systems, long-term accumulation of salts in the soils is resulting to aquifer pollution (Jampani et al., 2018; Kass et al., 2005). When the saline groundwater is used for domestic and drinking purposes, it creates adverse health impacts including hypertension and cardiovascular diseases (Drechsel et al., 2010). Saline irrigation waters also lead to decreased crop productivity and create infertile lands in the long run (Biggs and Jiang, 2009). Farmers are already adapting to salinity tolerant crops, for example, food crops such as paddy rice and vegetables are being shifted to bioenergy and cattle feeder crops such as paragrass and elephant grass (Amerasinghe et al., 2013; Buechler et al., 2002). Most of the farmers chose to practice flood irrigation, which is a water consuming irrigation practice and at the same time there will be a sturdy increase in the amount of salts dumped into the agricultural fields. Farm support and good knowledge on the amount of water and nutrient requirements for different crops can benefit the farmers for using less water and nutrient supplements, as wastewater itself is a rich nutrient source. Providing optimal water requirement information for the crops could aid decontamination of soil and groundwater resources.

Several studies around the world and in the Musi River basin area illustrated the wastewater irrigation and its impact on groundwater resources contamination (Jampani et al., 2018; Kass et al., 2005; K'oreje et al., 2016; Mahesh et al., 2015b; Perrin et al., 2010; Schmidt et al., 2013; Wang et al., 2014), but studies related to developing pollution controlling management strategies are lacking. With the intense wastewater irrigation, the salinity of groundwater is increasing over the years in the downstream of Hyderabad city of the Musi River basin (Biggs and Jiang, 2009; Jampani et al., 2018). To mitigate the saline conditions of the groundwater in the Musi River basin, sustainable irrigation management strategies need to be introduced for wastewater irrigation systems. The current chapter focuses on developing suitable and sustainable groundwater development strategies based on the amount of wastewater applied for irrigation and different qualities of the wastewaters used for irrigation. These management strategies can be useful for the efficient management of water resources and sustainable food production in the peri-urban environment.

6.2 Materials and Methods

6.2.1 Groundwater solute transport modelling

The groundwater contaminant transport modelling setup is based on the groundwater flow model setup explained in Chapter 5. Modular transport three dimensional multi-species (MT3DMS) is a finite difference groundwater mass transport model used for simulation of advection, dispersion and chemical reactions of contaminants in the groundwater flow system (Zheng and Wang, 1999). For the current study, Visual MODFLOW Flex 5.1 software is used for the groundwater solute transport modelling and the scenarios generation.

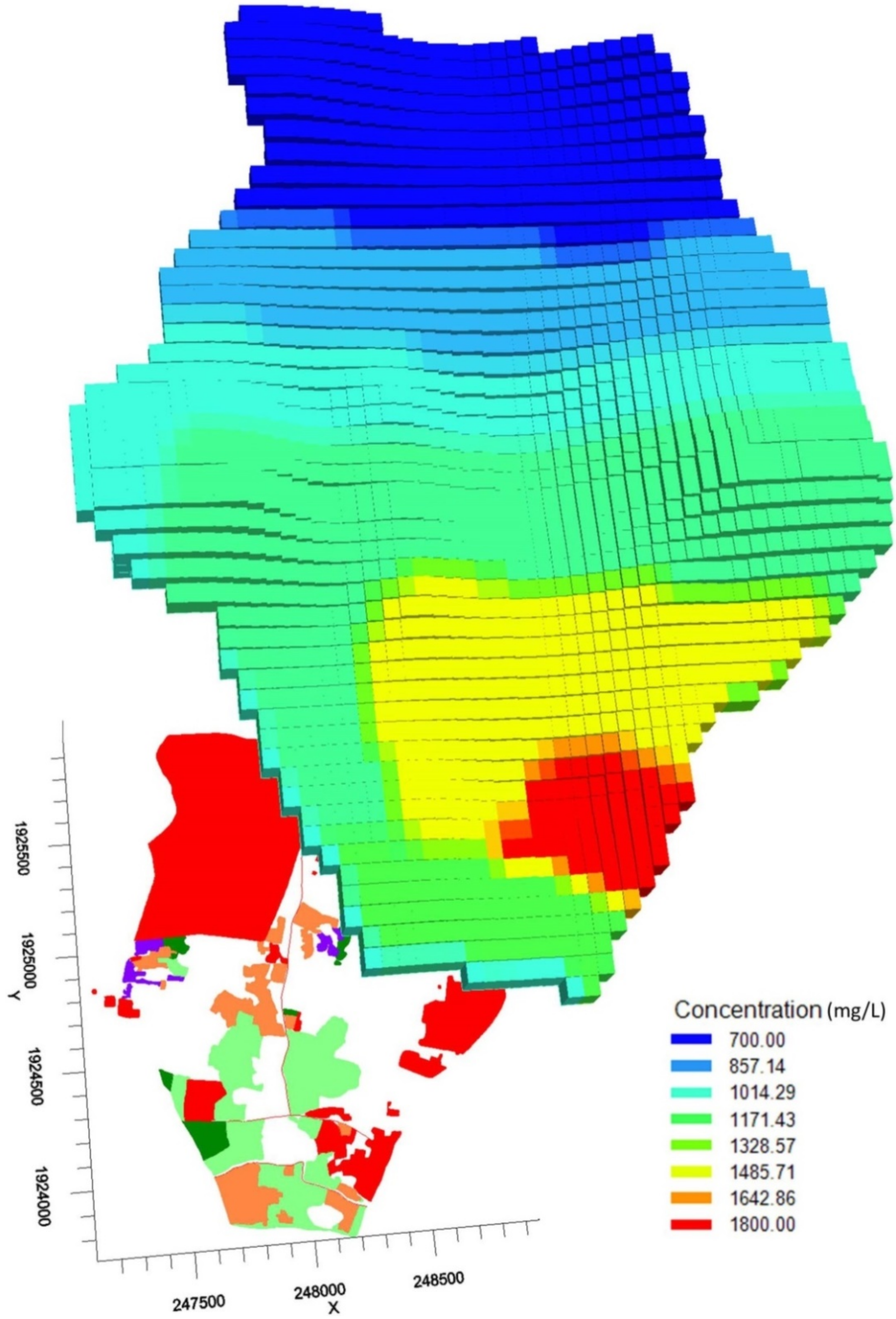

Figure 6.1 Spatial distribution of initial concentrations of TDS (in mg/L) in the mass transport model based on the land use of study watershed for the year 2013.

TDS is considered as the main contaminant to model the salinity conditions in the watershed. Defining aerial concentrations of TDS in the model is one of the key challenges for non-point source pollution distribution in the aquifer. Different polygons of land use data are used for constructing the source functions for TDS in the model. The model is based on the setup year 2013, where the maximum amount of data is available for groundwater chemistry. For the current study, the initial concentrations are assigned based on the land use and hydrogeology of the watershed (Figure 6.1). In the lower parts of the watershed, where wastewater irrigated areas are located in the river alluvium geology, lower TDS concentrations are assigned compared to the wastewater irrigated areas with granitic geological conditions. For wastewater irrigated areas ~1200 to 1800 mg/L of TDS concentrations are assigned and for groundwater irrigated areas ~900 to 1200 mg/L, considering the groundwater mixing of the subsurface layers. In the upstream of the watershed low initial concentrations of TDS are assigned, where no agricultural practice is observed.

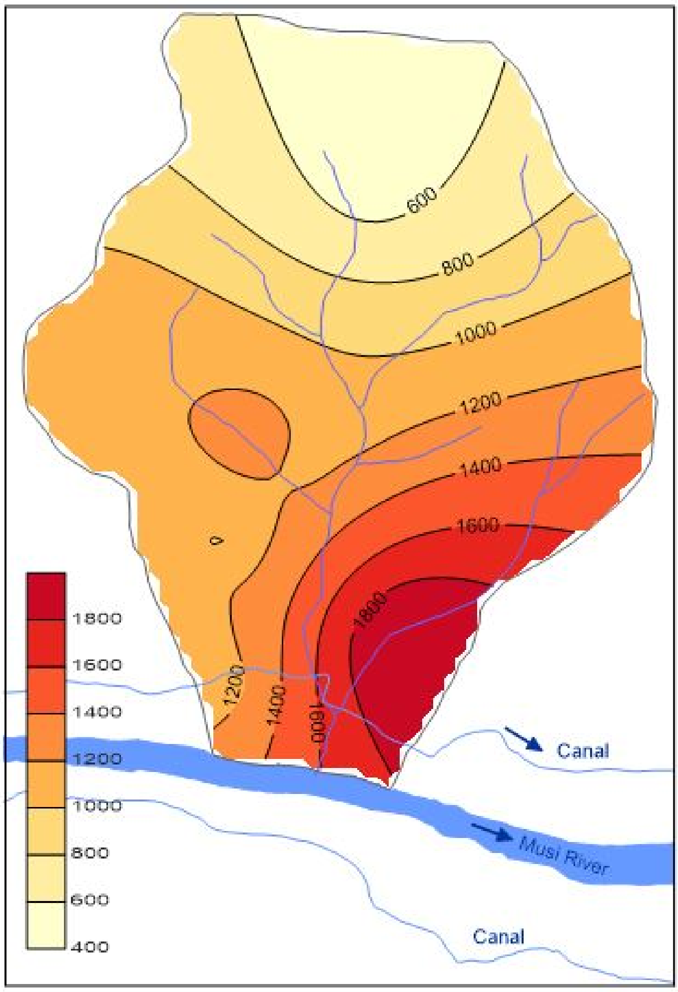

Figure 6.2 Observed TDS concentration in (mg/L) as business as usual (BAU) scenario

6.2.2 Management Strategies

Four suitable scenarios are considered in the modelling process for generating sustainable strategies under irrigation management and for groundwater development (Table 6.1) and the observed TDS concentration is presented as business as usual (BAU) scenario (Figure 6.2). At the current stage, groundwater conditions are already expressing high salinity values, which tell us that the groundwater quality is not suitable for drinking, domestic and irrigation purposes (Figure 2.2; Table 2.1) (Jampani et al., 2018). If the current trend continues, there is a risk of increasing groundwater salinity values in the aquifer. In this view, modelled four scenarios of groundwater salinity, in which two scenarios (W_{11} and W_{12}) are based on the amount of wastewater applied for irrigation with the current level of wastewater quality. W_{11} scenario is based on the assumptions that if the wastewater application rates are doubled or agricultural land use area under wastewater irrigation is increased, where farmers practice flood irrigation. This is an extreme condition assuming that farmers increase the amount of wastewater application rates or increase the irrigated areas with wastewater. In the case of W_{11}, rather than changing the initial concentrations, the amount of wastewater applied for irrigation in the model is doubled. W_{12} is an ideal condition considering that farmers chose to apply the amount of wastewater for irrigation based on the actual amount of water required by crops, this is a 30 to 40% less amount of wastewater application compared to current irrigation practice (Figure 5.4). The amount of wastewater application decreased by 40% compared to the current water application rates in the groundwater flow model.

Table 6.1 Scenarios based on wastewater qualities and quantity application rates for the groundwater development

Type of Scenario	Scenarios	Conditions	Description*	Action by
Wastewater application rates change	W_{11}	Extreme	Doubling the current amount of wastewater applied for irrigation	Farmers
	W_{12}	Ideal	Irrigating wastewater according to crop water requirements	
Wastewater quality change	W_{21}	Extreme	Doubling the current wastewater salinity concentration values	City water authorities or decision makers
	W_{22}	Ideal	Irrigating with treated wastewater (secondary level – ~750 mg/L) †	

*Considering the wastewater use is only for the areas where the wastewater irrigation is being practiced (except for scenario W_{11}); †TDS concentration based on the neighbouring (6 km) wastewater treatment plant from the study area after secondary level treatment.

The other two scenarios (W_{21} and W_{22}) are based on the qualities of wastewater used for application considering the amount of wastewater application rates constant at the current level. W_{21} is an extreme condition considered if the amount of wastewater generated by the Hyderabad city increased in future or there is no treatment of wastewater generated by the city (at the current stage, wastewater quality used for irrigation is a mixture of ~40% partially treated and ~60% untreated). The initial concentrations are doubled in the model for W_{21} scenario considering the extreme conditions of wastewater quality. W_{22} is an ideal scenario considered if the secondary level of treated wastewater used for irrigation. The initial concentrations are decreased in the model based on the wastewater quality values from the nearby wastewater treatment plant after secondary level treatment.

6.3 Results and Discussion

6.3.1 Contaminant transport modelling conditions and calibration

Salinity is considered as the main contaminant in the groundwater solute transport model, for which TDS concentrations are calculated at all the nodes of the modelled year, 2013. Trial and error calibration technique has been used to adjust the parameter values in the sequential model runs to match the calibration values within a range of error. The computed concentration values of TDS are correlated with the observed values of TDS from analytical results. There is a mismatch between the computed and observed values during the initial simulations of the model. To improve the matching, the background magnitude and distribution of initial salinity concentrations or pollutant loads are modified. An attempt also has been made to improve the calibration errors by changing the transport parameter values (dispersivity and effective porosity) of the model. The dispersivity is set to 55 m and effective porosity to 0.18 in the model. After all the changes, the observed and computed TDS values reached a good match with the standard error of estimate 6.48 mg/L, normalized RMS error 9.89% and correlation coefficient 0.90 (ten observation data points) in the model calibration. After the good match between the salinity concentration values of the model, the same model conditions are used to generate the four management scenarios by changing the initial groundwater quality concentration values or modifying the recharge conditions to observe the changes in the salinity values.

6.3.2 Sustainable management strategies

Generated four scenarios using the contaminant transport modelling are based on changing the amount of wastewater or wastewater quality used for irrigation, which has a visible impact on changing the groundwater salinity conditions in the watershed (Figure 6.3 and Figure 6.4). The predicted concentration values in the scenarios are dependent on the change in initial concentration or recharge values. In the W_{11} scenario, increasing the amount of wastewater use for irrigation has increased the return flows from irrigated areas to groundwater recharge and eventually increased the groundwater salinity values ~32%. Changing the amount of wastewater irrigation patterns in W_{11} scenario would certainly worsen the salinity conditions, which are already unsustainable under current conditions (BAU scenario). But it might stand true if the area under wastewater irrigation increased in the future. At the current stage, farmers are using way more than the water required for the crops. Changing this pattern by irrigating the crops with only the amount of water required can also improve the groundwater conditions in the watershed.

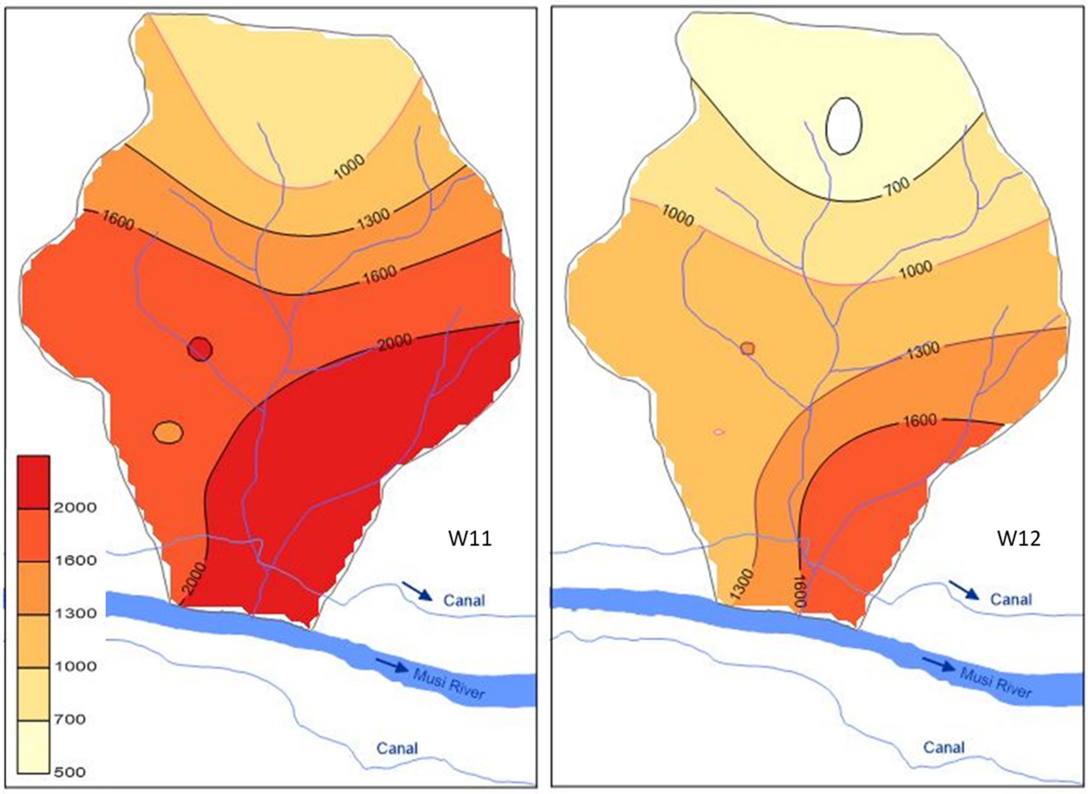

Figure 6.3 Modelled scenarios based on the amount of wastewater applied for irrigation with current wastewater quality. W_{11} – doubling the amount of wastewater applied for irrigation (extreme scenario); W_{12} – irrigating with wastewater based on the actual amount of crop water requirements (ideal scenario).

In the W_{12} scenario, switching the current wastewater application rates with the amount of water required by crops has no major influence on the groundwater recharge, but groundwater salinity decreased ~15% in the watershed (Figure 6.3 and Figure 6.5). The W_{12} scenario is a good and immediate sustainable management strategy for groundwater development with no costs involved for wastewater treatment, only farmers' motivation and capacity building on the amount of water required by crops to avoid any additional wastewater supply to the agricultural fields. In the W_{21} scenario, without changing any groundwater recharge conditions, input salinity concentration values are doubled, i.e., doubling the salinity values of the wastewater (doubling the initial concentration values) used for irrigation. With the increase in wastewater quality values, groundwater salinity values are also drastically increased by ~30% in the watershed. The W_{21} scenario is also not an option for sustainable groundwater management, rather this scenario is to implicate how the future groundwater salinity might change if the wastewater salinity values increases.

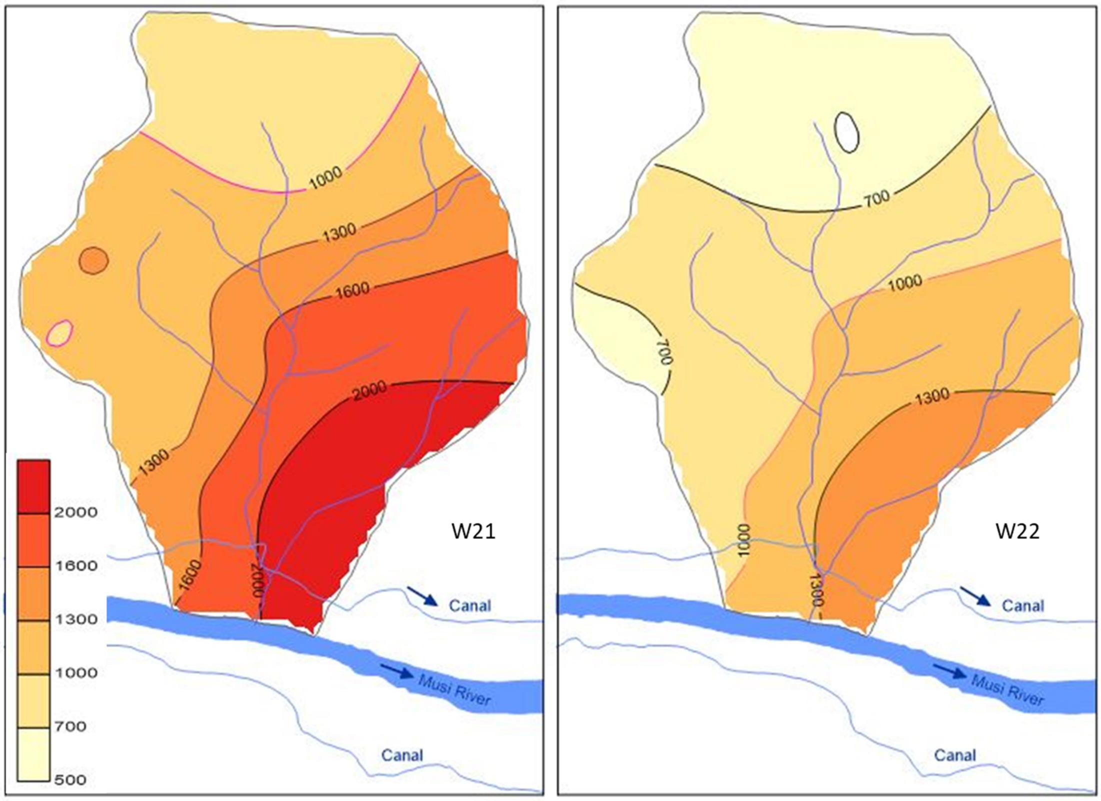

Figure 6.4 Modelled scenarios based on the qualities of wastewater used for irrigation. W_{21} – doubling the salinity values of the wastewater quality used for irrigation (extreme scenario); W_{22} – using the treated (secondary level) wastewater quality for irrigation (ideal scenario).

There is a possibility in the future, irrigated wastewater salinity might be increased with the increase in urban wastewater generation and lack of the infrastructure to treat that wastewater, which leaves more untreated wastewaters for the peri-urban farmers. In the ideal scenario of W_{22}, the secondary level treated wastewater quality concentrations are adopted as initial concentrations and groundwater salinity values are decreased by ~25% in the watershed (Figure 6.4). This scenario is a very good sustainable solution and management strategy for groundwater development and reversing the aquifer salinity, but adoption of this scenario might be difficult as the environmental agencies have other mandates such as river dilution, conservation etc. (CPCB, 2013), rather than supplying the treated wastewater to the peri-urban farmers.

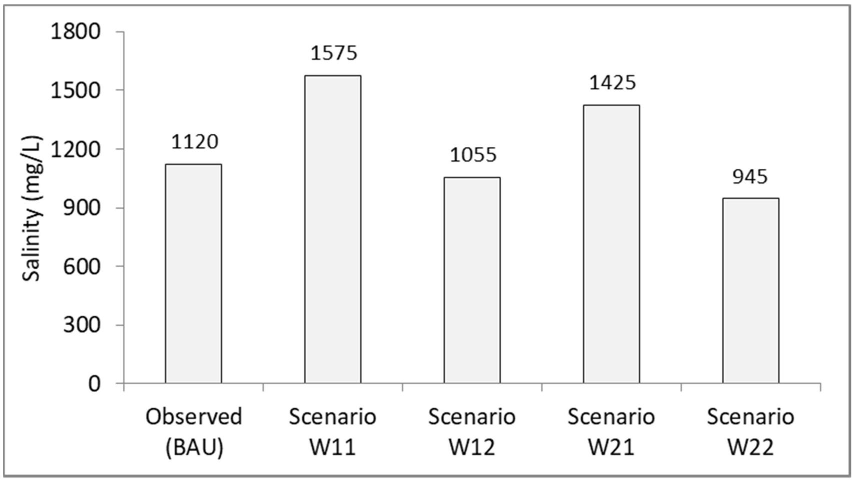

Figure 6.5 Mean weighted salinity values of the watershed area for all the scenarios

6.3.3 Implications of groundwater salinity conditions in peri-urban systems

Groundwater salinity conditions are afflicted by intense wastewater irrigation by smallholder peri-urban farmers in the watershed (Jampani et al., 2018). Peri-urban areas are economic transition zones and also multi-functional land use systems with a mixture of industry, built-up and agricultural areas, where peri-urban dwellers depend on the groundwater resources for drinking and domestic water supply. Use of polluted groundwaters can inflict human health problems such as hypertension and cardiovascular diseases in the peri-urban environment. Groundwater salinity values can be reversed by employing suitable and sustainable management strategies. Changing the wastewater amount for irrigation supply to the agricultural fields and providing the farmers with a good quality of wastewater for irrigation are good options for sustainable water resources management. The scenario W_{22} might be an ideal scenario, but in the developing world like India, the overall wastewater treatment stands at less than 70%. Policy makers are interested to funnel the treated wastewater into lakes and rivers for ecological conservation rather than supplying it to farmers or industry (Amerasinghe et al., 2012; CPCB, 2013). Supplying the treated wastewater for irrigation might improve the groundwater health in peri-urban environments, but this is a conflicting policy decision, which requires capacity development, moderation, co-design and good communication between stakeholders. Also more integrated research is needed to understand the synergies and trade-offs between supplying treated wastewater to farmers and industry or using it for river and lakes conservation. The current research work can be improved by adding more chemical parameters (Cl, SO_4, NO_3, PO_4, K etc.) in the solute transport modelling, so that the improved management strategies for groundwater development can be advised for the wastewater irrigated systems.

Based on the scenarios developed, there should be several practice and policy implications for the groundwater management in the wastewater irrigation systems.

- Planners and farmers should be made aware of the co-benefits of applying less amount of wastewater or partially treated wastewater supply for irrigation can increase agricultural productivity, and thus avoiding human and environmental health risks (Amerasinghe et al., 2016; Starkl et al., 2015).
- Onsite and farm based natural treatment systems such as wetlands, soil aquifer treatment, managed aquifer recharge etc. are also viable and cost effective options for wastewater treatment (Sonkamble et al., 2018).
- Research is needed to understand the options for applying more sophisticated soil and crop models to estimate the yields under different irrigation management scenarios.
- In water-soil-waste nexus view, integrated management of wastewater, land and groundwater resources in a sustainable manner can improve the crop, soil and aquifer conditions and also can create ecological benefits.

6.4 Conclusions

The results of groundwater contaminant modelling scenarios suggest that the increase in any wastewater concentration values or increasing any amount of wastewater application rates affect the groundwater salinity values in the watershed. Using the treated wastewater qualities for irrigation and decreasing the wastewater application rates according to crop water requirements are suitable management options that can improve the aquifer conditions with respect to salinity. In many parts of the developing world, it is the known case that most of the wastewater is untreated because of lack of financial resources. Providing the treated wastewater for irrigation can be a viable option but it is costly to treat the wastewater, which is a decision to be taken by financial planners and environmental decision makers together. A simple and sustainable management strategy is to motivate farmers to irrigate their agricultural fields according to the crop water requirements rather than applying the flood irrigation. Considering the increased groundwater chemical contamination and associated risks to human health, local decision makers should continuously monitor the wastewater and groundwater resource qualities in the watershed and inform the farmers of sustainable irrigation management strategies by suggesting suitable crops for the peri-urban areas with the corresponding wastewater application rates.

7 Conclusions and Policy Recommendations

7.1 Summary and Conclusions

The current research work is a very comprehensive study to understand the behaviour of the groundwater resources in the wastewater irrigated systems and to develop effective management strategies. The research study is carried out in a peri-urban micro watershed, which is located on the banks of Musi River in Southern India. The study watershed expresses the similar characteristics of the peri-urban wastewater irrigated systems in the developing world, where built-up spaces and agricultural areas coexist next to each other. In the peri-urban system, poor quality of wastewater (a mixture of untreated and partially treated wastewater) is used for irrigation.

Considering the research questions and objectives outlined in chapter 1 of the thesis, the following conclusions and policy recommendations can be drawn:

1. Spatio-temporal dynamics and chemical characterization of groundwater are analyzed to assess the influence of long-term wastewater irrigation using multivariate statistics, multi-way modelling and self-organizing maps. The results suggest that groundwater in the lower parts of the watershed is certainly influenced by long-term wastewater irrigation practices. Electrical conductivity, chloride and bicarbonate are dominant chemical variables in the groundwater, which represents the saline conditions of the aquifer. At the current stage, the quality of the groundwater is not suitable for irrigation, this might be the reason farmers tend to adapt for more salinity tolerant crops such as paragrass. The nutrients in the wastewater uptake by plants are the reason for no observed pollution levels with respect to nutrients in the groundwater.
2. Hydrogeochemical and mixing processes that control groundwater geochemistry in the wastewater irrigated system are evaluated using general hydrochemical indices, saturation indices and ionic delta ratios. The hydrogeochemistry of the study watershed is mainly controlled by evaporation and water-rock interactions. The result of the saturation indices tells that halite, gypsum and fluorite are with mineral dissolution and calcite and dolomite are with mineral precipitation. The mineralization processes and ionic delta analysis signifies the influence of monsoon rainfall, carbonate weathering and cation exchange associated with wastewater irrigation, which explains the groundwater geochemistry of the watershed. Accumulation of ions

in the soil medium over the years with long-term wastewater irrigation is also one other important factor in changing the aquifer geochemistry.

3. Multi-functionality of the land use change in the wastewater irrigated peri-urban watershed is mapped using finer spatial resolution images from Google Earth and modelled using land use change modeller to predict the future changes. The results suggest that wastewater irrigation influenced agriculture as the dominant land use. Crop choice is also highly influenced by the availability of wastewater for irrigation in the peri-urban system. Built-up areas are in rise continuously year by year because of the increase in land costs in the peri-urban Hyderabad. Predicted land use change results also suggest that built-up areas will dominate the watershed area in future. Even though built-up areas will increase in future, agricultural areas tend to exist with no major changes, maybe only change in the choice of crop type.
4. Land use change impacts on groundwater recharge conditions in the watershed with the influence of wastewater irrigation are assessed using the constructed finite difference groundwater flow model. The modelling results suggest that the groundwater levels in the lower parts of the watershed tend to increase with the surplus amount of return flows from wastewater irrigated areas. Particularly, return flows from paragrass cropping area is a major contribution to the groundwater recharge in the peri-urban environment. Even though there are increasing impervious surfaces over the last two decades, the recharge from the peri-urban agriculture is balancing the water levels in the micro-watershed.
5. For sustainable groundwater management, four scenarios are generated using groundwater contaminant transport modelling based on groundwater salinity values with the change in quantity and quality of wastewater application for irrigation in the watershed. Modelled groundwater salinity scenarios suggest that the treated wastewater reuse for irrigation can decrease the groundwater salinity and also have the possibility of reversing the aquifer salinity. Any other ideal scenarios only have limited impacts in mitigating the groundwater saline conditions in the watershed. Based on extreme scenarios, application of contaminated wastewater or increasing the amount of wastewater for irrigation can lead to further increasing the salinity values in groundwater.

Overall, wastewater irrigation systems are good examples for assessing the water-soil-waste nexus approach and with respect to the peri-urban systems, there are various synergies and trade-offs involved in managing the environmental resources. Even though the current conditions are providing social and economic benefits to the farmers but creating negative environmental effects in terms of groundwater pollution. For sustainable use of wastewater, land and groundwater resources in an integrated manner, efficient planning and management strategies should be introduced in the view of water-soil-waste nexus to avoid any aquifer contamination and human health risks.

7.2 Significance of the Research

The doctoral research study evaluates the complex flow of the resources and groundwater system dynamics in the wastewater irrigated systems by considering the water-soil-waste nexus approach for efficient management of resources. The extracted information regarding spatial and temporal variations in groundwater quality with analysis and modelling will be useful in developing long-term groundwater management strategies in the watershed. Estimated extent and level of contamination will be helpful to control the human health risks and to conserve the local hydrogeological environment. Farmers and local people around the wastewater irrigated systems are dependent on the local groundwater for domestic consumption and sometimes even for drinking water. So in view of water-soil-waste nexus, the developed sustainable integrated management strategies under irrigation management will be helpful in reducing the health risks, increasing the social benefits and restoring the degrading ecosystems. The study results will provide scientific evidence to decision makers and local people to manage the groundwater resources sustainably. The recommendations from the overall research study can be useful to attain the balance between food production, poverty alleviation and environmental conservation for sustainable development.

7.3 Recommendations and Outlook

1. Wastewater irrigation systems are one of the best examples to practice water-soil-waste nexus approach for sustainable environmental management solutions.
2. Even though the current research study endeavours to analyze the groundwater behaviour in a typical wastewater irrigated system, still more research needs to be carried out to understand the local aquifer's performance based on the irrigated crop or contaminant reactions (heavy metal, pharmaceuticals etc.) to wastewater irrigation.
3. The increasing urban trends around the world will certainly influence the growth of wastewater irrigated areas in urban and peri-urban systems. So, understanding the

social, economic and environmental conditions in an integrated manner is necessary for wastewater irrigated systems.

4. The local governments failed to treat the wastewater generated from the urban centres around the world, especially in the developing countries. The reasons for insufficient wastewater treatment include poor infrastructure and financial costs. Nature-based solutions such as wetland systems, river bank filtration, managed aquifer recharge etc. should be given more importance to treat the wastewater.
5. Further, in future, it is very important to understand different contaminants (heavy metals, nutrients, pharmaceuticals, pesticides, pathogens, antimicrobial resistant bacteria etc.) behaviour in the wastewater irrigation-plant-atmosphere-soil-groundwater continuum for mitigating the human health and environmental risks.
6. Future research should also focus on understanding the soil clogging, solute transport in soil-groundwater medium, nutrient dynamics, carbon emissions etc. in wastewater irrigated systems.
7. To mitigate the impairment of ecological environment in and around the urban agglomerations, measures should be taken to control the untreated or partially treated wastewaters funnelled directly into the oceans, rivers, lakes etc.
8. As the wastewater coming from the cities in the developing world is a mixture of domestic and industrial sewage, future research should also look at different organic and inorganic pollutants from the wastewater and their transport and resident times in the groundwater systems.
9. Future research also should concentrate on antimicrobial resistant bacteria and gene transport to groundwater, which is also an emerging risk associated with wastewater.
10. To avoid any health risks, farmers and local planners should take measures to monitor the food crops chemical and microbiological quality produced with wastewater.
11. Local planners and decision makers should continuously monitor the local aquifers in these peri-urban systems to avoid any contamination or to alleviate the current groundwater pollution levels, where peri-urban dwellers depend on the local aquifers for drinking and domestic purposes.
12. Sustainable measures and practices in wastewater irrigation can help to achieve water and food security as well as respective United Nations Sustainable Development Goals (UN SDGs) 2, 3, 6, 11 and 15 (Figure S7.1).

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Supplementary Material

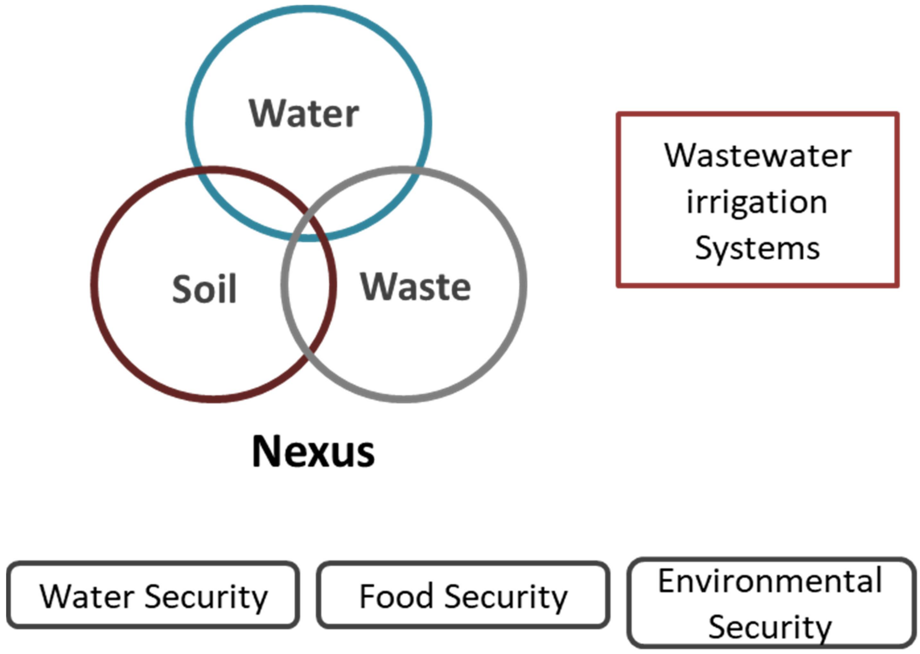


Figure S1.1 Wastewater irrigation systems link to Water-Soil-Waste nexus approach

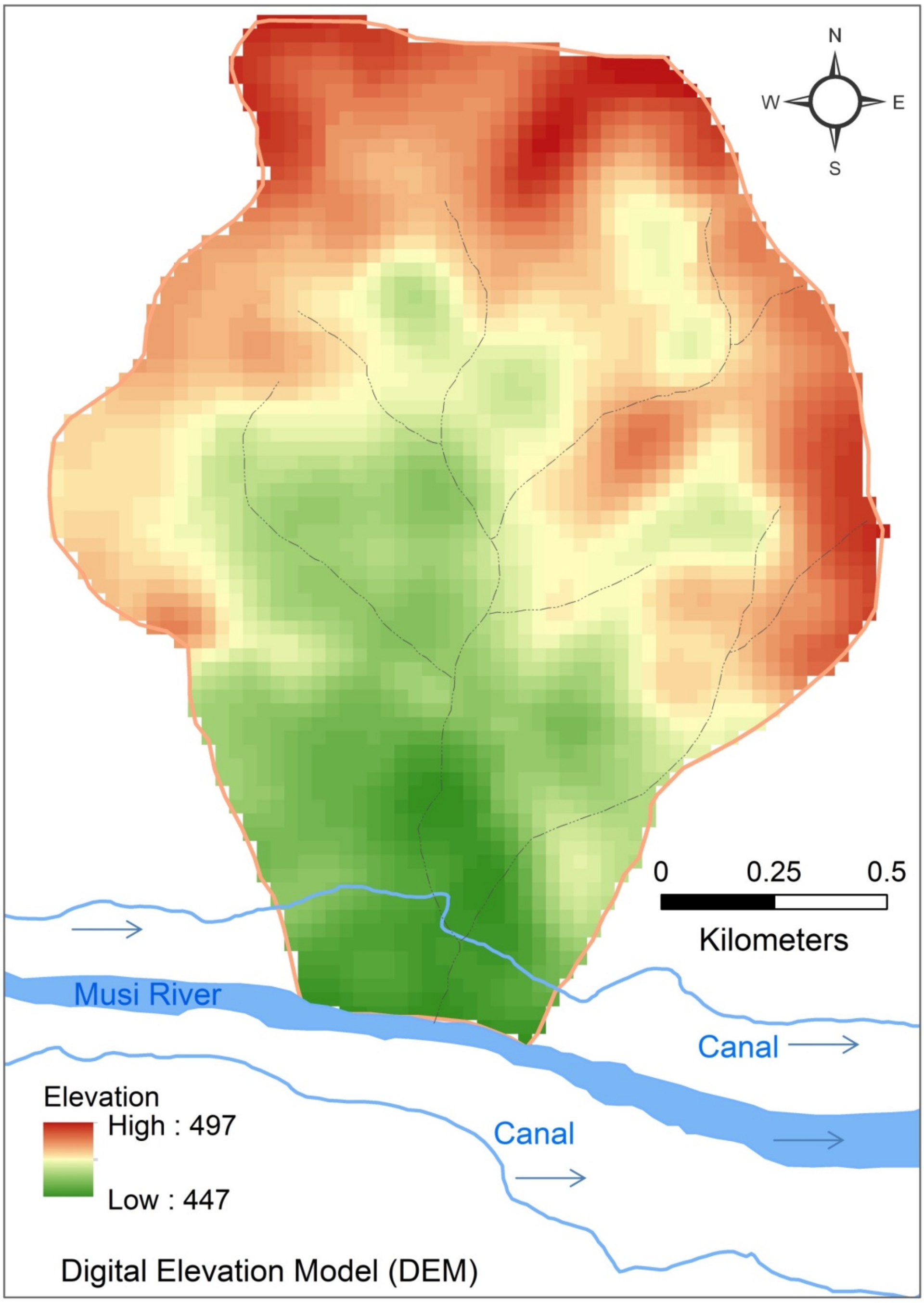


Figure S1.2 Kachiwani Singaram Micro-Watershed (KSMWS) with natural drainage and digital elevation model (DEM) in meters above mean sea level (amsl).

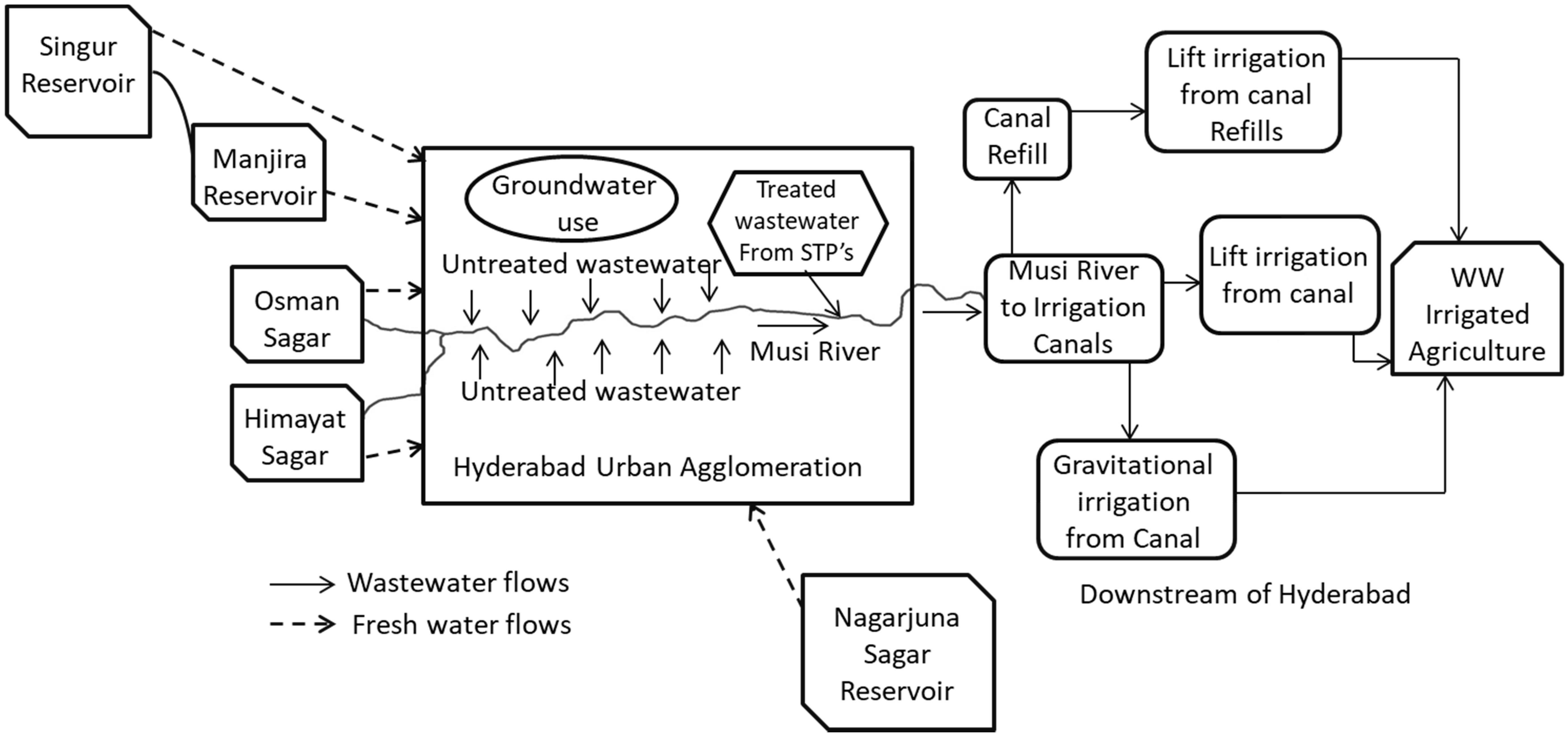


Figure S1.3 Schematic diagram of wastewater flows from Hyderabad urban agglomeration to peri-urban agriculture; adopted from Mahesh et al., 2015b.

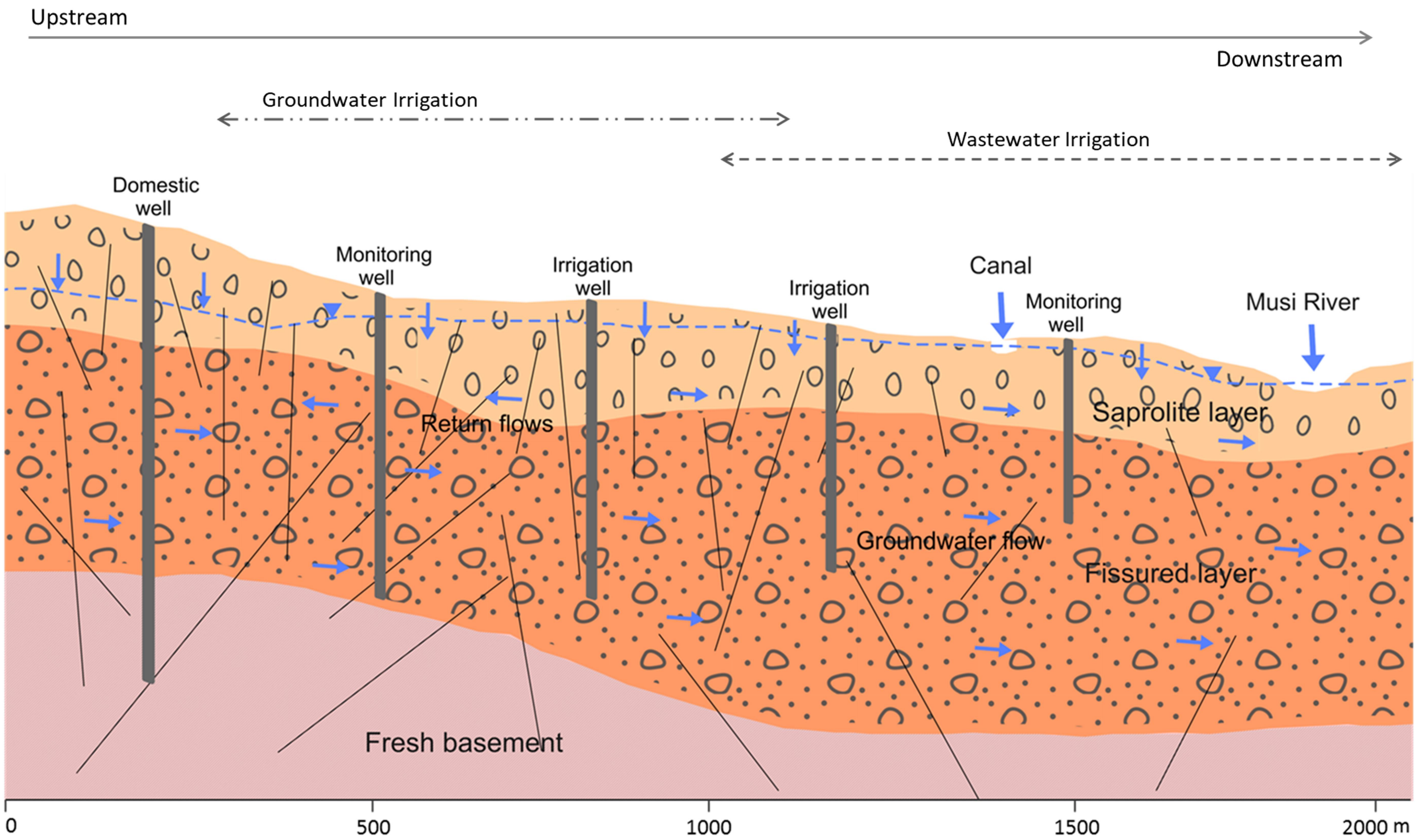


Figure S1.4 Hydrogeological cross-section profile of the study watershed; modified from Mahesh et al., 2015

Table S2.1 Factor loadings explained with percent variance for the months from May to April

Month	Component 1		Component 2		Component 3	
	Dominant parameters	% of variance	Dominant parameters	% of variance	Dominant parameters	% of variance
May	EC, TDS, Cl, SO_4, Na, Mg	37.4	pH, Ca	20.2	DO	13.8
June	EC, TDS, Cl, SO_4, Na, Mg	40.3	HCO_3, NO_3, K	21.0	pH, F, Ca, DO	20.0
July	EC, TDS, Cl, SO_4, Na, Mg	40.8	NO_3, K, PO_4,	23.6	HCO_3	13.9
August	EC, TDS, Cl, SO_4, Na, Mg	46.2	pH, HCO_3, NO_3, K	21.2	F, PO_4, DO	19.2
September	EC, TDS, Cl, SO_4, Na, Mg	42.5	F, PO_4, DO	19.6	pH, K, Ca	15.4
October	EC, TDS, Cl, SO_4, Na, Mg	40.6	pH, Ca	20.0	F, DO	16.1
November	EC, TDS, Cl, SO_4, Na, Mg	36.7	HCO_3, Na, K, NO_3	27.6	pH, F, DO	15.3
December	HCO_3, F, NO_3, Na	27.5	EC, TDS, Cl, Ca, Mg	26.9	pH, DO	16.0
January	EC, TDS, Cl, SO_4, Na, Mg	39.5	pH, PO_4, DO	17.6	K, Ca	17.5
February	EC, TDS, Cl, SO_4, Na, Mg	38.3	F, DO	19.7	HCO_3, K, PO_4	18.7
March	EC, TDS, Cl, SO_4, Na, Ca, Mg	39.3	F, DO	17.2	HCO_3, NO_3, PO_4	16.8
April	EC, TDS, Cl, SO_4, Na, Mg	38.4	pH, HCO_3, F, NO_3, Ca, DO	29.7	K, PO_4	16.6

Extraction method – Principal Component Analysis; Rotation method – Varimax with Kaiser Normalization

Table S2.2 Chemical variable clustering groups from HCA and SOM

Cluster group	Hierarchical cluster analysis	Self-organizing maps
Cluster I	EC, TDS	EC, TDS, Cl
Cluster II	F, PO4, pH, DO, K, NO3, Mg	NO3, PO4, K
Cluster III	SO4, Na, Ca	SO4, Na, Ca
Cluster IV	HCO3	HCO3
Cluster V	Cl	F, Mg
Cluster VI	-	pH, DO

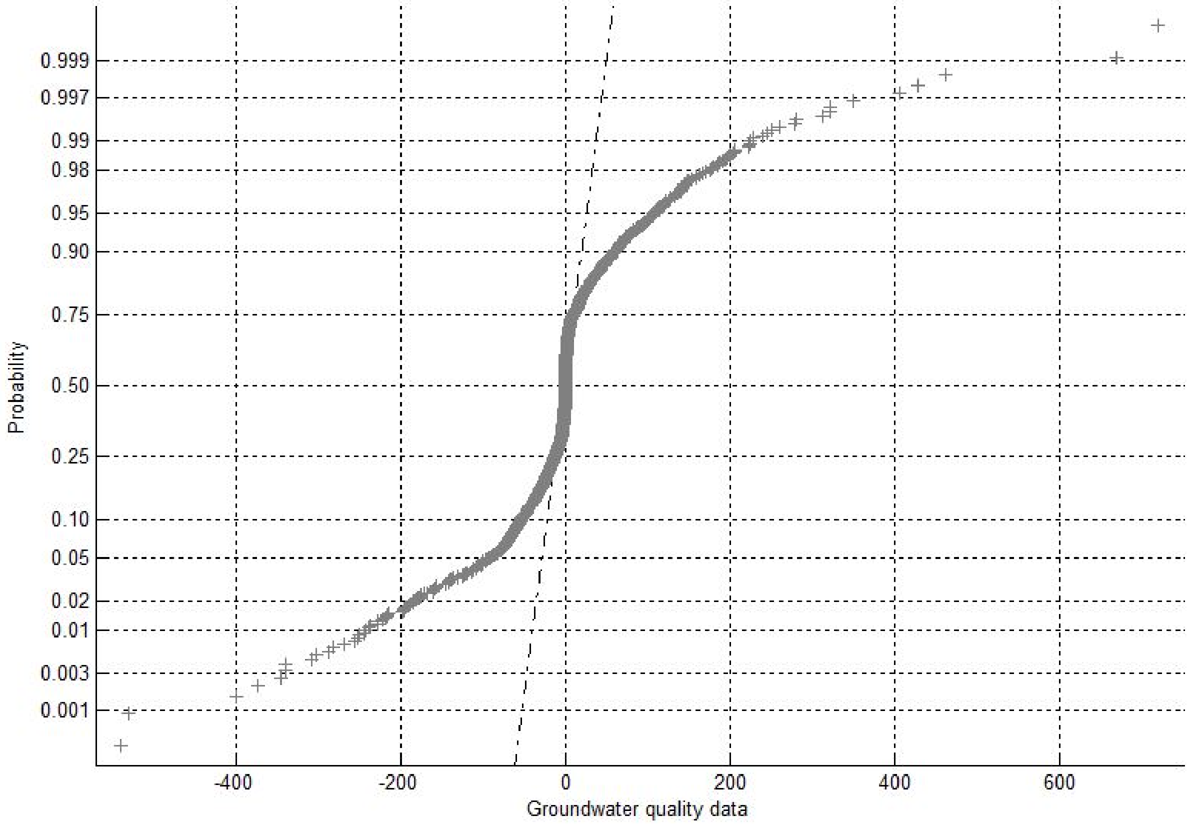


Figure S2.1 Normal probability map of the groundwater quality data in the PARAFAC model

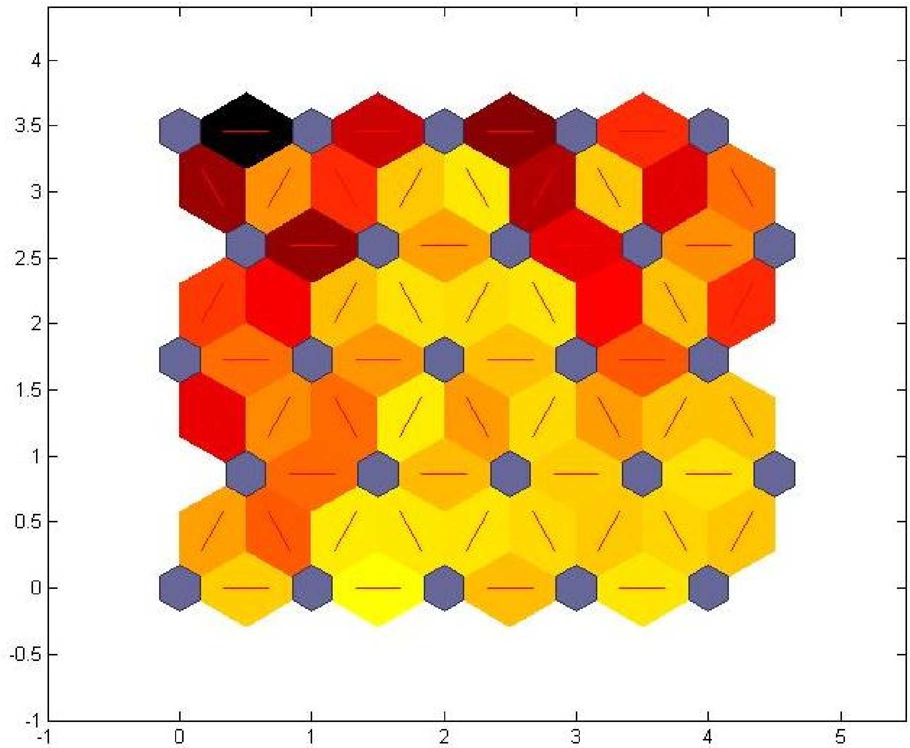
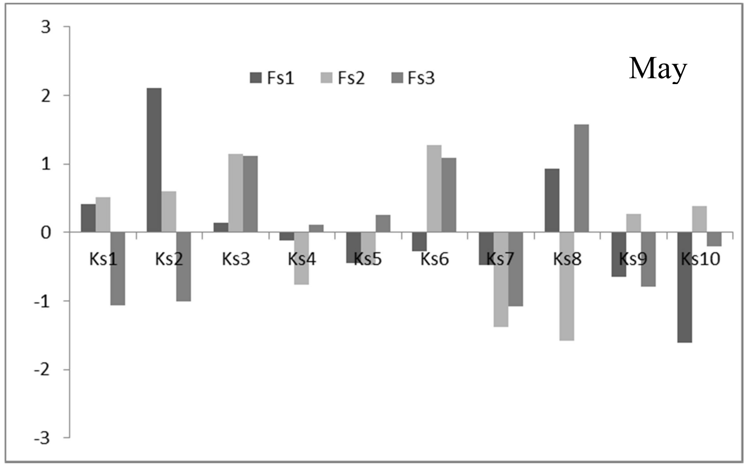
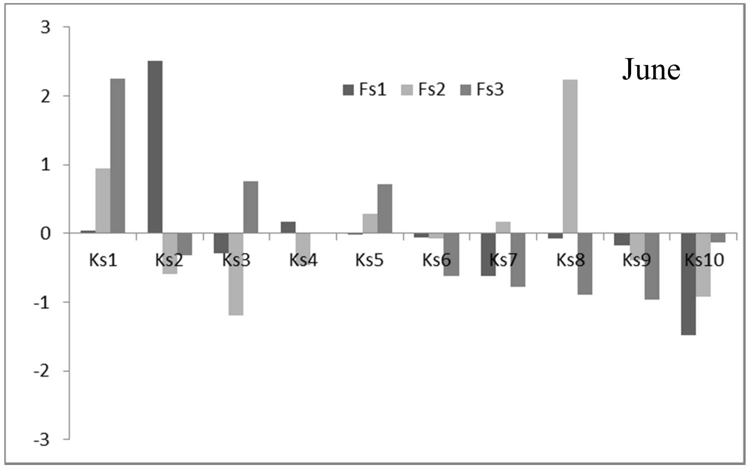


Figure S2.2 U-matrix - SOM neighbour weight distances. Red lines depict the connections between the neurons. The darker colours represent larger distances and the lighter colours represent smaller distances between the neurons.


May
Fs1
Fs2
Fs3
3
2
1
0
-1
-2
-3
Ks1
Ks2
Ks3
Ks4
Ks5
Ks6
Ks7
Ks8
Ks9
Ks10


June
Fs1
Fs2
Fs3
3
2
1
0
-1
-2
-3
Ks1
Ks2
Ks3
Ks4
Ks5
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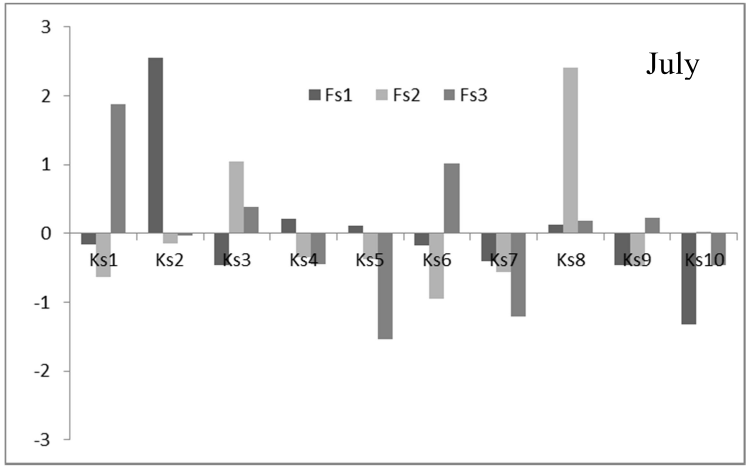
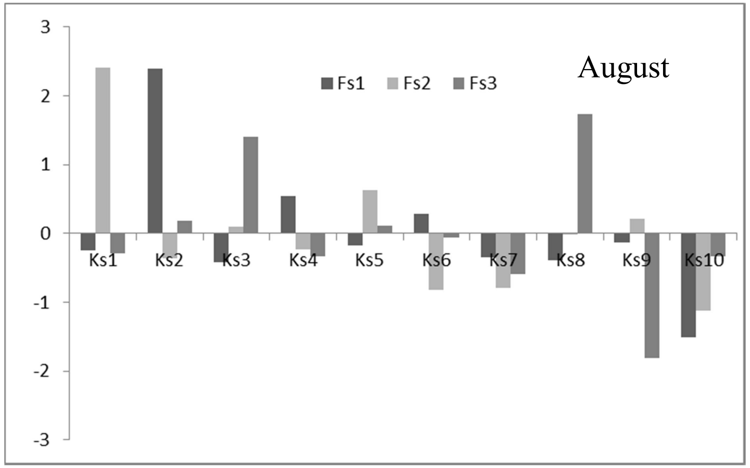
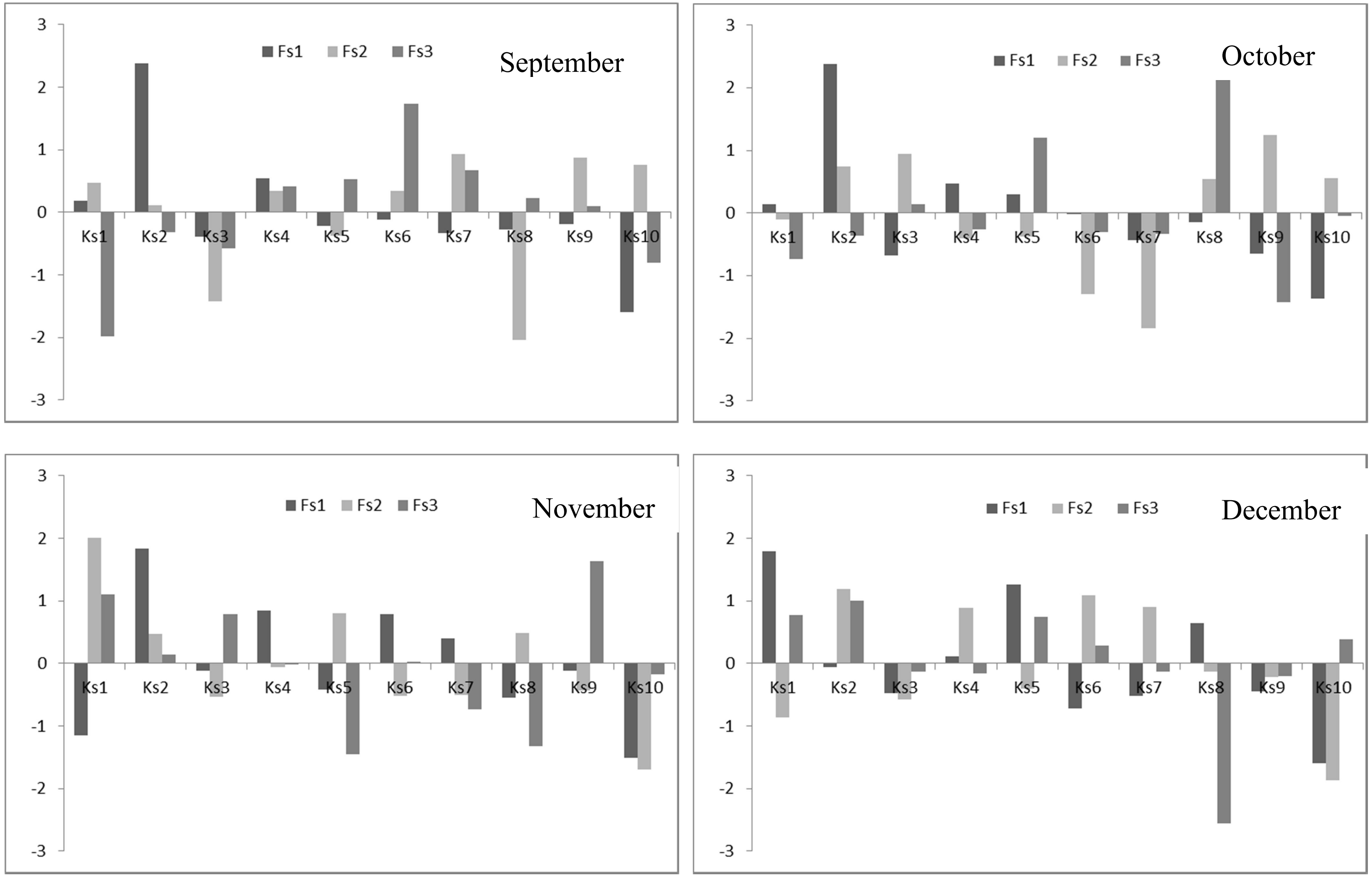
Ks7
Ks8
Ks9
Ks10


July
Fs1
Fs2
Fs3
3
2
1
0
-1
-2
-3
Ks1
Ks2
Ks3
Ks4
Ks5
Ks6
Ks7
Ks8
Ks9
Ks10


August
Fs1
Fs2
Fs3
3
2
1
0
-1
-2
-3
Ks1
Ks2
Ks3
Ks4
Ks5
Ks6
Ks7
Ks8
Ks9
Ks10


September
October
November
December
Fs1
Fs2
Fs3
Ks1
Ks2
Ks3
Ks4
Ks5
Ks6
Ks7
Ks8
Ks9
Ks10
3
2
1
0
-1
-2
-3

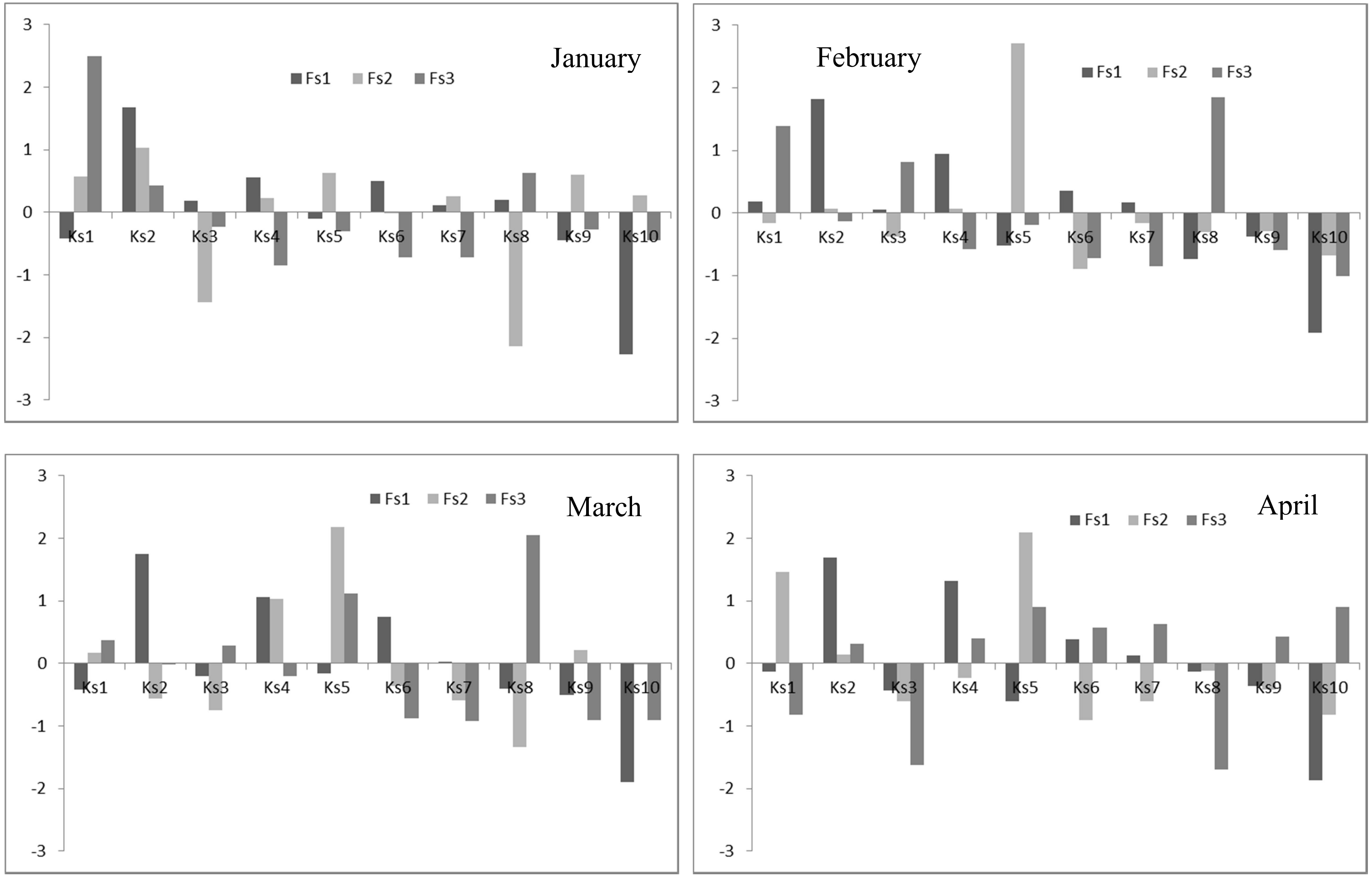


Figure S2.3 Three major dominant factor scores of the groundwater quality data in the corresponding months obtained by Anderson-Rubin method

Table S3.1 Detailed values of mineral saturation indices

May

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.48	1.13	-0.62	-1.55	-5.81
Ks2	0.23	0.53	-0.98	-2.14	-7.09
Ks3	0.48	1.03	-0.53	-1.73	-5.96
Ks4	0.24	0.19	-0.85	-1.90	-7.74
Ks5	0.71	1.26	-0.79	-1.16	-5.53
Ks6	0.79	1.84	-1.04	-1.79	-5.98
Ks7	0.54	1.12	-0.64	-1.32	-5.75
Ks8	0.83	1.27	-0.91	-1.55	-5.99
Ks9	0.67	1.23	-0.74	-1.26	-6.09
Ks10	0.34	0.63	-0.87	-1.39	-6.18

June

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.22	0.36	-0.99	-1.86	-6.03
Ks2	0.29	0.60	-1.03	-1.61	-6.23
Ks3	0.61	0.97	-0.52	-1.07	-5.68
Ks4	0.90	1.99	-0.64	-1.87	-5.84
Ks5	0.86	1.94	-0.92	-1.66	-5.98
Ks6	-0.12	-0.72	-0.85	-1.82	-7.45
Ks7	0.26	0.18	-0.98	-1.78	-6.12
Ks8	0.64	1.35	-0.87	-1.18	-5.57
Ks9	0.82	1.99	-0.76	-1.92	-6.03
Ks10	0.58	1.01	-0.82	-1.20	-6.23

July

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.33	0.48	-0.76	-1.27	-6.32
Ks2	0.51	0.85	-1.02	-1.69	-6.02
Ks3	0.52	1.23	-0.75	-1.73	-5.93
Ks4	0.43	0.61	-0.93	-1.45	-6.17
Ks5	-0.15	-1.00	-0.81	-1.90	-7.83
Ks6	0.54	1.31	-1.13	-1.07	-5.30
Ks7	0.50	1.00	-0.80	-1.41	-5.86
Ks8	0.35	0.63	-0.40	-1.74	-6.13
Ks9	0.84	1.50	-0.60	-1.30	-6.10
Ks10	0.32	0.65	-1.01	-1.47	-6.27

August

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.42	0.85	-0.90	-1.89	-5.83
Ks2	0.38	0.64	-1.04	-1.43	-6.08
Ks3	0.40	1.02	-1.01	-1.52	-5.84
Ks4	0.51	1.59	-1.26	-1.40	-5.57
Ks5	0.60	1.57	-0.85	-1.93	-5.93
Ks6	0.41	0.12	-0.55	-1.84	-7.61
Ks7	0.28	0.17	-0.97	-1.75	-6.02
Ks8	0.41	1.09	-1.15	-1.05	-5.26
Ks9	0.12	0.59	-1.49	-1.61	-6.05
Ks10	0.31	1.15	-0.99	-1.92	-5.98

September

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	-0.45	-0.60	-1.33	-1.70	-6.16
Ks2	0.83	1.28	-0.18	-1.57	-5.91
Ks3	0.39	0.50	-0.73	-1.66	-6.05
Ks4	0.02	-0.07	-1.09	-1.46	-6.12
Ks5	0.47	0.45	-0.70	-1.90	-7.22
Ks6	0.74	1.82	-0.91	-1.91	-5.84
Ks7	0.56	1.34	-1.24	-1.08	-5.31
Ks8	1.08	1.82	-1.00	-1.74	-5.79
Ks9	0.67	1.36	-0.84	-1.29	-5.80
Ks10	0.73	1.49	-0.63	-1.63	-5.90

October

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.50	0.82	-0.91	-1.26	-6.01
Ks2	0.26	0.61	-1.10	-1.49	-6.16
Ks3	0.42	1.06	-1.26	-1.11	-5.36
Ks4	0.24	0.41	-0.67	-1.86	-6.05
Ks5	0.51	0.98	-0.65	-1.38	-6.14
Ks6	0.64	1.59	-1.59	-1.57	-5.95
Ks7	0.95	2.14	-1.07	-1.84	-5.92
Ks8	-0.02	-0.09	-0.86	-2.06	-7.43
Ks9	0.31	0.69	-0.74	-1.60	-6.18
Ks10	0.46	1.04	-0.92	-1.34	-5.77

November

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.50	0.99	-0.57	-1.56	-5.98
Ks2	0.44	0.76	-0.88	-1.25	-6.09
Ks3	0.65	1.17	-1.03	-1.69	-5.89
Ks4	0.69	1.35	-0.97	-1.37	-6.16
Ks5	0.09	0.47	-1.67	-1.61	-6.01
Ks6	0.16	0.25	-0.63	-1.79	-6.02
Ks7	0.34	0.34	-1.04	-1.85	-7.27
Ks8	0.36	0.86	-0.63	-1.65	-6.27
Ks9	0.95	2.02	-1.19	-1.12	-5.38
Ks10	0.58	0.68	-0.84	-1.65	-6.28

December

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.82	1.30	-0.71	-1.15	-5.84
Ks2	0.66	1.02	-0.51	-1.45	-5.80
Ks3	0.58	0.51	-0.69	-1.80	-7.61
Ks4	0.73	1.03	-0.89	-1.17	-6.11
Ks5	0.85	1.50	-0.34	-1.88	-6.07
Ks6	0.71	0.48	-0.71	-1.13	-6.20
Ks7	1.11	2.26	-0.94	-1.89	-6.01
Ks8	0.27	0.50	-1.65	-1.73	-6.32
Ks9	1.04	1.80	-0.88	-1.65	-5.80
Ks10	0.74	1.55	-1.08	-1.15	-5.58

January

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.66	1.31	-0.99	-1.62	-6.21
Ks2	0.85	1.49	-0.67	-1.26	-5.76
Ks3	0.76	1.33	-0.82	-1.34	-6.39
Ks4	0.46	1.05	-0.69	-1.78	-5.87
Ks5	0.63	0.97	-0.78	-1.35	-6.07
Ks6	0.99	1.76	-0.91	-1.29	-5.99
Ks7	0.93	1.83	-1.17	-1.57	-6.02
Ks8	0.58	1.11	-0.51	-1.72	-6.19
Ks9	-0.05	-0.29	-0.92	-2.03	-7.34
Ks10	1.04	2.11	-0.81	-1.69	-5.92

February

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.94	2.08	-1.03	-1.83	-5.72
Ks2	0.42	0.77	-1.19	-1.24	-6.22
Ks3	0.39	0.97	-1.04	-1.69	-6.23
Ks4	0.78	1.24	-0.53	-1.44	-5.80
Ks5	0.95	1.90	-0.88	-1.56	-6.07
Ks6	0.98	1.86	-0.53	-1.45	-5.93
Ks7	0.08	0.26	-0.71	-1.86	-6.00
Ks8	0.66	1.22	-0.72	-1.30	-6.22
Ks9	0.18	0.33	-0.80	-1.28	-5.81
Ks10	0.52	0.89	-1.04	-1.93	-7.66

March

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.64	1.45	-1.21	-1.95	-6.08
Ks2	0.60	1.44	-1.07	-1.35	-5.68
Ks3	0.20	0.64	-0.74	-1.96	-6.00
Ks4	-0.29	-0.51	-1.08	-1.61	-6.03
Ks5	0.60	1.13	-0.45	-1.64	-5.96
Ks6	0.42	0.78	-0.58	-1.35	-5.88
Ks7	0.46	0.89	-0.78	-1.36	-6.20
Ks8	0.32	0.69	-0.91	-1.35	-6.06
Ks9	0.82	1.52	-0.85	-1.51	-6.27
Ks10	0.47	1.10	-1.24	-1.79	-6.20

April

Station	SI (Calcite)	SI (Dolomite)	SI (Fluorite)	SI (Gypsum)	SI (Halite)
Ks1	0.06	-0.31	-0.85	-1.72	-7.06
Ks2	0.18	0.39	-0.87	-1.61	-5.97
Ks3	0.25	0.57	-0.85	-1.44	-6.19
Ks4	0.48	1.21	-0.99	-1.27	-5.66
Ks5	0.65	1.33	-0.63	-1.28	-5.88
Ks6	0.94	1.81	-0.95	-1.47	-6.10
Ks7	0.77	1.56	-0.93	-1.35	-6.15
Ks8	0.69	1.08	-0.41	-1.68	-5.94
Ks9	0.66	1.26	-0.89	-1.83	-6.15
Ks10	0.56	1.07	-0.93	-1.55	-6.25

Table S3.2 Detailed values of wastewater fraction and ionic delta concentrations

May

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.62	-4.22	-1.75	-4.33	-0.96	-7.90	0.25	-1.13
Ks2	2.33	-6.79	2.75	-6.67	-1.34	-10.77	2.55	-1.24
Ks3	2.28	-0.64	-1.62	-10.56	-1.29	-11.94	-0.45	1.15
Ks4	1.33	-4.39	-0.67	-1.67	-0.78	-4.10	-0.65	-0.96
Ks5	0.95	-1.43	0.77	-3.31	-0.53	-2.79	0.14	-0.11
Ks6	1.33	-0.40	-0.67	-5.89	-0.76	-5.60	0.41	-0.39
Ks7	0.73	2.57	0.74	-1.96	-0.41	0.97	1.13	-0.28
Ks8	0.95	-1.43	-0.43	0.41	-0.43	-1.20	-0.75	0.01
Ks9	1.00	-2.40	-0.21	-3.16	-0.54	-3.83	-0.31	-0.19
Ks10	0.29	-1.03	0.50	-1.48	-0.14	-0.83	-0.28	0.75

June

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.00	-3.59	-1.41	3.41	-0.39	-0.44	0.23	-0.63
Ks2	2.95	-5.42	-1.20	-5.62	-1.66	-12.85	3.33	-1.76
Ks3	1.19	-1.08	-0.53	0.78	-0.06	1.69	-1.24	-0.84
Ks4	0.86	-1.08	-0.87	-0.37	-0.51	-1.08	-0.23	-0.10
Ks5	1.14	-2.51	0.05	-0.69	-0.64	-1.61	-0.20	-0.64
Ks6	1.38	-1.77	-1.25	-2.31	-0.82	-4.45	0.74	-0.63
Ks7	0.91	-0.48	0.28	-2.86	-0.51	-2.66	0.85	-0.47
Ks8	1.29	1.37	-0.89	-4.94	-0.68	-3.42	0.57	-0.65
Ks9	1.24	0.34	-0.71	-4.23	-0.70	-3.21	0.09	-0.54
Ks10	0.05	2.22	-0.20	0.15	-0.02	2.65	0.06	0.55

July

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	0.90	-3.25	0.55	2.73	-0.29	1.06	0.12	-0.60
Ks2	3.05	-6.16	-1.16	-5.04	-1.71	-13.65	4.25	-1.94
Ks3	1.10	-0.74	-2.97	0.37	-0.11	-1.85	-0.14	-0.03
Ks4	0.90	-1.25	-2.65	0.69	-0.08	-1.23	-0.42	-0.23
Ks5	1.33	-3.99	0.13	-4.96	-0.72	-7.82	0.80	-0.68
Ks6	1.43	-3.54	-0.63	-2.73	-0.86	-5.78	0.85	-0.72
Ks7	1.10	-3.54	-0.17	-3.75	-0.61	-6.29	0.57	-0.67
Ks8	1.10	-3.93	1.83	-0.96	-0.61	-1.65	0.02	-0.68
Ks9	1.18	0.22	-0.71	-4.10	-0.67	-3.12	0.05	-0.52
Ks10	0.19	1.31	-0.34	-0.75	-0.10	1.10	-0.16	0.56

August

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.14	-2.51	-3.55	2.21	0.17	-1.10	-0.41	-0.96
Ks2	2.81	-5.30	-1.46	-4.47	-1.59	-12.37	3.61	-1.47
Ks3	0.90	-1.25	-1.05	0.24	-0.23	-0.01	-0.71	-0.45
Ks4	0.90	-1.25	-2.25	0.15	-0.10	-1.46	-0.05	-0.48
Ks5	1.19	-1.88	-0.93	-0.76	-0.65	-1.84	-0.16	-0.87
Ks6	1.67	-1.20	-0.73	-4.55	-0.94	-5.52	0.90	-0.94
Ks7	1.12	-1.43	0.14	-3.91	-0.63	-4.23	0.61	-0.64
Ks8	1.38	1.03	-1.25	-4.77	-0.78	-4.11	0.76	-0.69
Ks9	1.43	-3.26	1.05	-4.54	-0.77	-5.03	0.38	-0.95
Ks10	0.10	0.46	1.22	-0.09	-0.05	2.51	-0.10	0.24

September

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.10	-3.14	-0.17	0.02	0.01	-0.93	-0.22	-0.92
Ks2	2.52	-4.68	-0.77	-2.81	-1.41	-10.02	3.77	-1.18
Ks3	0.95	-1.43	-1.23	0.14	-0.34	-0.55	-0.54	-0.54
Ks4	0.76	-1.54	-0.11	0.20	-0.46	-1.24	1.03	-0.20
Ks5	1.00	-0.80	-0.21	0.37	-0.56	0.21	0.31	-0.74
Ks6	1.67	-2.00	0.07	-3.83	-0.95	-5.05	0.96	-0.88
Ks7	1.14	0.68	0.45	-4.07	-0.65	-2.17	0.65	-0.61
Ks8	1.33	2.00	0.13	-5.25	-0.76	-2.67	0.83	-0.62
Ks9	1.33	-3.19	0.93	-4.33	-0.72	-4.77	0.31	-0.90
Ks10	0.19	0.91	0.06	-1.03	-0.10	-1.10	0.01	0.41

October

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.05	-1.77	-1.99	1.43	0.13	-0.16	-0.16	-0.84
Ks2	2.48	-3.71	-1.79	-2.91	-1.39	-9.16	2.68	-1.22
Ks3	0.86	-0.29	-1.27	0.56	-0.40	1.07	-0.88	-0.68
Ks4	0.67	-0.40	-2.15	-0.01	-0.16	-1.00	0.13	-0.31
Ks5	1.19	-0.29	-2.53	1.44	-0.66	-0.51	0.26	-0.77
Ks6	1.48	1.88	-2.81	-2.97	-0.85	-3.74	1.25	-0.69
Ks7	1.05	6.22	-3.59	-3.46	-0.58	-0.71	0.92	-0.44
Ks8	1.29	4.16	-2.09	-4.98	-0.73	-2.47	0.76	-0.50
Ks9	0.95	-2.22	-1.63	-4.00	-0.54	-5.35	-0.21	-0.45
Ks10	0.05	1.82	-0.20	0.15	-0.02	2.09	0.23	0.68

November

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.10	-1.94	-2.97	2.98	-0.21	-0.27	-0.04	-0.79
Ks2	1.86	-2.28	-1.05	-1.56	-1.04	-5.71	2.95	-1.47
Ks3	0.76	-0.34	-0.11	0.10	-0.24	0.88	-0.13	-0.32
Ks4	0.95	-1.03	-0.43	-2.75	-0.39	-2.42	0.06	-0.57
Ks5	1.10	-2.74	-0.97	1.01	-0.64	-1.08	-0.27	-0.76
Ks6	1.67	0.00	-2.73	-3.71	-0.97	-5.37	0.55	-0.77
Ks7	1.19	1.71	-1.73	-3.54	-0.68	-2.48	0.26	-0.51
Ks8	1.38	1.03	-1.65	-4.44	-0.80	-4.10	0.56	-0.57
Ks9	1.10	-0.74	-0.57	-1.62	-0.58	-2.23	0.00	0.21
Ks10	0.14	0.68	0.64	-0.61	-0.08	1.13	-0.18	1.10

December

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	0.95	-1.82	-0.83	2.30	-0.08	0.94	0.21	-0.73
Ks2	1.76	-3.54	-0.69	-3.95	-1.02	-5.21	-1.59	-0.64
Ks3	1.14	-2.11	-0.35	-1.97	-0.53	-1.99	-0.91	-0.62
Ks4	0.71	-0.57	-0.33	0.32	-0.25	-2.93	3.98	-0.18
Ks5	0.95	-1.03	-1.23	2.05	-0.56	0.39	1.31	-0.59
Ks6	1.57	1.14	-3.17	-3.47	-0.94	-4.78	-0.81	-0.87
Ks7	1.10	2.05	0.63	-4.21	-0.62	-1.45	0.92	-0.31
Ks8	1.48	-0.11	-0.01	-6.49	-0.86	-4.41	-0.83	-0.46
Ks9	0.90	-1.65	0.95	-2.40	-0.52	-1.78	0.16	-0.47
Ks10	0.05	1.43	1.40	-0.02	-0.03	2.35	0.13	1.48

January

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	0.86	-2.28	-0.07	0.40	-0.12	-0.16	-0.48	-0.24
Ks2	1.57	-3.25	0.43	-0.58	-0.91	-4.45	2.37	-0.60
Ks3	1.10	-2.34	1.03	-1.41	-0.54	-0.40	-0.97	-0.80
Ks4	1.19	-1.48	-0.13	-2.96	-0.66	-3.26	-0.16	-0.13
Ks5	1.10	-0.74	-0.97	-0.67	-0.65	-0.75	-0.37	-0.74
Ks6	1.43	-0.74	-1.03	-3.20	-0.85	-4.03	0.68	-0.79
Ks7	1.10	0.86	0.83	-4.07	-0.62	-1.97	0.80	-0.41
Ks8	1.43	-0.74	0.17	-5.82	-0.83	-5.52	0.70	-0.37
Ks9	0.95	0.97	-0.03	-3.26	-0.55	-1.13	0.04	-0.43
Ks10	0.29	1.37	-0.30	-1.35	-0.17	1.01	0.10	-0.21

February

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.00	-1.60	-0.61	0.43	-0.11	-0.37	0.37	-0.78
Ks2	1.62	-3.02	1.05	-0.70	-0.92	-4.49	3.08	-0.60
Ks3	0.81	-0.91	-0.69	-0.10	-0.34	0.19	-0.53	-0.54
Ks4	0.86	0.51	0.33	0.27	-0.44	0.88	0.78	-0.07
Ks5	1.00	-0.40	-2.21	1.11	-0.57	0.19	-0.50	-0.75
Ks6	1.52	-0.68	-0.19	-4.36	-0.91	-4.70	1.19	-0.87
Ks7	1.10	-0.34	1.03	-3.92	-0.62	-2.48	0.69	-0.50
Ks8	1.24	-0.06	0.49	-5.01	-0.71	-4.08	0.93	-0.29
Ks9	0.98	-0.11	-0.12	-3.36	-0.57	-2.22	0.04	-0.39
Ks10	0.05	1.82	1.00	-0.31	-0.02	2.69	0.06	0.84

March

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	1.00	-3.19	-0.21	0.28	-0.13	-1.43	0.31	-0.76
Ks2	1.95	-2.22	-3.81	-1.51	-1.09	-7.71	2.09	-0.87
Ks3	1.05	-0.97	-1.19	-1.27	-0.64	-1.09	-1.04	-0.65
Ks4	0.95	-0.23	-2.43	0.69	-0.49	-1.02	0.10	-0.21
Ks5	1.00	-1.60	-0.21	0.61	-0.60	0.05	-0.21	-0.61
Ks6	1.81	-2.11	-1.27	-5.15	-1.06	-6.98	0.64	-1.14
Ks7	1.14	-0.11	-0.35	-4.34	-0.65	-3.89	0.67	-0.36
Ks8	1.33	1.20	-0.27	-5.23	-0.76	-3.95	0.98	-0.36
Ks9	1.00	-1.20	-0.21	-3.46	-0.58	-3.30	0.05	-0.35
Ks10	0.10	1.65	0.02	-0.20	-0.06	2.05	0.12	0.46

April

S No	F_{ww}	ΔCa	ΔMg	ΔNa	ΔK	ΔHCO_3	ΔSO_4	ΔNO_3
Ks1	0.95	-2.22	0.77	1.06	-0.09	0.41	0.77	-0.71
Ks2	1.76	-1.94	-0.69	0.09	-1.00	-4.18	2.86	-0.82
Ks3	0.95	-1.03	-1.63	0.04	-0.35	-0.78	-0.27	-0.64
Ks4	0.90	-0.46	-2.25	-0.94	-0.24	-1.65	-0.73	0.02
Ks5	0.86	-2.28	1.93	1.46	-0.48	2.19	-0.21	-0.74
Ks6	1.71	0.63	-2.91	-2.39	-1.01	-5.12	2.30	-1.16
Ks7	0.81	3.08	0.91	-2.08	-0.45	1.24	1.36	-0.29
Ks8	1.00	2.79	0.59	-3.26	-0.56	-1.04	1.68	0.19
Ks9	1.05	0.63	-1.59	-3.31	-0.61	-2.77	0.11	-0.53
Ks10	0.00	2.40	-0.02	0.92	0.00	3.28	0.02	0.98

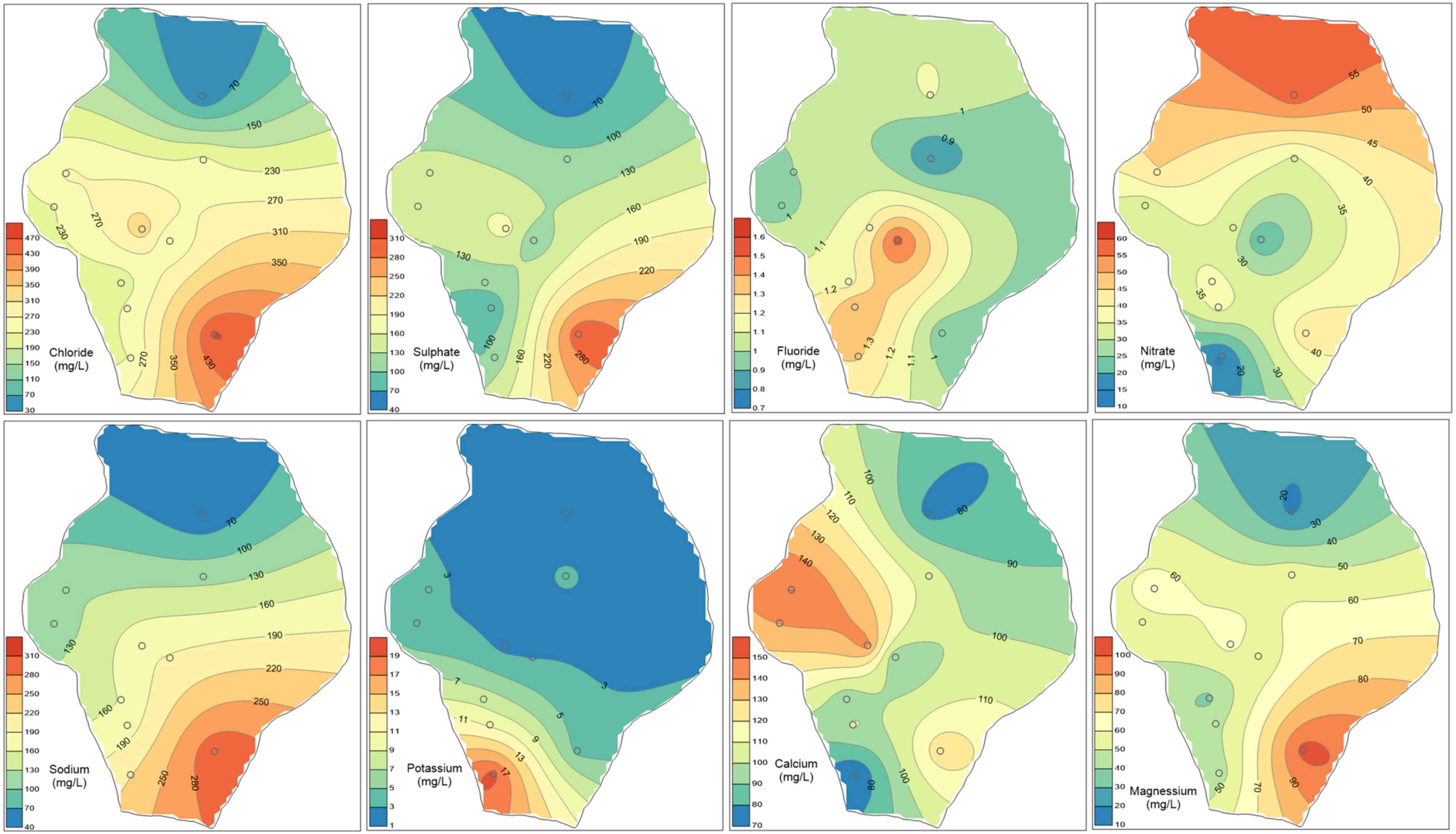


Figure S3.1 Average (one hydrological year) groundwater quality variations of major ions (Cl, SO_4, F, NO_3, Na, K, Ca, Mg).

Table S4.1 Google Earth imagery information used for land use analysis

Year	Number of Google Earth Images	Images with Cloud Cover	Clear images available between April to May	Date of the Google Earth Image
2000	2	0	1	27th May
2001	1	0	0	-
2002	0	0	0	-
2003	2	0	1	26th April
2004	0	0	0	-
2005	2	0	0	-
2006	1	0	1	08th April
2007	0	0	0	-
2008	1	0	0	-
2009	1	0	1	27th May
2010	2	1	1	05th April
2011	2	1	1	02nd April
2012	1	0	0	-
2013	5	1	1	31st March
2014	9	2	1	12th May
2015	5	0	1	27th April

Table S4.2 Farmer sample population according to gender, age, education, farm size and agricultural practice (adopted from Mahesh et al., 2015b)

Farmer distribution	Wastewater	Groundwater
Gender		
Male	36	9
Female	3	-
Age		
25 - 30	1	-
30 – 40	9	3
40 - 50	15	2
50 - 60	9	2
>60	5	2
Education		
Illiterate	9	1
Primary	12	2
Secondary	15	4
Intermediate	3	2
Graduate	-	-
Farm size in ha		
<0.5	17	2
0.5 - 1	10	1
1 - 3	10	5
3 – 5	-	1
5 - 10	1	-
Crop type*		
Paddy Rice	6	6
Paragrass	16	4
Vegetables	6	4

*Some farmers own the agricultural plots for more than one crop

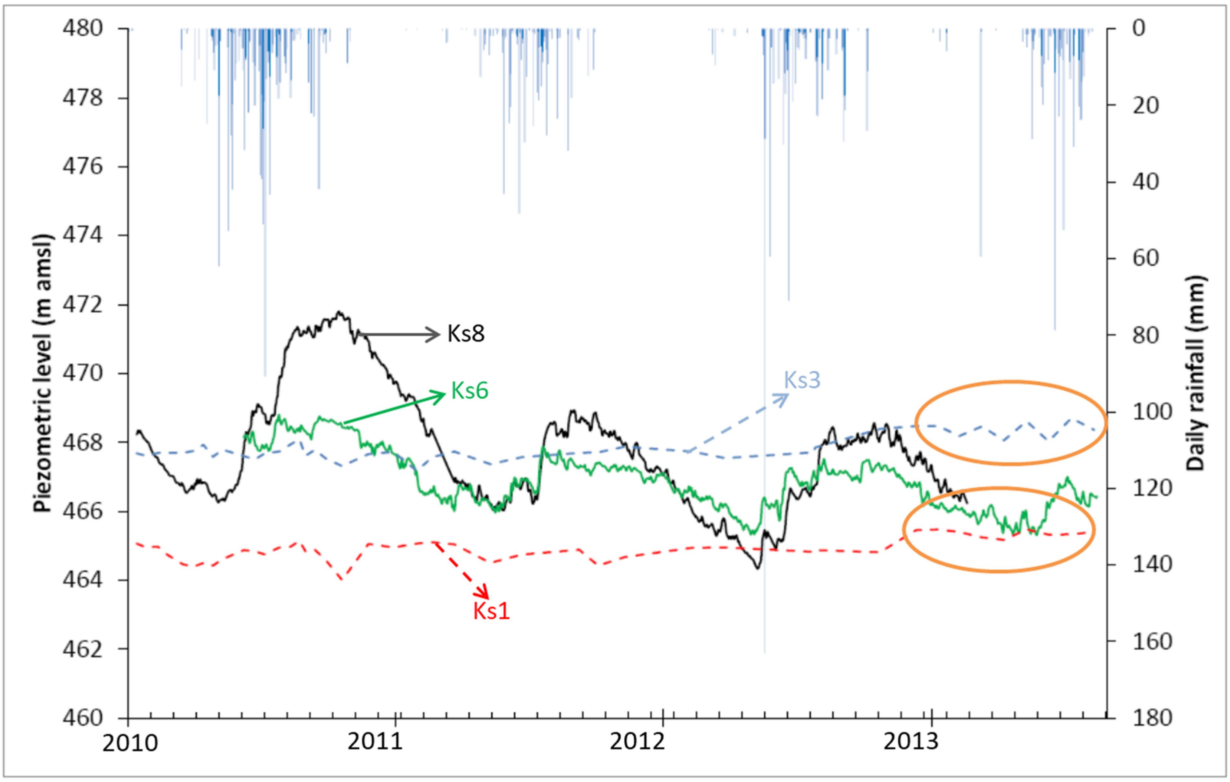


Figure S5.1 Observed groundwater levels in the watershed

Figure S7.1 Current research work direct and indirect relation with the United Nations Sustainable Development Goals (UN SDGs)