Addressing groundwater over-extraction in India: assessments, monitoring methods and interventions

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contribution

This manuscript-style thesis includes introductory material in Chapters 1 and 2, followed by manuscripts, prepared for publishing in peer-reviewed journals, in Chapters 3, 4, and 5. Thesis conclusions were presented in Chapter 6.

Tejasvi Hora was the sole author of the introductory material and conclusions in Chapters 1 and 2 and Chapter 6, written under the supervision of Richard Kelly and Nandita Basu. This material was not written or submitted for publication in its current form.

Chapter 3 has been submitted to the journal Ecological Indicators. Conceptualization was a collaborative effort between all the co-authors (Tejasvi Hora, Nandita Basu, Johanna Wandel and Richard Kelly). Tejasvi Hora wrote the manuscript with edits from Nandita Basu, Johanna Wandel and Richard Kelly.

Chaper 4 was published in the peer-reviewed journal Geophysical Research Letters (Hora et al., 2019). Nandita Basu and Tejasvi Hora conceived of the study. Tejasvi Hora led the processing of the data, and Tejasvi Hora, Veena Srinivasan and Nandita Basu contributed to the analysis and interpretation of the results. Tejasvi Hora, Veena Srinivasan and Nandita Basu all contributed to the writing and editing of the manuscript.

Chapter 5 will be submitted to the journal Agricultural Water Management. Conceptualization was a collaborative effort between all the co-authors (Tejasvi Hora, Nandita Basu and Richard Kelly). Tejasvi Hora wrote the manuscript with edits from Nandita Basu and Richard Kelly.

Abstract

Groundwater is a vast distributed source of water that is critical for meeting the demands of various socio-environmental systems globally. However, the management of groundwater resources has proven to be challenging with groundwater depletion being observed in many regions globally. As the country with the highest groundwater extraction (and depletion) rates in the world, India is currently at the forefront of this problem where national food security and the livelihoods of millions of households have grown to become dependent on the over-exploitation of groundwater resources. This dissertation consists of three studies to support the broad goal of addressing groundwater overexploitation in India. Specifically, these studies aim to improve understanding on: (1) assessments of stress on regional groundwater resources, (2) identification of groundwater depletion hotspots using monitoring data, and (3) the potential of rain-water harvesting systems as interventions to increase groundwater supplies.

The goal of the first study was to understand the effects of incorporating environmental considerations into large-scale groundwater assessments. Assessments of regional groundwater stress (measured here as the ratio of annual groundwater usage to renewable groundwater supply) are important for setting policy targets and guiding interventions. However, the threshold of yearly groundwater supply that is considered available for human use (especially in relation to environmental water demands) remains poorly defined at the regional-scale. In this study, groundwater extraction thresholds were estimated by scaling yearly groundwater recharge volumes based on different local and global environmental considerations. Focusing on India, district-scale groundwater use thresholds were developed based on: (a) no environmental considerations ('baseline'), (b) water requirements of 'local' groundwater-dependent ecosystems, (c) 'global' considerations using the current planetary boundary framework, and (d) a 'mixed' approach that is informed by both local and global considerations, but where a national groundwater use budget is disaggregated (top-down) to estimate thresholds based on current district-level extraction rates. This was followed by an assessment of how hotspots related to groundwater stress (i.e. regions where groundwater extraction rates exceed estimated thresholds) change in each scenario.

Compared to the baseline (where 26% of the districts were considered over-stressed in India), it was found that accounting for local environmental flow requirements results in 36% districts being classified as over-stressed with a groundwater stress hotspot emerging in Southern India. Under the global and mixed scenarios, results showed that nearly 70% of districts (where currently >801 million people live) are classified as over-stressed given current groundwater extraction rates. However, the effort required from over-stressed districts to stay within derived groundwater use thresholds in the mixed scenario (median groundwater stress = 143%) was found to be lower than the global scenario (median groundwater stress = 203%). Overall, the results from this analysis suggest that incorporating environmental considerations would significantly decrease the volume of groundwater resources available for human use in India (173-312 km^3/year; compared to 399 km^3/year in the baseline).

The aim of the second study was to improve how groundwater depletion hotspots are identified using monitoring data. Numerous recent studies have highlighted groundwater recovery in Southern India due to increasing rainfall rates and political interventions. However, these estimates of increasing groundwater storage trends obtained using hydrological data sources (monitoring wells, GRACE satellite) were found to be incongruent with reports of well failures from non-hydrological data sources (like census data and news articles). Results from this study revealed that previous trend estimates relying on monitoring well data were skewed by the presence of a survivor bias, where dry or defunct wells were excluded from trend analyses due to missing data. Upon further investigation, the timing of missing data and the location of wells with missing data were found to be strongly correlated with metrics of climate stress (i.e. dry periods) and groundwater irrigation intensity, which was indicative of a systemic exclusion. Two alternative metrics that better accounted for information from dry and defunct wells were developed to help augment analysis relying on water level measurement from monitoring wells. An assessment based on these metrics revealed increasing groundwater depletion rates in Southern India between 1996-2016.

In the third study, the potential of rain-water harvesting systems (RWH or tanks) as an intervention to increase groundwater supplies and provide farmers with an al-

ternative source of water was assessed in Southern India. Agricultural rain-water harvesting (RWH) structures remain a promising intervention for improving water availability for small-holder farmers in arid and semi-arid regions of the world. However, the feedback between RWH systems and the surrounding aquifer remains poorly understood in regions like Southern India where these structures are nestled within a landscape of intense groundwater development. In this study, a conceptual hydrological model was developed to answer fundamental questions about how RWH structures impact groundwater availability for irrigation, and in turn how groundwater irrigation impacts the outflow fluxes from RWH structures. Model simulations highlighted that agricultural RWH structures were able to increase groundwater availability in the surrounding area. However, these impacts were meaningful (in meeting agricultural water demands) under only a narrow spectrum of landscape and climate conditions. Specifically, the impact of tanks was found to decline significantly during drought spells or when the beneficiaries of tank-induced groundwater recharge was poorly regulated. Alternatively, results showed that groundwater irrigation in the surrounding aquifer positively impacted the efficiency of output fluxes from the RWH structures by reducing the percentage of evapotranspiration losses and increasing groundwater recharge, however, this came at the cost of reduced water available for surface irrigation. This study provides crucial information to understand the potential of RWH structures in contemporary small-holder dominated agricultural systems.

Overall, the results from this dissertation provide critical insights to support science-based decision-making to minimize environmental impacts of anthropogenic groundwater use, improve monitoring of regional groundwater resources, and better evaluate interventions aimed at increasing (ground)water availability. These insights can aid current efforts to improve groundwater management in India.

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Table of Contents

List of Tables xv

List of Figures xviii

1 Introduction 1

1.1 Research Context . . . 1

1.1.1 Global groundwater crisis . . . 1

1.1.2 Groundwater depletion in India . . . 3

1.2 Research Approach . . . 7

1.2.1 Specific Knowledge Gaps . . . 8

1.2.2 Objectives . . . 10

1.3 Dissertation Structure . . . 11

2 Background 13

2.1 Groundwater and the Water Cycle . . . 13

2.2 Basics of Groundwater Flow . . . 15

2.3 Impact of Groundwater Pumping . . . 17

2.4 Extraction Rates and Groundwater Sustainability 18
2.5 Groundwater Stress Assessments 19
2.6 Monitoring Groundwater Depletion 25
2.7 Interventions 29
2.7.1 Rain-Water Harvesting in India 30

3 Incorporating local and global environmental considerations into groundwater stress assessments in India 35

3.1 Introduction 36
3.2 Methods 40
3.2.1 Datasets 42
3.2.2 Estimating district-scale groundwater use thresholds 43
3.2.3 Estimating district-scale groundwater extraction 49
3.3 Results 50
3.3.1 Impact of environmental considerations on extraction thresholds . . 50
3.3.2 Assessing district-scale groundwater stress 53
3.4 Discussion 58
3.4.1 Incorporating environmental considerations 60
3.4.2 Staying within thresholds 61
3.4.3 Limitations 63
3.5 Conclusions 64

4 The Groundwater Recovery Paradox in South India 66

4.1 The paradox of rising farmer distress in areas of "groundwater recovery" . 67

4.2 Methods 68

4.3 Results and Discussion 70

4.3.1 Groundwater storage trends in India 70

4.3.2 Stable water levels are because of survivor bias 74

4.3.3 Alternative Metrics of Groundwater Stress 77

4.3.4 Summary and Implications 79

5 Examining rainwater harvesting structures in groundwater intensive irrigation systems 81

5.1 Introduction 82

5.2 Methods 87

5.2.1 General Description 87

5.2.2 Simulation of Tank Water Balance 89

5.2.3 Simulation of Water Table Dynamics 94

5.2.4 Simulation of Soil Moisture Dynamics 96

5.2.5 Forcing Data, Planting Date and Irrigation 97

5.2.6 Parameters, model validation and simulations 99

5.3 Results and Discussion 104

5.3.1 Sensitivity Analysis and Model Validation 104

5.3.2 Impact of tanks on groundwater storage 109

5.3.3 Understanding the influence of tank-recharge beneficiaries 113

5.3.4 Impact of groundwater irrigation on tank outflow dynamics 115
5.4 Discussion and Conclusions 117

6 Conclusions **123**
6.1 Summary of Work 123
6.2 Key Research Contributions 126
6.3 Recommendations for Future Work 127

References **130**

APPENDICES **169**

A Supplementary Material for Chapter 3 **170**

B Supplementary Material for Chapter 4 **178**

C Supplementary Material for Chapter 5 **189**

List of Tables

2.1 Average residence of different global water stores (adapted from Nelson et al., 2009) 15

2.2 List of groundwater resources sustainability indicators developed by UNESCO/IAEA/IAH (Vrba et al., 2007) 22

3.1 Different scenarios used to obtain district-level groundwater use thresholds in India 41

5.1 Tank constants based on field measurement by Steiff (2016) 94

5.2 Model parameter for the sub-surface used in the study 100

5.3 Different simulations used to understand the impact of tanks in our modelled system 103

A.1 Variables from the VDSA database used in the analysis. 171

A.2 Irrigation water requirements for different crops considered in the analysis. Irrigation water requirements assume conventional irrigation technology. . . 172

A.3 Ground water stress (in %) values in regions where base flow reductions have been observed. Overall, we find that groundwater stress is underestimated in the baseline scenario, which can potentially misclassify a district as 'safe' (even though it is overstressed). Please note that base flow reductions can be observed even when GSR values are under 100, and thus an assessment like this can at most complement field scale studies. 176

B.1 Description of the different data sources used to assess groundwater stress in South India 188

C.1 Crop parameters used in the analysis 190

C.2 Tank equation based on field observation by Steiff (2016) 190

C.3 Precipitation thresholds used to determine antecedent moisture conditions 191

C.4 Equation used to estimate runoff using the SCS-Curve Number approach . 191

C.5 Criteria used to estimate the baseline Groundwater Spread Area (GSA) parameter in modelled system. For a GSA of 3 surface command area units (surface command area for modelled system is equal to 27 ha), we found maximum NSE-coefficient and minimal differences between modelled and estimated (based on field data) tank outflow volumes. Note, that for GSA >4 surface command area units the tank behaviour remained unchanged for the single season with field measured tank data. 192

C.6 Pairwise comparisons using the Mann-Whitney U Test to assess difference in *monsoon* season %ET requirements met over the simulation period (54-years). Cells show the statistical significance of test results for each combination of model configuration (represented by rows and columns). $*$ represents results where the distribution means are significantly different ($p < 0.05$), while X represents results where distribution means are not statistically significant ($p > 0.05$). 196

C.7 Pairwise comparisons using the Mann-Whitney U Test to assess difference in *Dry* season %ET requirements met over the simulation period (54-years). Cells show the statistical significance of test results for each combination of model configuration (represented by rows and columns). $*$ represents results where the distribution means are significantly different ($p < 0.05$), while X represents results where distribution means are not statistically significant ($p0.05$). 197

List of Figures

1.1 Temporal trend of country-level groundwater extraction in km^{3}year^{-1} (Modified from Shah et al., 2007) 5

2.1 Components of the water cycle with volumetric estimates of water storage in each global water store (adapted from Trenberth et al., 2007) 14

2.2 Cross-section showing a groundwater flow system comprising of an unconfined and confined aquifer system (adapted from Dingman, 2015) 17

2.3 Impact of groundwater pumping on the water balance of a stream. The figure highlights the change from a discharging to recharging stream due to groundwater storage declines in the connected aquifer. (adapted from de Graaf et al., 2019) 20

2.4 Using groundwater stress values to categorize districts in India. (Figure from Sidhu et al., 2020 and CGWB, 2014) 23

2.5 A) Illustration showing the measurement of water table elevation in the unconfined aquifer using a tape measure (from Or et al., 2005), and B) Theoretical representation of a groundwater table timeseries highlighting: 1) seasonality in water level due to precipitation patterns and 2) groundwater depletion over time (adapted from Shah, 2009) 26

2.6 Important components of a tank irrigation systems (adapted from Van Meter et al., 2014) 31

3.1 Distribution of district-level groundwater extraction thresholds (area-averaged) estimated in each scenario (Table 3.1). The 3 horizontal lines from *top to bottom* in each violin represent the 75^{th}, 50^{th} and 25^{th} percentile values. 52

3.2 Incorporating local environmental consideration to estimate groundwater use thresholds in the 'local' scenario, a) Estimated percent environmental flow requirements (%EFR) of groundwater dependent ecosystems using modelled groundwater discharge and recharge data from the WaterGap hydrological model. b) Volume of annual groundwater recharge available for use in each district after subtracting %EFR. 54

3.3 District-level (volumetric) thresholds derived in the: a) global scenario, where the derived thresholds are consistent with the freshwater planetary boundary; and b) mixed scenario by developing a national ceiling of groundwater use in India before disaggregating to the district-level based on the 'grandfathering' allocative principle. The derived boundaries are then harmonized to be consistent with local hydrological limits while allowing regions to meet potential developmental requirements. 55

3.4 District-level groundwater stress for each scenario analyzed in the study (Table 3.1). Regions in shades of orange and red represent districts where groundwater extraction exceeds the estimated district-level groundwater availability threshold. 57

3.5 Categorization of districts based on the number of scenarios in which groundwater extraction rates there exceed the estimated groundwater availability thresholds. 59

4.1 Long-term groundwater storage trends in India: Yearly trend ($cm\ yr^{-1}$) in groundwater anomaly from GRACE analysis for (a) 2002-2016 and (b) 2005-2016. Dots represents areas with statistically significant trends ($p-value \leq 0.1$). (c) Percent change in the distribution of positive, negative and non-significant GRACE based trends as a function of the analysis timeframes in NI (above 23^o N) and SI (below 23^o N). Results highlight that the percent area with positive and no significant trends are highly sensitive to the period of analysis in Peninsular India. (d) Percentage of wells with non-significant trends ($p-value \geq 0.1$) in May aggregated at the State-scale. Each circle represents a State and the size of the circle represents the percentage of wells with positive trends ($p-value \leq 0.1$). Red circles represent States with greater negative trends than positive, while blue circles represent States with more positive trends. Results show that a majority of States have a large percentage of wells ($\geq$ 50%) have non-significant long-term trends. (e) Water level trends ($cm\ yr^{-1}$) in shallow monitoring wells. (f) Water level trends ($cm\ yr^{-1}$) in deep monitoring wells. Note that wells trends are presented for the pre-monsoon season (May), and all wells used have statistically significant trends ($p-value \leq 0.1$) 71

4.2 Dependence of dry well density on rainfall and groundwater irrigated area: a) Percentage of dry monitoring wells (after removing trend) aggregated at the NI and SI scale (referred to as $\%DryWells_t$) versus the 36-month standardized precipitation index (SPI) for each month between 1996-2016. SPI-values and percent dry wells were estimated for the each of the four months of available groundwater level data (January, May, August and November). b) Percentage of dry monitoring wells at the district scale, aggregated over the 1996-2016 timeframe (referred to as $\%DryWells_{1996-2000}$), versus the percentage of groundwater irrigation in the district. The %dry well aggregation was done by taking the median %dry well value between 1996-2016. The percent groundwater irrigation represents the median percent groundwater irrigation in each district between 2005-2010. Note that only districts where at least 20% of the gross cropped area is irrigated are shown. Figure highlights the dependence of %dry wells on natural (precipitation) and anthropogenic (groundwater irrigation) factors. 76

4.3 Spatial patterns of dry and defunct wells: a) Percent of defunct wells in 2016, calculated as the ratio of wells in grid cells (1^o by 1^o) that permanently stopped collecting data to all the long-term monitoring wells present. b) Median percentage of dry monitoring wells between 1996-2016 in each grid cell. c) Median percentage of dry monitoring wells between 1996-2000. d) Median percentage of dry monitoring wells between 2013-2016. The % dry wells was calculated as a ratio of wells in grid cells (1^o by 1^o) that did not record data in a time period to all the long-term monitoring wells present there. Higher percentage of dry/defunct wells (orange/red) indicates relatively more groundwater depletion, and lower percentage of dry/defunct wells (yellow/blue) indicates relatively less groundwater depletion. Please note that long-term monitoring wells in our analysis refers to monitoring wells active in 1996 78

5.1 Components of a typical tank structure, where runoff from a catchment area is collected behind a bund (earthen dam) and the stored water is released to a command area downstream. 84

5.2 Plan view showing the different components of a tank system. The groundwater spread area (GSA) is a region assumed to be linked to tank-based recharge and can be differentiated into two regions: (1) area that has access to the sluice released and groundwater recharged by tanks (brown); (2) region that has access to tank-based groundwater recharge (grey). 89

5.3 Schematic of the framework developed in this study highlighting the different components and processes of the modelled tank system. 90

5.4 Comparing modelled tank stage with measured data from the 2013 monsoon season collected by Steiff (2016) in tank 1 of the Thirumal Samudram cascade system. (a) measured rainfall data from the 2013 monsoon season, (b) measured and modelled daily tank stage data. An elevation of 0m represents groundwater level. 105

5.5 Comparing modelled tank dynamics with estimated tank outflow volumes from the 2013 monsoon season collected by Steiff (2016) in tank 1 of the Thirumal Samudram cascade system. The sluice, ET and groundwater exchange volumes were estimated using the White method that relies on diurnal fluctuations in water level (see Van Meter et al. (2016) for more details). (a) Estimated and modelled volumes (m^3) evapotranspiration loss, sluice outflow and groundwater exchange for the entire season; (b-d) comparison of estimated and modelled weekly fluxes of groundwater exchange, sluice outflow and evapotranspiration from the tank 106

5.6 Comparison between the annual groundwater table elevation (m) simulated by the model and groundwater level measured by the CGWB between 2000-2017 (Well ID: W02232). Results indicate a statistically significant correlation between measured and modelled groundwater storage. 108

5.7 The effects of tank water storage on mean monthly %ET requirements met in the *Groundwater Command Area* (CA_{gw}). The crops grown in CA_{gw} have access to only groundwater irrigation. Comparison of %ET requirements met in the Groundwater Command Area (CA_{gw}) for simulations with and without tank storage during: a) the dry (March-May) and monsoon season (August-January), b) drought and non-periods, where drought periods correspond to months with 12-month standardized precipitation index (SPI-12) less than -1, and c) in each cropping stage during the two growing seasons. In this figure, the %ET requirement met distributions were generated by aggregating daily %ET met. 111

5.8 Understanding the influence of tank storage on monthly groundwater table elevation (m) under the condition of: (a) no tank storage; (b-e) with tank storage and groundwater irrigation; (f) when there is no groundwater irrigation in the tank water spread area. 112

5.9 Understanding how groundwater spread area size affects tank impacts. a) Monthly water table elevation (m) distribution in simulations where GSA size is varied from 1 to 10 times the size of surface command area. b) Change in the mean and standard deviation of monthly %ET requirements met relative to the no tank scenario as function of groundwater spread area. Note: a baseline groundwater spread area of 3 surface command area units was used to generate results in other simulations. 114

5.10 Understanding how well density affects benefits from tank. a) Median %ET requirements met during the monsoon and dry season as a function well numbers in the groundwater spread. b) Median %ET requirements met during the monsoon and dry season as a function well numbers in the groundwater spread during *droughts*. Drought periods correspond to months with 12-month standardized precipitation index (SPI-12) less than -1. Note: the baseline well number corresponds to the value used to generate results in other simulations. 116

5.11 Tank outflow dynamics with and without groundwater irrigation in the groundwater spread area of the tank system. We find that tanks transition from sluice to groundwater recharge dominated structures due to irrigation in the groundwater spread area. *Note:* This comparison has been made using our baseline tank characteristics with groundwater spread area equal to 3 command area units (Table C.5). 118

A.1 a) District-scale annual groundwater recharge volume estimated by the Central Groundwater Board (CGWB) of India (Baseline Scenario), b) The ratio of groundwater recharge reserved for natural discharge in current CGWB estimates. Represents the amount of groundwater recharge not available for human use. 173

A.2 Estimated district-level groundwater extraction rates (in km^3) 174

A.3 Allocated district-level budgets in the Mixed scenario relative to the budgets estimated based on the local ($Agw_{EFR,d}$) and global scenario ($Agw_{PB,d}$). The category 'In-between' represents groundwater availability thresholds that were greater than the ($Agw_{PB,d}$) and less than ($Agw_{EFR,d}$). 175

A.4 Density plots highlighting the distribution district-level groundwater stress for the 4 scenarios analyzed in this study. The blue dashed line represents the median groundwater stress value in each scenario. 177

B.1 Yearly frequency of groundwater depletion related articles in major English language newspapers in South India. Results were obtained using the following query: ((groundwater or ground water) same (deple* OR scarc* or dry or dried or decli*)) and (farm* OR agri*) AND (Karnataka or KA or Maharashtra or MH or Andhra Pradesh or AP or Telangana or TG or Tamil Nadu or TN or Kerala or KL). Results show increasing mentions of groundwater stress in major newspapers in South India. 179

B.2 a) Percent change in the area irrigated by shallow wells from 1996-2001 to 2006-2011. b) Percent change in the area irrigated by deep wells from 1996-2001 to 2006-2011. The change was estimated at the district scale using the median value of irrigated areas for each time frame. A positive change (green) indicates an increase in irrigated area, while a negative change (brown) indicates a decrease in irrigated area. Figures show that shallow well irrigated area has been decreasing in South India but deep well irrigated area has been increasing. Please note that regions with no data availability are represented by hatched lines. 180

B.3 District map of India used in the analysis. Please note that the district boundaries are from 1966. 181

B.4 Comparison between GLDAS-1/NOAH forcing data and the IITM precipitation at the India-scale. The correlation between the data sources is strong (r = 0.99) 182

B.5 a) Location of all the shallow monitoring wells in the Central Groundwater Board (CGWB) database, b) Location of all the deep monitoring wells in the CGWB database. Figures show good coverage across India for both shallow and deep monitoring wells, with the shallow well network being more extensive 183

B.6 Standardized Precipitation Index (SPI) time series for India between 1996 and 2016. a) 12-month SPI, b) 24-month SPI and c) 36-month SPI. The shaded area represents the period between 2002-2004 and corresponds to a severe drought in large parts of the country. 184

B.7 Percentage of wells with non-significant trends ($p - value < 0.1$) in a) August, b) November and c) January. Each circle represents a State and the size of the circle represents the percentage of wells with ($p - value < 0.1$) positive trends. Red circles represents States with greater negative trends than positive, while blue circles represents states with more positive trends. The results are consistent with the May trends discussed in the main text. Results show that a majority of States have a large percentage of wells (>50%) have non-significant long-term trends. 185

B.8 Monthly trends ($cm\, yr^{-}1$) in groundwater from (a) GRACE anomaly, and (b) monitoring wells for the pre-monsoon season (May) between 2005-2016. Dots represents areas where the trends are significant ($p - value < 0.1$). Please note that the grid cell-level trends using monitoring well data was calculated after taking the median water level of all the monitoring wells falling within the grid cell. Results show that large regions in SI have non-significant trends. (c) Map of correlation coefficient between GRACE anomaly and monitoring well time series between 2002-2016. Results show that the correlation is stronger in Southern and Eastern India, and lower in North-West and Central India. 186

B.9 a) Percentage (cumulative) of defunct wells in South and North India b) Percentage of dry wells in South and North India. Results highlight that both dry and defunct wells have been monotonically increasing in India, indicative of groundwater stress. Please note that there was no data recorded in August 2012 in the entire database, so that point was removed from the %dry and %defunct well calculations. 187

C.1 Results from a partial sensitivity analysis using the FAST algorithm (Reusser et al., 2011). Plots highlight the extent of partial variance accounted for by each parameter. The plot shows the parameters exerting the greatest influence on the ratio of sluice and groundwater outflow from the tank. Results were obtained by running the model over a 20-yr period for 403 parameter sets. 193

C.2 Mean monthly precipitation amounts (in mm) for the 54-yr forcing data used in the analysis 194

C.3 12-month Standardized Precipitation Index (SPI) time series for the 54-yr forcing data used in the analysis. SPI-12 values less than -1 represent drought periods 195

Chapter 1

Introduction

1.1 Research Context

1.1.1 Global groundwater crisis

Groundwater is a partially renewable source of freshwater on the planet, and it has been estimated that nearly 96% of non-frozen freshwater is stored in groundwater systems. Over the last few decades, this vast and decentralized store of freshwater has become the preferred source of freshwater for billions of people around the world. Currently, it is estimated that nearly 50% of drinking water, 40% of irrigation water, and over 20% of industry water needs are sourced from groundwater globally (Smith et al., 2016; Margat and Van der Gun, 2013; Zektser and Everett, 2004). This increased dependence on groundwater extraction has played an important role in catalyzing progress toward multiple developmental goals. In particular, access to groundwater has been instrumental in helping meet the food demands of a growing population. It is estimated that nearly 70% of the groundwater extracted globally is being utilized for agricultural purposes (Van der Gun, 2012). This dependence is even higher in arid and semi-arid regions of the world where rainfall patterns are generally unreliable and surface water sources are often ephemeral. In these regions, groundwater can often serve as the only available perennial source of freshwater. In addition to supporting multiple human activities, groundwater is required for the functioning of numerous groundwater-fed ecosystems like lakes, springs, rivers, and wetlands. The importance of groundwater to the 'health' of socio-environmental systems is reflected in Sustainable Development Goals (SDG), where nearly 31% of developmental targets have linkages to groundwater (Guppy et al., 2018).

While increased accessibility to groundwater resources has transformed livelihoods and economies, efforts to manage this resource have been largely ineffectual globally. Numerous recent studies have questioned the long-term sustainability of current groundwater usage (Wada et al., 2010; Aeschbach-Hertig and Gleeson, 2012), with extraction rates far exceeding long-term groundwater recharge in many arid and semi-arid regions of the world (e.g. India, United States, China, Mexico). It is estimated that 1.7 billion people live in places where groundwater resources are considered to be stressed (Gleeson et al., 2012b). Recent satellite-based observations have shown that close to one-third of the 37 major aquifers around the globe, many underlying major agricultural belts, have unsustainable extraction rates (Richey et al., 2015). Non-renewable groundwater abstraction rates have increased 3 times over the last few decades alone with yearly abstraction increasing from 75 km^3yr^{-1} in 1960 to over 234 km^3yr^{-1} in 2000 (Wada et al., 2012). Nearly 11% of non-renewable groundwater abstraction is estimated to be embedded in the global food trade network, and a majority of countries are currently importing staple foods from regions with unsustainable groundwater extraction (Dalin et al., 2017).

The unsustainable extraction of groundwater has been shown to permanently lower groundwater tables which can ultimately result in decreased baseflow into surface-water bodies, cause land subsidence and increase salinization of groundwater resources through salt-water intrusion (Bierkens and Wada, 2019). Recent studies have highlighted that the increase in groundwater pumping has lead to sea-level rise at rate of 0.1-0.4 mm/yr between 1993-2010 (Wada et al., 2016; Intergovernmental Panel on Climate Change, 2014). In addition to environmental impacts, the negative effects of groundwater depletion are increasingly being perceived in socio-economic systems as well. Some documented socio-economic consequences of groundwater depletion include dry wells, loss of income and livelihood for farmers, increased pumping costs, a decrease in resilience to droughts, and increased inequalities within communities.

However, human dependence on groundwater is expected to continue to grow over the next few decades. Recent estimates suggest that food production will have to increase by approximately 25%-75% by 2050 to meet the demands of a growing population (Hunter et al., 2017). Improving access of rainfed farms to groundwater resources can play an important role in providing supplemental irrigation water necessary to reduce yield gaps between theoretical and actual crop yields (Rosa et al., 2018). Furthermore, groundwater resources can serve as a critical buffer that allows agricultural production to be more resilient to the expected increased frequency of extreme climate events and unreliable climate patterns (Taylor et al., 2013). Thus, with global groundwater resources already in a state of crisis (Famiglietti, 2014) and given groundwater's strategic importance to current and future generations, there is a fundamental need to improve our understanding of managing this vast "hidden" resource.

1.1.2 Groundwater depletion in India

Rise of Well Irrigation

The need for reforming the management of groundwater resources can be considered to be particularly pronounced in India, a country that has seen the most spectacular growth in groundwater extraction rates over the last half-century (Figure 1.1). Triggered by what has been termed the "silent revolution" (Llamas and Martínez-Santos, 2005), groundwater abstraction has increased exponentially since the 1960s as millions of farmers have acquired the ability to extract groundwater resources through the use of mechanized pumps in India. The number of wells tapping shallow and deeper aquifers has increased from 150,000 in 1960 to over 19 million in 2000 with groundwater extraction rates increasing from $\sim$25 km^3yr^{-1}

to >200 km^3yr^{-1} (Shah, 2009, 2005). By gaining access to groundwater, farmers that were traditionally vulnerable to uncertain monsoonal rainfall patterns prevalent in the country have now gained access to a reliable and self-manageable source of water. Groundwater irrigation has allowed many small-holder farmers to increase yields, grow multiple crops during the course of the year, and even risk growing more water-intensive but economically lucrative crops like rice and cotton (Sato and Duraiyappan, 2011; Jain et al., 2021). While initially centered around the Indo-Gangetic plains, groundwater irrigation has spread across the entire country and has gradually even replaced traditional and modern surface irrigation sources. The scale of the spread of groundwater irrigation in India combined with the spread of high-yielding variety (HYV) seeds and fertilizers (the latter two being promoted as part of the 'Green Revolution') have been instrumental in increasing food production in India. This has helped transform the country, from a region that was vulnerable to uncertainty of rainfall patterns and dependent on foreign aid, to a region that is largely self-sufficient (Parayil, 1992; Mukherji and Shah, 2005). Nearly 85% of groundwater extracted is used for irrigation purposes in India (Mukherjee et al., 2015), and groundwater has been estimated to currently support over 50% of drinking water and over 70% of agricultural production in India (Fishman et al., 2011; Mukherji and Shah, 2005).

Groundwater Depletion: Impacts

Exemplifying the global groundwater use narrative, while the short-term benefits associated with accessing groundwater have transformed livelihoods in India, the long-term consequences of unregulated groundwater extraction are being increasingly perceived in the country. India currently has the highest yearly groundwater extraction rates and some of the greatest groundwater depletion rates in the world (Rodell et al., 2009; Aeschbach-Hertig and Gleeson, 2012; Tiwari et al., 2009; Chinnasamy and Agoramoorthy, 2015). This uncontrolled groundwater extraction has resulted in a complex series of feedbacks with numerous negative environmental and socio-economic consequences. There is an increasing number of cases where intensive groundwater pumping has been shown to contribute to

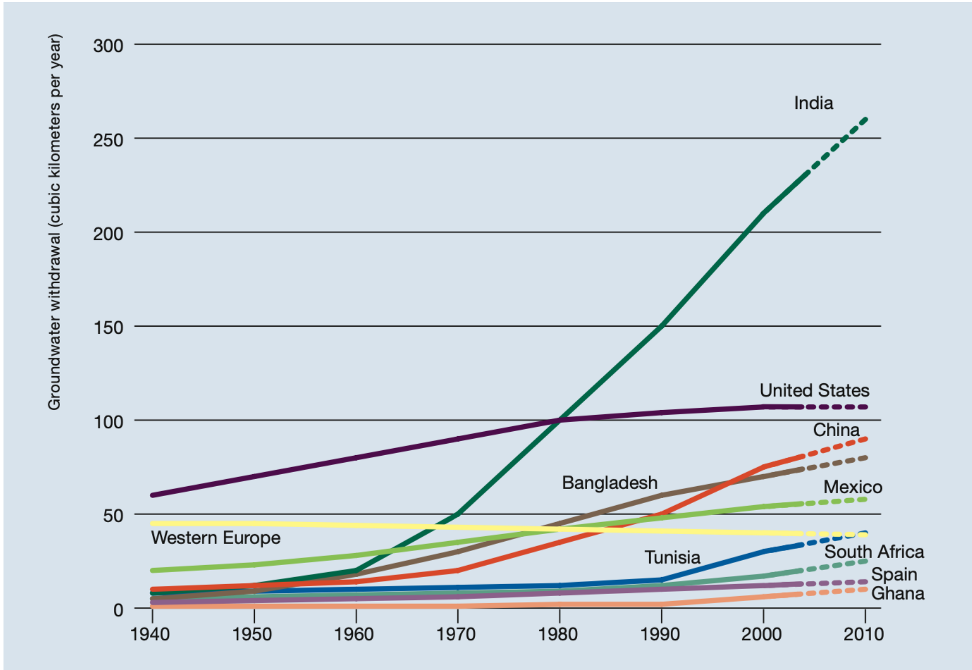

Figure 1.1: Temporal trend of country-level groundwater extraction in km^{3}year^{-1} (Modified from Shah et al., 2007)
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declining flow rates in rivers (including major rivers like the Ganges and Krishna) (Srinivasan et al., 2015; Biggs et al., 2007; Mukherjee et al., 2018). Groundwater depletion has resulted in increasing rates of land subsidence in the state of Gujarat (Choudhury et al., 2018), while deepening water tables elevation are causing water quality to deteriorate with high concentrations of arsenic, fluoride and uranium (Singh and Singh, 2002; Coyte et al., 2018). From a social perspective, groundwater depletion has disproportionately impacted the livelihood of smallholder farmers (< 2 ha of land), a farmer group that makes up nearly 80% of all farmers in the country (Mukherji and Shah, 2005; Shiferaw et al., 2008). These farmers are often unable to cope with the culture of competitive well deepening that has been observed in systems with declining groundwater tables (Janakrajan and Moench, 2006; Pahuja et al., 2010; Sarkar, 2012). While small-holder farmers work on only 32%

of agricultural land in the country, nearly 73% of the wells abandoned due to a lowering water table belong to them (Mukherji and Shah, 2005). As a result, a lack of reliable sources of irrigation water is forcing farmers in large parts of India to leave lands fallow for extended periods, grow less economically viable crops, take on crippling loans, and migrate to urban centers (Sato and Duraiyappan, 2011; Narayanamoorthy, 2014; Fishman et al., 2013; Birkenholtz, 2009). Moreover, groundwater usage has also contributed to the physical and institutional degradation of traditional irrigation structures like tanks (Van Meter et al., 2014). Recent estimates suggest that rural poverty in regions with groundwater table depths greater than 8m has been estimated to be 10% higher (Sekhri, 2014). Given current groundwater depletion rates, it has been estimated that the national cropping intensity will reduce by 20% in India by 2050 (Jain et al., 2021).

Groundwater Depletion: Causes

Numerous reasons have contributed to the current state of groundwater resources in India. At a fundamental level, there is a growing mismatch between the agricultural water demands and supply that has been exacerbated by overpopulation and changing diets (de Fraiture and Wichelns, 2010). Recent estimates suggest that annual water demands in South Asia, driven primarily by irrigation, have increased from 200 km^3 to nearly 600 km^3 between 1960 and 2001 (Wada et al., 2011). With these growing water demands and limited scope to expand agricultural lands, the depletion of groundwater resources in India has often been considered to be justifiable in academic and political circles given the socio-economic benefits that have been accrued from increased groundwater use (Mukherji, 2006a; Molle and Closas, 2020). In addition to rapidly growing demands, part of the crisis can be attributed to the inherent difficulty in managing groundwater given its 'invisible, slow and decentralized' nature (Villholth et al., 2018). This partially explains the current extent of groundwater depletion globally irrespective of the wealth of a region or the sector that dominates groundwater use. Groundwater systems often operate at timescales of 50-100 years, and thus require longer-term planning and management strategies which are often challenging to formulate and operationalize (Gleeson et al., 2012a). Similarly, given the decentralized nature of the resource, monitoring and regulating groundwater use of millions of users has been ineffectual in India (and other parts of the world) often due to the sheer scale of logistical coordination required (Molle and Closas, 2020).

In India, these challenges have been amplified by poor groundwater governance frameworks, slow acknowledgement of hydrogeological science to inform management, data scarcity, and by interventions that have been ineffectual or had unintended negative consequences. The legal framework in India currently treats groundwater as a private resource linked to property, and as a result, landowners have excessive control over the usage of groundwater (Cullet, 2014). Policies introduced by State governments have often worsened groundwater depletion, but have been difficult to revoke or modify given the potential for an electoral disaster (Phansalkar and Kher, 2006; Molle and Closas, 2020). An example related to this is power subsidies provided to farmers in India at no-cost or flat rates that have been shown to encourage overpumping and introduce a culture of competitive well deepening (Sarkar, 2012; Mukherji, 2006c). Additionally, despite trends suggesting the continued dominance of groundwater irrigation in India, the government in India has been shown to have a preference for investing large sums of money into surface irrigation structures despite these investments yielding poor returns (Amarasinghe et al., 2008; Shah, 2011). Further, the Central Groundwater Board, India's primary groundwater regulatory body, has been shown to inadequately acknowledge the hydrological continuum between surface and groundwater resources, ultimately resulting in an over-allocation of groundwater available for human use and a failure to identify groundwater stressed regions (Srinivasan and Lele, 2017).

1.2 Research Approach

The research presented in this thesis focuses on improving our knowledge about three broad aspects of groundwater management in systems experiencing long-term groundwater depletion: (1) regional *assessments* of groundwater stress, (2)*monitoring* of groundwater depletion, and (3) *interventions* aimed at improving groundwater storage and agricultural water availability. Each of these aspects are explored within the context of Indian agricultural and groundwater systems. In the next sections, a summary of the specific knowledge gaps addressed by this thesis are first highlighted, and this is followed by a section outlining the research objectives.

1.2.1 Specific Knowledge Gaps

Indicator-driven groundwater assessments are important science-based tools that are used to inform policies and management plans (Gleeson and Wada, 2013). Assessing how groundwater extraction rates compare with the groundwater supply at a given location can be considered to be one of the most fundamental (and widely used) measures of regional groundwater stress. Globally, the groundwater supply has most commonly been assumed equal to the yearly/monthly groundwater recharge of a given region. However, this assumption has been shown to inadequately account for the impacts of groundwater pumping on environmental systems (Bredehoeft, 2002; Zhou, 2009), and as a result, regions have experienced negative environmental effects of groundwater extraction despite pumping rates being less than the long-term groundwater recharge (Sophocleous, 2000). Recent research has advocated developing regional groundwater extraction thresholds that better account for environmental needs by taking into consideration local groundwater-discharge rates into surface-water bodies (Gleeson and Richter, 2018) and global environmental limits (Zipper et al., 2020). However, the application of these principles into groundwater stress assessments has been lacking in the Indian as well as global groundwater context.

Water level data from monitoring wells represents the most direct measure of groundwater storage at a given location. Monitoring well data serves as a fundamental dataset to identify regions experiencing long-term groundwater depletion. In India, recent monitoring well data collected by the government has been used to assess trends in groundwater storage at different spatial scales. Results from these analyses suggest that groundwater levels are dropping in the deep alluvial aquifers of Northern India, and rising in the shallow hard-rock aquifers of Southern India between 1996-2016 (Asoka et al., 2017; Bhanja et al., 2017). These findings have been further supported by studies utilizing data collected by the GRACE satellite system (Panda and Wahr, 2016). While there is general consensus regarding groundwater depletion in North India, the results from these recent studies seem to be at odds with on-the-ground reports of groundwater depletion in Southern India (Sato, 2013; Srinivasan et al., 2015). Furthermore, recent studies have highlighted the potential of a publication bias where studies have a tendency of reporting positive groundwater storage outcomes in India (Chindarkar and Grafton, 2019), while others have highlighted the need to re-define how groundwater depletion and sustainability are measured in hard-rock aquifer systems (Fishman et al., 2011). Thus, there exists an opportunity to resolve potential discrepancies regarding groundwater storage trends in Southern India, and critically evaluate how monitoring data is used to draw conclusions with regard to groundwater storage in hard-rock aquifer systems.

Rain-water harvesting is generally considered to be a promising intervention to reduce climate vulnerability and improve groundwater storage in semi-arid regions globally. In India, rain-water harvesting has a long history where village-level structures have been used to store monsoonal runoff for agricultural and domestic purposes for millennia now (Van Meter et al., 2014). However, a large number of these structures have gone into disrepair over the last few decades as groundwater has gradually become the preferred source of irrigation water across India. With groundwater levels falling, the government in India and international development agencies (like the World Bank) are investing large funds into rehabilitating existing and/or constructing new rain-water harvesting structures across the country. However, there is considerable uncertainty surrounding the impact of these structures in contemporary agricultural systems with studies either viewing rain-water harvesting structures as 'panaceas' (Palanisami et al., 2010; Reddy and Behera, 2009) or as 'mirages' (Kumar et al., 2008; Batchelor et al., 2002). Hydrological modelling can serve as an important tool to understand the effects of these structures in data-scarce systems. However, previous modelling approaches have inadequately conceptualized important hydrological processes in agricultural rain-water harvesting systems, especially with regards to the surface water-groundwater interactions (Glendenning and Vervoort, 2011; Jayatilaka et al., 2003). Thus, there is a need to improve our modelling approaches to help answer critical questions regarding the role rainwater harvesting systems can play in improving water availability in heavily groundwater irrigated systems.

1.2.2 Objectives

The primary goal of this dissertation was to improve the state of knowledge related to groundwater assessments, monitoring methods and interventions such that the knowledge generated can be used to support current efforts to address groundwater over-exploitation in agriculture-dominated, data-scarce systems like India. With an overarching view to improve the sustainability of groundwater usage, an attempt was made to answer the following research questions: (1) What are the implications of developing regional groundwater extraction thresholds that take into account local and global environmental considerations?, (2) How can the congruence between 'hard' groundwater monitoring data sources and 'soft' data sources (like census data and newspaper articles) be improved to better identify groundwater storage trends?, (3) What influence can agricultural rain-water harvesting structures have on water availability in groundwater-dependent agricultural systems? These questions were approached with the following set of objectives:

1. Estimate groundwater extraction thresholds that: (a) takes local environmental flow requirements into account and (b) are consistent with the freshwater planetary boundary, and assess how stressed groundwater resources are in India with respect to these thresholds.

2. Demonstrate challenges in identifying groundwater depletion hotspots using monitoring well data, and develop metrics to reliably capture groundwater storage trends

in hard-rock aquifer systems.

3. Develop a conceptual hydrological modelling framework to improve our understanding of how agricultural rain-water harvesting structures (RWH) function in contemporary groundwater-dependent agricultural systems, and assess how the dynamics between RWH structures and groundwater storage impact water availability for farmers.

1.3 Dissertation Structure

This manuscript-based dissertation has been structured into 6 chapters. The first chapter provides the context and motivation for this research and outlines the objectives of this study. Chapter 2 provides the relevant background information, and Chapters 3-5 form the body of this thesis.

In Chapter 3, district-scale groundwater use limits are estimated in India by taking local and global environmental considerations into account. The implications of setting targets based on these considerations are then assessed across the country. This chapter has been submitted for peer review to the journal Ecological Indicators:

Hora, T., Basu, N. B., Wandel, J., & Kelly, R., Examining rainwater harvesting structures in groundwater intensive irrigation systems

Chapter 4 addresses the second objective of the thesis where discrepancies between groundwater storage trends in Southern India using monitoring well data and 'soft' non-hydrological data (census surveys, field reports, newspaper searches) are addressed. The potential of a survivor bias in the long-term monitoring of groundwater systems is uncovered, and alternative metrics to capture groundwater depletion in hard-rock systems are developed. Chapter 4 has been published in the peer-reviewed journal *Geophysical Research Letters*:

Hora, T., Srinivasan, V., and Basu, N. B. (2019). The Groundwater Recovery Paradox in South India.Geophysical Research Letters, 46(16):9602–9611.

In Chapter 5, a conceptual hydrological model has been developed to capture the dynamics of rain-water harvesting systems (called tanks) in Southern India. The model is validated using data from a single tank system in Tamil Nadu, India. The model is then used to evaluate the hydrological impacts of rain-water harvesting structures in groundwater-intensive irrigation systems. This chapter will be submitted for peer review to the journal Agricultural Water Management:

Hora, T., Basu, N. B. & Kelly, R., Examining rainwater harvesting structures in groundwater intensive irrigation systems (in preparation for Agricultural Water Management)

In the final chapter, the major findings of this research are highlighted and recommendations are made for future work.

Chapter 2

Background

2.1 Groundwater and the Water Cycle

The global water cycle is the continuous circulation of water between land, atmosphere and the oceans (Figure 2.1). Driven by solar radiation, the water stored in the ocean (and other surface stores like lakes) evaporates and rises to reach the atmosphere. Water also evaporates from plants into the atmosphere through transpiration. This water vapour then circulates, cools, and condenses based on the temperature gradients present in the atmosphere, and falls back onto land and into the oceans in the form of precipitation. Precipitation falling on land then either gets sent back into the atmosphere from plant canopies or makes its way to the land surface. The precipitation reaching the surface then either gets stored temporarily as snow/ice, or, if temperature conditions permit, makes its way into the subsurface or drains as surface flow back into the oceans through rivers and lakes. A portion of the water entering the subsurface is then either returned from the unsaturated zone in the soil as evapotranspiration or makes its way into surface water bodies, while the rest of the infiltrated precipitation reaches the water table as recharge (or more specifically as *diffuse* recharge). Groundwater recharge also takes through the beds of lakes and rivers (or more specifically as *focused* recharge). Under natural conditions, groundwater ultimately discharges into surface water stores (lakes or rivers) or directly into the ocean in coastal areas.

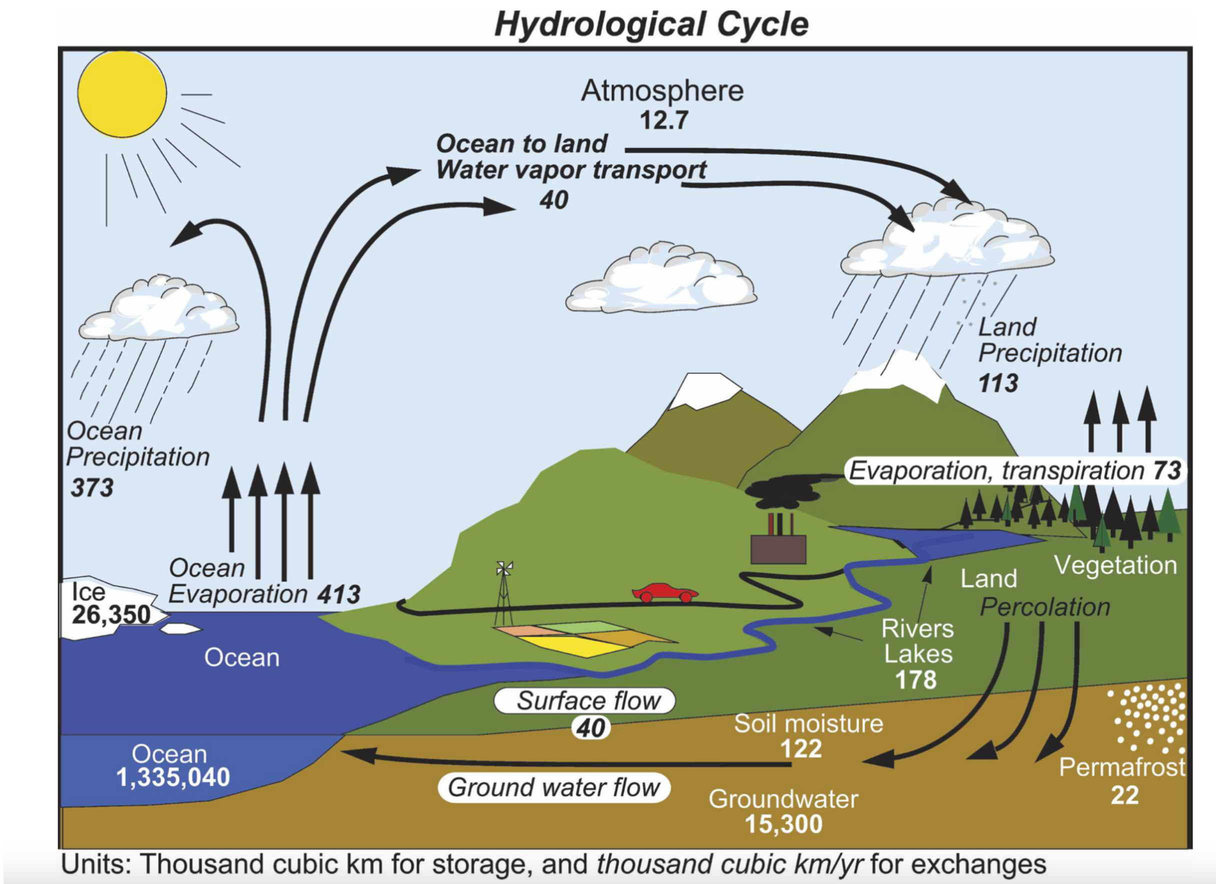

Figure 2.1: Components of the water cycle with volumetric estimates of water storage in each global water store (adapted from Trenberth et al., 2007)

Groundwater plays a key role in the continental part of the water cycle accounting for nearly 30% of freshwater (while the remaining two-thirds is stored in ice caps/glaciers), and almost 99% of all non-frozen freshwater stored on the planet (Dingman, 2015). An important characteristic of groundwater is that it typically moves at a much slower rate compared to most other components of the hydrological cycle (Table 2.1). The global residence of groundwater (i.e. a measure of time spent in storage) has been estimated at 235

Table 2.1: Average residence of different global water stores (adapted from Nelson et al., 2009)

Reservoir	Residence Time (mean)
Oceans	~3000 years
Glaciers	20-100 years
Seasonal Snow Cover	2-6 months
Soil Moisture	1-2 months
Groundwater: Shallow	100-200 years
Groundwater: Deep	10,000 years
Lakes	50-100 years
Atmosphere	9 days

years (Dingman, 2015), however, some deeper groundwater reservoirs can have residence times >1000 years. Groundwater systems where the residence time far exceeds a human lifespan are generally referred to as fossil groundwater, and these stores of groundwater are considered to be a non-renewable resource (Nicolas, 2019). Despite lower velocities, groundwater still serves as a large source of water for surface water systems (like lakes and rivers), especially in humid/tropical regions (Nicolas et al., 2019; Dingman, 2015).

2.2 Basics of Groundwater Flow

Like all components of the water cycle, the water stored as *groundwater* is in constant motion. Geological formations in the sub-surface that can store and transmit significant volumes of water are termed *aquifers*, while sub-surface formations that poorly transmit water are called *aquitards*. Aquifers are generally further classified into two additional classes: unconfined and confined aquifers (Figure 2.2). Unconfined aquifers are bounded by the water table at the top, and are predominantly recharged by precipitation that infiltrated through the unsaturated zone. In contrast, confined aquifers are bounded between two confining formations (aquitards or aquicludes) and are recharged in areas where the aquifer outcrops on the surface (Ge and Gorelick, 2015). The ability of a porous medium to store water is generally measured by its porosity, which is the proportion of void space in a given volume of a material. Alternatively, the ability of a material to transmit fluid is measured by its permeability $[L^2]$, which is a property that reflects the inter-connectedness of its pore spaces. Combining the permeability of a geological material with the properties (like density, viscosity) of the fluid it is transmitting provides a measure of the hydraulic conductivity of a porous medium $[LT^{-1}]$. The hydraulic conductivity of a given aquifer is one of the primary drivers controlling the flow of groundwater. Hydraulic conductivity of geological material varies over many orders of magnitude ranging from 10^{-10} cm/sec (for material like clay) to over 10^2 cm/sec (for karst systems and gravel).

The second key driver controlling the flow of water through an aquifer is the hydraulic head. The hydraulic head (h) is a measure of the total mechanical energy of fluid at any point in the aquifer such that $h = \Psi + z$, where Ψ is the pressure head (i.e. the internal static pressure, Ψ) and z is the elevation head (i.e. gravitational energy). Estimates of hydraulic head at a given location are generally obtained by measuring the depth-to-water using piezometers or wells.

The flow of water through an aquifer is commonly estimated using an empirical relationship known as Darcy's Law. In one-dimension, Darcy's law can be given as:

$$Q_x = -K_x \frac{dh}{dx} A_x \tag{2.1}$$

where, Q_x is the volumetric flow rate $[L^3T^{-1}]$, K_x is the hydraulic conductivity of the porous medium (in the x-direction), A_x is the cross-sectional area of the medium and $\frac{dh}{dx}$ is the hydraulic gradient which is a measure of the change in hydraulic head per unit distance. The negative sign in the equation highlights that the flow of groundwater takes place from high to low hydraulic head.

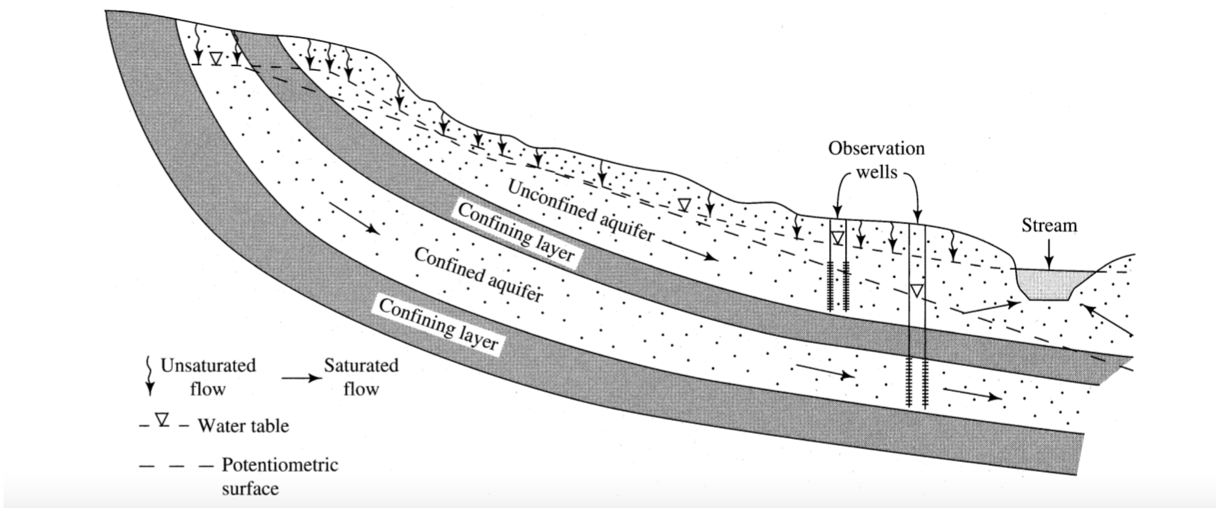

Figure 2.2: Cross-section showing a groundwater flow system comprising of an unconfined and confined aquifer system (adapted from Dingman, 2015)

2.3 Impact of Groundwater Pumping

In a system without anthropogenic influence, the groundwater storage can be considered to be in long-term equilibrium such that water input (recharge) into the groundwater system is equal to the output (e.g. discharge into lakes) from the groundwater system (Alley et al., 1999). Therefore, under these conditions:

$$Natural\ Recharge\ (R_o) = Natural\ Discharge\ (D_o) \tag{2.2}$$

When groundwater is pumped out using wells for human consumption, the water table in the pumping well declines and water starts flowing into the well from the surrounding area. The groundwater system responds to this change in storage by either increasing the recharge into the aquifer or by decreasing the discharge from the aquifer. To ensure the conservation of mass, the response of the groundwater system to pumping can be given by:

$$(R_o + R_i) - (D_o + D_i) - P = dV/dt \tag{2.3}$$

where, R_i is the induced recharge caused by pumping, D_i is the reduction in discharge from the groundwater system, P is the volume pumped out of the system, and dV/dt is the potential change in groundwater storage.

In arid and semi-arid systems, groundwater is often extracted from deep aquifer systems where the replenishment periods far exceed an average human lifespan. These systems are considered to be non-renewable sources of groundwater (Bierkens and Wada, 2019), where the impact of pumping is predominantly viewed in terms of relative permanent declines in groundwater storage (termed groundwater mining) (Bierkens and Wada, 2019).

2.4 Extraction Rates and Groundwater Sustainability

A core question around groundwater and its relation to society relates to how much groundwater can be pumped to meet human water demands. Traditionally, the management of groundwater with regards to establishing pumping rates aimed to limit regional pumping volumes (P in equation 2.3) to be less than the natural recharge rate (R_o in equation 2.3). The natural recharge rate was considered to be the renewable supply that a region could safely extract (*safe yield*) without leading to a net loss in long-term groundwater storage. Terming this as the *water budget myth*, numerous studies have challenged this assumption as negative consequences associated with groundwater pumping have been observed despite policies that aimed to limit groundwater pumping to be less than the natural recharge rate (Zhou, 2009; Bredehoeft, 2002). These consequences have brought to light the fact that groundwater pumping results in the groundwater system balancing the water lost by either increasing recharge (R_i in equation 2.3) or decreasing the discharge (D_i in equation 2.3) from the system. These changes often modify the water balance of surface water bodies (like lakes or rivers) and as a result, negatively impact groundwater-dependent ecosystems (Kløve et al., 2011). Figure 2.3 highlights the effects of intensive groundwater pumping on the water balance of a stream, where the stream transitions from a discharging to a recharging stream. Furthermore, other impacts of groundwater pumping like land subsidence, salt-water intrusion, and degradation of water quality are also becoming important to take into consideration when developing thresholds on regional groundwater pumping (Alley et al., 1999).

Therefore, the question about how much groundwater can be pumped has evolved from looking at the renewable groundwater recharge rate to: *(a)* understanding how pumping impacts groundwater recharge/discharge in the system, and *(b)* setting thresholds on what changes associated with groundwater pumping are acceptable based on region-specific social, economic and environmental considerations. For systems where the pumping is occurring predominantly from non-renewable sources of groundwater, it is useful to view extraction rates in terms of total groundwater reserve, and how that groundwater can serve to meet anthropogenic needs over a prolonged period (Bierkens and Wada, 2019). Additionally, the sustainable extraction of groundwater resources is increasingly being considered in relation to the governance arrangements required to achieve desired long-term outcomes related to groundwater systems (Elshall et al., 2020), while also viewing groundwater sustainability at a national/global scale (Gleeson, 2020).

2.5 Groundwater Stress Assessments

Assessing the long-term sustainability of water usage often relies on the use of indicators to measure the state of human-influenced water systems. Good indicators serve the role of simplifying complex hydrological phenomena effectively in a form that is easy to communicate to decision-makers, while allowing for spatial and temporal comparisons (Vrba et al., 2007). Indicators used in water resources range from simple ratios (e.g. the Falkenmark Indicator) to multi-criteria composite measures (e.g. the Water Poverty Index). The

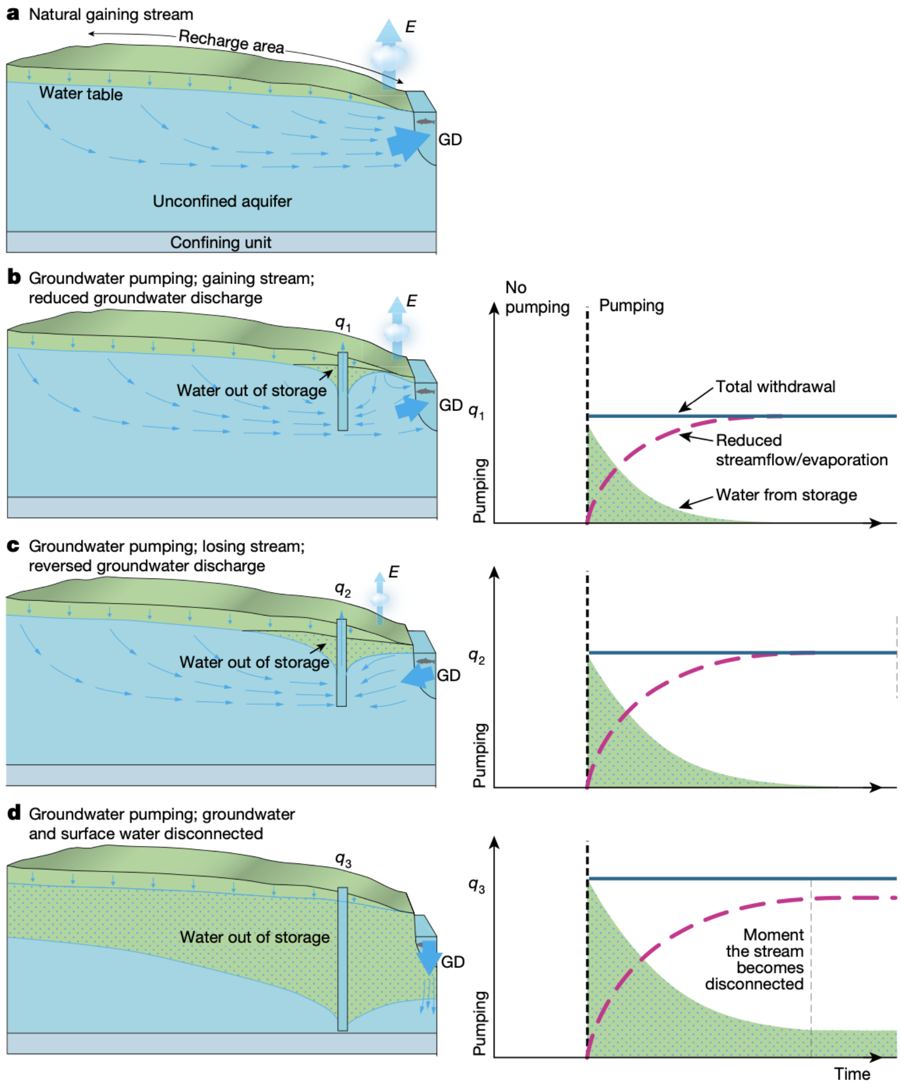

Figure 2.3: Impact of groundwater pumping on the water balance of a stream. The figure highlights the change from a discharging to recharging stream due to groundwater storage declines in the connected aquifer. (adapted from de Graaf et al., 2019)

importance of developing indicators to assess groundwater systems recently resulted in the development of a list of 10 indicators to capture different aspects of groundwater processes by a multi-institutional working group comprising of UNESCO, IAH (International Association of Hydrogeologists) and the IAEA (International Atomic Energy Agency)(Table 2.2).

One of the core indicators associated with groundwater systems aims to capture demand-driven *water scarcity* by measuring the groundwater stress at a given location (Kummu et al., 2016). Groundwater stress can be considered to be similar to the Water Stress index that is currently part of the Sustainable Development Goals (SDG; Indicator 6.4.2). In its most fundamental form, groundwater stress can be estimated by:

$$Groundwater\ Stress = \frac{Groundwater\ use}{Groundwater\ availability} \tag{2.4}$$

where, groundwater use is commonly estimated by extraction rates at the regional- or national-scale, and groundwater availability is assumed to be the long-term average groundwater recharge rate (Indicator 2 in Table 2.2). Thus, groundwater stress provides a relatively intuitive measure of how groundwater extraction rates at a given location compare with its renewable groundwater supply, and helps identify regions with unsustainable extraction relative to supply. Despite limitations in capturing complex groundwater dynamics (Alley et al., 2018), groundwater stress assessments continue to form the basis of national-level policies in various regions around the world often due to their ease of computation and relatively low data requirements. In India, the Central Groundwater Board (CGWB) has been releasing periodical reports (every 2-4 years), where districts are classified based on groundwater stress values (referred to as Stage of Groundwater Development in the reports; Figure 2.4).

Expanding Groundwater Stress Assessments

The base form of the groundwater stress index can be modified to better take into account regional socio-environmental factors (Indicator 3 in Table 2.2). One of the most important modifications to groundwater stress involves scaling the groundwater recharge

Table 2.2: List of groundwater resources sustainability indicators developed by UNESCO/IAEA/IAH (Vrba et al., 2007)

Indicator	Description
1) Renewable groundwater resources per capita	Total annual amount of renewable groundwater resources (m^3y^{-1}) per capita at the national or regional level
2) Total groundwater abstraction/ Groundwater recharge	Ratio of groundwater abstraction to total groundwater recharge
3) Total groundwater abstraction/ Exploitable groundwater resources	Total groundwater abstraction as a fraction of total water availability based on socio-economic constraints
4) Groundwater as a percentage of total use of drinking water	The present state and trends of groundwater use for drinking purposes at a national level.
5) Groundwater depletion indicator	Fraction of areas with groundwater depletion problems to the total studied area
6) Total exploitable non-renewable groundwater resources/Annual abstraction of non-renewable groundwater resources	Ratio of total exploitable non-renewable groundwater and total non-renewable abstraction
7) Groundwater vulnerability	Fraction of aquifer area that is considered to be vulnerable
8) Groundwater quality indicator	Fraction of area with groundwater quality problems (natural or anthropogenic) to the total studied area
9) Groundwater usability with respect to treatment requirements	Usability of abstracted groundwater that is publicly distributed with respect to treatment requirements.
10) Dependence of agricultural population on groundwater index	Ratio of the total population using groundwater to enhance the productivity of agriculture or livestock enterprise.

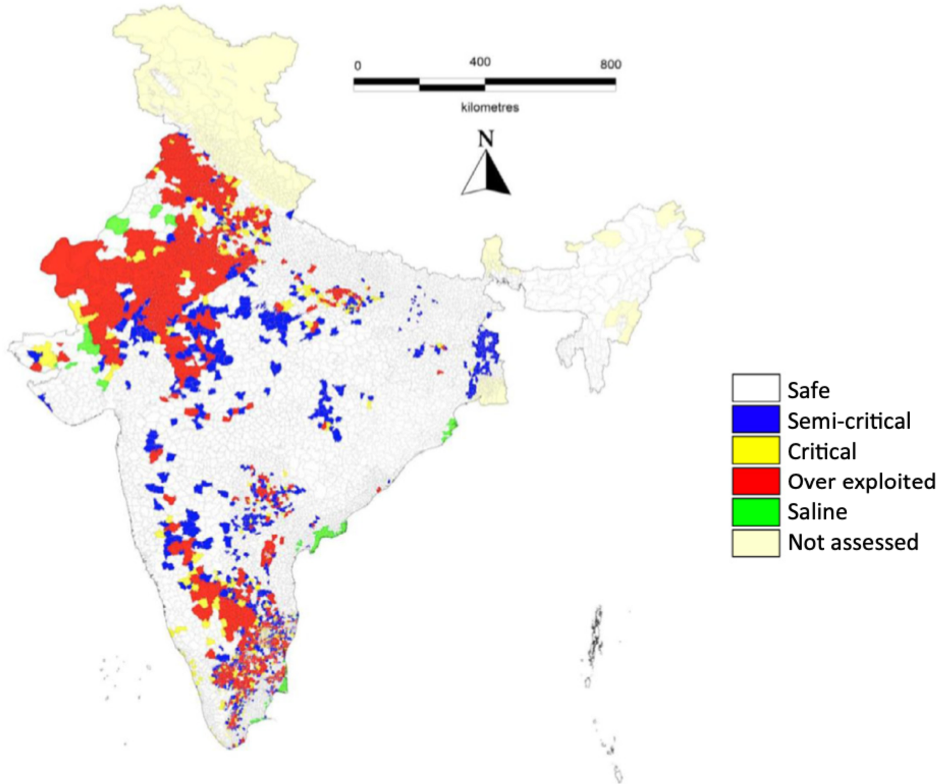

Figure 2.4: Using groundwater stress values to categorize districts in India. (Figure from Sidhu et al., 2020 and CGWB, 2014)

deemed available for extraction (the denominator in equation 2.4) to better account for the water requirements of groundwater-dependent ecosystems. This allows groundwater stress assessments to better represent our understanding of the effects of groundwater pumping on the environment. However, most national/regional assessment frameworks either ignore or under-represent environmental factors (Gleeson and Richter, 2018; Srinivasan and Lele, 2017). Part of the reason is that estimating environmental water requirements at a given location remains challenging even in well-monitored systems. To overcome these constraints, recent research has relied on using - (1) presumptive standards (Gleeson and

Richter, 2018), where a fixed percentage of regional recharge is assumed to be allocated for environmental needs, or (2) the output from global-scale hydrological models (Gleeson et al., 2012b). Other modifications to groundwater stress assessments involve estimating groundwater stress at monthly and/or daily time-scales (groundwater stress is most commonly measured at the annual-scale) to help differentiate between the magnitude of 'acute' and 'chronic' groundwater stress being experienced at a given location (Devineni et al., 2013; Fishman et al., 2015).

A growing perspective with regards to groundwater sustainability calls for the need to view groundwater problems from a global and/or national lens (Gleeson, 2020). The rationale for this viewpoint is the preponderance of groundwater issues globally, a need for knowledge sharing, and groundwater's strategic importance to current and future sustainability goals (Gleeson, 2020). This represents a fundamental shift in hydrological management that has traditionally been viewed from the perspective of local/regional socio-hydrological factors. In particular, the recently proposed Planetary Boundary framework has provided a basis for integrating freshwater management with other important Earth-system processes at the global scale. The Planetary Boundary framework sets global thresholds for 9 interlinked Earth-System processes under the assumption that crossing these thresholds can have disastrous consequences for socio-environmental systems (Rockström et al., 2009). With regards to fresh water, the current planetary threshold has been set to limit freshwater consumption to 4000 km^3y^{-1} globally. As the concept of planetary boundary evolves (Steffen et al., 2015), especially concerning freshwater systems (Heistermann, 2017; Gleeson et al., 2020), there is a need to better understand the implications of managing groundwater with a perspective that extends beyond just the regional hydrology of a system.

2.6 Monitoring Groundwater Depletion

With groundwater depletion occurring in many regions around the globe, the capacity to monitor groundwater resources is a fundamental component needed to improve the management of groundwater resources. Groundwater monitoring can be defined as *"as the scientifically-designed, continuing measurement and observation of groundwater"* resources (Jousma et al., 2006). Typical monitoring programs collect measurements related to groundwater quantity (e.g groundwater levels) and groundwater quality (e.g chemical composition), which then form the basis for a multitude of analyses that includes storage trends, model calibration/validation, and delineation of contamination. Groundwater storage at a given location is typically monitored using *in-situ* depth-to-water measurements from monitoring wells, however, more recently, the use of data from remote sensing systems like the Gravity Recovery and Climate Experiment (GRACE) is increasingly being used to assess changes in regional groundwater storage.

Monitoring Well Data

Representing the most direct measure of groundwater storage at a given location, water level data from monitoring wells is the principal source of data used to formulate the behavior of groundwater systems to anthropogenic influence (Bierkens and Wada, 2019; Ha et al., 2015). These measurements are typically taken from piezometers using instruments ranging from manually operated tape measures to automatic data logging pressure transducers (Jousma and Roelofsen, 2004) (Figure 2.5a). Groundwater level measurements in phreatic aquifers represent the depth of the water table, and depth-to-water measurements in confined aquifers represent the hydraulic head at that location. Individual point measurements of groundwater levels from multiple sites are commonly combined to help understand the regional distribution of hydraulic head, while repeated measures of the groundwater level over the course of a year/multiple-years helps provide a sense of the temporal evolution of groundwater storage. Figure 2.5b shows a theoretical depth-to-water time-series of a groundwater system that experiences intra-annual changes in storage due to precipitation patterns (short-term changes), but that is also seeing a decline in groundwater storage over the long-term.

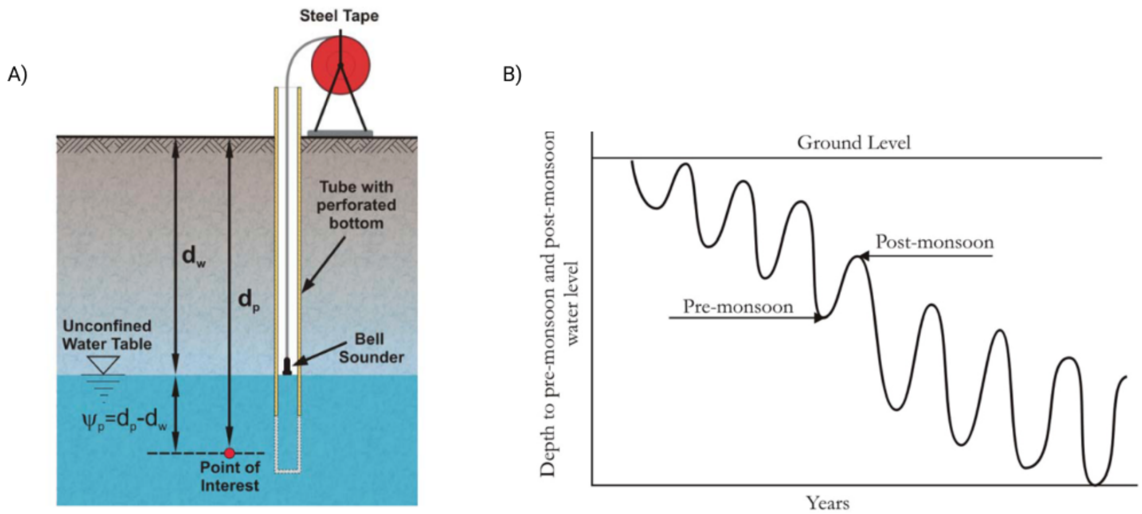

Figure 2.5: A) Illustration showing the measurement of water table elevation in the unconfined aquifer using a tape measure (from Or et al., 2005), and B) Theoretical representation of a groundwater table timeseries highlighting: 1) seasonality in water level due to precipitation patterns and 2) groundwater depletion over time (adapted from Shah, 2009)

The accurate monitoring of groundwater depletion is heavily dependent on the spatio-temporal coverage of water level data. Most groundwater monitoring programmes need to take into account the heterogeneity of aquifer systems, the distributed nature of anthropogenic and environmental influence, and the need for high-frequency long-term data ($>$ 20 years). Therefore, the collection of monitoring well is data often challenging and requires a significant institutional capacity to invest and then manage the logistics of data collection (IGRAC, 2020). As a result, groundwater data needed to monitor depletion is considered to be severely limited around the globe (Lall et al., 2020). Additionally, the data collected by different government agencies is often either poorly controlled for quality or not made available on public domains (IGRAC, 2020).

GRACE Data

Remote sensing data (e.g. from satellites) is increasingly being used to overcome data scarcity commonly associated with groundwater systems. Over the last few years, data from remote sensing source has been utilized to estimate regional groundwater potential (e.g. Machiwal et al., 2011; Maskooni et al., 2020), groundwater extraction volumes (e.g. Rodell et al., 2009), and to obtain data that can serve as proxies for groundwater usage (e.g. Barron et al., 2014). In particular, data collected as part of the Gravity Recovery and Climate Experiment (GRACE) satellite mission has become indispensable in monitoring regional groundwater storage dynamics. Launched in 2002, GRACE is a satellite system that predominantly measures changes in the gravitational field caused by the redistribution of water mass over the Earth's surface at a monthly scale (Tapley et al., 2019). GRACE data for hydrological applications is generally provided as a vertically integrated measure of changes in terrestrial water storage (TWS) at a given location. Thus, for studies aiming to measure changes in groundwater storage, there is a need to decompose this TWS estimate and isolate the GW component by subtracting the non-GW components such that:

$$\Delta GWS = \Delta TWS - (\Delta SM + \Delta SWE + \Delta SW) \tag{2.5}$$

where, SM, SWE, SW and GWS represent changes in soil moisture, surface water, snow, and groundwater storage respectively. ΔTWS is the GRACE-measured change in terrestrial water storage.

Over the last 15 years, most of the major global aquifer systems have been assessed using GRACE data. Numerous studies have found a good correlation between groundwater storage dynamics from GRACE and in-situ measurements (e.g. monitoring wells) (Scanlon et al., 2012; Bhanja et al., 2016; Huang et al., 2015). As a result, GRACE has been used to assess groundwater depletion in systems with intensive groundwater extraction like North-Western India (e.g. Rodell et al., 2009; Asoka et al., 2017), Central Valley and High Plains Aquifer in USA (e.g. Brookfield et al., 2018), North-China Plain (e.g. Feng et al., 2013) and the Murray-Darling Basin in Australia (e.g. Chen et al., 2016b). In one of the most comprehensive assessments of groundwater storage using GRACE data, Shamsudduha and Taylor (2020) recently utilized GRACE to estimate changes in groundwater storage across the 37 largest aquifers around the world from 2002-2016.

While providing researchers and government agencies an alternative data source to monitor groundwater storage dynamics (especially in unmonitored regions of the world), GRACE data has some inherent limitations that prevent it from being extensively used to monitor groundwater resources. The most important limitation associated with GRACE-data is that it has a spatial resolution of >100,000 km^2, and therefore, can only (at best) provide a 'big-picture' overview of groundwater storage dynamics in a given region (Scanlon et al., 2016). Therefore, most GRACE-based assessments need to be complemented with assessments relying on detailed local-scale groundwater datasets (e.g. from monitoring wells) to inform regional policies. Additionally, the isolation of the groundwater signal from GRACE TWS signal requires data on other hydrological components (like soil moisture, and surface water) from external sources. However, in most systems, this data is either unavailable at the spatial resolution of GRACE or is highly uncertain (Chen et al., 2016a; Shamsudduha and Taylor, 2020). Recent studies in the High Plain aquifer (Breña-Naranjo et al., 2014) and Tigris-Euphrates Region (Darama, 2014) showed that GRACE-based groundwater storage changes were erroneous due to an inadequate accounting of surface water components. Most studies presently rely on the outputs of global hydrological models or land-surface models (LSM) to estimate these non-groundwater components, however, there is considerable uncertainty in how different models estimate these components (Shamsudduha and Taylor, 2020).

2.7 Interventions

As problems associated with groundwater depletion have grown in frequency and severity around the globe, numerous interventions have been proposed to reverse and/or mitigate the effects of groundwater overexploitation. These interventions can be broadly grouped into two categories: technological and institutional (Giordano, 2009). Technological interventions generally consist of engineering-based solutions, and can further be broken down into demand or supply-focused interventions. Demand-focused solutions generally aim to reduce the rate of extraction at a given location by decreasing the demand for groundwater. This is commonly achieved through the spread of more efficient irrigation technology (like drip or sprinklers over traditional flood irrigation) (e.g. Fishman et al., 2015) or by promoting a change in cropping patterns towards less water-intensive crops (e.g Davis et al., 2019). Supply-focused interventions focus on increasing the supply of water available for human consumption at a given location. These solutions have included the development of water harvesting infrastructure that captures locally-generated runoff for percolation into the subsurface (e.g. Sakthivadivel, 2007), or where stored water is purposefully injected the into the aquifer for use during dry spells (Khan et al., 2008). Other supply-based solutions focus on providing users with alternative sources of water through surface irrigation schemes, or by importing out-of-basin water to meet demands (e.g. Amarasinghe et al., 2008).

In contrast, institutional interventions focus on improving the governance arrangements required to better manage groundwater resources at a given location. These measures have commonly focused on developing local-scale Water User Associations and/or programs that catalyze collective action in communities with known groundwater depletion (e.g. Meinzen-Dick et al., 2018). Other institutional interventions aim to regulate groundwater use patterns through a system of water rights or licenses (e.g. Khan and Brown, 2019), or by developing groundwater markets that incorporate pricing and trading of usage-rights (Ayres et al., 2021; Bruno and Jessoe, 2021). Other measures aim to limit groundwater use indirectly by influencing the pricing of electricity that is necessary to operate pumps or by providing incentives to farmers to reduce cropped area (Giordano, 2009).

2.7.1 Rain-Water Harvesting in India

The development of rain-water harvesting systems has been a widely applied strategy to increase groundwater storage and/or provide farmers with an alternative source of water globally. Though 'modern' society is only recently acknowledging the value of rainwater harvesting (Cain, 2014), such techniques have been a part of the local landscape in India for millennia now (Agarwal and Narain, 1997; Van Meter et al., 2014). In peninsular India, farming communities as far back as 505 AD (Pandey et al., 2003) have captured water in natural depressions in the landscape by building bunds on the downstream end. These ancient structures, commonly referred to as tanks, survive in these landscapes till present-day, and help store water from monsoonal periods for use in the dry season (Agarwal and Narain, 1997; Keller et al., 2000; Shah, 2009). There are currently over 200,000 tanks in India (Palanisami et al., 2010) with around 60% of them being concentrated in the Southern States of Tamil Nadu, Andhra Pradesh, Karnataka, and Kerela (Sakthivadivel, 2007).

Given the seasonal nature of precipitation in India, tanks have traditionally been built to help meet irrigation and domestic water demands of the local community during the dry season (Bitterman et al., 2016). These structures are on average about 20-40 hectares in size, and consist of crescent-shaped earthen bunds that help hold the collected water back (Gunnell and Krishnamurthy, 2003; Van Meter et al., 2014). During the monsoon season, runoff from the tank catchment area inundates the tank bed. Sluice gates present in the tank bund are then used to manage the outflow of water from the tanks to the irrigation channels, and into the agricultural fields in the tank command area (Figure 2.6). Tanks are often linked in cascades with overflow from upstream tanks routed through surplus channels into downstream tanks. These cascades can consist of a few to hundreds of tanks, often forming a complex hydrological network of anthropogenic wetlands in the region (Bitterman et al., 2016).

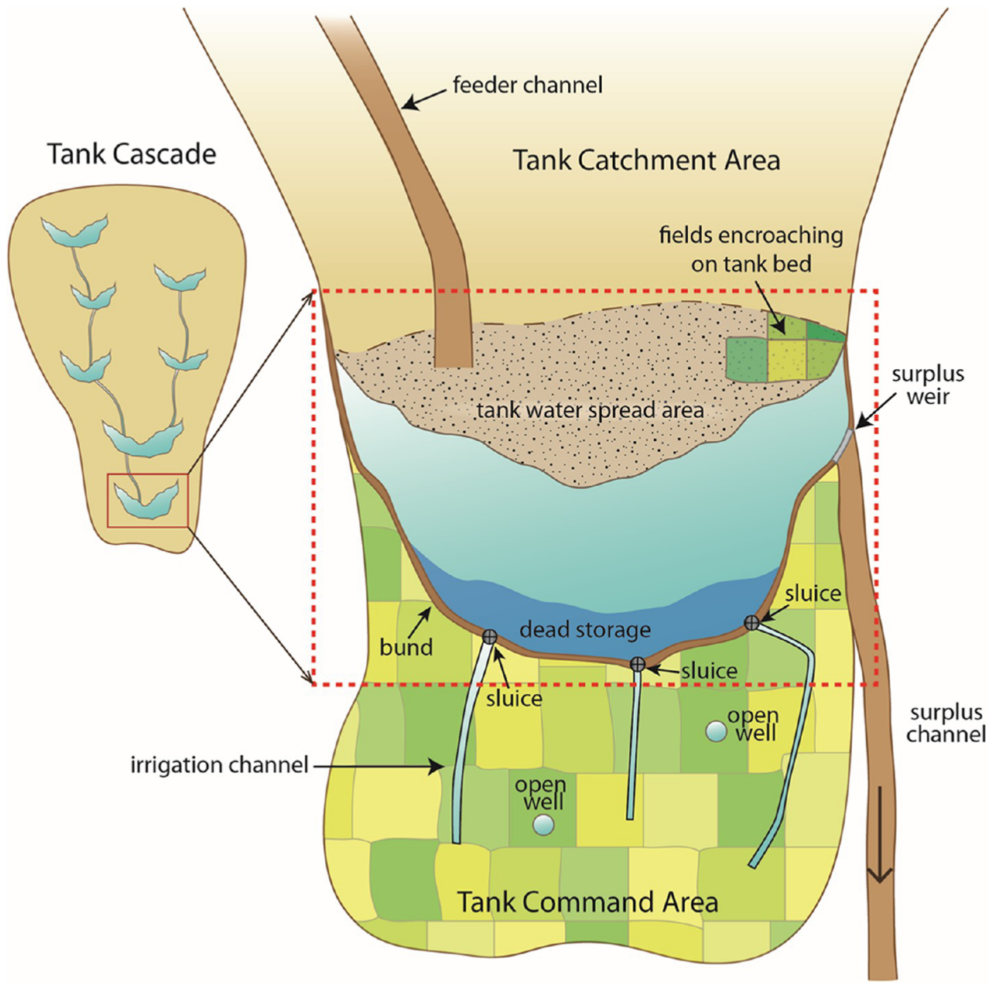

Figure 2.6: Important components of a tank irrigation systems (adapted from Van Meter et al., 2014)

In the state of Tamil Nadu, tank irrigation systems currently support an agricultural area covering 61% of the State, and enable the cultivation of subsistence crops like rice and market crops like maize, sugarcane, and chilli pepper (Van Meter et al., 2014). However, the functionality of tanks has evolved to extend beyond just agricultural water provisioning. Numerous studies have highlighted that tanks provide ecological as well as socio-cultural services to the local communities (Ariza et al., 2007; Palanisami et al., 2010). Tanks are known to recharge the shallow aquifer, while providing flood control and preventing soil erosion (Sakthivadivel, 2007). They also provide farmers with fertilizers (silt), while supporting local biodiversity. Culturally, tanks are sites of numerous rituals and festivals for the local population (Van Meter et al., 2014). The management of the stored water and maintenance of these structures has traditionally been carried out by informal village-level organizations that governed tanks as common-pool resources. These informal institutions ensured tank functionality through numerous operational rules including provisions for proper sluice operation and water distribution amongst the command area farmers (Egadeesan and Koichi, 2011). An essential part of tank management was the coordination of local labor for regular maintenance (e.g. repair work and desiltation) of the tank bed and supply channels. While tanks had traditionally dominated agriculture in Southern India, there was a trend away from them during the Colonial period in India (starting in the 18th Century). Numerous authors have now documented the increasing disrepair – in both the physical condition and management practices – of tanks during this era (Ludden, 1979; Sengupta, 1985). Part of the reason for this decay has been attributed to a mismatch arising from the administrative needs of a decentralized irrigation structure in a rapidly centralizing state (Ludden, 1979; Mosse, 2003), while others have pinned it onto numerous actions taken by the colonizers to increase rule and profit (D'Souza, 2006). Ultimately, the poor returns from tanks combined with the advancement of modern irrigation science and dire needs arising from a series of famines led to a preference for larger scale, centralized irrigation structures in the late-19th Century by the colonial government (Ludden, 1979; Sengupta, 1985). Tanks as a source of irrigation were further marginalized with the rapid rise of groundwater irrigation in India. The transition towards well irrigation has been shown to impact the community-level investments needed to keep the tank systems functioning, and this has led to rapid tank degradation with numerous instances of structural failures, over-siltation and illegal encroachment (Narayanamoorthy, 2014; Sato and Duraiyappan, 2011; Sivasubramaniyan, 2006; Van Meter et al., 2014). The overall share of tank irrigated area has fallen from 18.5% in 1960-61 to about 4.6% in 2001-02, while the well irrigated share has increased from 30% to 61% across India (Palanisami et al., 2010).

The combined effects of falling groundwater levels and uncertainty associated with climate change have led to a renewed interest in tank structures across India. Studies have shown that rain-water harvesting and artificial recharge have the potential to add almost 125 km^3yr^{-1} to the water supply, which can help reduce the expected mid-century water shortfall of $475 - 950$ km^3yr^{-1} in India (Gupta and Deshpande, 2004). From a climate change adaptation perspective, water harvesting is considered to be a 'low regret' adaptation that can improve the resilience of the ecosystem and rural community in India (Carabine et al., 2014; Shanmugasundaram et al., 2017). Studies have also shown that conjunctive use of surface and groundwater can be an important strategy in meeting seasonal demands (Siderius et al., 2015). Consequently, numerous organizations ranging from small non-governmental organizations (NGOs) to the World Bank are looking to restore the traditional rain-water harvesting structures in India (Van Meter et al., 2014). In Tamil Nadu, for example, tank rehabilitation in gained momentum in the 1980s and 1990s through engineering-dominated ventures funded by European Economic Community (EEC) (Palanisami et al., 2008). Similarly, the World Bank has invested over $189 million for tank restoration in Andhra Pradesh (Reddy and Behera, 2009). Most recently, the Groundwater Recharge Master Plan (GRMP) released by the Central Groundwater Board of India (CGWB) in 2005 has allocated close to $6 billion to improve the groundwater situation in the country through measures that aim to increase recharge through a variety of structures including percolations ponds (comprising of rehabilitation of old tanks and construction of new ones) and check dams (Shah, 2008). Overall, the interest in tanks has been rekindled in recent years for a variety of reasons including increasing groundwater recharge, adapting to climate change, and addressing equity concerns. However, despite the enhanced investment in tank rehabilitation, the understanding of how these ancient structures function in the current landscape is still limited (Glendenning et al., 2012; Van Meter et al., 2014).

Chapter 3

Incorporating local and global environmental considerations into groundwater stress assessments in India

Numerous regions are currently facing the socio-environmental consequences of depleting groundwater resources globally. Assessing regional groundwater stress (measured here as the ratio annual groundwater usage to groundwater supply) is important for setting policy targets and guiding interventions. However, the threshold of renewable groundwater supply that is considered available for human-use remains poorly defined at the regional-scale. In this study, we estimate district-scale extraction thresholds in India and then assess groundwater stress based on: a) no environmental considerations ('baseline'), (b) water requirements of 'local' groundwater-dependent ecosystems, (c) 'global' considerations using the current planetary boundary framework, and (d) a 'mixed' approach that is informed by both local and global considerations, and where a national groundwater use budget is disaggregated (top-down) to estimate thresholds. Compared to the baseline where 26% of the districts are considered over-stressed in India, we find that accounting for local environmental flow requirements results in 36% districts being classified as over-stressed with a hotspot emerging in Southern India. Under the global and mixed scenarios, we find that nearly 70% of districts (where currently >801 million people live) are classified as over-stressed given current groundwater extraction rates. However, we find that the effort required from over-stressed districts to stay within derived groundwater use thresholds in the mixed scenario (median groundwater stress = 143%) is considerably lower than the global scenario (median groundwater stress = 203%). Our findings suggest that incorporating environmental considerations would substantially lower the volume of groundwater resources available for human use in India (173-312 $km^3/year$; compared to 399 $km^3/year$ in the baseline scenario). The results from this study can help policymakers understand the implications of prioritizing environmental needs into groundwater management.

3.1 Introduction

Agriculture is considered to be a dominant driver of global environmental change with multiple negative planetary-scale impacts (Rockström et al., 2009; Campbell et al., 2017). Yet, further intensification of agricultural systems is still considered necessary for making progress towards multiple developmental goals like eradicating hunger (SDG2) (Godfray and Garnett, 2014; Pretty and Bharucha, 2014). A key part of achieving agricultural sustainability relies on the ability to meet water requirements associated with contemporary food production practices. Globally, agriculture accounts for nearly 70% of all freshwater withdrawals (Shiklomanov, 2000), a number that is even higher in regions like South Asia and Sub-Saharan Africa (Campbell et al., 2017). In particular, an expansion in the use of groundwater to meet agricultural water requirements has been instrumental in increasing food production globally over the last few decades. Recent estimates suggest that nearly 43% of global consumptive water use for irrigation comes from groundwater (Siebert et al., 2010). However, the increase in groundwater use for agriculture (nearly 70% of groundwater abstracted is intended for irrigation) has resulted in rapid groundwater depletion (Margat and Van der Gun, 2013), and recent estimates suggest that nearly 1.7 billion people live in areas where groundwater resources are under threat (Gleeson et al., 2012b).

The measurement of progress towards sustainability goals often relies on setting targets using relevant indicators. With regards to groundwater systems, this has often been done by quantifying 'stress' using the ratio of annual groundwater withdrawals to renewable supply in a region (Gleeson and Wada, 2013). At its most fundamental level, different forms of groundwater stress ratio (GSR) serve to compare human groundwater abstraction to groundwater availability in a region, and thus provide a basis for evaluating the long-term sustainability of regional groundwater use (Margat and Van der Gun, 2013). Groundwater stress assessments also highlight the degree to which negative outcomes (e.g. decrease in base flow, degeneration of wetlands, land subsidence) associated with groundwater pumping can be expected in a region. As a result, despite limitations in capturing the dynamics of complex aquifer systems (Margat and Van der Gun, 2013; Brauman et al., 2016), different forms of groundwater stress ratio (GSR) continue to be used to assess groundwater resources at aquifer, national and even global scales (e.g. Gleeson and Wada (2013); Richey et al. (2015); Herbert and Döll (2019); CGWB (2014); Forstner et al. (2018)).

A critical aspect of estimating groundwater stress depends on defining a threshold against which regional groundwater use can be compared. Traditionally, assessments of groundwater resources have assumed this threshold to be the regional renewable groundwater recharge volume such that pumping rates are considered to be 'sustainable' if they do not exceed total recharge rates. But numerous studies have emphasized the importance of separating the water requirements of groundwater-dependent ecosystems from estimates of groundwater recharge available for human use (Vrba et al., 2007; Gleeson et al., 2012b; Smakhtin et al., 2004; Srinivasan and Lele, 2017). This is driven by growing evidence of current anthropogenic groundwater pumping rates decreasing groundwater flow into water bodies, thereby negatively impacting the health of aquatic ecosystems (Kløve et al., 2011; Srinivasan et al., 2015; de Graaf et al., 2019; Rohde et al., 2021). Previous studies have estimated that nearly 41% of the global irrigation water use is extracted at the expense of the water requirements of environmental systems (Jägermeyr et al., 2017). As a consequence, researchers have developed groundwater stress indicators that either explicitly account for the water requirements of groundwater-dependent ecosystems (Gleeson et al., 2012b), or that measure stress based on the deviation of groundwater discharge under natural and human-impacted conditions (Herbert and Döll, 2019). Yet, despite growing recognition of the need to respect environmental flow requirements like The Brisbane Declaration in 2007 (Arthington et al., 2018), the integration of such measures into policies and regulations is considered to be severely lacking (Srinivasan and Lele, 2017; Gleeson and Richter, 2018; Liu et al., 2017).

In addition to incorporating local environmental water requirements, there has also been a recent push towards considering the global consequences of water use (Zipper et al., 2020). There is growing evidence of hydrologic processes in a basin both influencing and being influenced by global socio-economic factors and Earth-system processes (Hoekstra, 2010; Jaramillo and Destouni, 2015; Wang-Erlandsson et al., 2018). Some known examples include the irrigation in India supporting precipitation in East Africa (de Vrese et al., 2016), evapotranspiration change due to deforestation in the Amazon impacting rainfall patterns in the United States (Avissar and Werth, 2005) and nations like Jordan importing 'virtual' water embedded in trade products (Hoekstra and Hung, 2005). Yet, the scope of water management frameworks rarely extend past basin boundaries, and in particular, there is limited evidence of any global-scale policy or institutional coordination to inform the management of local freshwater systems (Hoekstra, 2010; Häyhä et al., 2016; Biermann, 2012). Recognizing the importance of considering the global consequences of freshwater use, the recently developed planetary boundary (PB) framework provides a basis to look at the management of freshwater systems through a global lens (Rockström et al., 2009; Gerten et al., 2013; Steffen et al., 2015). The PB framework sets a global limit to freshwater use (and other important Earth-System processes) under the assumption that transgressing this threshold would increase the chances of disrupting Holocene-like Earth System conditions. The current PB for freshwater use is set at 4000 km^3/yr of blue water consumption (i.e. consumption of water stored in lakes, rivers and aquifers) (Steffen et al., 2015). Given that groundwater accounts for nearly 36% of all freshwater consumption globally (Wada et al., 2014), the proper management of groundwater can play an important role in ensuring that freshwater consumption remains within the planetary boundary.

In this study, we ask: how would incorporating environmental considerations impact the groundwater available for human extraction across a nation? We explore this question by estimating safe groundwater use thresholds based on different local and global environmental considerations. Specifically, we estimate regional groundwater use thresholds based on: (a) no environmental considerations to set a 'baseline', (b) water requirements of 'local' groundwater-dependent ecosystems, (c) 'global' considerations using the current planetary boundary framework, and (d) a 'mixed' approach that is informed by both local and global considerations, but where a national groundwater use budget is disaggregated (top-down) to estimate thresholds based on current district-level extraction rates. We then evaluate how the thresholds derived in each scenario would influence hotspots related to groundwater stress (i.e. regions where groundwater extraction rates exceed estimated thresholds) based on current extraction rates.

We focus our analysis on regional agricultural systems in India, where 85% of the groundwater extracted is used for irrigation purposes (Mukherjee et al., 2015) and where groundwater irrigation supports 90 million households (Government of India, 2014b). India has the highest groundwater abstraction rates in the world ($\sim$250 km^3/year), and large-parts of the country are already facing the socio-environmental consequences of groundwater depletion (Mukherji and Shah, 2005; Rodell et al., 2009; Hora et al., 2019). Previous assessments of groundwater stress in the country have either focused on understanding the influence of water-saving irrigation technologies without considering the effects of environmental limits (Fishman et al., 2015) or have only taken local environmental considerations into account (Gleeson and Wada, 2013). Furthermore, most assessments have been conducted at the aquifer (Gleeson and Wada, 2013) or grid-scale (Herbert and Döll, 2019), neither of which are align with the political and institutional accounting units in India. Additionally, recent groundwater stress assessments using the Gravity Recovery and Climate Experiment (GRACE) satellite data, while enabling the identification of multiple characteristic groundwater stress regimes, are generally valid at only coarse spatial resolutions ($\geq$63,000 km^2)(Richey et al., 2015; Vishwakarma et al., 2018). Most importantly, despite playing an important part in directing support and interventions, groundwater stress assessments in India have been shown to over-allocate groundwater resources available for extraction, while inadequately accounting for environmental water demands (Srinivasan and Lele, 2017). Therefore, our analysis can serve a practical purpose of aiding current efforts to regulate groundwater resources and map groundwater stress in India, while contributing to the ongoing conversation of understanding the implications of incorporating environmental considerations into regional resource management.

3.2 Methods

In this study, regional groundwater stress is characterized using the ratio of annual district-scale groundwater extraction (km^3/yr) to renewable groundwater supply (km^3/yr) in India. We assume the renewable groundwater supply to be equal to the long-term annual groundwater recharge volume in a district and scale this recharge volume based on different environmental factors to estimate district-level groundwater use thresholds. In this analysis, we estimate groundwater stress in a district by:

$$\%GSR_d = \frac{Ext_d}{Agw_d^*} * 100 \tag{3.1}$$

where, $\%GSR$ is the groundwater stress in district 'd', Ext is the current annual groundwater extraction in a district, and Agw^* is the threshold of groundwater recharge available for use (in km^3) that is estimated under four scenarios: baseline, local, global and mixed (Table 3.1). In the following sections, we first describe the datasets used in this study, and then explain the methodology used to estimate groundwater use thresholds (Agw^*) in each scenario. Finally, we describe the methodology used to estimate district-level groundwater extraction, Ext.

Table 3.1: Different scenarios used to obtain district-level groundwater use thresholds in India

Scenario	Description
Baseline ($Agw_{base,d}$)	The entire district-level long-term annual groundwater recharge ($Rch_{total,d}$) volume is assumed to be available for human extraction
Local Consideration ($Agw_{EFR,d}$)	The groundwater recharge required to meet environmental water requirements (EFR) is subtracted from the $Rch_{total,d}$; i.e. $Agw_{EFR,d} = Rch_{total,d} - EFR$. EFR was calculated by estimating the monthly groundwater discharge value that is exceeded 90% of the time based on de Graaf et al. (2019).
Global Considerations ($Agw_{PB,d}$)	District groundwater budgets are derived to be consistent with the current freshwater planetary boundary target of limiting freshwater use to 40% of accessible freshwater flows (Rockström et al., 2009). In this scenario, district groundwater thresholds are set to 40% of the district groundwater recharge; i.e. $Agw_{PB,d} = 0.4 * Rch_{total,d}$
Mixed Considerations ($Agw_{MC,d}$)	A national-level groundwater use target is set based on 'global' considerations using the current freshwater planetary boundary. District-level thresholds are then derived from this national budget based on current district-level groundwater extraction levels (grandfathering). Budgets derived are harmonized to ensure districts can meet their development needs while being within 'local' hydrological limits; i.e. $Agw_{MC,d} = [Agw_{PB,d}, Agw_{EFR,d}]$

3.2.1 Datasets

We obtained district-level data on agro-economic variables prepared by the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT) through their Village Dynamics in South Asia (VDSA) project from publicly accessible domains (http://data.icrisat.org/dld/index.html). The VDSA database combines census datasets from multiple National and State-level agencies of India, and provides agro-economic data for 311 districts (from 18 States) between 1966 and 2015. As the number of districts in India has nearly doubled from 1966 to 2015, the database aggregates data from newer districts into their parent districts (1966 boundaries) which enables comparisons over longer periods. In this study, we estimate environmental pressures from agricultural systems for a single period (between 2005-2015) by taking the median value of different variables (Table S1) used in the analysis. This helped to overcome data gaps in any single year while also correcting for year-to-year variability. Due to the absence of data in some regions, our analysis focused on 287 districts for which complete data was available (covering 87% of the land area and 90% of the population) (Figure S1).

We extracted district-level recharge data compiled by the Central Groundwater Board (CGWB), which is the national groundwater regulatory agency in India (CGWB, 2014). These estimates form the basis of national-scale groundwater values commonly used globally as part of FAO's AQUASTAT database (FAO, 1999; Margat et al., 2005). The CGWB estimates recharge using water level changes, precipitation patterns, and simplified assumptions about hydrogeological and irrigation parameters (Fishman et al., 2015). The CGWB district-scale recharge estimate includes recharge due to leakage from irrigation structures (e.g. canals) and return flow from the application of irrigation water. We specifically obtained the total renewable groundwater recharge volume (TRGW) estimates from CGWB reports. Since the data released was for district boundaries in 2011, we aggregated the extraction and recharge estimates in the dataset to their parent districts based on 1966 boundaries.

Finally, we used publicly available gridded monthly groundwater discharge and recharge outputs from the WaterGAP global hydrological model (https://www.uni-frankfurt.de/45218063/WaterGAP) (Herbert and Döll, 2019). WaterGAP quantifies the global terrestrial storage and flow of groundwater and surface water resource at a $0.5^o * 0.5^o$ scale by taking into account human modifications to the water cycle. We specifically obtained monthly groundwater recharge data for model runs from 1981-2010 under conditions of human water use. To estimate environmental water requirements from groundwater discharge, we utilized grid-cell level groundwater discharge data for model runs from 2001-2010 under 'naturalized' conditions. The model output from these 'naturalized' conditions provided estimates of groundwater discharge in the system without human water use (Herbert and Döll, 2019; Müller Schmied et al., 2021).

3.2.2 Estimating district-scale groundwater use thresholds

Baseline

In the baseline scenario, the entire district-scale annual groundwater recharge volume was assumed to be available for human consumption ($Agw_{base,d} = Rch_{total,d}$). $Rch_{total,d}$ was set equal to the total annual groundwater recharges (TRGW) estimates published in the CGWB reports. This represents the scenario under which there is no explicit consideration of environmental water requirements.

Using local environmental flow requirements

Due to a lack of data on groundwater-contribution to environmental flow requirements (EFR) at the district-scale in India, we relied on modelled outputs from the WaterGAP global hydrological model (Herbert and Döll, 2019) to estimate EFR requirements. We specifically used the modelled groundwater discharge outputs between 2001-2010 under naturalized conditions from WaterGAP (Herbert and Döll, 2019), and adapted the methodology developed by de Graaf et al. (2019) to estimate EFR of groundwater dependent ecosystems using a low-flow index. For each grid cell, the groundwater needed to maintain ecosystem services was estimated by calculating the monthly modelled groundwater discharge value that is exceeded 90% of the time (Q90) (Gleeson et al., 2012b; de Graaf et al., 2019). The monthly Q90 groundwater discharge values were then summed to obtain yearly targets (in km^3). Previous studies have shown that groundwater recharge and discharge estimates vary considerably between datasets (Gleeson and Wada, 2013), and therefore, instead of directly subtracting the estimated EFR volumes from the CGWB-based recharge estimates, we instead used WaterGAP-based recharge estimates to obtain the ratio of EFR to total groundwater recharge. We estimated the modelled long-term groundwater recharge values for each grid cell by taking the median annual groundwater recharge rate ($GW_{rch,WG}$) between 1981-2010. Finally, by taking the ratio of grid-cell specific annnual EFR volumes to $GW_{rch,WG}$, we estimated the proportion of grid-cell groundwater recharge (EFR_{WG}) that is required to sustain groundwater-dependent ecosystems.

Due to a mismatch in district-scale boundaries and grid-boundaries, we resampled the EFR-target grid to a finer 0.1-degree resolution, and then estimated district-scale EFR targets by taking the median value of all the grid points falling within the district-boundary. District-scale estimates of groundwater available for anthropogenic use under the EFR scenario were calculated by subtracting the district-scale EFR target from current CGWB estimates of district-scale groundwater recharge such that:

$$Agw_{EFR,d} = Rch_{Total_d}(1 - EFR_d) \tag{3.2}$$

where, $Agw_{EFR,d}$ is the estimated district-level groundwater use threshold ($km^3/year$) and EFR_d is the district-level EFR requirements estimated using the WaterGAP hydrological model (Herbert and Döll, 2019), and Rch_{Total_d} is annual recharge volumes obtained from CGWB (section 3.2.1).

Incorporating global freshwater standards

We used the planetary boundary (PB) framework to help develop district-level groundwater use thresholds in India that are consistent with global limits. The PB framework (Rockström et al., 2009), built on the concepts of tipping points and biophysical limits, offers a powerful science-policy tool in approaching the growing need to manage freshwater systems through a global lens (Hoekstra, 2010; Biermann, 2012). The need for global governance related to freshwater is generally justified given the scale of human modifications to the water cycle (Abbott et al., 2019), increasing amounts of 'virtual' water in trade products globally (Dalin et al., 2012) and growing evidence that highlights the teleconnections between regional and large-scale hydrological processes (Vorosmarty et al., 2015). The current planetary limit for annual freshwater use (specifically blue water) is set at 4000 km^3 (Steffen et al., 2015), although there are on-going debates regarding the hydrological basis for such a boundary (Heistermann, 2017) and the methodology used to develop the boundary (Gerten et al., 2013). One of the appeals of the original approach was the development of a boundary that was measurable, but that also respected the scientific uncertainty inherent in any such quantification (Biermann, 2012). The original approach quantified the PB by restricting the total renewable (and accessible) blue water supply for the entire planet ($\sim$12,500 km^3) to $40 \pm 8\%$ – which yielded a PB between 4000-6000 km^3, while the remaining 60% of the supply was considered to be reserved to meet environmental flow requirements and to avoid physical water stress (Rockström et al., 2009). The 40% value represented a prescriptive threshold beyond which water resources are generally assumed to be severely water-stressed (Vorosmarty et al., 2000; Alcamo et al., 2003), and can be seen in a similar vein to other prescriptive approaches developed in hydrological sciences (Richter et al., 2012; Gleeson and Richter, 2018).

While the original PB was developed for blue water consumption, Gleeson et al. (2020) recently proposed to split the freshwater PB into sub-boundaries for different water stores (which includes a groundwater sub-boundary) relating to multiple core functions of water systems. This was aimed at addressing concerns regarding the oversimplification of the water cycle and its linkages to different Earth-System processes. However, there is currently no estimate for a global/national limit for groundwater use based on linkages to different Earth-system processes. Thus, while we treat groundwater as a separate sub-boundary in our analysis, in the absence of a global limit of groundwater use, we chose to develop a groundwater sub-boundary for India that is consistent with the current planetary boundary for freshwater use.

The operationalization of the PB concept into decision-making requires a way to disaggregate these global boundaries to regionally relevant environmental thresholds (Häyhä et al., 2016; Lucas et al., 2020; Zipper et al., 2020; Fanning and O'Neill, 2016). Previous studies working on disaggregating the 4000 km^3 global boundary to sub-global scales highlighted its limitation with regards to a lack of consideration of regional water availability (Nykvist et al., 2013). Instead, Nykvist et al. (2013) developed national boundaries by limiting consumption to 40% of the renewable blue water supply of a country. We followed a similar approach in this scenario, but instead of working with the cumulative renewable blue water supply for India, we restrict our approach to the district-level renewable groundwater supply. Therefore, we developed district-level thresholds by limiting the annual groundwater recharge available for human consumption to 40% of the total annual district-level groundwater recharge ($Agw_{BM,d} = Rch_{Total_d} * 0.4$).

Mixed considerations

An alternative approach to operationalize the PB concept into decision-making has focused on ways to disaggregate global boundaries to regionally relevant thresholds based on different environmental, socio-economic, and ethical principles (Häyhä et al., 2016; Lucas et al., 2020; Zipper et al., 2020). A top-down allocation offers the advantage of accounting for inherent differences in development, attitudes, and capacities while setting regional targets, and thus offers a pathway towards fairness in resource distribution and allocation. This top-down disaggregation differs from the bottom-up approach used by Gerten et al. (2013) (and the threshold derived in the 'global' scenario), where global freshwater boundaries are obtained by aggregating local hydrological limits. While the application of a top-down approach has most commonly been utilized in the context of Earth-system processes (e.g. climate change, ocean acidification) where the location of emissions can be considered to be less important, recent research has explored the potential of applying these principles to freshwater systems (Zipper et al., 2020; Nykvist et al., 2013).

Most top-down approaches rely on the use of distributional approaches that help disaggregate a global/national budget to local resource use thresholds. Distributional approaches can aim to prioritize principles like capacity, equality, or cost-effectiveness (see van den Berg et al. (2019) for more detailed discussion), and different approaches can favour different regions in terms of the budget allocated (Lucas et al., 2020). While the distributive principle used depends on regional preferences or ethical considerations, for freshwater systems, it is critical for any resource use threshold derived to be within the hydrological limits of a region. Alternatively, it is also important for any top-down derived threshold to not be overly restrictive in terms of resource use such that it prevents a region from making progress towards sustainable development to fulfill basic human rights (Raworth, 2012). Hence, estimating regional resource use thresholds using a top-down approach requires additional harmonization to ensure local hydrological limits are not transgressed (Zipper et al., 2020), and freshwater requirements for regional development are met. In this analysis, we estimate district-level groundwater use targets in India based on a top-down allocative approach using three steps: (1) setting a national-level groundwater use threshold; (2) disaggregating the national threshold to the district-scale based on allocative principles, (3) ensuring the budgets derived fall within the local hydrological conditions of a district, but that also allowing regional development.

To establish a groundwater use boundary for India, we chose to set a national threshold that was consistent with the current freshwater planetary boundary. Adapting the approach used by Nykvist et al. (2013) to develop national freshwater thresholds, we set the national groundwater threshold by restricting groundwater use to 40% of the annual renewable groundwater supply in India. Based on the data compiled by the CGWB, the current renewable groundwater supply is estimated to be 432 km^3 for India, and thus by restricting groundwater available for use to 40%, we obtained an annual national groundwater budget of 173 km^3. This value should be considered an initial effort to derive a national threshold that is consistent with the current freshwater planetary goal.

The next step of analysis involved disaggregating the national groundwater budget into district-level thresholds based on an allocative principle. Following the approaches developed by van den Berg et al. (2019) and Lucas et al. (2020), we applied a 'grandfathering' allocative approach in our analysis which assumes that regions with greater current/historic resource use are entitled to a greater proportion of the resource budget. In the case of groundwater use thresholds, we disaggregated the national groundwater use budget based on current district-level groundwater extraction rates. Therefore, the district-level share of the national groundwater budget was estimated by:

$$GF_d = \frac{Ext_d}{Ext_{total}} * GWA_{NC} \tag{3.3}$$

where, Ext_{total} is the total groundwater extraction rate in India, GWA_{NC} is the national groundwater use budget and Ext_d is the extraction rate of district d.

In the final step, we harmonized the boundaries derived to be within thresholds based on local hydrological limits, which we assume to be equal annual recharge values after subtracting minimum environmental needs ($Agw_{EFR,d}$). To ensure that the allocated groundwater budget also enables regions to meet water requirements needed for human development, we ensured that all districts would have access to at minimum 40% of their annual groundwater recharge volume ($Agw_{PB,d}$). Thus, using the district groundwater thresholds derived based on local and global considerations (Table 3.1), we harmonized the district-level groundwater budgets by:

$$Agw_{MC,d} = \begin{cases} Agw_{EFR,d}, & \text{if } GF_d \geq Agw_{EFR,d} \\ Agw_{BM,d}, & \text{if } GF_d \leq Agw_{BM,d} \leq Agw_{EFR,d} \\ GF_d & \text{otherwise} \end{cases} \tag{3.4}$$

where, $Agw_{MC,d}$ is the district groundwater budget based on mixed considerations.

3.2.3 Estimating district-scale groundwater extraction

We estimated groundwater extraction at the district-scale by estimating the irrigation water needed to produce major crops in India. Irrigation currently accounts for over 85% of groundwater withdrawals in India (Mukherjee et al., 2015). We followed the methodology proposed by Fishman et al. (2015) to estimate agricultural groundwater extractions for different districts. This involved estimating the district-scale total irrigation water requirement (TIR) as:

$$TIR_d\ (m^3) = \sum_{c=1}^{n} A_{c,d} * i_c \tag{3.5}$$

where, TIR is the total yearly irrigation water requirement in district d (m^3/year), $A(m^2)$ is the irrigated area under crop c; n is the total number of irrigated crops; i is the crop-specific seasonal irrigation volume assuming flood irrigation listed in Table S2. Using the ratio of groundwater irrigated area in each district, we estimated groundwater extraction values as:

$$TGW_{irr,d}(m^3) = TIR_d * \frac{GIA_d}{TIA_d} \tag{3.6}$$

where, GIA is the total irrigated area by groundwater (from open-wells and borewells) in district d; TIA is the total irrigated area from all sources.

Additionally, we obtained the district-level groundwater extraction volumes $TGW_{cgwb,d}$ (m^3) estimated by the CGWB. The CGWB estimates groundwater extraction based on the number of extraction structures present in any district (like dugwells, tubewell) and assumptions on the structure-specific annual extraction volumes CGWB (2014).

Previous analysis by Fishman et al. (2015) found that groundwater irrigation volumes by taking crop-specific irrigation volumes tend to be 15% higher than the groundwater extraction volumes estimated by the CGWB of India. However, both methods can be considered to have a high degree of uncertainty as they tend to assume spatially uniform patterns of groundwater extraction. Therefore, to prevent our estimates of groundwater extraction to be biased towards either method, we estimated the district-level groundwater extraction volumes as:

$$TGW_d = \frac{TGW_{irr,d} + TGW_{cgwb,d}}{2} \tag{3.7}$$

where, TGW_d is the mean district-level annual groundwater extraction volume (km^3)

3.3 Results

3.3.1 Impact of environmental considerations on extraction thresholds

For each scenario outlined in Table 3.1, district-scale groundwater use thresholds were estimated across the study area. Figure 3.1 highlights the distribution of (area-averaged) volumetric thresholds estimated for districts in each scenario. As expected, we find a reduction in groundwater use thresholds relative to the baseline as constraints that take environmental considerations into account are introduced. For the 'local' considerations scenario, where the water requirements of groundwater-dependent ecosystems are subtracted from the district-level recharge volumes, we find that the median annual district-level groundwater use (area-averaged) thresholds reduces from 130mm [Interquartile range(IQR): 86-274mm] to 101mm [IQR: 66-219mm]. The median annual district-level environmental flow requirements (EFR) requirements constituted 20% [IQR: 17-24%] of the annual groundwater recharge volume in our study area. Spatially, we find that the Eastern portion of the Ganges Basin and parts of the Deccan Plateau, on average, have higher %EFR compared to regions in North-Western (Figure 3.2). Regions of North-Western India generally have the highest groundwater irrigation intensity in India (Rodell et al., 2009), and the lower EFR as a percent of the total groundwater recharge can potentially be explained by an increase in groundwater recharge induced by return flow from irrigation and greater groundwater storage availability in the alluvial aquifer found in the region (Bhanja et al., 2019). Overall, in our study area, the cumulative groundwater use thresholds bounded by EFR considerations (312 km^3) are much lower than the baseline scenario (399 km^3). While the CGWB assessments do not explicitly consider the groundwater contributions to environmental flow requirements, they subtract a 'natural-discharge' term from the groundwater recharge estimates of a district. These values are stored under the term net renewable groundwater recharge volume (NRGW) and are obtained by scaling the total renewable groundwater recharge volume (TRGW) by a fixed percentage (5-10%). But we found that the 'natural discharge' values were generally much lower (Figure A.1b) than our estimates of %EFR (Figure 3.2a). This results in higher volumes of groundwater being made available for human consumption at the district-scale across the country at the expense of environmental considerations.

For the global and mixed considerations scenarios, we find that the cumulative groundwater use thresholds in our study area are 60% and 47% lower than the baseline scenario respectively (Figure 3.1 and Figure 3.3). We find that the median volumetric districts-level thresholds are relatively close in the Global and Mixed scenarios (0.48 km^3 and 0.61 km^3). However, the thresholds derived in the Mixed scenario (IQR = 0.67 km^3) have a greater spread than the Global scenario (IQR = 0.44 km^3), and the threshold distribution in the Mixed scenario skews similarly to the Local scenario (Figure 3.1). This is

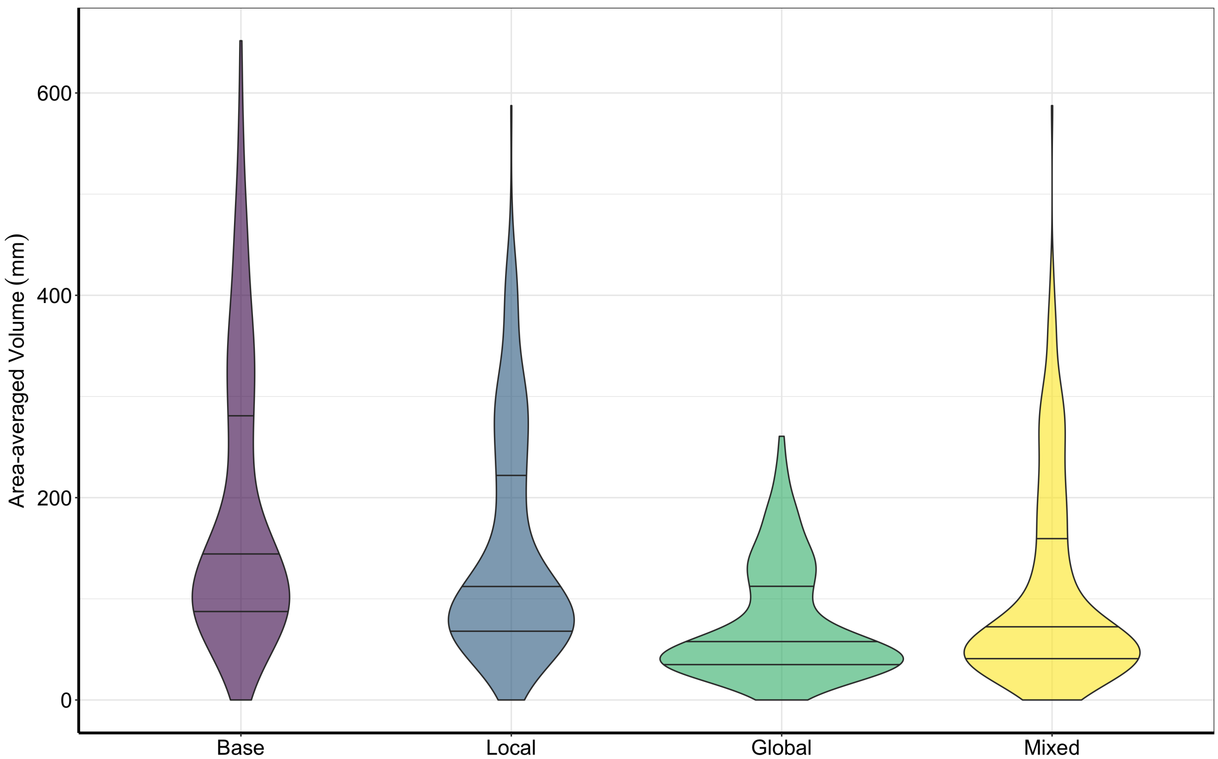

Figure 3.1: Distribution of district-level groundwater extraction thresholds (area-averaged) estimated in each scenario (Table 3.1). The 3 horizontal lines from *top to bottom* in each violin represent the 75^{th}, 50^{th} and 25^{th} percentile values.

driven primarily by the relatively high extraction rates in North-Western India (e.g Punjab, Haryana) which results in districts in these regions being allocated a groundwater use budget that exceeds local hydrological limits (Figure A.3). In contrast, districts in large parts of Central and Southern India get allocated budgets that are often less than 40% of their renewable groundwater recharge due to relatively lower groundwater extraction rates. As a result, given the harmonization process developed in this analysis (Equation 3.4), these districts get allocated budgets equal to the thresholds derived as part of the Global scenario ($Agw_{PB,d}$) (Section 3.2.2). Overall in the Mixed scenario, we find that out of the 287 districts analyzed, 58 have allocations equal to thresholds derived based on EFR requirements (relatively high extraction zones), 131 have allocations equal to the thresholds derived based on global considerations (to meet development needs), and 98 districts have thresholds in-between $Agw_{EFR,d}$ and $Agw_{PB,d}$ (Figure A.3).

3.3.2 Assessing district-scale groundwater stress

In Figure 3.4, we map district-level groundwater stress for each scenario analyzed in this study. Based on the volumetric thresholds derived in the Local scenario, where district-level thresholds were derived after subtracting environmental flow requirements, we find that 103 districts (36%) fall in the over-stressed ($\%GSR \geq 100$) category. This corresponded to nearly 390 million people living in regions where (in the long-term) we would expect to see the environmental impacts associated with substantial reductions in groundwater discharge into surface water bodies. Comparing these estimates of district-scale %GSR to estimates derived in the baseline scenario, we find a smaller proportion (26% of the districts) of the country to be over-stressed in the baseline scenario (Table 1). Spatially, the hot-spots in North-Western India (and the Indo-Gangetic plains) are similar in these scenarios, while a notable hot-spot in Southern India emerges in the Local scenario (Figure 3.4). Although a large-scale analysis tracking the change in groundwater discharge into surface water bodies is lacking in India, we compared the results from these scenarios (by

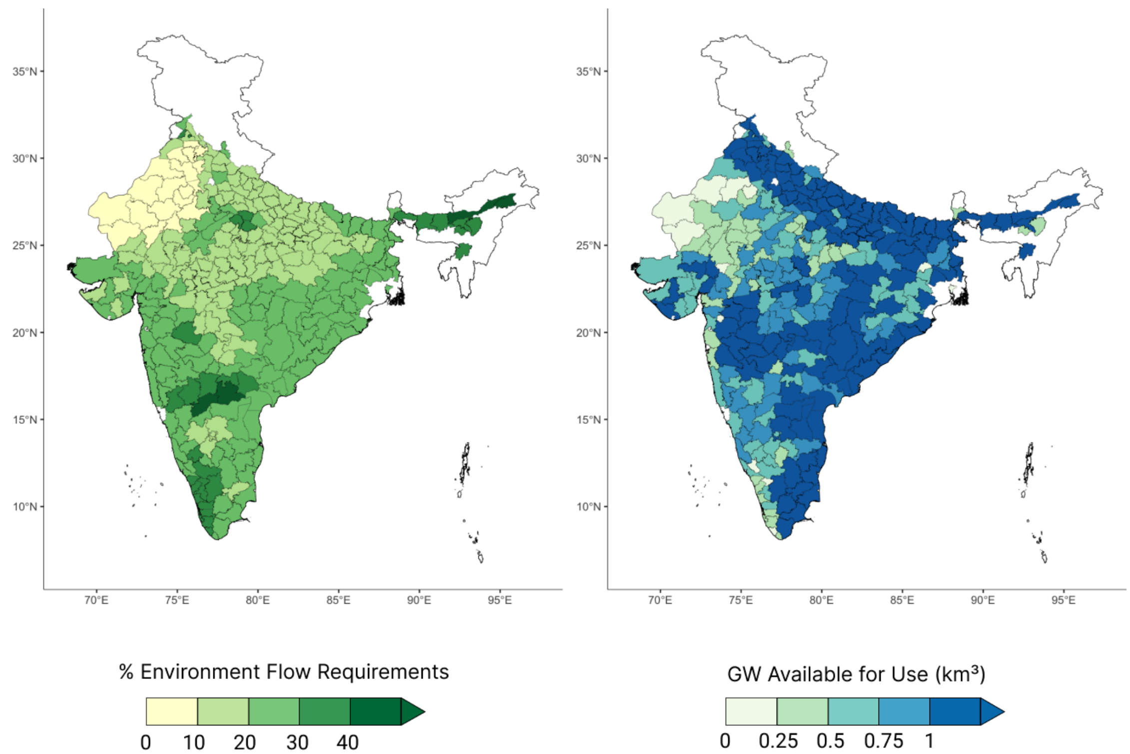

Figure 3.2: Incorporating local environmental consideration to estimate groundwater use thresholds in the 'local' scenario, a) Estimated percent environmental flow requirements (%EFR) of groundwater dependent ecosystems using modelled groundwater discharge and recharge data from the WaterGap hydrological model. b) Volume of annual groundwater recharge available for use in each district after subtracting %EFR.

aggregating the %GSR values in districts/basins) to where other studies have observed base flow reductions. Overall, we find that there is good agreement between the two methods, however, the baseline scenario resulted in the misclassification of two study regions as safe (Table A.3).

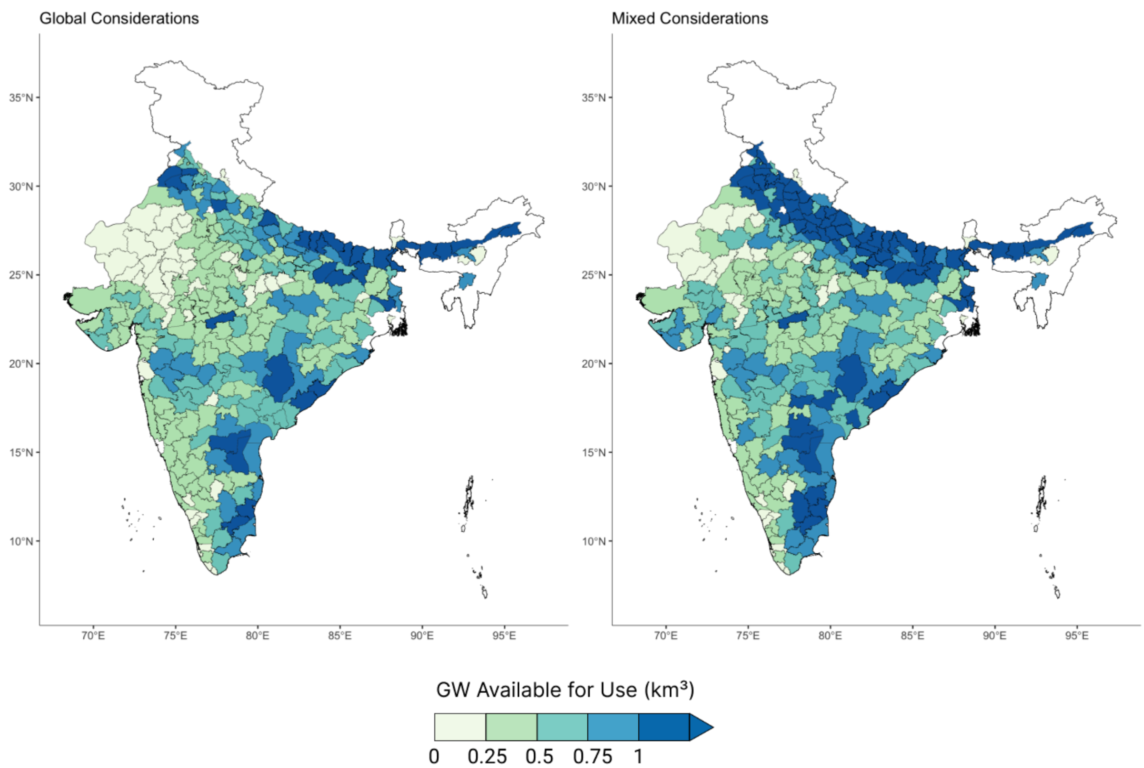

Figure 3.3: District-level (volumetric) thresholds derived in the: a) global scenario, where the derived thresholds are consistent with the freshwater planetary boundary; and b) mixed scenario by developing a national ceiling of groundwater use in India before disaggregating to the district-level based on the 'grandfathering' allocative principle. The derived boundaries are then harmonized to be consistent with local hydrological limits while allowing regions to meet potential developmental requirements.

In the Global and Mixed consideration scenarios, we find that nearly 70% of the districts fall in the over-stressed category. Spatially, the hot-spots related to groundwater extraction in India spreads from being centered around North-Western India (in the baseline scenario) to large parts of the Deccan plateau and Southern India in these scenarios. Based on these results, we find that nearly 801 million people would be classified to be living in districts where groundwater extraction exceeds the derived thresholds. Although the number of over-stressed districts between the Global and Mixed scenarios overlap exactly, we find that the distribution of district-level groundwater stress (%GSR) values varied considerably between these scenarios (Figure A.4). This was predominantly due to the allocative principle used to derive thresholds in the Mixed scenario, where regions with greater extractions rates were allocated a larger share of the national groundwater use budget. As a result, the groundwater stress values were attenuated in these high extraction regions resulting in a distribution with a lesser spread compared to the Global scenario. We find that the median %GSR in the Global and Mixed scenarios was 169% and 129%, while the interquartile range (IQR) was 166% and 67% respectively.

Based on the estimated volumetric thresholds (Figure 3.1) and groundwater stress results (Figure 3.4, the four scenarios analyzed in this study (i.e. base, local, mixed and global) can be seen as progressively safer in terms of the groundwater available for human extraction (and more in-line with different environmental considerations). Combining the results from each scenario, we can group districts into 4 categories based on if a district transgresses the estimated groundwater use threshold in: a) all scenarios, b) local scenarios (implying stress in the global/mixed), c) only in Global or Mixed scenarios, and d) no scenario. We find that 74 districts fall in the category where groundwater extraction rates exceeds the estimated threshold in all scenarios (Figure 3.5). These districts are primarily centered around North-Western India, and our results suggest that these regions would require even more stringent regulations to ensure extractions levels are within environmental limits. In the second category, we have 29 districts where groundwater availability thresholds are transgressed in the Local scenario (and Global/Mixed scenario) and not the baseline scenario. These districts are predominantly found in parts of Southern and Western India. Alternatively, we find that 97 districts fall in the over-stressed category only in

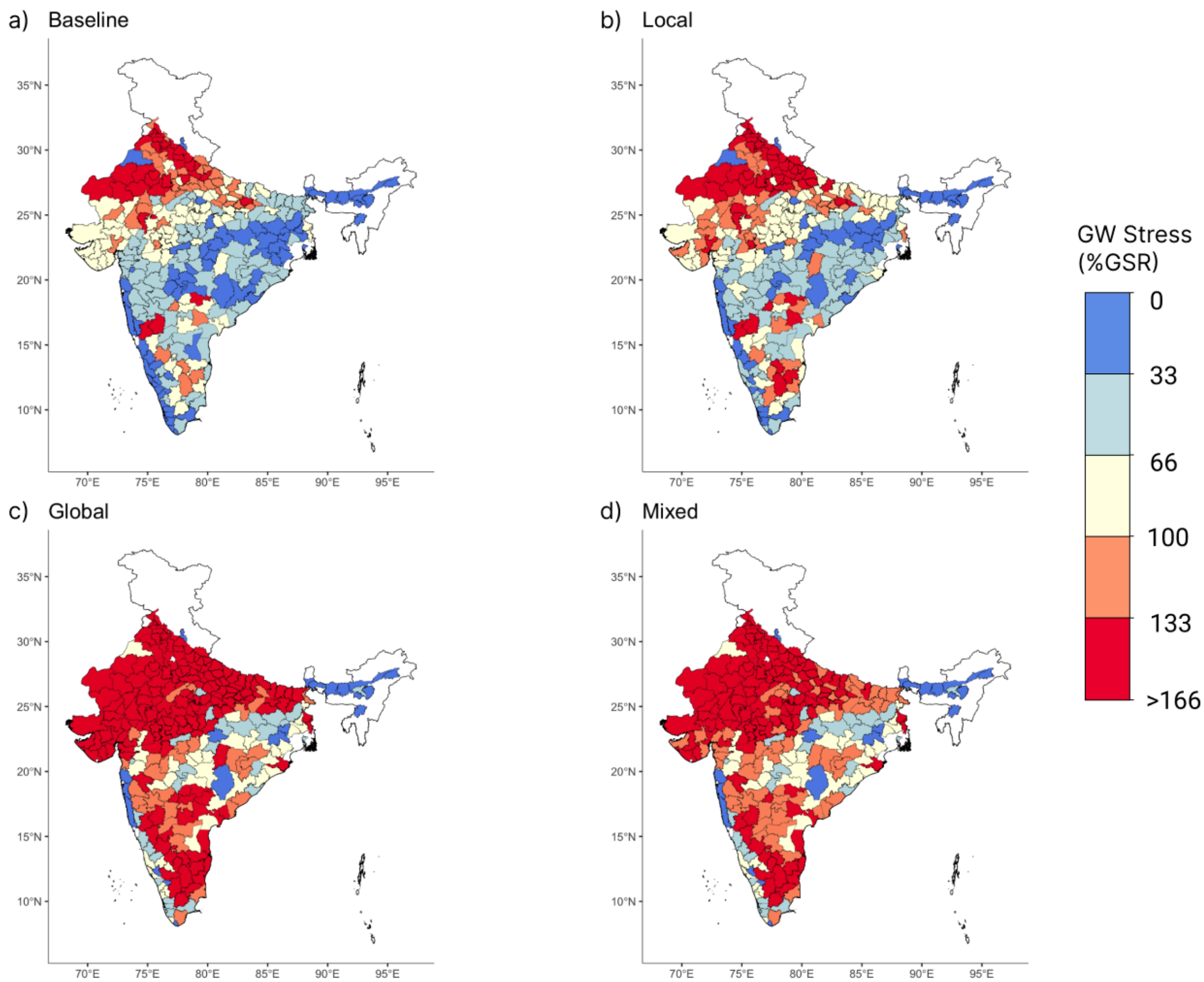

Figure 3.4: District-level groundwater stress for each scenario analyzed in the study (Table 3.1). Regions in shades of orange and red represent districts where groundwater extraction exceeds the estimated district-level groundwater availability threshold.

the Global and Mixed scenarios. These districts are located in the Indo-Gangetic plains and large regions of Peninsular India. For districts in categories 2 and 3 (i.e not over-stressed in the baseline but in the Local or Global/Mixed scenarios), developing thresholds based on environmental limits would have the biggest impact as current groundwater extraction rates would have to be lowered to stay within the derived thresholds. This is in contrast to the interpretation of groundwater stress in these districts under the baseline scenario, where stabilizing or even an increase in extraction rates would be considered an option. Finally, we find that 86 districts fall into the category where the groundwater availability thresholds are not exceeded in any scenario. These districts are situated mainly along the Western coast of India and in Eastern India. Our results in these regions suggest that there is still a possibility of carefully increasing extraction rates to aid agricultural production and regional development, albeit to a lower level to ensure environmental boundaries are not transgressed.

3.4 Discussion

This analysis addresses the growing need to incorporate environmental considerations when establishing anthropogenic groundwater use thresholds. In particular, this study provides insights on how environmental requirements at different scales can be estimated with respect to groundwater systems across a nation, and how current groundwater extractions rates compare to estimated groundwater supply based on different environmental considerations. We focus our analysis at the district-scale in India – a country with some of the highest groundwater extraction (and depletion) rates in the world, and where groundwater systems are expected to play a crucial role in minimizing the vulnerability of agricultural systems to the combined effects of climate change and population increase. In this section, we explore the challenges associated with prioritizing environmental considerations into groundwater management in India (Section 3.4.1), discuss approaches that can help reduce current extraction rates (Section 3.4.1), and highlight the limitations of the current

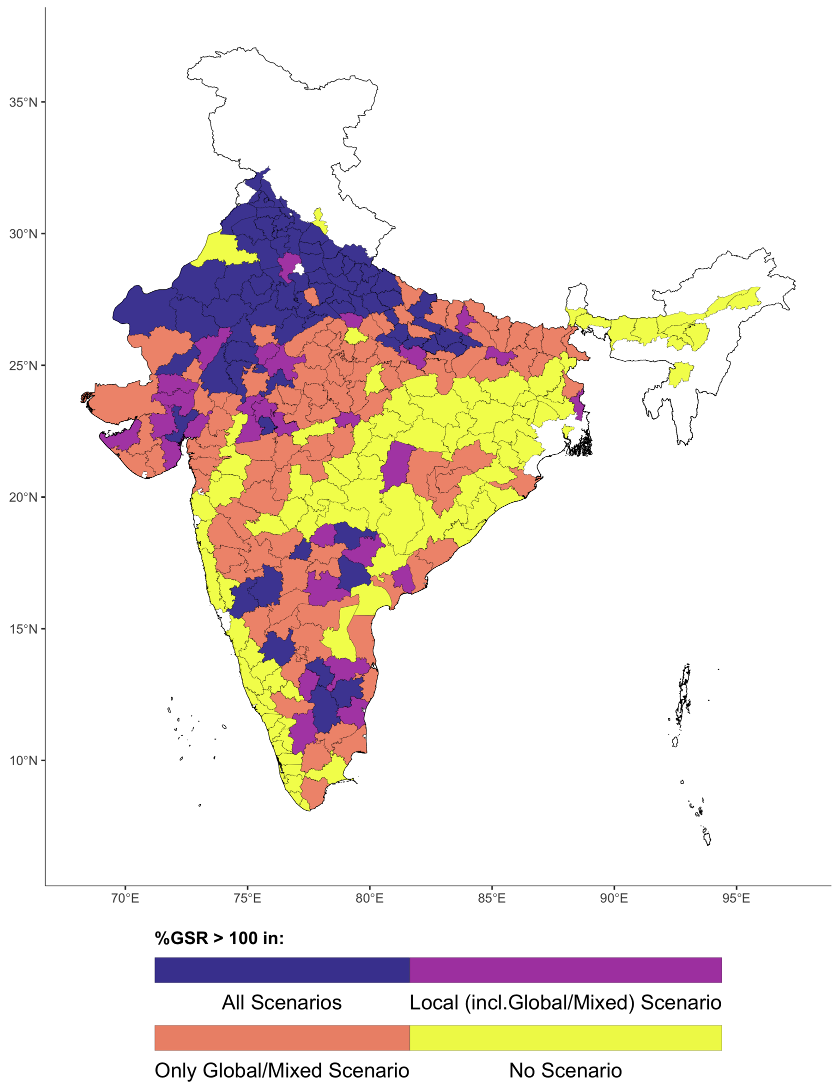

Figure 3.5: Categorization of districts based on the number of scenarios in which groundwater extraction rates there exceed the estimated groundwater availability thresholds.

analysis (Section 3.4.3).

3.4.1 Incorporating environmental considerations

Our findings suggest that the groundwater resources available for human extraction in India are considerably lower than generally assumed (i.e. using the baseline scenario) when local and global environmental considerations are taken into account. Yet, justifying the prioritization of environmental needs into groundwater management in the current Indian context can be considered to be particularly challenging. Globally, the focus on environmental considerations has often followed after some degree of human water security has been achieved. In contrast, regions like India (and many other middle/lower-income regions of the world) face the dual threat of high water insecurity with respect to human and environmental needs (Vörösmarty et al., 2010). With nearly 60% of the cropped area falling under rainfed agriculture (Suresh et al., 2014) and irrigated areas being forced to contend with reducing water supplies, improving access to groundwater can still be considered to be a key strategy toward stabilizing agriculture-based livelihoods and achieving overarching development targets. Furthermore, about 43% of the population has been estimated to be employed in the agricultural sector in India (The World Bank, 2019), and drastic water restrictions to account for environmental needs when these users already have limited access to water resources can further increase their vulnerability. In addition to low human water security in India, barriers to incorporating environmental needs are compounded by a general lack of consensus around what constitutes environmental needs and how these needs are estimated given limited available data. As a result, one school of thought concerning groundwater management has advocated the temporary over-exploitation of groundwater resources to aid the trajectory towards sustainable development (Custodio, 2002), while increasing capital (e.g. institutional, social) and improving the adaptive capacity of regions to deal with future environmental problems (Moench, 2007).

In contrast, a valid case for incorporating environmental needs into current groundwater management frameworks can also be made as the environmental impacts of groundwater exploitation are increasingly being documented in large parts of India (e.g. Table A.3). Additionally, experience in other parts of the world has shown that incorporating environmental needs proactively can be a less expensive option over the long-term than addressing them reactively when issues reach a certain threshold (Palmer et al., 2008). Furthermore, groundwater systems often operate at slow time scales ($\geq$20 years) compared to typical policy horizons, and thus it would necessary for interventions related to environmental considerations to be introduced at the present-day for desirable outcomes to manifest themselves in the future (Aeschbach-Hertig and Gleeson, 2012). Finally, studies analyzing the relationship between resource use, environmental sustainability and developmental indicators have shown uneven results in India (Hauff and Mistri, 2015; Alam et al., 2016), and there is a possibility for environmental requirements to not be taken into account even after human water security improves in the country.

With arguments to support either side of whether environmental considerations are taken into account, ultimately, the decision to account for environmental requirements at the current stage would come down to the level of environmental protection desired by socio-political entities in India. Based on our findings, the incorporation of environmental demands results in extraction rates exceeding derived groundwater use thresholds in large parts of the country, and, therefore, ensuring that environmental boundaries are not transgressed will require a fundamental shift in how groundwater resources are managed in India.

3.4.2 Staying within thresholds

There are several proposed options for limiting current groundwater extraction rates in India. On the technological front, Fishman et al. (2015) recently explored the potential effect of water-saving irrigation technologies (like drip, sprinkler) at the district-scale in India. Based on their assessment, the adoption of these technologies can reduce groundwater extractions rates by up to 30% across the country. Alternatively, Davis et al. (2018) analyzed the potential of switching major cereal crops types in India from rice to coarser cereals like millets and maize. Based on their results, a transition towards alternative cereals could reduce the demand for irrigation water by up to 33% (Davis et al., 2018), while improving the climate resilience of the crops produced and overall nutritional supply (Davis et al., 2019). Other proposed solutions include increasing recharge and/or providing alternative sources of irrigation to farmers through the development of rain-water harvesting structures. Studies have estimated rain-water harvesting can potentially increase water supply by 125 km^3 in India (Gupta and Deshpande, 2004). Further still, recent research has also highlighted the potential of energy policies to indirectly influence pumping behaviour. This option predominantly relies on the introduction of meters to measure and then charge users based on electricity usage (instead of the current flat tariff scheme), or by limiting the power (i.e. number of hours) supplied to the agricultural sector (Giordano, 2009; Sidhu et al., 2020).

The incorporation of solutions that have the desired effect of reducing groundwater extraction rates depends on the institutional and regulatory framework under which the solutions are introduced. Experience from attempts to regulate groundwater over-abstraction globally has shown that state-centered governance has been often been ineffective for reasons that includes a lack of data, logistical challenges, financial constraints, and a lack of political motivation (Molle and Closas, 2020). Thus, the introduction of top-down regulatory solutions without an ability to, for example, monitor use or enforce regulations can potentially produce no tangible effects. For example, village-scale studies on the introduction of drip irrigation technologies in Rajasthan found that farmers often intensify crop production after adopting the technology to the extent that there is no net reduction in (and even an increase in some cases) water usage (Bierkens and Wada, 2019). Other than the ability to regulate groundwater extraction, proposed solutions need to also take into the socio-cultural preferences of groundwater users. For example, policies to incentivize the consumption of orange-fleshed sweetpotato in Sub-Saharan Africa, which in theory seemed to have a high potential to bridge nutritional deficiencies, resulted in a limited uptake in some target regions due to the consumption preferences of the local population (Hummel et al., 2018). Additionally, there may be a need to introduce multiple intervention measures concurrently to see the desired effects. Fishman et al. (2016) found that the introduction of energy policies to promote groundwater conservation saw limited success partially due to a lack of technological options for farmers to adopt. Finally, any proposed solution would also require an holistic consideration of biophysical and social impacts. For example, research has shown that introducing metered connections can disproportionately impact the poorer farmers (Sidhu et al., 2020), or that constructing rain-water harvesting structures to increase groundwater supplies can negatively impact surface water flows to downstream users (Bouma et al., 2011).

3.4.3 Limitations

The major contribution of this study is to highlight how incorporating different environmental considerations might affect the estimation of anthropogenic extraction limits across India. While we rely predominantly on data collected by national agencies in India, previous studies have highlighted the importance of using an ensemble of data sources to estimate regional groundwater stress as a way to mitigate the uncertainty associated with large-scale groundwater analysis (Gleeson and Wada, 2013). Therefore, the approach developed in this analysis can be extended to incorporate data from alternative sources like satellites and global/regional hydrological models. Likewise, while we focus on estimating groundwater stress at the annual scale in India, recent studies have indicated that estimating annual water stress can often mask stress at the seasonal or monthly time-scale (Brauman et al., 2016). Thus, future groundwater assessments that measure stress at shorter timescales can complement this assessment. Further, we recognize that our estimates of environmental needs derived are highly uncertain, and therefore should be viewed as an initial step towards establishing more robust EFR measures in India. Our approach for estimating environmental flow requirements relies on a fairly simplistic metric that inadequately accounts for regional hydrological characteristics predominantly due to a lack of data. It may be beneficial for future analysis to use additional methods to estimate EFR that, for example, better incorporate the seasonal patterns of environmental water requirements (Pastor et al., 2014).

With growing calls to utilize a global lens to help guide regional groundwater management (Gleeson, 2020), we estimated regional thresholds that would be consistent with the original freshwater planetary boundary in this analysis. However, we acknowledge that the planetary boundary framework has been shown to be contentious in multiple ways especially in relation to freshwater systems (Gerten et al., 2013; Heistermann, 2017; Gleeson et al., 2020). There are on-going efforts to redefine the boundary based on more rigorous hydrological considerations (Gleeson et al., 2020), and thus, further analysis will be necessary to better incorporate groundwater-specific planetary boundaries. Furthermore, although popular in the climate change literature, there is currently a lack of precedent of developing regional targets based on a national budget for groundwater systems. Therefore, approaching management with such a lens might require fundamental shifts in the current institutional and regulatory framework in India. Furthermore, in our analysis, we have only considered a single disaggragating approach to allocate our estimated national groundwater budget. Previous studies have shown that the allocation approach used can lead to distinct winners and losers (Lucas et al., 2020), and thus approaching budget-setting exercises would require an in-depth analysis of what an agreeable principle might be from a biophysical, ethical, and socio-economic perspective in India (Häyhä et al., 2016).

3.5 Conclusions

Improved access to groundwater resources has played a crucial role in improving food security and livelihoods in many regions of the world over the last few decades. However, environmental considerations have often been inadequately incorporated into the management of groundwater resources with negative impacts of groundwater over-abstraction being increasingly observed in many parts of the world (Rohde et al., 2021; Srinivasan et al., 2015). With calls to better integrate environmental factors growing, the potential implication of incorporating environmental demands into groundwater management remains unclear especially in systems with high human water insecurity. In this study, we focus on improving our understanding of how incorporating environmental considerations would affect the groundwater extraction volumes available for human use across a nation. Overall, our results show that the incorporation of different environmental considerations would substantially impact the volume of groundwater resources available for human use globally. In regions like India, where groundwater extraction rates already exceed the annual supply volumes (despite a limited consideration of environmental needs), a recognition of environmental limits would render large parts of the country over-stressed with respect to current extraction rates. Therefore, staying within these thresholds would require a fundamental change in how groundwater resources are regulated and managed in these regions. The scenarios developed in this analysis can provide perspective on the efforts required for regions to stay within derived groundwater use thresholds if local and global environmental limits are taken into account.

Chapter 4

The Groundwater Recovery Paradox in South India

Reported groundwater recovery in South India has been attributed to both increasing rainfall and political interventions. Findings of increasing groundwater levels, however, are at odds with reports of well failure and decreases in the land area irrigated from shallow wells. We argue that recently reported results are skewed by the problem of survivor bias, with dry or defunct wells being systematically excluded from trend analyses due to missing data. We hypothesize that these dry wells carry critical information about groundwater stress that is missed when data is filtered. Indeed, we find strong correlations between missing well data and metrics related to climate stress and groundwater development, indicative of a systematic bias. Using two alternative metrics, which take into account information from dry and defunct wells, our results demonstrate increasing groundwater stress in South India. Our refined approach for identifying groundwater depletion hotspots is critical for policy interventions and resource allocation.

4.1 The paradox of rising farmer distress in areas of “groundwater recovery”

Food security is inextricably tied to water availability for irrigation. In arid and semi-arid regions with high inter-annual variability in rainfall, a significant fraction of the irrigation water demand is met by mining groundwater. Increasing rates of groundwater usage have led to the drying up of major aquifers around the world (Giordano, 2009; Mukherji and Shah, 2005; Srinivasan and Kulkarni, 2014). India has some of the highest groundwater extraction rates in the world, with annual abstraction increasing from 25 km3/year to 200 km3/year between 1950-2000 (Giordano, 2009; Shah, 2005). These high rates of abstraction are especially alarming as groundwater accounts for 60% of irrigation water and 85% of drinking water in India (World Bank, 2010). Given the societal dependence on this critical resource, accurate identification of hotspots of groundwater depletion is imperative, so as to design effective intervention strategies.

Recent analyses have identified North India (NI) as a hotspot for groundwater depletion (Asoka et al., 2017; Panda and Wahr, 2016; Rodell et al., 2009). In contrast, water levels are reported to be rising in South India (SI) (Asoka et al., 2017; Bhanja et al., 2017; Panda and Wahr, 2016). These conclusions are based on a variety of methods, including satellite observations such as those made by the NASA Gravity Recovery Climate Extraction (GRACE) satellites, water-level measurements in groundwater monitoring wells, and global hydrological models. The problem is that these findings contradict on-the-ground field reports and farm surveys that report increasing well failures during the same time period (Merriott, 2015; Srinivasan et al., 2015)

Indeed, using a keyword search on the Dow Jones Factiva database, we found that the number of newspaper articles reporting groundwater depletion in SI have increased by an order of magnitude since the 2000s (Figure B.1). Furthermore, farm surveys that are done as a part of the agricultural and minor irrigation census (Government of India, 2014a) provide evidence that farmers are drilling deeper wells. Census reports suggest a decline in shallow well irrigated area from 1996-2010, and an increase in the deep well irrigated area over the same timeframe (Figure B.2). The median percent decline in shallow well irrigated area from 1996-2000 to 2006-2010, across the South Indian states of Andhra Pradesh, Telangana and Karnataka is 52%, while the median percent increase in deep well irrigated area over these states is 98% over the same timeframe. If groundwater rejuvenation is indeed occurring in SI, as claimed by earlier studies (Asoka et al., 2017; Bhanja et al., 2017), why are farmers switching from shallow to deep wells, and why are newspaper mentions of groundwater stress increasing?

We hypothesize that the lack of agreement between large-scale analysis of groundwater vulnerability and on-the-ground reports in South India can be attributed to methodological constraints arising from the unique characteristics of hard rock aquifers; viz. low storage capacity and heterogeneous spatial patterns of storage. Our objective in this paper is (1) to assess groundwater storage trends in India using traditional data sources (monitoring wells, satellite data), (2) to explain differences in groundwater stress inferred from traditional "hard" hydrological data (monitoring wells, satellite data) versus "soft" non-hydrological data (census surveys, field reports, newspaper searches), and (3) to identify alternate large-scale metrics of groundwater depletion that are consistent and reliable.

4.2 Methods

District-wise irrigated area from 1996-2011 was obtained from the agricultural census database (http://vdsa.icrisat.ac.in/) compiled by the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT). This database contains information on district-scale (Figure B.3) irrigated area that has been grouped based on the source of irrigation (Table B.1). As the number of districts in India has changed significantly since the 1960's, the database aggregates data from new districts into (parent) districts from 1966 for consistent temporal comparisons. Monthly precipitation data was obtained from the Indian Institute of Tropical Meteorology (IITM) from 1980-2016 (https:

//www.tropmet.res.in/Data%20Archival-51-Page). Monthly precipitation values were converted to standardized precipitation index (SPI) values to represent region specific dry and wet periods. SPI for 12, 24 and 36 months was calculated by fitting a gamma distribution to the cumulative precipitation amounts for the same time period.

We used monthly Gravity Recovery Climate Experiment (GRACE) data from 2002-2016 to estimate the satellite derived groundwater storage anomalies (GWSA). Specifically, GWSA was estimated by subtracting surface water storage from the GRACE-derived terrestrial water storage anomaly (TWSA). The surface water storage (canopy storage, soil moisture, snow) was estimated using Noah land surface model, available from the Global Land Data Assimilation System (GLDAS) (Rodell et al., 2004). We obtained level-3 TWSA version RL05 (Swenson and Wahr, 2006; Landerer and Swenson, 2012) from the Centre for Space Research (CSR) at the University of Texas, Austin (ftp://podaac-ftp.jpl.nasa.gov/allData/tellus/L3/land_mass/RL05). The accompanying scaling factors based on the Community Land Model v4.0 (CLM) were applied to reduce the signal loss from sampling and post-processing. We ensured that the GRACE-derived TWSA and GLDAS-derived surface water storage were relative to the same baseline period (2002-2016 in our analysis). The GLDAS-forcing data showed strong correlation with the IITM precipitation used in the study (Figure B.4).

Observation well data from 1996-2016 was obtained from the Central Groundwater Board (CGWB) database (Figure B.5) that contains water level measurements recorded four times a year (January, May, August and November) for 29,513 wells (http://www.india-wris.nrsc.gov.in/wris.html). Of these, 12,279 wells were active in 1996, representing wells with the longest possible records in the database. Based on census reports (Government of India, 2014a), we categorized ‘Dug Wells’ as shallow wells, and ‘Tube Wells’ (or ‘Bore Wells’) as deep wells for a region. We used shallow wells with >18 (out of 21) years of data to estimate groundwater trends. For deep wells, this condition was relaxed due to a lack of long-term monitoring wells, and wells with >10 years of data were used. The number of wells used for trend analysis ranged from 6350 to 7532, depending on the month of analysis.

We assessed statistical significance of trends in groundwater storage for both satellite and monitoring well data using the non-parametric Mann-Kendall trend test (Mann, 1945), while slopes were estimated using the Theil-Sen slope estimation method (Sen, 1968). We used the trend estimation techniques suggested by (Yue et al., 2002), which corrects for the influence of autocorrelation in the data series. Trends and slopes in GRACE data were estimated at the yearly time scale after averaging the monthly GWSA estimates. Trends in observation well data was obtained separately for each of the four months to avoid the influence of seasonality.

4.3 Results and Discussion

4.3.1 Groundwater storage trends in India

We used GRACE satellite data to estimate trends in groundwater storage (GWS) in India for two overlapping time periods: 2002-2016, and 2005-2016. Consistent with previous studies, the analysis showed widespread significant ($p - value < 0.1$) decreasing trends in groundwater storage anomalies in NI (above 23^o N) for both time periods (Figure 4.1a and 4.1b). The SI (below 23^o N) regions, however, showed a noteworthy difference in trends between the two periods. A large fraction of the wells that showed a positive trend with a 2002 start date showed non-significant trends ($p - value > 0.1$) with a 2005 start date. Specifically, the trend analysis from 2005 reveals 23% of the region with significant positive trend, 19% with negative trend and 58% with no trend, in contrast to 51%, 11% and 38% of the region with positive, negative and no trend, respectively with a 2002 start date (Figure 1c). In fact, only 16% of the cells identified by (Bhanja et al., 2017) as improving due to political interventions in Andhra Pradesh based on a 2002-2014 GRACE analysis still showed a significant positive trends in the 2005-2016 analysis. This sensitivity to the choice of initial date potentially arises due to the extreme drought that occurred in India

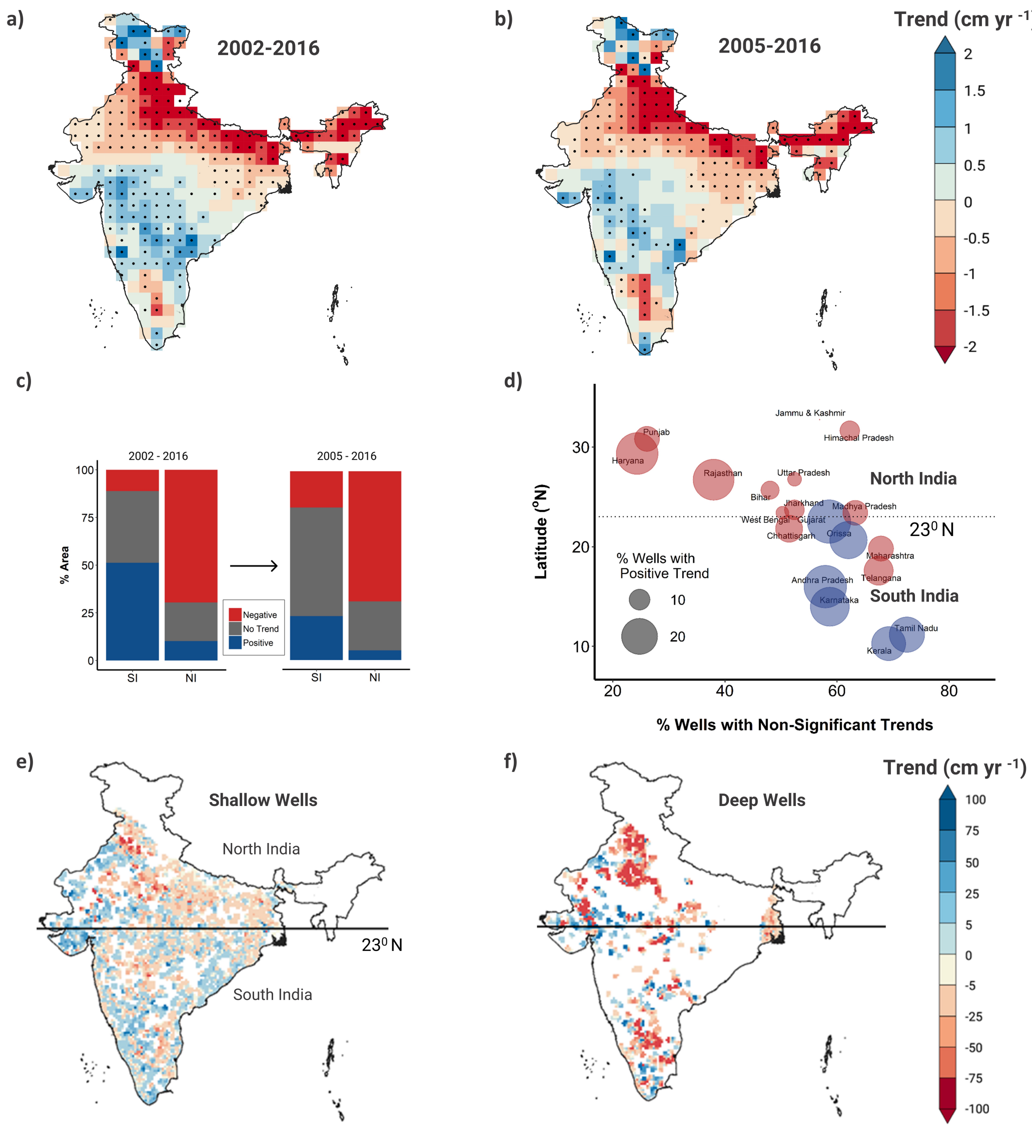

Figure 4.1: Long-term groundwater storage trends in India: Yearly trend ($cm\,yr^{-1}$) in groundwater anomaly from GRACE analysis for (a) 2002-2016 and (b) 2005-2016. Dots represents areas with statistically significant trends ($p-value \leq 0.1$). (c) Percent change in the distribution of positive, negative and non-significant GRACE based trends as a function of the analysis timeframes in NI (above 23^o N) and SI (below 23^o N). Results highlight that the percent area with positive and no significant trends are highly sensitive to the period of analysis in Peninsular India. (d) Percentage of wells with non-significant trends ($p-value \geq 0.1$) in May aggregated at the State-scale. Each circle represents a State and the size of the circle represents the percentage of wells with positive trends ($p-value \leq 0.1$). Red circles represent States with greater negative trends than positive, while blue circles represent States with more positive trends. Results show that a majority of States have a large percentage of wells ($\geq$ 50%) have non-significant long-term trends. (e) Water level trends ($cm\,yr^{-1}$) in shallow monitoring wells. (f) Water level trends ($cm\,yr^{-1}$) in deep monitoring wells. Note that wells trends are presented for the pre-monsoon season (May), and all wells used have statistically significant trends ($p-value \leq 0.1$)

during 2002-2004 (Figure B.6), and highlights the sensitivity of the trends to the initial date, especially when the period of analysis is relatively short (10-20 years). Thus, we argue that in a monsoonal climate with high inter-annual variability in precipitation it is important to consider the starting point of analysis when evaluating the significance of various groundwater trends. The other significant limitation of GRACE analysis is that while it provides an excellent estimate of regional trends in groundwater depletion, it is aggregated at a larger scale, and provides no information on local depletion hotspots. This becomes especially important in a highly heterogeneous, hard rock aquifer system that characterizes South India (Dewandel et al., 2011; Perrin et al., 2011a).

To address this issue and identify local depletion hotspots, we analyzed groundwater monitoring well data available from 1996-2016 (Figure 4.1d-f). Interestingly, other than the North-Western states of Punjab and Haryana, we found that majority of wells in India show no significant trends ($p - value > 0.1$), and all the south Indian states have >50% wells with non-significant trends (Figure 4.1d and Figure B.7). Past studies have tended to draw conclusions based on either wells with only significant trends (Asoka et al., 2017) or by not testing for significance (Bhanja et al., 2017). Further, by only considering wells with significant trends, we find that both shallow and deep wells in NI show declining trends (median trend = -7.6 cm/y for shallow wells and -51 cm/y for deep wells). In contrast, the deep wells in SI show a declining trend (-14 cm/y), while the shallow wells there show an increasing trend (4.4 cm/yr). Interestingly, when the wells are not segregated by depth, the median water level trends in all wells more closely mimics that of the shallow wells, given the more extensive network of shallow wells (median trend = -10.8 cm/year for NI and 2.7 cm/year for SI), explaining why previous studies (Bhanja et al., 2017; Asoka et al., 2017) that did not segregate wells by depth found an increase in water levels in SI wells. Our analysis of groundwater level data highlights two critical points. First, there are significant depth variations in water level trends in a multi-aquifer system that characterizes the Indian subcontinent, and thus aggregating information with depth conceals critical trends and stresses in the aquifer system. Second, in shallow, hard-rock aquifer systems with low storage buffers and high inter-annual variability in water levels that characterizes most of South India, a large fraction of the wells have non-significant trends and there is critical information that is missed by not considering them. The GRACE data and the groundwater monitoring data corroborate each other over most of India, except parts of North Western and Central India (Figure B.8). Furthermore, the GRACE data highlights patterns of large areas with non-significant trends in SI, similar to the monitoring well data.

While these issues have never been evaluated at the scale of SI, other local studies have questioned the value of long term water level trends in highly dynamic hard rock aquifer systems (Pavelic et al., 2012; Maheswaran et al., 2016), and argued for groundwater sustainability to instead be evaluated in terms of short term water provisioning (Fishman et al., 2011). In a recent field study in the Arkavathy river watershed in Karnataka, SI, (Srinivasan et al., 2015) argued that the groundwater monitoring well data that was available from the CGWB reports was not representative of on-the-ground reality. The two wells recording data in the CGWB reports showed water levels at 10-30 m BGL, and no significant temporal trends. In contrast, their survey of wells in the region found no visible open wells with water. A more focused analysis based on a detailed survey 472 borewells in a small (26 km^2) sub-catchment of the Arkavathy watershed, revealed that the median depth of the borewells farmers were digging increased from <50m in 1970s to over 200m in 2000s (Srinivasan et al., 2017). Deeper borewells are a clear indication of disappearing groundwater, despite CGWB wells recording no significant trends in this region. The lack of trend in this local study is analogous to the large fraction of wells showing no significant trend. Since most studies have ignored wells with no significant trend, they have missed asking the question, given that a majority of wells show a lack of any significant declining trend in South India, why do irrigation statistics allude to groundwater stress?

4.3.2 Stable water levels are because of survivor bias

We contend that this conundrum can be explained using the concept of survivor bias, which is a type of selection bias (Hernán et al., 2004; Simundic, 2013). Survivor bias is an artefact that arises in statistical analysis of data by focusing on data that were filtered based on some selection criteria. Subsequent analysis on the data that "survived" has the potential to significantly skew the interpretation (Aggarwal and Jorion, 2010; Ton et al., 2018). Examples of survivor bias are commonly found in finance where failed hedge funds are excluded from performance studies (Aggarwal and Jorion, 2010), or occupational health studies with inadequate consideration of workers with poorer health status (Buckley et al., 2015).

With groundwater well data, we often use a selection criterion where wells with a certain proportion of missing data are routinely eliminated from the analysis of long term trends. Missing data in a well record can occur in two ways: (1) the well goes defunct and stops collecting data permanently during the analysis time-frame, (2) the well records no data in multiple intermediate months within the timeframe. The underlying cause of such missing data can be either (1) physical, where the water level in the well falls permanently or temporarily below the well screen depth, or, (2) logistical, where operators neglect maintaining monitoring wells, or they collect or record the data inadequately. We argue that while the latter (operator errors) can contribute to missing data, the former is one of the key underlying reasons for missing data. The aquifer system of SI is known to experience large inter-annual fluctuations in water level such that a dry well can recover and become functional relatively quickly during wet periods (Fishman et al., 2011; Reddy et al., 2009). Paradoxically, wells that go dry are usually excluded from time series analyses of water level trends that tend to select wells that have the most complete datasets (Asoka et al., 2017; Bhanja et al., 2017).

To test the above claim, we explore the relationship between the %dry wells and metrics that capture climate variability and anthropogenic stress on the groundwater resources. We designate wells as "dry" in a given season and year if they did not record any data in that timeframe, and estimate the %dry wells in a region as ratio of the number of dry wells to the total number of monitoring wells in that region. For climate analysis, we aggregate the % wells at the regional scale of NI and SI to obtain % dry wells per year between 1996 - 2016 (Figure B.6), and de-trend the data using the least-squares approach to reduce the effect of systematic changes (e.g. changing technology or groundwater irrigated area) in a region (Hosseini-Moghari et al., 2019; Kumar et al., 2016). Overall, we find a significant negative correlation between the 36-month Standardized Precipitation Index ($r = -0.44$, p-value<0.1; Figure 4.2a) for each year, and the residuals of the % dry well timeseries. The higher percent dry wells in years with a low SPI confirms that the missing data is most likely related to dry spells when water level in wells fall below the screen level. These wells might recover in wet years, but the lack of water availability in dry years is an indication of groundwater stress.

To understand the relationship between the % dry wells and the degree of groundwater development at the district scale, we estimate the median % dry wells by aggregating the % dry wells per year over the 20 years of analysis (1996-2016) in that district. We contend that the median percent of "dry wells" over the 1996-2016 timeframe is indicative of the degree of groundwater stress in the region. We find a significant positive correlation between the median % dry wells and the % area under groundwater irrigation in a district, aggregated over the same time frame ($r = 0.58$, $p - value < 0.1$; Figure 4.2b). A higher proportion of dry wells in areas with higher groundwater extraction is a clear indication that the occurrence of missing data is not random, as it would be if driven solely by human error. This analysis highlights that missing data in well records carry critical information on groundwater stress due to climate and/or anthropogenic factors that is completely missed in long term trend analyses that routinely filters out this information. Here lies the artefact of survivor bias – the wells that survive the filtering process are wells with the smallest percent of missing data, and are thus preferentially wells at deeper depths, or in pockets of stagnant water in a highly heterogeneous aquifer system. The lack of

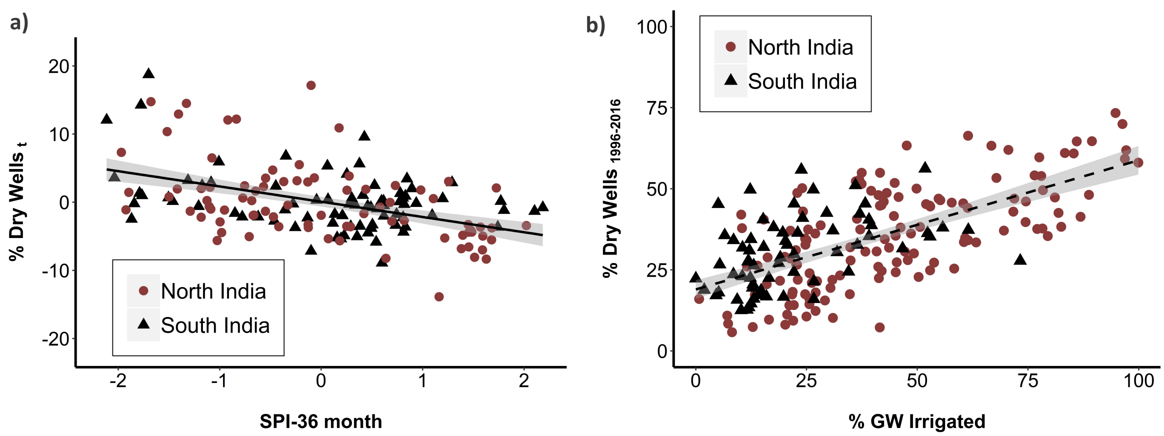

Figure 4.2: Dependence of dry well density on rainfall and groundwater irrigated area: a) Percentage of dry monitoring wells (after removing trend) aggregated at the NI and SI scale (referred to as $\%DryWells_t$) versus the 36-month standardized precipitation index (SPI) for each month between 1996-2016. SPI-values and percent dry wells were estimated for the each of the four months of available groundwater level data (January, May, August and November). b) Percentage of dry monitoring wells at the district scale, aggregated over the 1996-2016 timeframe (referred to as $\%DryWells_{1996-2000}$), versus the percentage of groundwater irrigation in the district. The %dry well aggregation was done by taking the median %dry well value between 1996-2016. The percent groundwater irrigation represents the median percent groundwater irrigation in each district between 2005-2010. Note that only districts where at least 20% of the gross cropped area is irrigated are shown. Figure highlights the dependence of %dry wells on natural (precipitation) and anthropogenic (groundwater irrigation) factors.

trend we see in most existing groundwater well data, despite reported increasing stress on groundwater resources is most likely due to such survivor bias. Thus, the validity of trends using only wells with continuous long-term records as an indicator of groundwater stress in low-storage, hard rock aquifers needs to be questioned.

4.3.3 Alternative Metrics of Groundwater Stress

Given the uncertainty associated with long-term water level trends in hard-rock systems, there is a need for metrics that can measure groundwater sustainability in terms of short-term reliability (Fishman et al., 2011). Based on the above analysis, we propose two metrics: defunct wells and dry wells. We designate monitoring wells as defunct if they stop collecting data permanently within the 1996-2016 timeframe, and the % defunct wells in 2016 is estimated as the proportion of wells that started collecting in 1996 that stopped collecting data permanently before 2016. The %dry well is the proportion of wells that started collecting in 1996 that have missing data in the time step of consideration. While other studies have explored trends in groundwater levels, no study so far has explored the trend in the inactivity of the monitoring wells. We find a steady increase in the % defunct and dry wells in both NI and SI highlighting the increased stress in groundwater (Figure 4.3 and Figure B.9). On average, we see an increase in dry wells at a rate of 104 wells/year in SI, with higher percentages in the drought period of 2002-2003 (Figure B.6). Furthermore, we found that nearly 93% of the defunct wells (with at least 8 years of data) had either a significant negative trend or no long-term trend. This supports the validity of these metrics as an appropriate indicator for evaluating groundwater depletion.

Spatially, dry and defunct wells were found to be uniformly distributed in north and south India, but absent in Central India (Figure 3). The distribution of hotspots (areas with >50% dry and defunct wells) in NI correlates well with declining water levels in the monitoring wells, as expected (Figure 4.3a-b). In contrast, there is a significant density of these hotspots in SI despite increasing or stable water level trends in monitoring wells

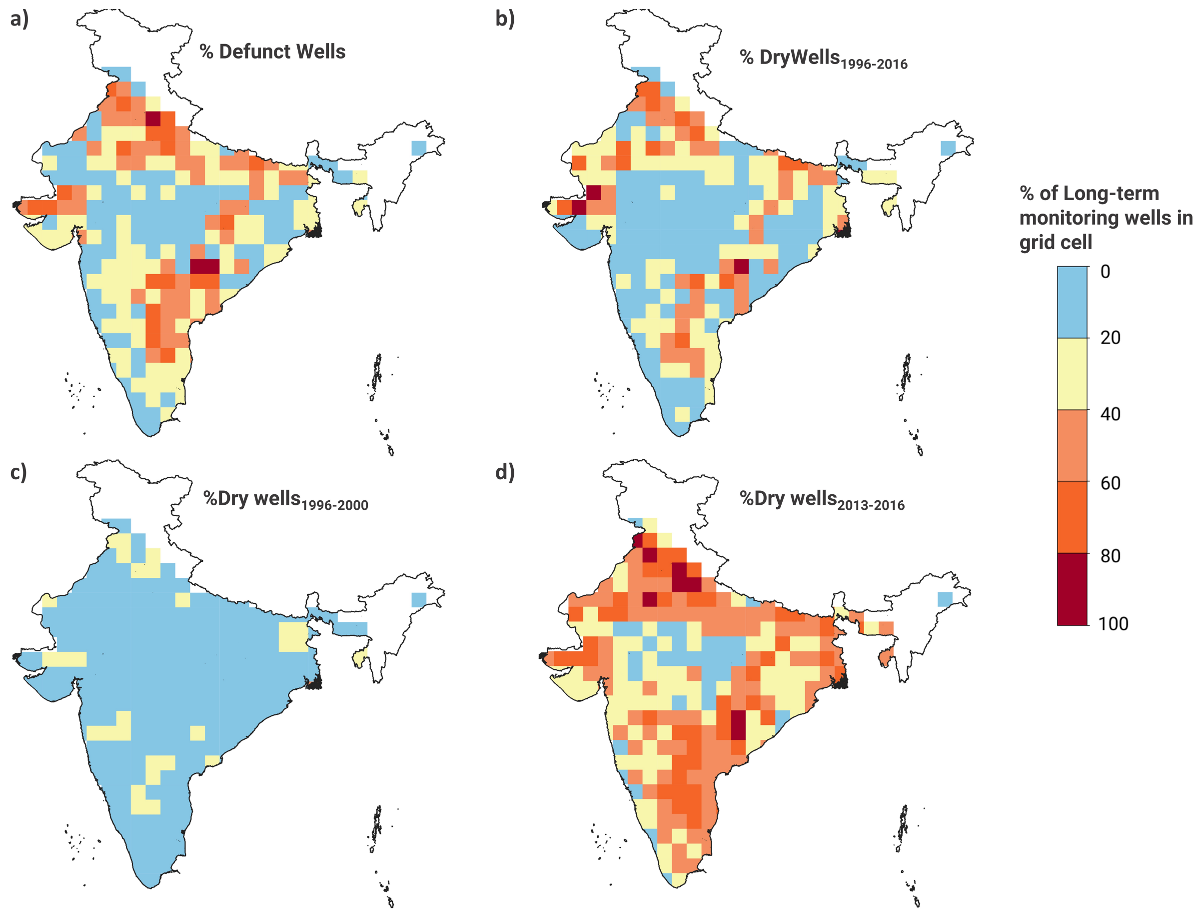

Figure 4.3: Spatial patterns of dry and defunct wells: a) Percent of defunct wells in 2016, calculated as the ratio of wells in grid cells (1° by 1°) that permanently stopped collecting data to all the long-term monitoring wells present. b) Median percentage of dry monitoring wells between 1996-2016 in each grid cell. c) Median percentage of dry monitoring wells between 1996-2000. d) Median percentage of dry monitoring wells between 2013-2016. The % dry wells was calculated as a ratio of wells in grid cells (1° by 1°) that did not record data in a time period to all the long-term monitoring wells present there. Higher percentage of dry/defunct wells (orange/red) indicates relatively more groundwater depletion, and lower percentage of dry/defunct wells (yellow/blue) indicates relatively less groundwater depletion. Please note that long-term monitoring wells in our analysis refers to monitoring wells active in 1996

(Figure 4.3a-b, and Figure 4.1e), that is an artefact of survivor bias. These hotspots in SI also correspond to areas where the proportion of shallow well irrigated area has been decreasing (Figure B.2), and where other local studies report occurrences of water stress (Anantha, 2013; Perrin et al., 2011a). These results corroborate our initial claim that exploring patterns in monitoring wells going defunct and dry wells may provide critical information on the degree of groundwater stress in a region.

4.3.4 Summary and Implications

Our study was motivated by the need to answer why farmer distress is increasing in regions experiencing apparent groundwater recovery. Specifically, analysis of both GRACE and groundwater monitoring data highlight large areas in South India with stable or increasing water level trends (Asoka et al., 2017; Bhanja et al., 2017), which is at odds with findings from social survey data that include reports of well failure and decreases in land area irrigated from shallow wells. Our results highlight that this discrepancy arises due to the problem of survivor bias, where wells with too much missing data are routinely excluded from long-term trend studies. We find that aquifer dryness can manifest itself as data gaps in monitoring well records; thus, these wells carry critical information on the degree of groundwater stress in a region. Given that monitoring well data often underpins government reports and modelling studies, any regional assessment of long-term trends in groundwater systems relying on this information might be prone to such survivor bias. While satellite-based data products can help overcome such biases, the current spatial resolution makes it difficult to highlight local hotspots in heterogeneous aquifer systems.

We argue that in hardrock aquifer systems with large precipitation-driven inter-annual variability and significant spatial heterogeneity, water level trends are not an adequate metric to measure groundwater stress. We provide two metrics, % dry and defunct wells, derived from data gaps in monitoring wells, that might be better suited for assessing groundwater stress in these regions. In South India, both the % of defunct and dry wells have been increasing over time, and the hotspots identified by them correspond well with areas that have local reports of well failures. Correct interpretation of monitoring well data shows that SI is indeed facing significant groundwater stress, and thus requires improved regulations to help meet local demands for groundwater in a more sustainable manner. Interventions promoting the conjunctive use of surface water from rain-water harvesting structures and groundwater (Siderius et al., 2015), along with policies that change the current structure of agricultural power subsidies (Shah et al., 2012) have the potential to improve the situation in SI. Some practical implications of this analysis are that data collection agencies should prioritize collecting information on these dry/defunct wells, while making available details on why data was not collected from a monitoring well at any time step (using flags in the database).

Groundwater storage and its linkage to socio-environmental demands in India is a complex, inter-disciplinary issue. As we improve our ability to monitor groundwater systems, the interpretation of the data collected and its linkage to on-the-ground reality remains important in developing sustainable groundwater management practices. Government agencies have tended to argue that monitoring data are the only reliable source of data and farmer surveys that rely on recall are unreliable. By triangulating across different official datasets at the national scale, our study uncovers methodological limitations in conventional long-term trend analysis of groundwater levels. Though the study is focused on the Indian subcontinent, our finding related to the analysis of long term trends in groundwater levels is applicable to hard-rock aquifers that underlie substantial areas in arid and semi-arid regions of the world, including the Arabian-Nubian shield (Sultan et al., 2008) and parts of West Africa (Lapworth et al., 2013).

Chapter 5

Examining rainwater harvesting structures in groundwater intensive irrigation systems

Agricultural rain-water harvesting (RWH) structures such as tanks remain a promising intervention for improving water availability for smallholder farmers in arid and semi-arid regions of the world. However, the feedbacks between RWH systems and the surrounding aquifer remains poorly understood especially in regions like Southern India where these structures are nestled within a landscape with intensive groundwater development. Here, we develop a conceptual hydrological model to answer fundamental questions about how RWH structures impact groundwater availability for irrigation, and in turn how groundwater irrigation impacts the outflow fluxes from RWH structures. Our results show that agricultural RWH structures can increase groundwater availability in the surrounding area. However, this impact is meaningful (in meeting of agricultural water demands) under only a narrow spectrum of landscape and climate conditions. Specifically, the impact of tanks declines significantly during drought spells or when the beneficiaries of tank-induced groundwater recharge are poorly regulated. Alternatively, groundwater irrigation in the surrounding aquifer positively impacts the efficiency output fluxes from the RWH structure by reducing annual evapotranspiration losses and increasing groundwater recharge of the stored water, however, this comes at the cost of reduced water released from RWH structures for surface irrigation. Our results provide important insights into understanding the potential of RWH structures in contemporary smallholder dominated agricultural systems.

5.1 Introduction

A significant proportion of the world's population is currently estimated to live in water-scarce regions, where the water demands exceed supply (Mekonnen and Hoekstra, 2016; Brauman et al., 2016; Oki and Kanae, 2006). Water scarcity has been shown to increase risks related to poverty, human health, energy supply, and food production (Falkenmark, 2013; Hanjra and Qureshi, 2010). The effects of climate change, population increase, and dietary changes are expected to further exacerbate the stress on global water resources (Foley et al., 2011; Giordano et al., 2019). The agricultural sector remains by far the largest user of freshwater in the world (Siebert et al., 2010; Foley et al., 2011), and agricultural water withdrawals are exceeding environmental limits in many parts of the world. However, increasing water availability for farmers is still seen as a key strategy for increasing agricultural productivity to meet growing societal demands (Molden et al., 2010). Studies estimate that an additional 5600 km^3 of water will be required to sustain food production in 2050 compared to 6800 km^3 of water currently being used globally (Hanjra and Qureshi, 2010). An inability to meet crop water requirements directly impacts crop yields and increases the chances of crop failure. Smallholder farmers (operating on <2 ha area) in semi-arid and arid landscapes with unreliable and low rainfall, such as those found in South Asia and Sub-Saharan Africa, are considered to be particularly vulnerable to the effects of water scarcity (Ngigi, 2003; Burney and Naylor, 2012; Pande and Savenije, 2016). There are currently 450-500 million smallholder farmers worldwide constituting 85% of the world farms (Lowder et al., 2016; Harvey et al., 2014). These farms are generally characterized by low productivity and the farmers operating them face multiple forms of poverty traps (Hanjra et al., 2009; Burney and Naylor, 2012)

Rainwater harvesting (RWH) techniques are commonly seen as an effective way to increase agricultural productivity of smallholder farms in semi-arid and arid landscapes (Burney and Naylor, 2012; Rockström and Falkenmark, 2015). These techniques involve collecting runoff to increase local water storage, thereby helping agricultural systems overcome some of the temporal mismatches between water supply and demands. In rain-fed farms, RWH techniques can lead to large increases in water productivity (i.e the net benefit per unit of water) of farms (Rockström et al., 2003; Molden et al., 2010; Biazin et al., 2012), and are thus seen as an important intervention to help improve agricultural productivity and farmer livelihoods (Rockström and Falkenmark, 2015). Many researchers currently consider water harvesting as a 'low regret' adaptation to climate change that can help build the resilience of the ecosystem and rural community in regions like India (Carabine et al., 2014; Shanmugasundaram et al., 2017).

In India, farmers have utilized RWH structures called tanks (Figure 5.1) to deal with unpredictable monsoonal rainfall patterns for millennia now (Van Meter et al., 2014). There are currently over 208,000 tank structures in India, most of which are found in the southern part of the country (Palanisami et al., 2010). Taking advantage of natural depressions in the landscape, these village-level tank structures have been an integral part of the socio-ecological fabric of agricultural communities in the region (Mosse, 2003). The benefits associated with tanks have traditionally extended beyond irrigation water provisioning to other ecosystem services like flood protection, groundwater recharge and local forestry (Bitterman et al., 2016; Palanisami et al., 2010). However, a significant proportion of these tanks are currently considered to be in a state of disrepair, both physically (broken sluice gates, excessive siltation) and institutionally (loss of water sharing and resource management traditions). While the reasons for the decline of tank systems have often been related to the socio-political goals of the State (Mosse, 2003), most recently, the declining condition of these ancient tank structures has been attributed to the rise of well irrigation, which is seen as a more effective form of irrigation by farmers (Gunnell and Krishnamurthy, 2003). The share of well-irrigated area increased from 30% to 61% between 1960-2001 in India, while tank irrigated reduced from 18% to 5% in the country (Palanisami et al., 2010).

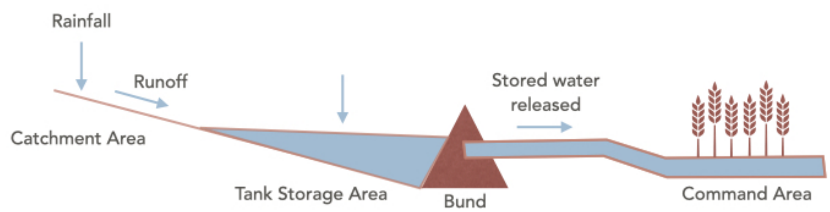

Figure 5.1: Components of a typical tank structure, where runoff from a catchment area is collected behind a bund (earthen dam) and the stored water is released to a command area downstream.

Recently there has been a renewed interest in the revival and construction of tanks in India. The negative consequences of unregulated groundwater extraction in large parts of the country (Rodell et al., 2009; Cullet, 2014; Hora et al., 2019) combined with the growing threat of climate change to smallholder farmers (Palanisami et al., 2010) has resulted in increasing amounts funds being allocated to revive these RWH structures by government agencies, local NGO's and international development agencies (Sakthivadivel, 2007; Shah, 2003). Tanks and other rain-water harvesting structures are widely seen as an ecologically sustainable and socially equitable alternative to large water storage options (Mosse, 2003). Moreover, tanks are also viewed as an appealing option given recent paradigms related to the decentralization in water management from the State to local farmer groups, and are considered important in transitioning back towards more traditional and sustainable forms of resource management (Mosse, 2003; Shah, 2003).

However, it is unclear whether these efforts are yielding substantial benefits in contemporary Indian agricultural systems. Some studies have found that reviving tanks have improved agricultural revenues (Palanisami and Meinzen-Dick, 2001; Reddy et al., 2018), water productivity (Siderius et al., 2015) and farmer livelihoods (Reddy and Behera, 2009). Others have shown that the downstream externalities of tanks outweigh their positive impacts (Bouma et al., 2011), while the benefits of tanks are often seen to be locally concentrated (Boisson et al., 2015) potentially increasing inequalities prevalent in farming communities (Batchelor et al., 2002). Others have argued about the incompatibility of tank structures given the current agricultural and hydrological setup of India (Kumar et al., 2006). Given the uncertainty around tank structures in academic and policy circles, there is a need to better understand the compatibility between the benefits associated with these structures and the needs of the smallholder farming systems. The uncertainty around the influence of tank structures in contemporary Indian agroecosystems is further compounded by a scarcity of available data related to these structures. Field-data associated with tanks has been challenging to obtain due to the high costs involved, high heterogeneity in both the management and physical characteristics of tanks, and difficulty in quantifying the tank hydrologic fluxes (Glendenning and Vervoort, 2011; Van Meter et al., 2016). However, despite the large uncertainties regarding the usefulness of tank structures in current Indian conditions, water managers and policy-makers are being forced to make decisions without a sufficient analysis of the hydrological trade-offs associated with these structures.

Hydrological modelling offers a relatively inexpensive method to extend our ability to use limited data to improve our understanding of these systems, while serving as tools to link science and management decisions (Glendenning et al., 2012). However, most previous modelling efforts either inadequately conceptualize these tank systems or rely on data available only in heavily instrumented systems. Tanks have most commonly been modelled as surface irrigation structures with minimal interactions with the subsurface aquifers (Jayatilaka et al., 2003; Calder et al., 2008; Pandey et al., 2011; Li and Gowing, 2005). These models fail to incorporate the dynamics of the surrounding shallow aquifer, and the only interaction the tanks have with the subsurface is via loss of tank water to the aquifer as groundwater recharge. A second group of less commonly used models assume tanks to act as groundwater recharge structures, and model the surrounding shallow aquifer, but the role of tanks as surface irrigation structures is not represented in these models (Glendenning and Vervoort, 2011; Garg et al., 2012). This limits the use of these models in regions where there is both significant tank and well irrigation. For example, in Tamil Nadu, surface water irrigation from tanks is still responsible for 20% of the total irrigated area in the state (Palanisami et al., 2010). Additionally, the open-source Soil and Water Assessment Tool (SWAT), with its ability to simulate surface, sub-subsurface and management processes, has been commonly used to model RWH systems around the world (Ouessar et al., 2009; Ferrant et al., 2014; Wambura et al., 2018). However, the representation of groundwater processes (especially related to water table dynamics and groundwater-surface water interactions) in SWAT is considered to be inadequate (Melaku and Wang, 2019), while its integration with groundwater-specific modelling frameworks like MODFLOW is considered to over-parameterized and data-intensive for some RWH dominant regions (Glendenning and Vervoort, 2011).

The goal of this paper is to develop a conceptual modelling framework using existing data to improve our understanding of how RWH structures (tanks) function in contemporary groundwater-dependent agricultural systems. Using the Southern Indian smallholder agricultural system as a backdrop, we aim to understand the interplay between groundwater extraction, rainfall and tank water storage, and assess how the dynamics between RWH and groundwater storage impacts water availability for farmers. Specifically, we look to address the following research questions:

1. How does the presence of tanks affect the groundwater storage available for extraction? And, how does this impact vary under different climate conditions?

2. How do the number of users of tank-induced groundwater recharge water affect the overall benefits from tanks (in terms of crop water requirements met)?

3. How are seasonal outflow dynamics of tanks impacted by intensive groundwater extraction for irrigation?

5.2 Methods

5.2.1 General Description

Our modelling approach falls in the recently described category of 'stylized' hydrological models (e.g. Srinivasan and Kulkarni, 2014; Fishman et al., 2011), where the goal is to not develop a model capable of making precise predictions at a given location, but instead to capture the key hydrological processes, generate hypotheses and replicate generally observed dynamics. The model landscape represents a village-level tank-dominated agroecosystem found commonly in Southern India. These semi-arid regions are underlain by low-storage hard-rock aquifers with large inter-annual and seasonal water table fluctuations (Fishman et al., 2011). The cropland in these regions has been traditionally separated into tank-irrigated and rain-fed areas. However, with the availability of well technology, groundwater irrigation is now found in both the tank irrigated and rain-fed areas in many of these regions (Sato and Duraiyappan, 2011). The expansion of groundwater irrigation has gradually caused groundwater depletion in the shallow aquifer, while the tanks themselves are considered to be in poor condition due to siltation, broken sluices gates and poor management (Sato and Duraiyappan, 2011; Van Meter et al., 2016).

The model developed in this study focuses on capturing the daily interactions between tank water and the tank-influenced cropping area (i.e. the area benefiting directly from tank water storage). Tank systems have traditionally been studied in relation to the region that has access to the sluice released from tanks (i.e. surface command area). Since the influence of tanks extends beyond just surface water provisioning, we extend the concept of surface command area by linking tanks to their groundwater spread area. The groundwater spread area corresponds to the region in the landscape that is influenced by tank recharge, and we assume that it can extend to an area greater than the surface command area (Figure 5.2). We differentiate the groundwater spread area into a region that has access to both the surface discharge and groundwater recharge from the tank (CA_{sw-gw}), and a region that only has access to groundwater recharge from the tank (CA_{gw}). In each of these regions, our model simulates daily fluctuations in soil moisture separately with percolation from each region feeding into a common shallow aquifer layer spanning the groundwater spread area.

While the model has been conceptualized in an idealized landscape mimicking Southern Indian agricultural systems, we also incorporate location-specific data for a single RWH tank system in Tamil Nadu, India, where field data was collected during the 2013 monsoon season (Steiff, 2016). Data included daily tank-water fluxes like evapotransipiration, recharge and surface discharge (for the 2013 season), stage-volume and stage-discharge relationships, and data on tank storage, catchment area and command areas. Using this measured data allowed us to parameterize the tank system using field data, while enabling us to measure the ability of the modelled daily tank water balance to reproduce measured tank water fluxes. In contrast, data on the subsurface systems was generally unavailable in these hard-rock regions. Therefore, we instead rely on insights from field studies carried out in representative hard-rock systems in Andhra Pradesh to conceptualize the subsurface component of the model (Dewandel et al., 2008; Nicolas, 2019). We assumed that the sub-surface is composed of a clay-rich soil layer (0-1 mbgl) and a weathered hard-rock shallow aquifer (extending to 20mbgl) with a confining layer underneath. The hydraulic properties in these hard-rock aquifer systems shows significant vertical and horizontal heterogeneity with flow driven through preferential flow paths. Past studies reporting a radius of influence ranging between 100-1000m from tanks in these hard-rock systems (Metha and Jain, 1997; Massuel et al., 2014; Sethi et al., 2020). The area of tank recharge influence is a modelled output that can be considered to be linked to aspects like aquifer type, tank-condition and down-stream extraction intensity. However, given a lack data and high uncertainty about this in tank systems, the groundwater spread area (measured in discrete units of surface command area) was considered to be fixed in our modelled system and a simulation to explore its impact was developed for assessing tank-aquifer dynamics.

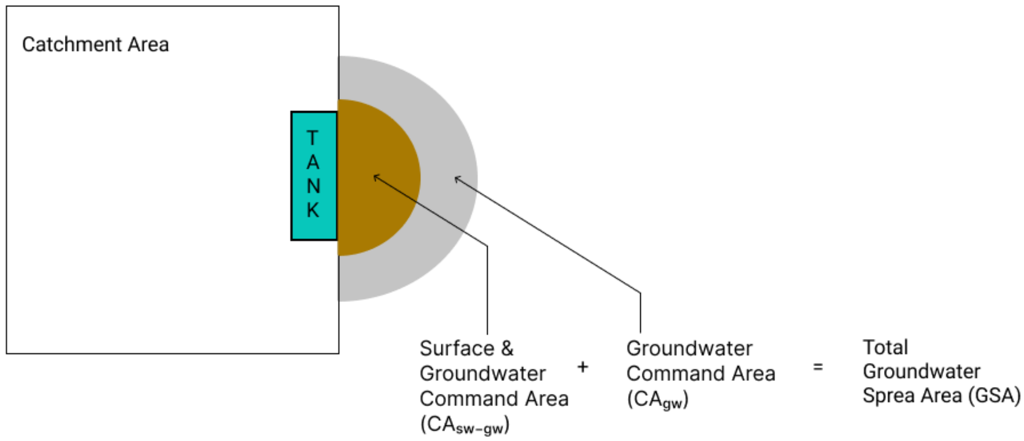

Figure 5.2: Plan view showing the different components of a tank system. The groundwater spread area (GSA) is a region assumed to be linked to tank-based recharge and can be differentiated into two regions: (1) area that has access to the sluice released and groundwater recharged by tanks (brown); (2) region that has access to tank-based groundwater recharge (grey).

5.2.2 Simulation of Tank Water Balance

A water balance at the daily-scale was used to simulate the water storage dynamics in tanks based on the framework developed by Pandey et al. (2011). The tank model was assumed to be triggered following a runoff-generating rainfall event that leads to water storage. The daily tank water balance was given by:

$$\frac{dy_r}{dt} = RO + P - ET_{tank} - GW_{ex} - Sl - Ov \tag{5.1}$$

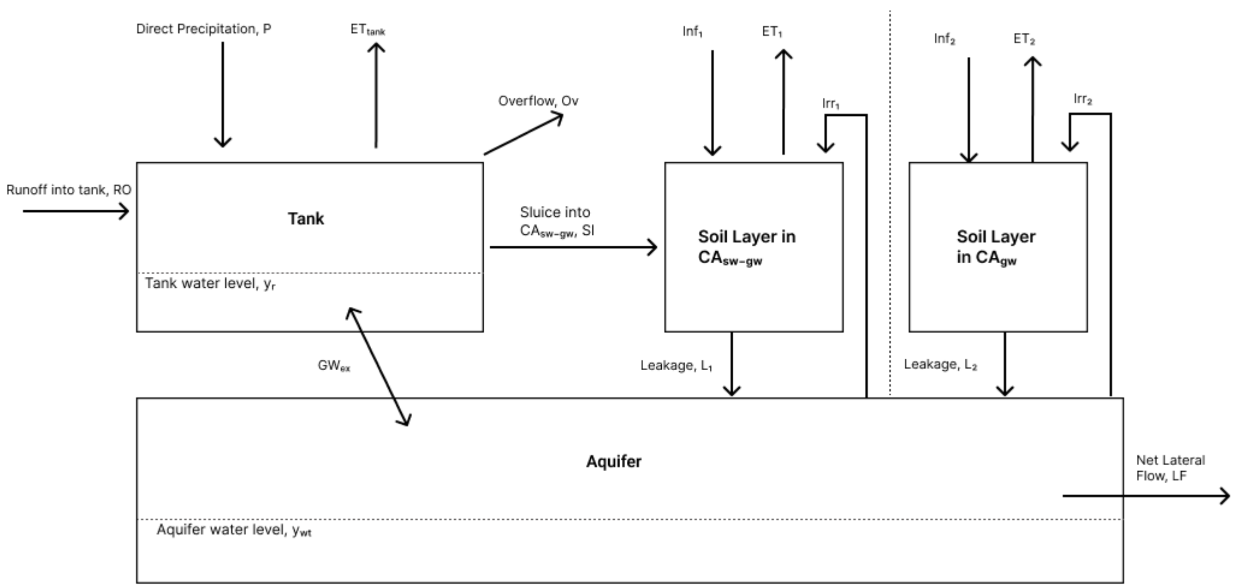

Figure 5.3: Schematic of the framework developed in this study highlighting the different components and processes of the modelled tank system.

where y_r is elevation of the water level in the tank (m); RO is the inflow into the tank structure due to runoff generation in the catchment; P direct precipitation on the tank structure (m/d); ET_{tank} is the water loss through evapotranspiration (m/d); $GW_{ex}(y_r, y_{wt})$ is the groundwater exchange between the tank and the surrounding aquifer; Sl is the sluice released (m/d) from the tank into the surface command area (Figure 5.2); Ov is the spillage from the tank when the tank water storage exceeds the maximum storage of the tank structure under consideration (m/d).

Stage-volume and stage-area relationships were estimated using power relationship following previously derived bathymetric relationships in tank systems (Liebe et al., 2005; Steiff, 2016; Vanthof and Kelly, 2019), with: $Volume = \alpha_1 * Area^{\beta_1}$ and $Volume = \alpha_2 * Stage^{\beta_2}$, where α and β are tank specific constants dependent on geometry. We used previously determined area-volume and volume-stage relationships developed by Steiff (2016) based on field data collected during the 2013 monsoon season for our simulations. These have been highlighted in Table 5.1.

Runoff generation into the tank was estimated using the curve number method developed by the USDA-Soil Conservation Service (SCS). We used a single curve number (CN) to represent the entire basin, while adjusting its value based on the antecedent moisture conditions (Table C.3 and C.4) (Dingman, 2015; Hawkins et al., 1985). While the maximum catchment area of the tank was fixed based on field measurement (Table 5.1), the catchment area for runoff generation into the tank at any timestep was estimated by subtracting the tank water spread area from the maximum catchment area.

ET_{tank} was assumed equal to the potential evapotranspiration rate, PET. Daily PET values were estimated using the the temperature-based PET estimation method developed by Oudin et al. (2005) using the 'airGR' R package (Coron et al., 2017). This method has been shown to provide satisfactory results in catchment-scale daily rainfall-runoff models, while having greater computational efficiency and lower data requirements.

The $GW_{ex}(y_r, y_{wt})$ was estimated by adapting the hydrodynamic function developed by

Massuel et al. (2014) that takes into account the impact of hydraulic connection between the tank and surrounding aquifer. The tank-aquifer interaction was modelled for 3 scenarios when:

1. the tank water level is hydraulically connected to the aquifer. Following the analysis by Bouwer (2002), we assumed that the tank was hydraulically connected to the water table (y_{wt}) when the water level in the aquifer was less than 1m below the tank bed (z_{tb}) (assuming an 1m capillary fringe). In this scenario, the groundwater exchange was primarily controlled by the difference in hydraulic head between the water table and tank water surface. We assumed the maximum infiltration rate ($Inf_{tank\,bed}$) to be at the point of hydraulic connection (i.e. aquifer water level = $|z_{tb}| + 1$) and when the tank is at maximum storage. This infiltration rate was then assumed to decrease linearly based on the hydraulic head difference between the water table and tank water surface such that the recharge rate is equal to 0 when the water table is at the same elevation as the tank water surface. The recharge rate was estimated as:

$$GW_{ex}(y_r, y_{wt}) = \left(\frac{Inf_{tank\,bed}}{|z_{tb}| + 1}\right)(y_r - y_{wt}) \tag{5.2}$$

   where, y_{wt} is the water table elevation, y_r is the tank water surface elevation, $Inf_{tank\,bed}$ is the infiltration rate of the tank bed, and z_{tb} is the depth of the tank bed. $\left(\frac{Inf_{tank\,bed}}{|z_{tb}|+1}\right)$ can be considered to be an effective resistor parameter that is related to clogging effects and hydraulic conductivity of the surrounding aquifer.

2. the tank is hydrologically disconnected from the aquifer. When the difference in elevation between the tank bed (z_{tb}) and the aquifer water level was greater than 1m, the groundwater exchange was assumed to be primarily controlled by the hydraulic conductivity of the tank bed rather than groundwater levels. Under this scenario, the recharge was set to be a constant infiltration rate through the tank bed with $GW_{ex} = Inf_{tank\,bed}\,(m/d)$. Infiltration rates in hydrologically disconnected systems

have been shown to increase linearly with water depth in the tank (or basin), however, this effect has been shown to variable in systems with extensive clogging due to compaction (Bouwer, 2002). Due to the lack information on these effects in the modelled system, the assumed constant infiltration rate in this scenario should be considered to be an average infiltration rate.

3. dry period surge. Following a prolonged dry period, we assumed that the clay-rich clogging layer in the tank bed develops cracks that greatly increase the infiltration capacity of the tank bed (Pathak et al., 2013). Under this scenario, we set the recharge rates equal to the maximum measured infiltration rate (Van Meter et al., 2016) with $GW_{ex} = 0.17\,(m/d)$. This was triggered during runoff-generating rainfall events when the water storage in the tank was equal to zero.

We used the stage-sluice relationship developed by Steiff (2016) based on field data collected during the 2013 monsoon season to estimate the sluice release from the tank. These relationships have been highlighted in Table 5.1. Based on field observations, we assumed that the sluice gates are left open at all times due to a combination of broken sluice gates and poor tank water management (Van Meter et al., 2016). Additionally, it was observed that once the water levels in the tank fall below the sluice invert elevation (m/d), farmers in the system extract this dead storage using pumps. Based on the dead-storage extraction rates measured by (Steiff, 2016), we assumed that water was pumped directly out of the tank at a daily rate of 12.6 m^3 when the tank water level fell below the sluice invert elevation. This extracted volume was added as sluice into the tank command area with access to surface water (CA_{sw-gw}).

Finally, as tanks are often connected in cascades (Van Meter et al., 2016), the spillage Ov from the tank is designed to feed into a downstream tank. However, since our current modelling effort focuses on a single tank structure, this volume was considered to be lost from the system.

Table 5.1: Tank constants based on field measurement by Steiff (2016)

Catchment Area	5 km^2
Surface Command Area	0.27 km^2
Max. Tank stage	3.2 m
Max. Tank Storage	221382 m^3

5.2.3 Simulation of Water Table Dynamics

We simulated daily water table elevation changes in the tank groundwater spread area (GSA) which was bounded between the soil layer (z_{sl}) and confining layer (z_{cl}). The daily water level change was simulated using the following water balance equation:

$$sy\,\frac{dy_{wt}}{dt} = Rch - ET_{wt} + GW_{aq,ex} - LF - Irr_{GW} \tag{5.3}$$

where y_{wt} is water table elevation (m); sy is the specific yield of the shallow hard-rock aquifer system; Rch is the recharge reaching the water table based on rainfall rate and irrigation return flow; ET_{wt} is the evapotranspiration rate from the saturated zone (assumed to be 0 as we are not modelling the phreatophyte vegetation found in these systems); $GW_{aq,ex}$ is the change in aquifer storage due to groundwater exchange between the aquifer and tank; LF is the lateral flow out of the modelled aquifer system, and Irr_{GW} is water extracted from groundwater storage using wells (see Section 5.2.5). We assumed vertical flow through the confining layer to be negligible in our system.

Groundwater recharge, Rch, was triggered when the water content in the soil layer overlying the aquifer exceeded field capacity (s_{fc}). Percolation from the soil layer was assumed to be instantaneously redistributed and reach the groundwater table. As the soil moisture dynamics in the surface-groundwater (CA_{sw-gw}) and groundwater only (CA_{gw})

command area were simulated separately (Figure 5.2 and 5.3), Rch into the aquifer was estimated using the combined leakage as:

$$Rch = \frac{L_{CA_{sw-gw}}\, CA_{sw-gw} + L_{CA_{gw}}\, CA_{gw}}{GSA} \tag{5.4}$$

where, Rch is the recharge to the aquifer layer, $L_{CA_{sw-gw}}$ is the leakage (m/day) from the soil layer simulated in the surface-groundwater command area, $L_{CA_{gw}}$ is the leakage (m/day) from the soil layer simulated groundwater only command area (Figure 5.2). The equations used to estimate leakage from the soil are described in Section 5.2.4.

The change in aquifer storage due to water exchange between the tank and aquifer system was estimated by:

$$GW_{aq,ex} = \frac{GW_{ex}\, WSA}{GSA} \tag{5.5}$$

where, GW_{ex}, was estimated using the hydrodynamic function described above (Section 5.2.2); WSA is the wetted surface area of the tank at a given time which we assumed equal to tank water spread area, and GSA is the groundwater spread area of the tank system (Figure 5.2).

Previous studies have shown that lateral outflow across tank-recharged areas can be non-negligible in the aquifer systems found in Southern India (Boisson et al., 2014). However, due to a lack of information on this exchange, we used the exchange rate observed by Boisson et al. (2014), where they found the lateral flow rate to vary between $0.02-0.7mm/d$ in their tank system. The lateral flow rate at any given timestep was estimated as: $LF = LF_{max} * (1 - \frac{y_{wt}}{z_{cl}})$, where LF_{max} is the maximum lateral flow rate and z_{cl} is the assumed aquifer thickness.

Finally, given the preponderance of groundwater irrigation in these systems, we assumed that the entire groundwater spread area is irrigated with well irrigation, and that farmers irrigate during the monsoon and dry season. Irrigation withdrawal (Irr_{GW}) from groundwater storage was considered to be a function of well intensity and cropping period (see Section 5.2.5 for details).

5.2.4 Simulation of Soil Moisture Dynamics

We simulated the water balance of a soil layer of fixed depth at a daily time step separately for the CA_{sw-gw} and CA_{gw} (Figure 5.2 and 5.3). We adapted the soil moisture model developed by Laio et al. (2001) for the tank system by including an irrigation component. The water balance equation for the soil layer can be expressed as:

$$AWS * \frac{ds}{dt} = IF - ET_s - L + Irr_{GW} + Sl \tag{5.6}$$

where, AWS is the available water storage in the soil layer equal to the product of soil porosity and depth of soil (i.e. $AWS = \phi * y_{sl}$), s is soil saturation ($0 \geq s \geq 1$), IF is the infiltration water into the soil layer following a rainfall event; ET_s is the ET flux from the soil based on the soil saturation at a given time and the land-use; L is the leakage from the soil layer to the aquifer based on the soil saturation level at a given time; and Sl and Irr_{GW} is addition of irrigation water (area-averaged) from surface and sub-surface sources respectively.

In the situation that the infiltration into the soil layer exceeds AWS, we assumed the water to be lost from the system as runoff. Leakage from the soil was triggered when the soil saturation exceeds field capacity (s_{fc}) at any timestep. Following Laio et al. (2001), the leakage at any timestep was given by:

$$L = \begin{cases} \frac{K_{s,soil}}{e^{\beta*(1-s_{fc})-1}} * \left[e^{\beta*(s-s_{fc})-1}\right] & \text{if } s > s_{fc} \\ 0 & \text{if } s < s_{fc} \end{cases} \tag{5.7}$$

where, $K_{s,soil}$ is the saturated hydraulic conductivity of soil; $\beta = 2b + 4$, where b is pore size distribution index and dependent on soil type. For the two simulated soil layers, the combined leakage was estimated based on equation 5.4.

The irrigation added to the soil layer differed between CA_{sw-gw} and CA_{gw}. The soil layer in CA_{sw-gw} had access to sluice discharge (Sl), while in CA_{gw} this input was set to 0. The irrigation input from groundwater sources Irr_{GW} was assumed to be equal for the two modelled soil layers (See section 5.2.5).

The evapotranspiration loss, $ET_s(s)$, from the soil was estimated using a piece-wise linear function, where:

$$ET_s = \begin{cases} ET_c & \text{if } s > s_{fc} \\ ET_c * \left(\frac{s - s_w}{s_{fc} - sw_w}\right) & \text{if } s_w < s < s_{fc} \\ 0 & \text{if } s < s_w \end{cases} \tag{5.8}$$

where, ET_c is the crop water requirements and s_w is the soil saturation at wilting point. ET_c was estimated following the FAO method (Allan et al., 1998) with $ET_c = K_c * ET_o$, where K_c is the crop coefficient (Supplementary C.1) based on the crop growth stage, and ET_o is the potential evapotranspiration rate (mm/day).

5.2.5 Forcing Data, Planting Date and Irrigation

We obtained high-resolution gridded rainfall data from Indian Meteorological Department (IMD) between 1951-2019 (Pai et al., 2014). The precipitation data from the grid point closest to our study site ($77.5^oE, 9.5^oN$) was used to run long-term simulations. We aggregated this daily precipitation data to estimate drought periods using the 12-month Standardized Precipitation Index (SPI-12) (McKee et al., 1993), where periods with SPI-12 less -1 were demarcated as droughts. For temperature data to estimate PET, we obtained daily gridded maximum and minimum temperature data from the IMD between 1951-2018 (Srivastava et al., 2009).

The crop rotation was assumed to be constant throughout the simulation period. Based on field observations made in these systems (Steiff, 2016; Sato and Duraiyappan, 2011), we assumed that the farmers with access to well irrigation in the simulated region plant paddy in the rainy season (*Kharif*; August to January) and sorghum in the dry season (*Rabi*; March to May). While the cropping pattern was fixed, we accounted for the uncertainty of the Indian monsoon by modifying the planting date of the rainy season crop. This involved identifying the onset of the monsoon in any given year, and then adjusting the plant date based on that. We used the method developed by Mathison et al. (2018) to identify the monsoon onsets in the Indian sub-continent. For each water year, assumed to run between August 1-July 31, we estimated the onset of the North-East monsoon by estimating the Normalized Pentad Precipitation Index (NPPI):

$$NPPI = \frac{P - P_{i,min}}{P_{i,max} - P_{i,min}} \tag{5.9}$$

where P is the pentad precipitation estimated by summing the precipitation over a 5-day period, P_{min} and P_{max} are the minimum and maximum non-zero pentad precipitation of the monsoon season in water year i. Following Mathison et al. (2018), we assumed the monsoon onset to be equal to the first pentad in which the $NPPI > 0.618$. Given that farmers with access to irrigation often plant before the first major rainfall event (REF), we adjusted the plant date using a PD parameter. We varied this between 10-30 days, and assessed the sensitivity of the model results to PD. For years where the onset of the monsoon season was significantly delayed (i.e. the major first rainfall event occurred post-November), the plant date was assumed to be October 15. In contrast, the planting date for crops in the dry season was considered to be relatively fixed, and set to March 1 for the entire simulation period.

Following observation by previous studies (Maréchal et al., 2006; Sato and Duraiyappan, 2011) in similar landscapes, we assumed that farmers with access to wells irrigate their crops during both the monsoon and dry season. Further, given our focus on the shallow aquifer system, we assumed that farmers irrigate with dug-wells (or open-wells) with depths equal to z_{cl}, i.e. farmers can pump water from all depths of the shallow aquifer. Daily pumping rates were estimated using the per well regional annual pumping volumes provided by the Central Groundwater Board of India (CGWB,1997). The annual pumping rates can vary significantly based on well type and aquifer system. For Tamil Nadu, the CGWB estimates an average pumping volume of 7000 $m^3.well^{-1}.yr^{-1}$ (CGWB,1997) for dug wells. This translated to a pumping volume of 18 $L.well^{-1}.min^{-1}$, which fell within the range measured by Maréchal et al. (2006) in similar hard-rock aquifer systems. Average irrigated area per well estimates were obtained using the field observations made by Janakrajan and Moench (2006) in similar systems (average irrigated area per well = 1.8 hectares). We used this information on well density to obtain the number of wells in the groundwater spread area under the assumption that the entire groundwater spread area is irrigated with groundwater. Finally, we estimated the daily groundwater extraction volume during the cropping period in the groundwater spread area (GSA) as: $Extraction = \big(WellDensity * GSA * Daily\ Pumping\ Rate\big)$. During the non-cropping period and/or if the groundwater storage was empty, we set $Extraction = 0$. The $Extraction$ volume was divided by the GSA to obtain an area-averaged groundwater irrigation input (Irr_{GW}) into the soil layer.

5.2.6 Parameters, model validation and simulations

The goal of this study was to develop a parsimonious coupled tank–shallow aquifer model able to capture key processes and generate hypothesis related to Southern Indian tank systems. Given a lack of data and our modelling philosophy, the model was not developed to make predictions for any specific site. We utilized region-specific parameter values to characterize the modelled sub-surface system from previous field studies where possible. For example, aquifer characteristics were obtained from the extensive field studies conducted in South Indian hard-rock systems by Maréchal et al. (2006), Dewandel et al. (2008) and Briz-Kishore (1993). For other parameters, we used representative values well-established in the literature, for example, soil hydraulic parameters from Clapp and Hornberger (1978).

Table 5.2: Model parameter for the sub-surface used in the study

Symbol	Name	Value	Source
ϕ_1	Porosity: soil	0.42	Dingman (2015)
ϕ_2	Porosity: hard-rock	0.015	Boisson et al. (2014)
$sy_{aquifer}$	Specific Yield	1.5%	Boisson et al. (2014)
$K_{sat,soil}\ (m/d)$	Soil-layer Hydraulic Conductivity	0.021	Van Meter et al. (2016)
$Inf_{tank\ bed}\ (m/d)$	Tank-bed Infiltration rate	0.024	Van Meter et al. (2016)
$Max\ Inf_{tank\ bed}\ (m/d)$	Dry-period tank-bed Infiltration rate	0.17	Van Meter et al. (2016)
$LF_{max}(mm/d)$	Maximum lateral flow rate	0.3	Boisson et al. (2014)
s_{fc}	Soil Saturation at Field Capacity	0.71	Calculated using Dingman (2015)
s_w	Soil Saturation at Wilting Point	0.41	Calculated using Dingman (2015)
b	Pore Size Distribution	7.12	Dingman (2015)
$z_{sl}\ (m)$	Soil Layer depth	1	-
$z_{cl}\ (m)$	Aquifer Thickness	20	-

Further, we relied on field data collected for a single tank system by Steiff (2016) to characterize our modelled system. Data obtained included tank characteristics like bathymetric relationships and sluice-stage relationships; tank-specific constants like command area size (Table C.2 and 5.1). We also performed a limited sensitivity analysis to understand parameter impacts on model outputs. Our limited sensitivity analysis involved varying parameters (from Table 5.2, GSA and Planting Date) by $\pm 50\%$, and then using the Fourier amplitude sensitivity test (FAST) method to conduct a variance-based, low-budget global sensitivity analysis (Reusser et al., 2011). The sensitivity analysis was performed by running the model over 20-year periods, and assessing how different parameter sets impacted the total volume and ratio of tank sluice discharge and groundwater exchange.

Due to limited data, the exercise of calibrating model parameters to precisely match the available data was not undertaken. Instead, we assessed how well the paramterized model based on literature values was able to capture the observed tank dynamics. An exception to this was the parameter estimate for groundwater spread area (GSA) which we found was poorly constrained in literature. Therefore, we estimated GSA using measured tank dynamics by Steiff (2016) to formulate a baseline value (by varying GSA between 1-10 surface command area units), while developing a separate scenario to understand how changes in GSA impacts the modelled system. Overall, we tested the ability of the model to replicate daily tank water storage over the 2013 monsoon season, and compared field estimated and modelled partitioning of tank water storage into sluice, evapotranspiration and groundwater recharge at the weekly and seasonal timescale. It should be noted that Steiff (2016) adapted the White method to estimate daily groundwater exchange and evapotranspiration losses as a function of measured diurnal change in tank water level, and as such do not represent directly measured fluxes of groundwater exchange or evapotranspiration in the tank. Moreover, the sluice-tank stage relationship utilized in the model was based on field measurements of sluice by Steiff (2016). Thus, while we would expect close correspondence between the measured and modelled discharge volume for a given tank water level, the evolution of sluice discharged over the season could still vary depending on the relative magnitudes of groundwater exchange and evapotranspiration.

Additionally, we also obtained groundwater observation data between 2000-2017 collected by the Central Groundwater Board for monitoring wells in the vicinity of the tank system for which we had field data. This served as an independent data source that was used to validate the ability of the model to capture general groundwater storage patterns observed in the region.

Simulations

Model runs were designed to improve our understanding about the dynamics between tank and groundwater storage (Table 5.3) in a system with intensive groundwater irrigation. Our first set of runs assessed the impact of tanks on the groundwater availability in a region, and involved running simulations where the agricultural system was simulated with and without tanks. We simulated the effect of no tanks by assuming a maximum tank storage equal zero, such that the water that would otherwise flow into the tank exits the watershed for downstream users. System impacts were measured using indicators related to average groundwater table depth in the groundwater spread area, and %crop ET requirements met with and without tanks under different climate conditions (i.e. drought vs non-drought).

In our second simulation set, we assessed how the influence of tanks with respect to groundwater recharge varies as a function of the number of beneficiaries with access to that recharge. This set of runs consisted of two parts. In the first part, we assessed how the influence of tanks varies when the size of the groundwater spread changes in the simulated landscape. With the dominance of preferential flow paths driven by fractures in the hard-rock aquifers found in these systems, previous studies have reported a recharge radius between 100-1000m from tanks (Metha and Jain, 1997; Massuel et al., 2014; Sethi et al., 2020). Therefore, we assume that recharge from a particular tank configuration (with respect to capacity, command/catchment area size) can have varying levels of spread depending on the flow paths present in the system. In the second part, we asked a comple-

Table 5.3: Different simulations used to understand the impact of tanks in our modelled system

	Simulation	Description
1	Impact of tanks	We explore how the presence of tanks affects water availability in the modelled system. Simulations were run with and without (i.e. inflow into the tanks set to 0) tanks, and outcomes were assessed with regards to %crop water requirements met and groundwater table depth
2	Influence of recharge beneficiaries	We assess how the influence of tanks changes in landscapes with varying levels of groundwater spread area (GSA) and well density. These simulations consisted of two parts: a) runs where the groundwater spread area was varied between 1-10 times the measured surface command area ($CA_{sw,gw}$), and the well density was fixed, and b) runs where the number of wells irrigating the groundwater spread was varied between 5-300 wells but where the size of tank groundwater spread area was kept constant.
3	Influence of Groundwater Irrigation	We assess how groundwater irrigation in a tank system impacts the tank outflow dynamics. Simulations were run with and without groundwater irrigation in the modelled system, and outcomes were assessed with regards to the changes in the proportion of tank sluice, groundwater exchange and evapotranspiration volumes

mentary question: how does a changing well density affect the benefits from tanks? These runs aimed to help improve our understanding of the potential increase in groundwater irrigation intensity (which we capture by varying the area irrigated by a well) on crop water requirements met in the system.

Finally, our last set of runs focused on understanding how intense groundwater extraction in the shallow aquifer surrounding the tanks impacts tank water storage and tank output fluxes. For these runs, we assessed the performance of the tank systems using metrics related to the number of days with sluice outflow, and tank inefficiency (measured as the volume of annual evapotranspiration loss) with and without groundwater irrigation. For each set of simulations, the model was run using forcing data over a 54-year time span.

5.3 Results and Discussion

5.3.1 Sensitivity Analysis and Model Validation

We assessed the capability of the model to replicate the field observed patterns of tank storage and tank fluxes from the 2013 monsoon season. Based on our limited sensitivity analysis, we found the model output (with respect to total volumetric outflow from the tanks) to be most sensitive to changes in CN. This is similar to results obtained by (Glendenning and Vervoort, 2011) for water harvesting structures in semi-arid regions of North-Western India. This highlighted the importance of utilizing accurate land-use data in modelling these systems, and the influence upstream regions in the catchment area have on the functionality of these tank structures. We also found that hydraulic conductivity of the tank bed (K_{sat}), shallow aquifer thickness (z_{cl}), and the size of the groundwater basin (GSA) had the greatest influence on the partitioning of tank outflow into sluice and groundwater exchange. These parameters accounted for nearly 60% of the output variance in our sensitivity analysis (Figure C.1).

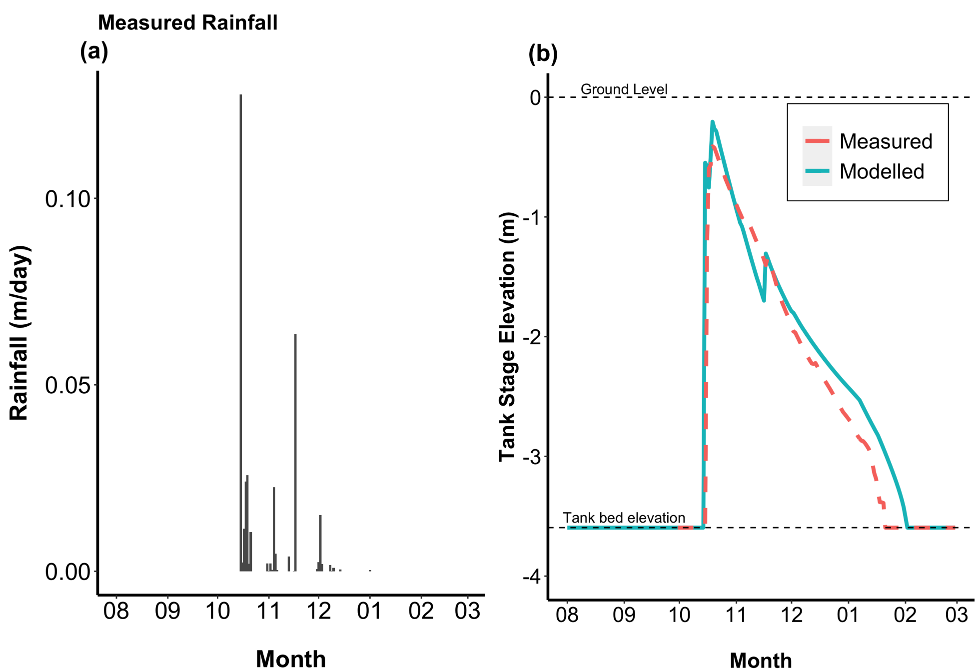

Figure 5.4: Comparing modelled tank stage with measured data from the 2013 monsoon season collected by Steiff (2016) in tank 1 of the Thirumal Samudram cascade system. (a) measured rainfall data from the 2013 monsoon season, (b) measured and modelled daily tank stage data. An elevation of 0m represents groundwater level.

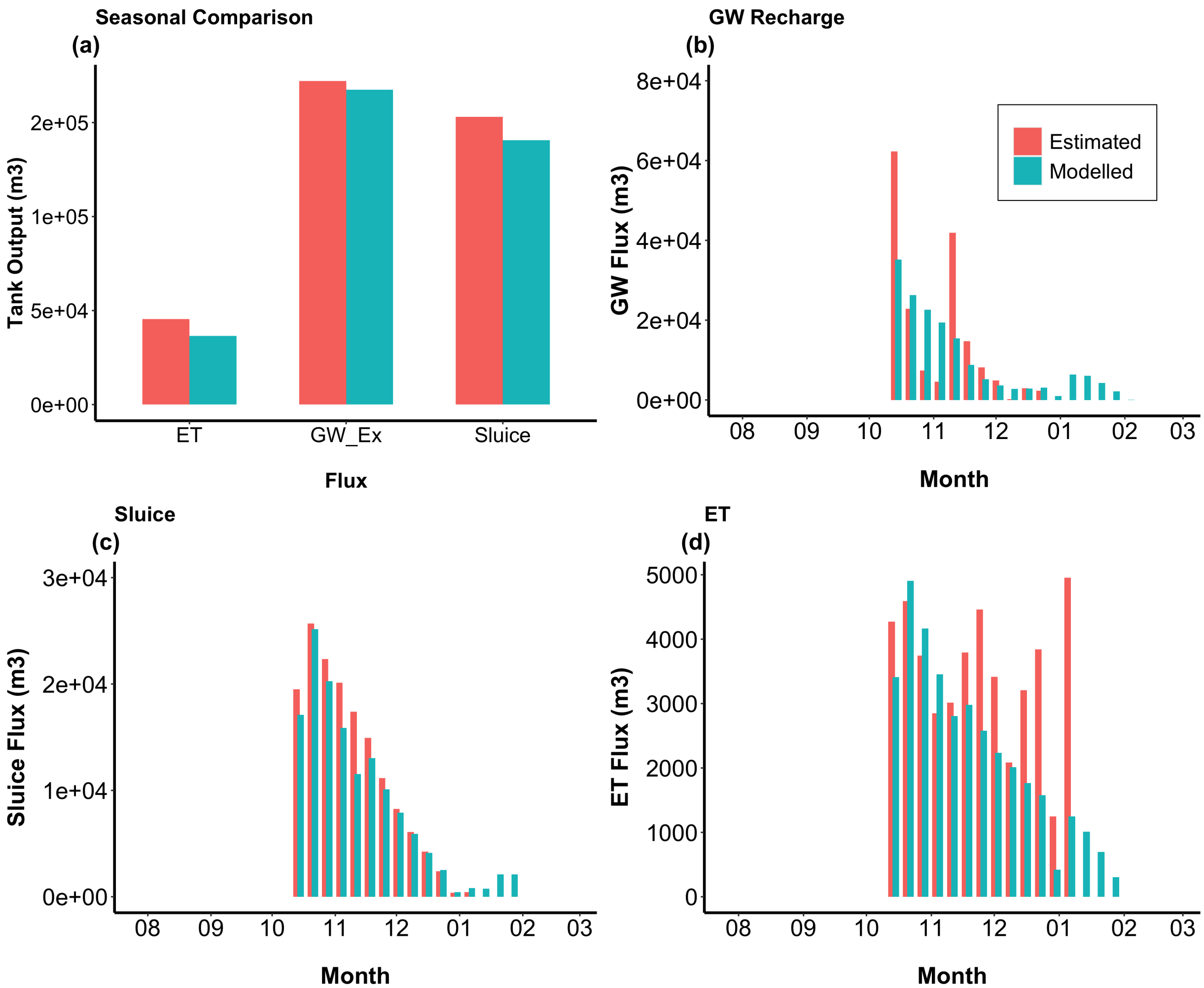

Figure 5.5: Comparing modelled tank dynamics with estimated tank outflow volumes from the 2013 monsoon season collected by Steiff (2016) in tank 1 of the Thirumal Samudram cascade system. The sluice, ET and groundwater exchange volumes were estimated using the White method that relies on diurnal fluctuations in water level (see Van Meter et al. (2016) for more details). (a) Estimated and modelled volumes (m^3) evapotranspiration loss, sluice outflow and groundwater exchange for the entire season; (b-d) comparison of estimated and modelled weekly fluxes of groundwater exchange, sluice outflow and evapotranspiration from the tank

Figure 5.4 shows the results of modelled tank stage at the daily-scale for the 2013 monsoon season. Given the number of model parameters and limited field data to validate tank storage, our results showed that the model was able to replicate the measured daily tank water level relatively easily (Nash-Sutcliffe co-efficient = 0.90; Figure 5.4b). Instead, assessing the partitioning of tank outputs helped better understand the performance of the model in capturing estimated tank output fluxes from field values. Using the tank outflow partitioning, we found that a groundwater spread area of 3 surface command area units (Figure 5.2) best replicated the field tank output fluxes estimated by Van Meter et al. (2016) (Table C.5). Our results showed that the model performed relatively well over the course of the 2013 monsoon season with inflows into the tank (and as a result outflows) being underestimated by 7.0% by the model over the course of the season (Modelled total inflows = 344629m^3; Measured total inflows = 370616 m^3). In terms of individual outflow components, we found that the groundwater recharge, sluice, and ET loss were 2.7%, 8% and 20% less than the outflow volumes estimated by Van Meter et al. (2016) over the course of the season (Figure 5.5a) respectively.

Assessing the model results at a weekly scale showed that the model had difficulty in capturing two major processes estimated from the field data. The first was related to evapotranspiration patterns estimated in the field, where the ET losses did not decrease with declining tank surface area (Figure 5.5d). This can be attributed to the likely influence of invasive vegetation, *Prosopis Juliflora*, in the tank bed (Sato, 2013) that was not included in the current model framework. However, there is also significant uncertainty associated with the methodology used by Van Meter et al. (2016) to estimate ET rates from measured tank water levels, and therefore there is a need for more field studies to better inform how ET is modelled in these systems. The model was also unable to capture focused recharge events measured in the tank system, where nearly 40% of the tank-induced groundwater recharge took place over two days–October 19 and November 17 (Figure 5.5b). Given the limited information on the recharge dynamics of focused recharge events in these systems (Nicolas, 2019), we were unable to incorporate processes to capture recharge from such events in our model. However, our results highlight that the inclusions of processes controlling these focused recharge events are important for understanding tank-induced recharge dynamics.

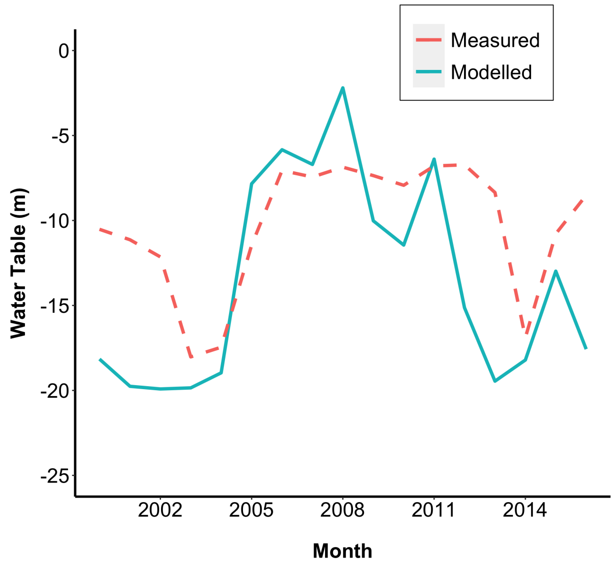

Figure 5.6: Comparison between the annual groundwater table elevation (m) simulated by the model and groundwater level measured by the CGWB between 2000-2017 (Well ID: W02232). Results indicate a statistically significant correlation between measured and modelled groundwater storage.

Additionally, we also assessed the ability of the model to capture measured groundwater storage patterns using the data collected by the CGWB in the region. Figure 5.6 shows a comparison between the time-series of modelled groundwater table elevation and the measured groundwater level data at the annual timescale. We found that the model was able to replicate the measured groundwater storage pattern reasonably well (r = 0.62, $p - value < 0.05$). However, it is worth noting that we did not attempt to precisely match the modelled groundwater storage to the measured groundwater data. This was driven by our conceptual modelling approach and the uncertainty related to groundwater storage measurements in hard-rock aquifer systems (Hora et al., 2019). Overall, given our data constraints, we found that our model performed well enough relative to measured data with regards to both the internal partitioning of tank water storage and the regional groundwater storage dynamics.

5.3.2 Impact of tanks on groundwater storage

To assess the impact of tanks on groundwater availability, we ran model simulations over a 50-yr period under the configuration of groundwater irrigation with and without tank storage. Without tank storage, given current groundwater demands, our simulation shows that the shallow hard-rock aquifer found in the Southern Indian system has a limited ability to support extended groundwater irrigation. We find that the median water table (y_{wt}) is equal to 20 mbgl (i.e. the depth of the confining layer z_{cl}) under these conditions, and the shallow aquifer recovers only during exceptionally wet periods (Figure 5.8a). Thus, our results suggest that in the absence of tanks the infiltration from rainfall alone is unable to overcome the water deficit in the vadose zone to reach the saturated zone in these semi-arid systems. This is similar to the results obtained by Nicolas (2019) who found that diffuse recharge in the Southern Indian hard-rock systems was very low in years with annual rainfall amounts less than 1000mm (average rainfall in our simulated landscape 818mm). Under these conditions of no tank storage, we find that the median daily crop %ET requirements met is 64%[IQR = 35-100%] during the monsoon season in our simulated landscape, and only 24%[IQR = 13-40%] during the dry season (Figure 5.7). This suggests that farmers attempting to cultivate crops during the dry season would have to contend with poor yields and crop failure under these conditions. Looking closer at ET requirements being met during different crop growth stages, we find that the initial and late growing seasons are more water-stressed in both seasons (Figure 5.7). These crop stages generally correspond with the driest periods in these systems (February-March; and June-July), while the early and mid crop growth stage in the monsoon season coincided with the wettest period in the landscape (Figure C.2), and the early/mid-crop stage in the dry growing season benefiting from supplemental rainfall during the Indian South-West monsoon. As expected, these limitations get even more pronounced during multi-year drought spells, where the median daily crop ET requirement met is equal to 22%[IQR = 11-47%], as opposed to 41%[IQR = 21-76%] during non-drought periods. Overall, we find that in the absence of tank structures, most wells tapping the shallow aquifer remain dry in these systems with farmers (despite once having access to wells) essentially practicing rain-fed agriculture.

In comparison, our simulations with tank water storage show that recharge from tanks can have a positive overall effect on groundwater availability in the region. In our simulations with tanks, a higher median groundwater table elevation was observed in the groundwater spread area (Figure 5.8b) which allowed farmers to irrigate using the groundwater resulting in a greater percentage of crop ET requirement being met in the system (Figure 5.7a). We find that the median groundwater table elevation is equal to -13m [IQR = -20m - -6m] in our simulated landscape with tanks. Due to a greater volume of groundwater available for extraction, approximately 88% [IQR = 56-100%] of the daily crop water requirements are met during the monsoon season (38% increase from the no tank scenario) and 48%[IQR = 28-66%] of the daily ET requirements are met during the dry season (101% increase from the no tank scenario) in the groundwater command area of the tank (CA_{gw}). Further, we find that in areas with access to both groundwater and sluice released from tanks (CA_{gw-sw}) an even greater percentage of the daily crop water requirements are met in the monsoon (median: 100%; IQR = 76-100%) and dry seasons (median: 61%; IQR = 34-100%). Overall, our results show that tanks can increase the reliability of crop water requirements met during both growing seasons across the groundwater spread of the tank, while having a higher relative impact on dry season agriculture in the system.

However, despite the increased water availability for irrigation compared to the no-tank scenario, we find the benefits from tanks reduce substantially during drought periods. During these dry spells, 54%[IQR - 28-81%] and 20%[IQR = 11-35%] of the daily crop water requirements are met during the monsoon and dry growing seasons respectively in

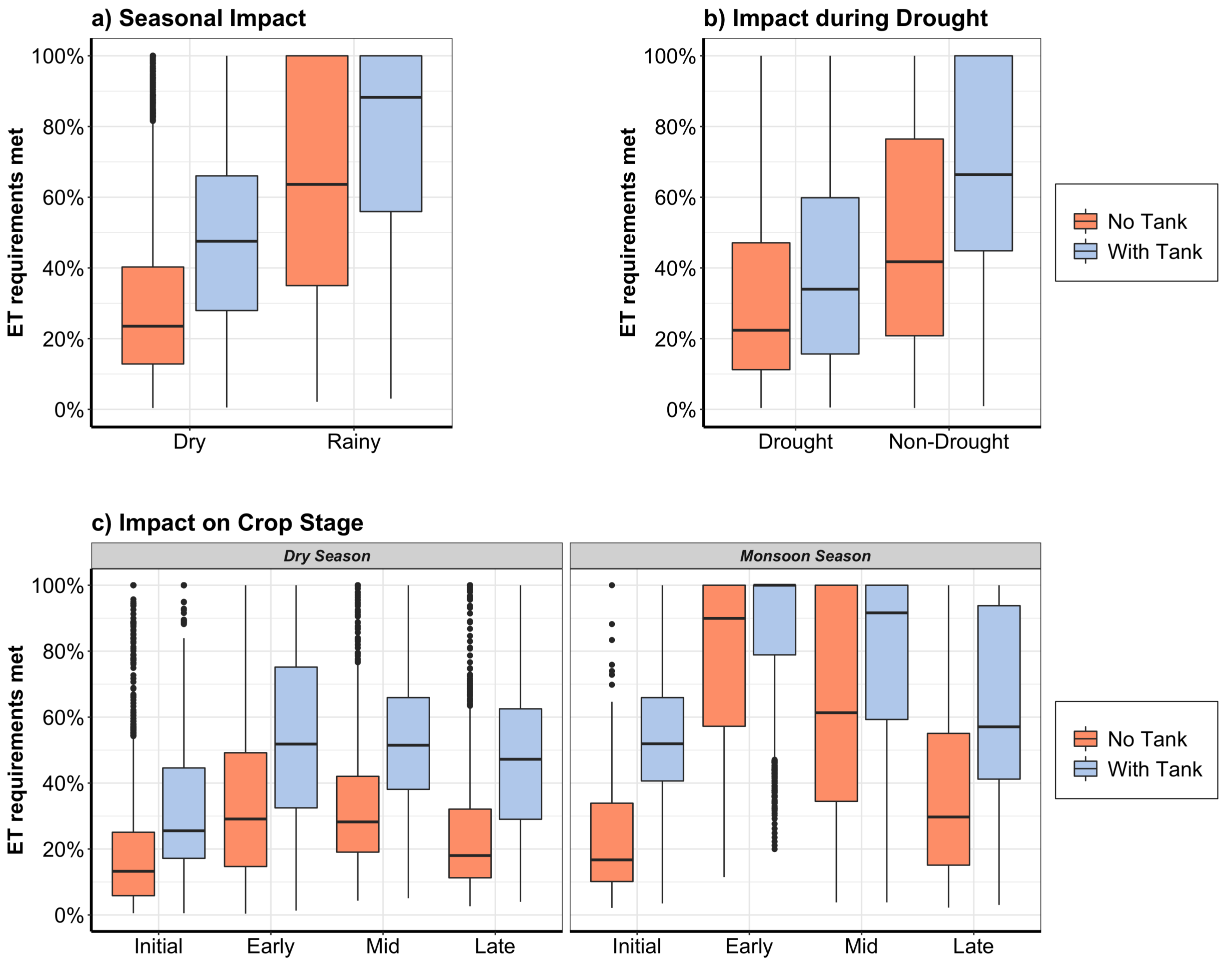

Figure 5.7: The effects of tank water storage on mean monthly %ET requirements met in the *Groundwater Command Area* (CA_{gw}). The crops grown in CA_{gw} have access to only groundwater irrigation. Comparison of %ET requirements met in the Groundwater Command Area (CA_{gw}) for simulations with and without tank storage during: a) the dry (March-May) and monsoon season (August-January), b) drought and non-periods, where drought periods correspond to months with 12-month standardized precipitation index (SPI-12) less than -1, and c) in each cropping stage during the two growing seasons. In this figure, the %ET requirement met distributions were generated by aggregating daily %ET met.

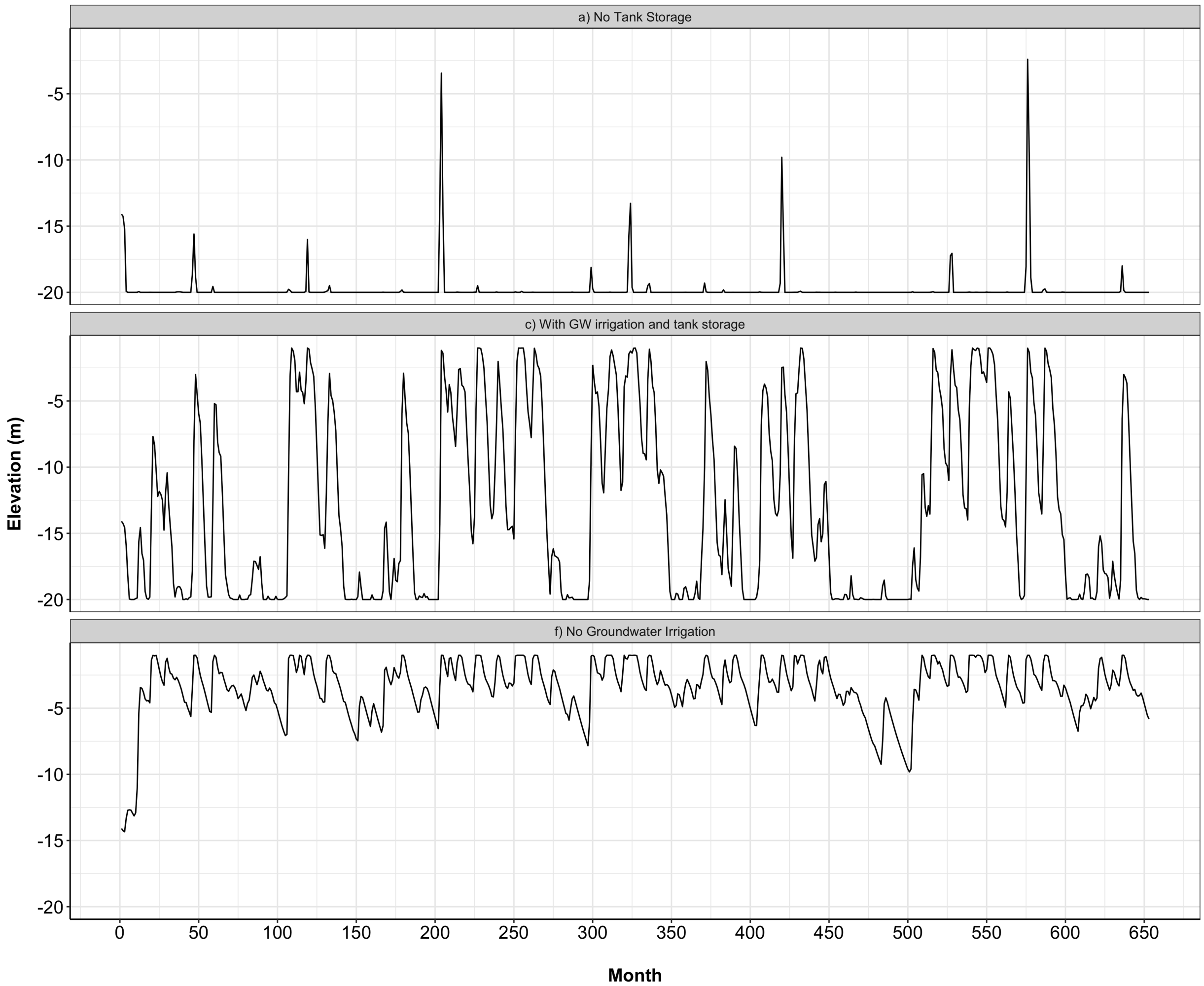

Figure 5.8: Understanding the influence of tank storage on monthly groundwater table elevation (m) under the condition of: (a) no tank storage; (b-e) with tank storage and groundwater irrigation; (f) when there is no groundwater irrigation in the tank water spread area.

the groundwater command area of the tank (CA_{gw}). Similarly, we find that only 65% [IQR - 29-100%] and 21% [IQR - 11-38%] of the crop water requirements are met in areas with access to both surface and groundwater from tanks (CA_{gw-sw}) during the monsoon and dry growing seasons respectively.

5.3.3 Understanding the influence of tank-recharge beneficiaries

Changing groundwater spread area

In our second set of simulations, we assessed how the influence of tanks changes with varying beneficiaries of tank-induced groundwater recharge (Table 5.3). Using the flexibility of the model developed in this study, we first explored how the spread of tank-based groundwater recharge affects the perceived impact of tank structures by varying the GSA linked to our tank system between 1-10 surface command area units (1 surface command area = 27ha in the simulated tank system). Our results show that median monthly groundwater table elevation over the simulation period declines non-linearly with increasing groundwater spread area, and is equal to -20m in simulations with GSA greater than 5 command area units (Figure 5.9a). As a result, the % crop ET requirements met on farms relying on well irrigation in the GSA of the tank declines due to a greater number of users accessing the tank-induced groundwater recharge (Figure 5.9b). We find that as the GSA increases the system experiences a greater frequency of 'outlier' events (Figure 5.9a) instead of sustained increases in water table elevation. However, we see that the shallow aquifer empties quickly in the months following the 'outlier' events due to a greater number of wells now sharing the recharge from tanks in the groundwater spread area. This is reflected in an assessment of differences in distribution (measured using the Mann-Whitney U-test) of %ET crop water requirements met in each season (Table C.6-C.7). Our results show no statistically significant ($p < 0.05$) difference (compared to the no tank scenario) in the distribution of %ET requirements when GSA is greater than 6 and 8 command area units in the monsoon and dry season respectively.

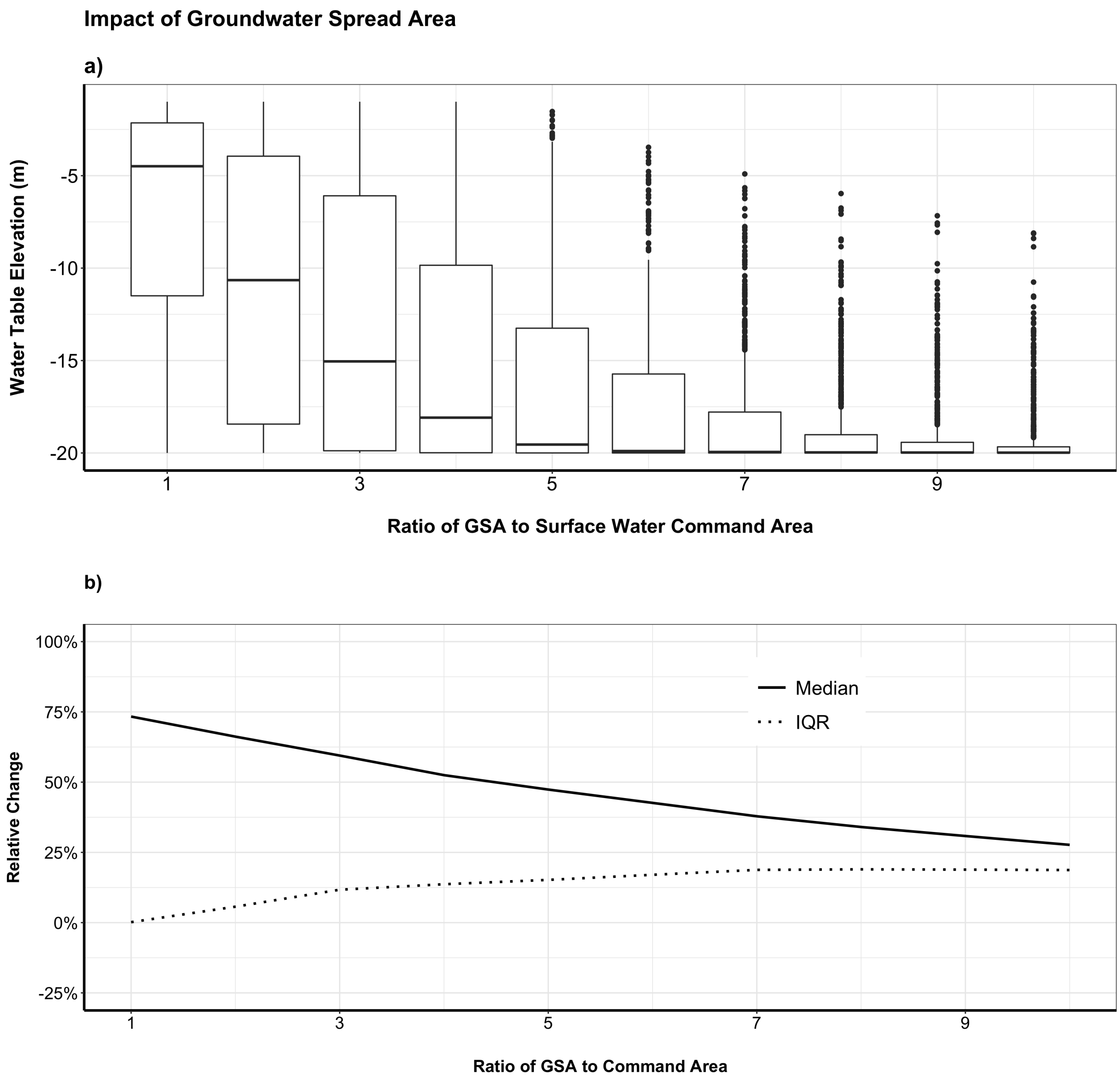

Figure 5.9: Understanding how groundwater spread area size affects tank impacts. a) Monthly water table elevation (m) distribution in simulations where GSA size is varied from 1 to 10 times the size of surface command area. b) Change in the mean and standard deviation of monthly %ET requirements met relative to the no tank scenario as function of groundwater spread area. Note: a baseline groundwater spread area of 3 surface command area units was used to generate results in other simulations.

Changing well density

A complementary question aimed to assess how changing the well density (measured by the number of wells per hectare of irrigated area) in the groundwater spread area affects the overall impacts of tanks. Our results suggest that increasing well density can positively impact the %ET requirements of crops in the groundwater spread through increased irrigation during both cropping seasons (Figure 5.10a). In the monsoon season, these impacts of increased well density plateau as the median % crop water requirements met approaches 100%. This suggests that the marginal benefits associated with operating more wells approaches zero in the monsoon season after a threshold equal to 1 well/ha in our simulations. In the dry season, we find a similar increase in %crop ET requirements met in the simulated system with benefits peaking when the well density is closer to 1 well/ha (Figure 5.10a). However, we find that a continually increasing well density starts negatively impacting the % crop water requirements met during the dry season, predominantly due to increasing volumes of groundwater being extracted during the monsoon season. Furthermore, we find that increasing well density has a limited impact on the % crop water requirements during drought period (Figure 5.10). This highlights that neither the influence of tanks nor an increase in well density (assuming current extraction patterns) can lead to a notable increase in the reliability of agricultural practices during droughts in these systems.

5.3.4 Impact of groundwater irrigation on tank outflow dynamics

In the final scenario, we looked into how groundwater irrigation impacts the outflow dynamics of tank structures. To explore this scenario, we ran simulations with and without groundwater irrigation in the groundwater spread area of the tank. Our results showed that without groundwater extraction, the water table in the simulated shallow aquifer stabilizes close to the surface with a median elevation of -3.2m [IQR = -4.3m - -1.8m] (Figure 5.8c). Under these conditions, we found that sluice releases dominate the tank output fluxes (Figure 5.11) with the annual median number of days with sluice in the surface command

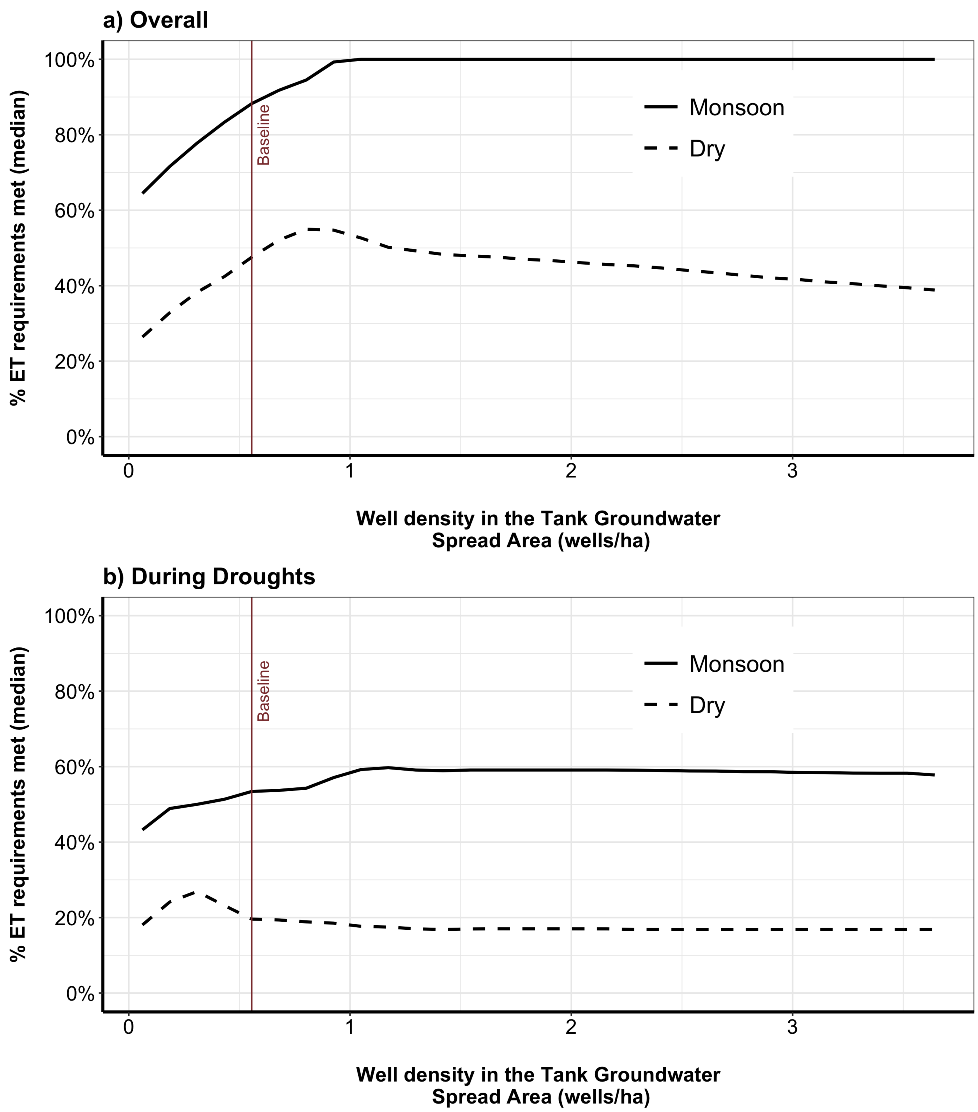

Figure 5.10: Understanding how well density affects benefits from tank. a) Median %ET requirements met during the monsoon and dry season as a function well numbers in the groundwater spread. b) Median %ET requirements met during the monsoon and dry season as a function well numbers in the groundwater spread during *droughts*. Drought periods correspond to months with 12-month standardized precipitation index (SPI-12) less than -1. Note: the baseline well number corresponds to the value used to generate results in other simulations.

area equal to 295 [IQR = 195-349 days]. We find the efficiency of the tank structure, measured as the ratio of productive water uses (sluice discharge and groundwater exchange) to evapotranspiration losses, is equal to 3.7 [IQR = 2.82 - 4.4] in this situation.

In comparison, intensive groundwater irrigation in the system results in a transition from sluice-dominated to recharge-dominated tank outflow dynamics. We find that tank-induced recharge increases significantly (compared to recharge in the scenario without irrigation) constituting 50% [IQR: 38%-60%] of the yearly outflow from the simulated tank. This is predominantly driven by a lowering of the water table which results in the tank remaining hydraulically disconnected from the water table for a greater percentage of the time, which in turn increases gravity-dominated recharge through the tank bed. As a result, this transition from sluice to recharge-dominated tank outflow dynamics results in tanks holding water for fewer days in a year. The median number of days with sluice in the scenario with groundwater irrigation is equal to 160 [IQR: 106-207 days]. We also find that the efficiency of tank structures improves in this scenario with yearly ET flux reducing to 11% [IQR: 10%-15%] of total tank outflow, and the ratio of productive water use (sluice discharge and groundwater exchange) to evapotranspiration losses increasing to 7.8 [IQR: 5.52 - 8.63]. Additionally, we find that the transmittance of water to downstream tanks due to overflow spillage reduces by 12% under the condition of groundwater irrigation.

5.4 Discussion and Conclusions

The discussions around the role of rain-water harvesting structures (like tanks) in modern agricultural systems have been ongoing for many decades now. These structures are generally believed to have the potential of transforming rain-fed agricultural systems by reducing the yield gap (i.e. the difference between the theoretical and observed crop yields) and improving resilience against the unreliable rainfall patterns (Dile et al., 2013; Piemontese et al., 2020). In India, interest in RWH has been rekindled due to declining groundwater storage, a need to increase food production, and as part of governance paradigms to

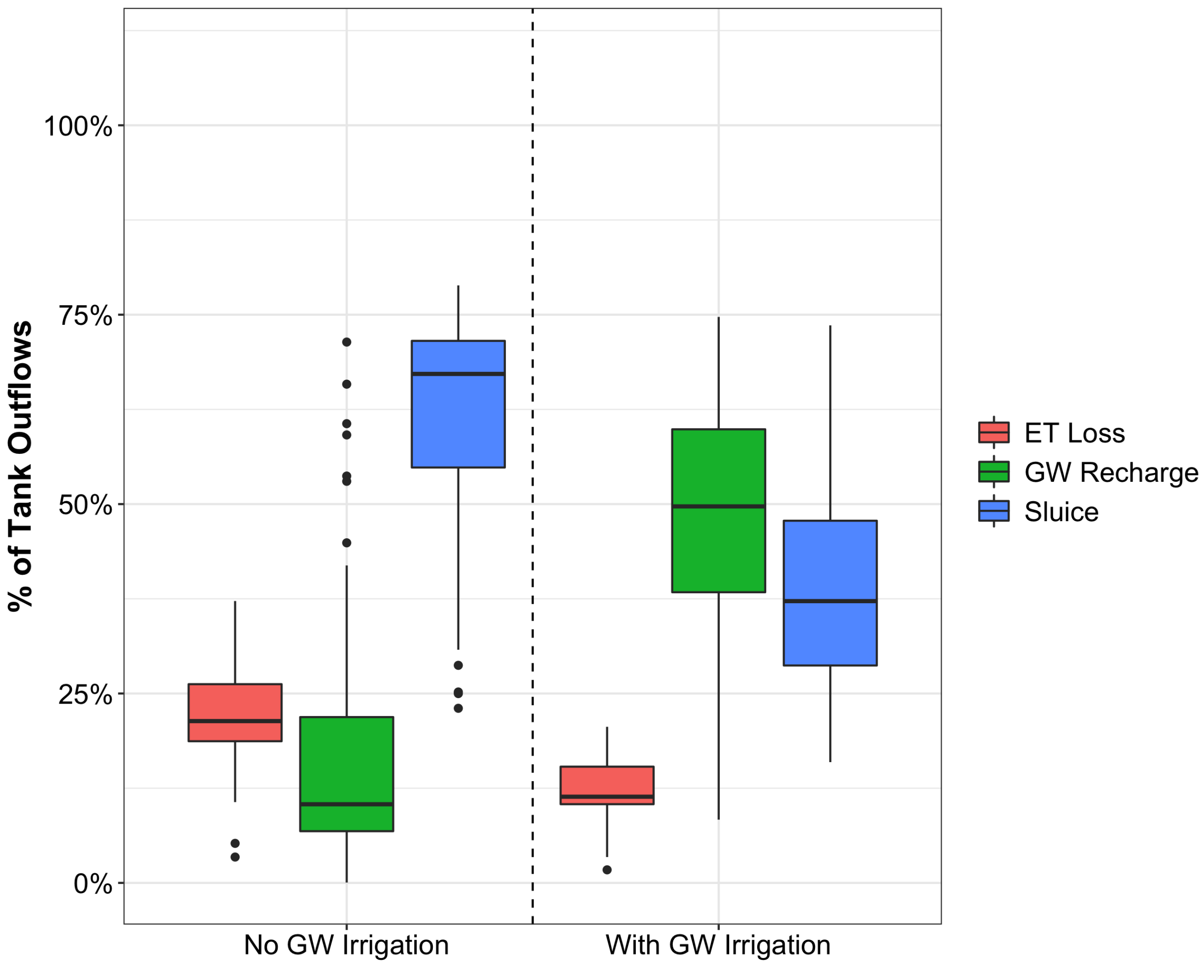

Figure 5.11: Tank outflow dynamics with and without groundwater irrigation in the groundwater spread area of the tank system. We find that tanks transition from sluice to groundwater recharge dominated structures due to irrigation in the groundwater spread area. *Note:* This comparison has been made using our baseline tank characteristics with groundwater spread area equal to 3 command area units (Table C.5).

decentralize irrigation management. The rehabilitation of existing rain-water harvesting structures and the construction of new ones are a central part of India's Groundwater Artificial Recharge Master Plan (2013), where 11 billion USD has been allocated for improving groundwater storage in the country. However, the role of tanks in post-green revolution agricultural systems with a long history of water harvesting and where a clear preference for groundwater irrigation over surface irrigation methods has emerged is unclear. The analysis presented here aims to improve our understanding of the hydrological impact and dynamics of RWH structures in contemporary Southern Indian agricultural systems through the development of a conceptual hydrological model.

Our first research question (and simulation) assessed how tanks impact groundwater availability in a Southern-Indian landscape. This question has been evaluated in previous modelling studies (Glendenning and Vervoort, 2011; Perrin et al., 2012; Boisson et al., 2015; Nicolas, 2019), and our results complement findings from these analyses where we find that tanks improve groundwater availability in the vicinity of the tank through recharge. Without tanks and given current extraction rates, our results show that the shallow aquifer remains dry for almost the entire simulation period in our analysis, and farmers despite having access to wells have to contend with essentially practicing rain-fed agriculture. Similar results have been observed by Sato and Duraiyappan (2011) through farmer surveys in the Gundar Basin (Tamil Nadu) where systems with poorly performing tanks forced well-owning farmers to leave lands fallow for extended periods. The introduction of tanks helps improve the groundwater available for extraction. However, we find that these benefits from tanks are unable to extend to drought periods. Thus, in addition to tanks, farmers in these systems would require alternative adaptation strategies to cope during these dry spells.

The second part of our analysis looked at previously understudied questions related to the relationship between tank-induced groundwater recharge and the beneficiaries utilizing this recharge. First, we analyzed the importance of delineating the groundwater spread area of a tank, and the number of users utilizing the tank-induced recharge. Unlike the homogeneous alluvial aquifer systems of Northern India, the number of beneficiaries of tank-induced recharge in complex hard-rock aquifer systems with preferential flow paths may be variable for tanks with an otherwise similar configuration (in terms of size, command area, catchment area, etc.). Our results suggest that the agricultural benefits from tanks are closely tied to the groundwater spread area of a tank, with benefits declining as the groundwater spread area increases. Thus, our findings highlight the possibility of no net benefits associated with groundwater recharge from tanks with large groundwater spread areas as the volume of water recharged gets distributed over too many wells. Results from previous field studies in other Southern Indian tank systems have highlighted the localized impact of tank structures on groundwater storage as a drawback (Massuel et al., 2014). Our results suggest that a landscape configuration with localized impacts related to tank structures might be preferential with regards to actualizing groundwater benefits from these structures in terms of crop yields.

Additionally, we have also assessed how an increase in well density might affect the perceived benefits from tanks. As expected, our results suggest that compared to a scenario with limited or no well irrigation, an increase in well density can lead to an increase in the crop water requirements met in the system. However, our results also show that beyond a threshold well density (1 well per hectare of irrigated land in our simulations), the benefits from additional wells in the system plateau in the monsoon season as a greater proportion of crop water requirements are met. However, we find that this increase in % crop water requirements met during the monsoon season impacts the groundwater available for use in the dry season. This highlights the need for improved intra-annual groundwater management within these systems. Furthermore, our results highlight that an increase in well density has a limited impact in helping agricultural systems meet crop water requirements during drought periods. Strategies aiming to regulate groundwater extraction amongst farmers that have access to tank-based sluice discharge during the monsoon season can be promising in these systems to improve groundwater availability in the dry season (Siderius et al., 2015). However, there may also be a need for management arrangements that coordinate groundwater extraction between years to improve the reliability of water requirements during drought periods.

The last part of our analysis explored how groundwater irrigation in the surrounding aquifer impacts tank outflow dynamics. Our results show that tanks nestled within a landscape with intensive groundwater irrigation see a large decline in the number of days with water storage compared to tanks in systems with no groundwater irrigation. With fewer days of water storage, we find that the total sluice outflow volume from tanks declines over the course of the season. Furthermore, we find that the decline in water storage days is predominantly driven by an increase in groundwater recharge to the surrounding aquifer, and so groundwater irrigation in the system transforms the surface water benefits from tanks to groundwater benefits. As a result, we would expect this transition in tank dynamics to negatively impact farmers relying predominantly on surface irrigation water from tanks in these systems. Further, we find that a reduction in the number of days with tank water storage results in lower evapotranspiration losses from tanks. Thus, tanks become more efficient storage structures (assuming efficiency is linked to a reduction in ET losses) when there is groundwater irrigation in the landscape. We also find that overflow spillage from upstream tanks to downstream tanks reduces when there is groundwater irrigation. As a result, this reduction in spillage could have negative impacts on water availability in downstream tanks that depend on this spillage. Overall, our results suggest that tank outflow dynamics are fundamentally altered due to groundwater irrigation in these systems, and thus, policy discussions and management scenarios relying on historical notions of tanks might not be applicable in contemporary groundwater irrigated agricultural landscapes from a hydrological perspective.

In this study, we have developed a conceptual hydrological that captures key processes associated with rain-water harvesting structures and the surrounding aquifer for application in data-scarce agricultural systems. However, the framework presented in this paper has limitations. Our modelling approach focuses on a single tank, while tanks are generally embedded in larger catchments and are part of linked cascades in Southern Indian agricultural systems. Previous studies have highlighted how cascade-specific hydrological processes can impact tank inflow and outflow dynamics (Van Meter et al., 2016), while economic analyses have shown limited benefits associated with tanks structures by considering downstream impacts (Bouma et al., 2011). However, few modelling studies have explored the trade-offs associated with these structures in systems with multiple connected tanks. With the availability of high-resolution remotely sensed data to monitor these structures (Vanthof and Kelly, 2019), there remains an urgent need to explore the dynamics of these structures at larger spatial scales. Additionally, the model presented in this study has been developed and validated using sparse field data. While data scarcity is generally the norm in these systems, there remains a need to complement modelling efforts with in-depth field studies to improve our understanding of these systems. Furthermore, our modelling framework has over-simplified the management side at the individual farmer and collective system-level. For example, we have assumed a constant daily pumping rate from wells regardless of soil moisture conditions and cropping stage. Such an approach is based on field observations where farmers are known to leave their water pumps constantly switched on due to low/no energy costs given current energy subsidies provided by the State (Sidhu et al., 2020). However, there is a need for further research that aims to link different irrigation strategies and heterogeneous decision-making leads to emergent outcomes in these systems. The modelling framework developed in this analysis can serve as a platform to develop different nuanced management scenarios.

Rain-water harvesting structures remain intriguing interventions to increase water availability (both surface and groundwater) in semi-arid and arid agricultural systems. However, the impact of these structures appears to be meaningful under a specific spectrum of landscape and climate conditions. Thus, there remains a need to critically assess the socio-hydrological conditions under which investments in rain-water harvesting are most impactful.

Chapter 6

Conclusions

6.1 Summary of Work

Groundwater represents a vast distributed water source that is currently critical for meeting the demands of various socio-environmental systems globally. However, the management of groundwater has proven to be challenging with groundwater over-extraction being observed in many regions around the world. Current patterns of groundwater use are negatively impacting environmental systems, while endangering the ability of future generations to access this strategically important resource. As the region with the highest groundwater extraction rates in the world, India is currently at the forefront of this problem where national food security and the livelihoods of millions of households have grown to become dependent on the over-exploitation of groundwater resources. This dissertation consists of three studies to support the broad goal of addressing groundwater overexploitation in India. Specifically, these studies aimed to improve understanding on the following research questions:

1. *Assessments:* What are the implications of developing regional groundwater use thresholds that take local and global environmental limits into account?

2. *Monitoring methods:* How can the congruence between 'hard' groundwater monitoring data sources and 'soft' non-hydrological data sources (like census data and newspaper articles) be improved to identify groundwater depletion hotspots?

3. *Interventions* Can agricultural rain-water harvesting improve water availability in groundwater-dependent agricultural systems?

The introductory chapter provides some context on the problem of groundwater overextraction in India, while chapter 2 provides a background to the specific issues related to groundwater systems addressed in the dissertation. In chapter 3, environmental limits at the local and global scales were used to develop groundwater extraction thresholds at the district-level across India. Traditionally, regional groundwater stress assessments (measured as the ratio of annual groundwater usage to renewable groundwater supply) have assumed the entire groundwater recharge volume to be available for human consumption when assessing the long-term sustainability of groundwater use. However, these assessments inadequately account for the water requirements of groundwater-dependent ecosystems or the recently estimated planetary limits, and as a result over-allocate the water available for human extraction. Using four scenarios, this chapter showed how patterns of over-stressed districts vary when different environmental considerations are taken into account. Results showed that incorporating the groundwater requirements of local environmental systems results in 36% districts being classified as over-stressed in India compared to 26% when no environmental considerations are taken into account. With regards to global considerations, the current freshwater Planetary Boundary was applied in two contrasting ways to estimate district-level groundwater extraction thresholds. In the first approach (bottom-up), district-level groundwater use thresholds were set to 40% of groundwater recharge to be consistent with the current freshwater Planetary Boundary of restricting global freshwater use to 40% of the renewable supply. In the second approach, an India-level groundwater use threshold was derived that was consistent with the Planetary Boundary, and then this national threshold was disaggregated (top-down) at the district-scale based on current district-extraction patterns (grandfathering). Results from these approaches showed that nearly 70% of the districts would be considered over-stressed when global considerations are taken into account. However, the results from these two approaches differed significantly in relation to the efforts required by over-stressed districts to stay within the derived thresholds. Overall, results from this analysis suggested that incorporating environmental considerations would significantly lower the volume of groundwater resources available for human use in India (173-312 km^3/year; compared to 399 km^3/year under no restrictions). The methodology and results from this chapter can help policy-makers understand the implications of prioritizing environmental needs into groundwater management in India, while contributing to literature focusing on the incorporation of environmental limits into regional resource management.

Chapter 4 aimed to improve understanding on how monitoring data can be used to better identify regions with groundwater depletion in hard-rock aquifer systems. This analysis was primarily driven by a discrepancy in findings in Southern India, where physical (like monitoring wells) and remotely sensed (from the GRACE satellite) data suggested increasing groundwater storage trends in the region between 1996-2016, while data from non-hydrological sources like census data and news articles suggested depleting groundwater storage over the same period. Results from this study revealed that previous trend estimates relying on monitoring well data were skewed by the presence of a survivor bias, where dry or defunct wells were systematically excluded from trend analyses due to missing data. The timing of missing data and the location of wells with missing data were found to be strongly correlated with metrics of climate stress (in the form of dry periods) and groundwater development, which was indicative of a systemic exclusion. Using two alternative metrics that took into account information from dry and defunct wells, groundwater stress was found to be increasing in South India between 1996-2016. The approach developed in this analysis can provide critical information on how groundwater depletion hotspots are identified and monitored in regions with extensive groundwater extraction.

In Chapter 5, the potential of rain-water harvesting systems (RWH or tanks) to increase groundwater supply and provide farmers with an alternative source of water was assessed in Southern India. A conceptual hydrological model was developed to better capture important tank-specific hydrological processes (like tanks can serve as sources of surface water irrigation while recharging the aquifer) and considerations (like separating the surface command area from the groundwater command area) for application in data-scarce agricultural systems. The fundamental questions asked in this chapter revolved around the feedback between RWH systems and the surrounding aquifer in a region with intensive groundwater irrigation. Simulations over a 65-year period showed that tanks were able to improve the water available for farmers (measured by the percentage of crop water requirements met), however, these benefits were found to be negligible during drought spells and/or when the number of beneficiaries of tank-induced groundwater recharge was poorly controlled in the system. Results also showed that groundwater irrigation in the surrounding aquifer positively impacted the efficiency of output fluxes from RWH structures by reducing the percentage of evapotranspiration losses and increasing groundwater recharge. However, it was found that this increase came at the cost of reduced water available for farmers relying on surface irrigation water. The model developed in this study provides important insights into understanding the role RWH structures can play in improving water availability in contemporary groundwater-dependent agricultural systems.

6.2 Key Research Contributions

The challenge of transitioning towards a more sustainable trajectory related to groundwater use is inherently complex, and as a result, there is a need for research that transcends traditional boundaries (like spatial scales, disciplines, hydrological components). Respecting this need for 'cross-boundary' research, an attempt to cross traditional boundaries was made in each of the studies presented in this dissertation. Transcending the boundary of spatial scales, Chapter 3 represented a step forward with regards to how environmental considerations at multiple scales (global and local) are used to inform the regional-scale resource use and management. In Chapter 4, information from non-traditional data sources was used to improve how findings from hydrological data sources, thereby highlighting the importance of utilizing data that crosses traditional disciplinary boundaries. Finally, Chapter 5 aimed to better acknowledge the surface water-groundwater continuum in RWH structures to improve understanding on how interventions aimed at augmenting groundwater supplies might impact other hydrological processes.

Additionally, each of the chapters also contributed to domain-specific knowledge. First, an attempt was made to quantify the volume of annual groundwater recharge needed to respect local and global environmental water considerations across India. An assessment of groundwater stress highlighted that a greater proportion of districts are over-stressed in India when these environmental considerations are taken into account. Additionally, the potential of setting regional targets by disaggregating a national groundwater budget was also introduced. Second, the potential of a survivor bias in relation to how monitoring well data is utilized was revealed. This bias showed that reliance on monitoring wells with the most complete records (over those with missing data) can mask groundwater depletion hotspots in hard-rock aquifers. New indices (%dry wells and %defunct wells) that more reliably captured groundwater depletion in hard-rock aquifers were also developed. Third, a conceptual hydrological model was developed to better understand how agricultural rain-water harvesting systems operate in contemporary agricultural systems of Southern India. The model was used to highlight the circumstances under which rain-water harvesting systems can improve the percentage of irrigation water requirements met. Model results were also used to better understand the water storage dynamics of rain-water harvesting structures in systems with extensive groundwater irrigation.

6.3 Recommendations for Future Work

The studies presented in this dissertation can serve as a source for future research that can aim to address the limitations of the current analyses and/or build on the contribution of this work. Here are some recommendations for future research from this dissertation:

- In Chapter 3, groundwater extraction thresholds that took different environmental considerations into account were developed in India. Future research can look into the potential of technological interventions (e.g. more efficient irrigation technologies, less water-intensive crops) to reduce groundwater extraction in regions currently transgressing derived groundwater-use thresholds.

- There is considerable uncertainty on how environmental water requirements are estimated at the local scale. While the analysis presented in this dissertation relied on a global hydrological model to estimate groundwater discharge values, future research can integrate outputs from global hydrological models with region-specific modelling results to improve estimates of local environmental flow requirements.

- Currently, there only exists an integrated freshwater Planetary Boundary, however, it may be more appropriate to develop a groundwater-specific framework at the Planetary scale. Additionally, there is a need to better understand how regional actions to limit groundwater use impact hydrological processes at a local and global scale.

- Using the dry-well indices developed in Chapter 4, future research can further investigate the factors (e.g. policy interventions) that cause missing records in the water level times series.

- There is a need to extend the conceptual modelling framework developed for rain-water harvesting systems in Chapter 5 from a single tank to multiple cascading tanks to better understand the impact of tanks at larger spatial scales.

- The conceptual model developed can be applied to systems with different socio-environmental characteristics like climate, hydrogeological properties, and land-use patterns to better scope out the dynamics and impacts of rain-water harvesting systems.

- The model developed in chapter 5 can be used to investigate how rain-water harvesting systems could function under different climate change scenarios.

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APPENDICES

Appendix A

Supplementary Material for Chapter 3

Table A.1: Variables from the VDSA database used in the analysis.

Crop-Wise Irrigated Area	Land use	Source-wise Irrigated Area	Population Census
Rice	Geographical area	Net irrigated area by Canals	Total population
Sorghum (total)	Forest area	Tanks	Male population
Minor pulses	Barren and uncultivable land	Tube wells	Female population
Total Fruits and Vegetables	Land put to non agricultural uses	Other wells	Total rural population
Wheat	Cultivable waste	Total wells	Total cultivators
Pearl millet	Permanent pastures & other grazing land	Other sources	Male cultivators
Barley	Other fallow land		Female cultivators
Groundnut	Current fallow land		
Linseed			
Sorghum (kharif)			
Maize			
Chickpea			
Sesamum			
Sugarcane			
Sorghum (rabi)			
Finger Millet			
Pigeonpea			
Rapeseed and Mustard			
Cotton			

Table A.2: Irrigation water requirements for different crops considered in the analysis. Irrigation water requirements assume conventional irrigation technology.

Crop	Irrigation Water Requirement (in mm)
Rice	1000
Wheat	450
Sorghum	400
Pearl Millet	400
Maize	650
Finger Millet	400
Barley	400
Chickpea	240
Pigeon Pea	500
Other Pulses	300
Groundnut	600
Sesame	150
Rapeseed and Mustard	150
Linseed	150
Sugarcane	1600
Cotton	450
Fruits and Vegetables	600

[a] All irrigation water requirement values obtained from Fishman et al. (2015). Irrigation water requirement for fruits and vegetables obtained from Sivanappan (1994) and for oil crops from Tamil Nadu Agricultural University (TNAU; (https://www.agritech.tnau.ac.in/agriculture/agri_irrigationmgt_waterrequirements.html).

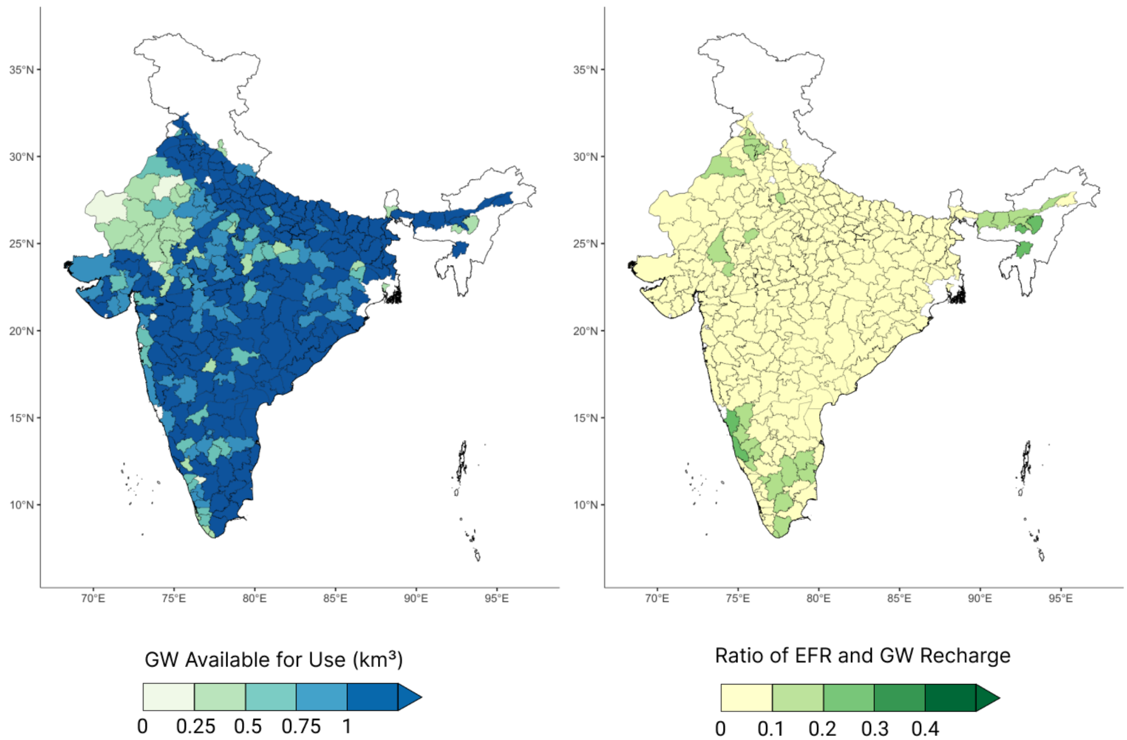


Figure A.1: a) District-scale annual groundwater recharge volume estimated by the Central Groundwater Board (CGWB) of India (Baseline Scenario), b) The ratio of groundwater recharge reserved for natural discharge in current CGWB estimates. Represents the amount of groundwater recharge not available for human use.

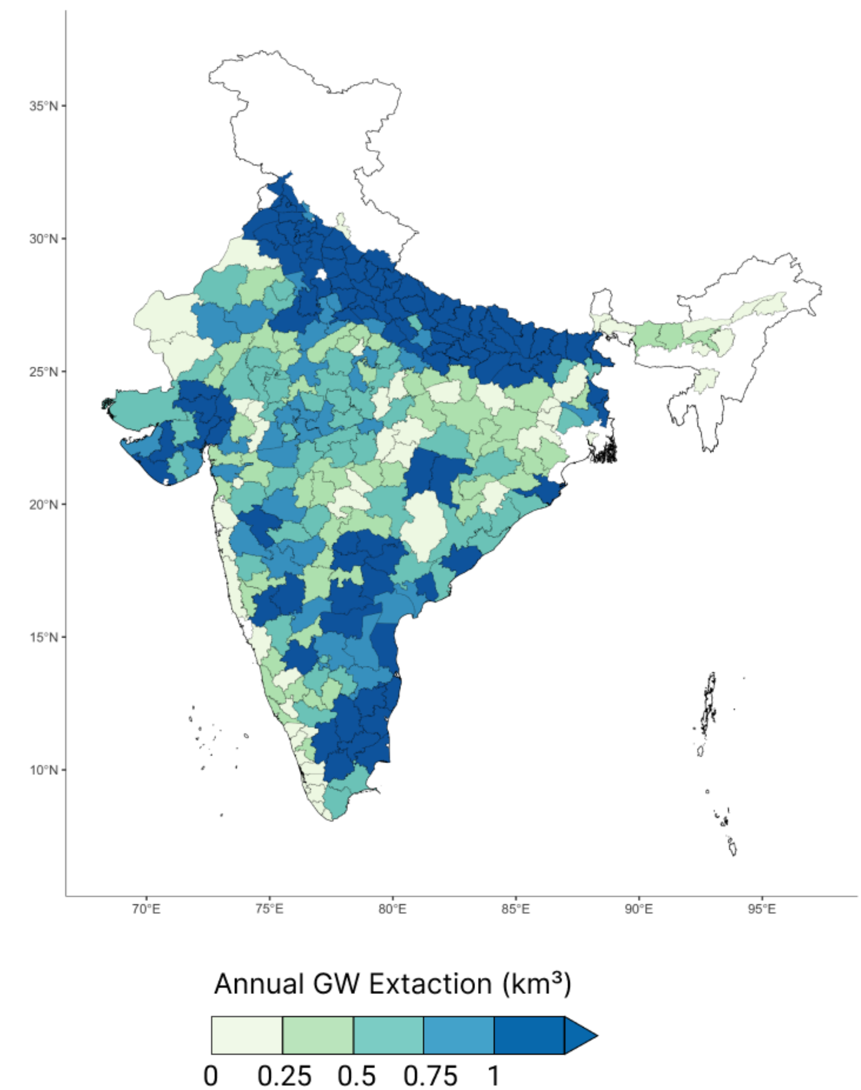


Figure A.2: Estimated district-level groundwater extraction rates (in km^3)

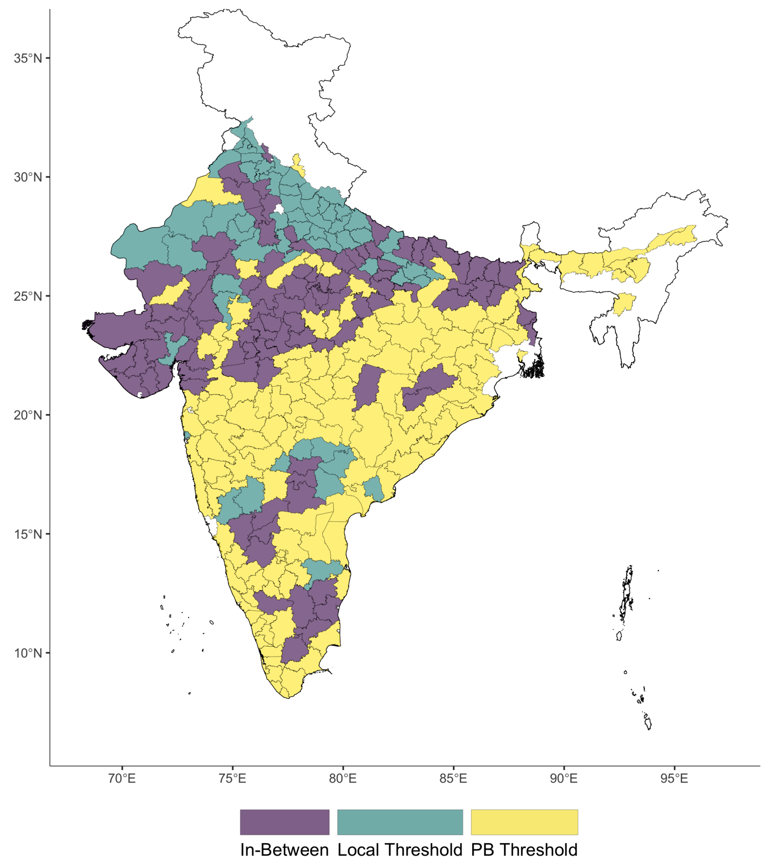


Figure A.3: Allocated district-level budgets in the Mixed scenario relative to the budgets estimated based on the local ($Agw_{EFR,d}$) and global scenario ($Agw_{PB,d}$). The category 'In-between' represents groundwater availability thresholds that were greater than the ($Agw_{PB,d}$) and less than ($Agw_{EFR,d}$).

Table A.3: Ground water stress (in %) values in regions where base flow reductions have been observed. Overall, we find that groundwater stress is underestimated in the baseline scenario, which can potentially misclassify a district as 'safe' (even though it is over-stressed). Please note that base flow reductions can be observed even when GSR values are under 100, and thus an assessment like this can at most complement field scale studies.

District/Region	Source	%GSR (Baseline)	%GSR (Local)
Ujjain-Indore	Galkate et al. (2020)	108	130
Namakkal (Salem)	Ballukraya (2000)	116	154
Chamarajnagar (Mysore)	Collins et al. (2020)	62	88
Hyderabad-Medak	Perrin et al. (2011b)	74	98
Gomti River Basin	Dutta et al. (2015)	108	124
Indus River Basin	Yang et al. (2014)	117	132
Ganges River Basin	Mukherjee et al. (2018)	83	102
Ramganga Basin	Maheswaran et al. (2016)	142	173
Bangalore	Srinivasan et al. (2015)	81	105

[a] Web of Science query used to obtain relevant field scale studies in India: ((“groundwater” or ”ground water”)) AND TOPIC: ((abstrac* OR extrac* OR pump* OR ”deple*” OR ”decli*”)) AND TOPIC: ((”baseflow” OR ”low flow” OR ”base flow” OR ”lowflow” OR ”surface flow”)) AND TOPIC: (India). This yielded 20 results out of which a subset was selected based on relevance.

[b] %GSR values were obtained by taking the median GSR values of district(s) spanned by the area studied in the research articles

[c] Gomti and Ramganga are sub-basins of the Ganges river basin

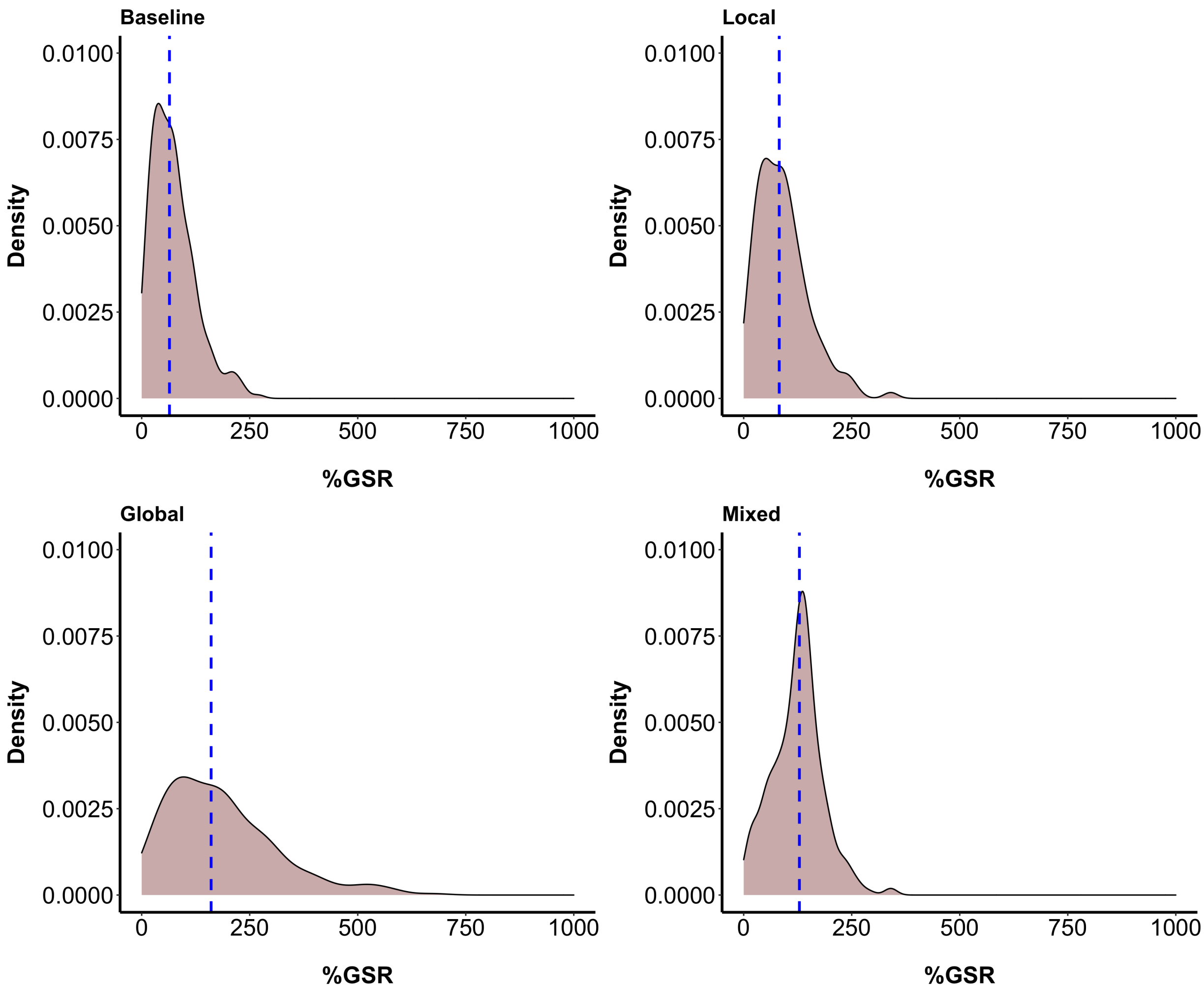


Figure A.4: Density plots highlighting the distribution district-level groundwater stress for the 4 scenarios analyzed in this study. The blue dashed line represents the median groundwater stress value in each scenario.

Appendix B

Supplementary Material for Chapter 4

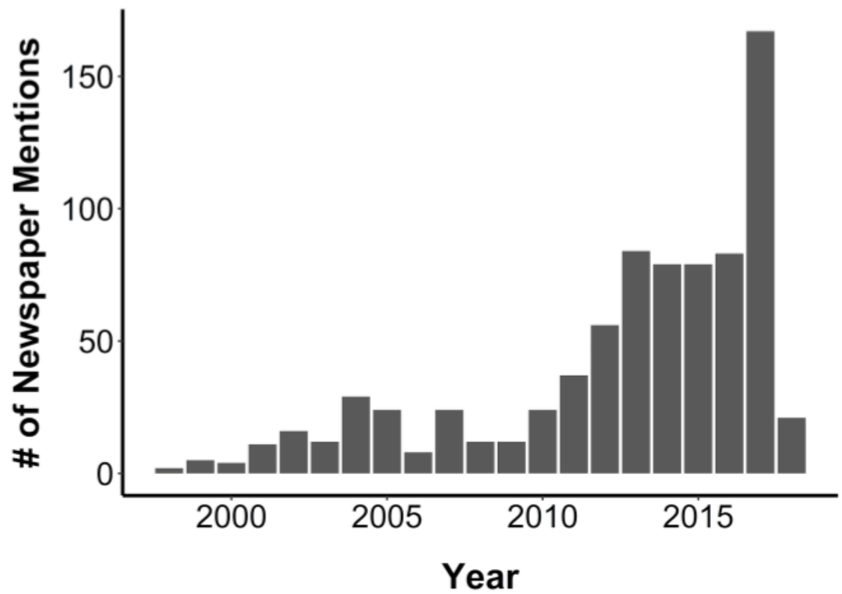


Figure B.1: Yearly frequency of groundwater depletion related articles in major English language newspapers in South India. Results were obtained using the following query: ((groundwater or ground water) same (deple* OR scarc* or dry or dried or decli*)) and (farm* OR agri*) AND (Karnataka or KA or Maharashtra or MH or Andhra Pradesh or AP or Telangana or TG or Tamil Nadu or TN or Kerala or KL). Results show increasing mentions of groundwater stress in major newspapers in South India.

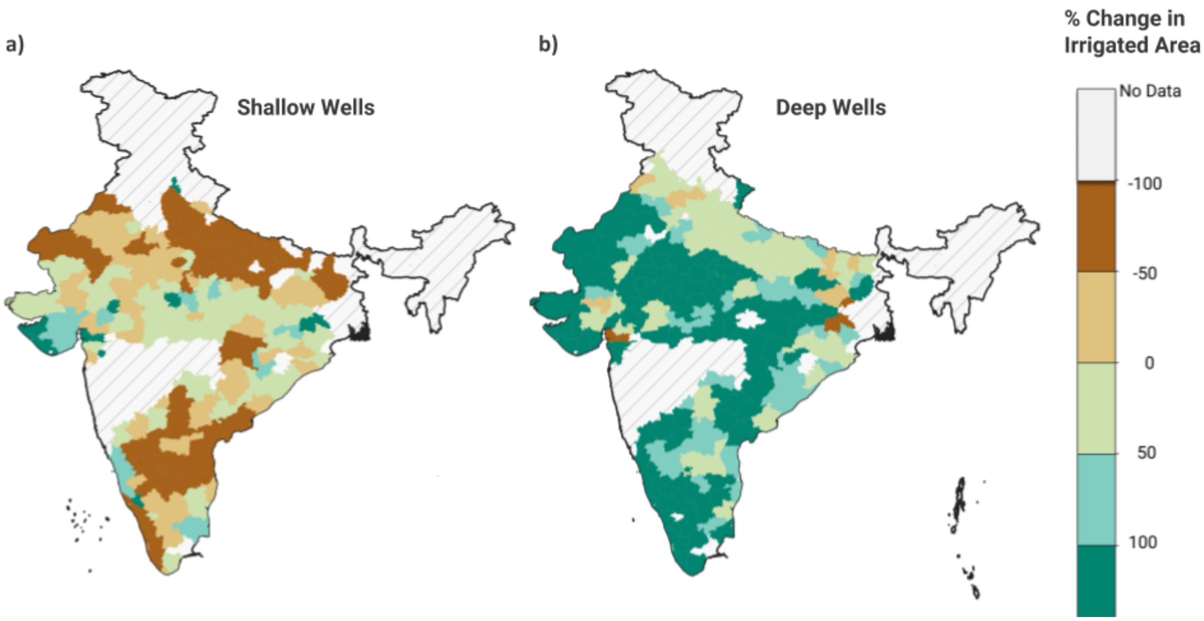
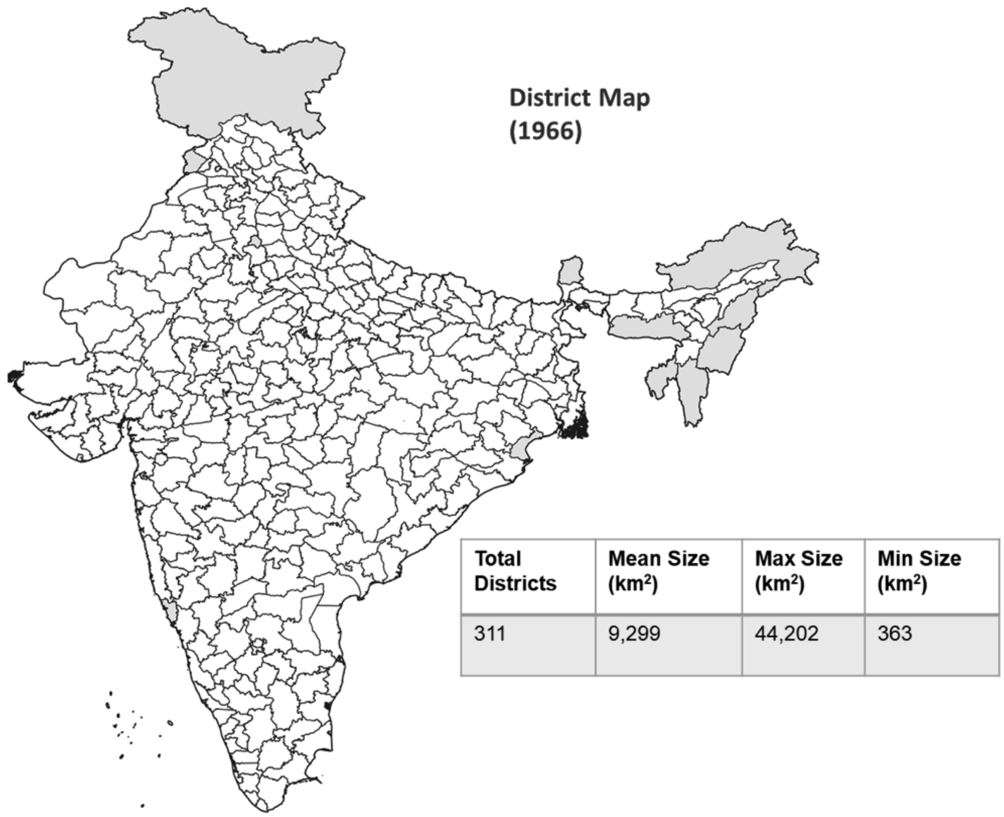


Figure B.2: a) Percent change in the area irrigated by shallow wells from 1996-2001 to 2006-2011. b) Percent change in the area irrigated by deep wells from 1996-2001 to 2006-2011. The change was estimated at the district scale using the median value of irrigated areas for each time frame. A positive change (green) indicates an increase in irrigated area, while a negative change (brown) indicates a decrease in irrigated area. Figures show that shallow well irrigated area has been decreasing in South India but deep well irrigated area has been increasing. Please note that regions with no data availability are represented by hatched lines.



Total Districts	Mean Size (km^2)	Max Size (km^2)	Min Size (km^2)
311	9,299	44,202	363

Figure B.3: District map of India used in the analysis. Please note that the district boundaries are from 1966.

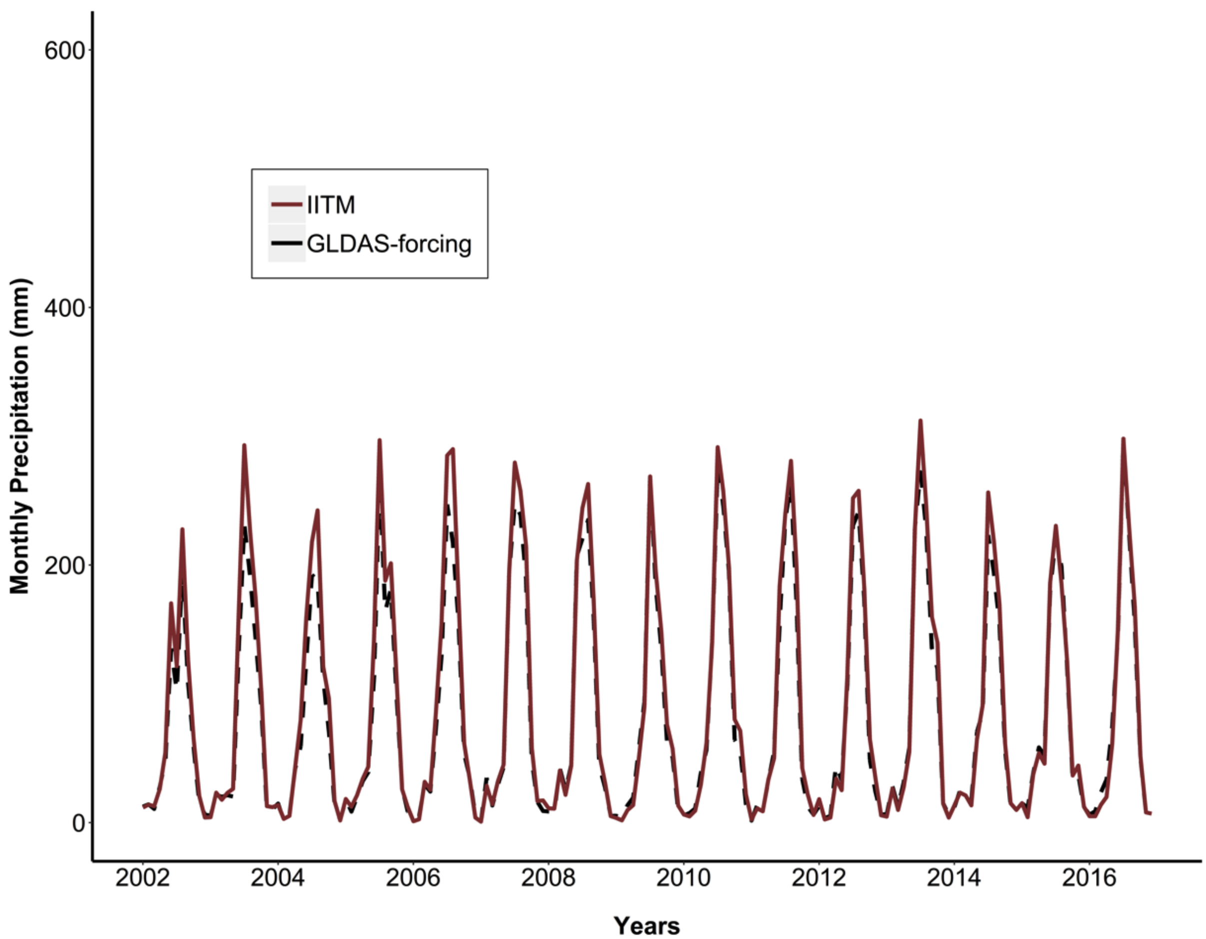


Figure B.4: Comparison between GLDAS-1/NOAH forcing data and the IITM precipitation at the India-scale. The correlation between the data sources is strong (r = 0.99)

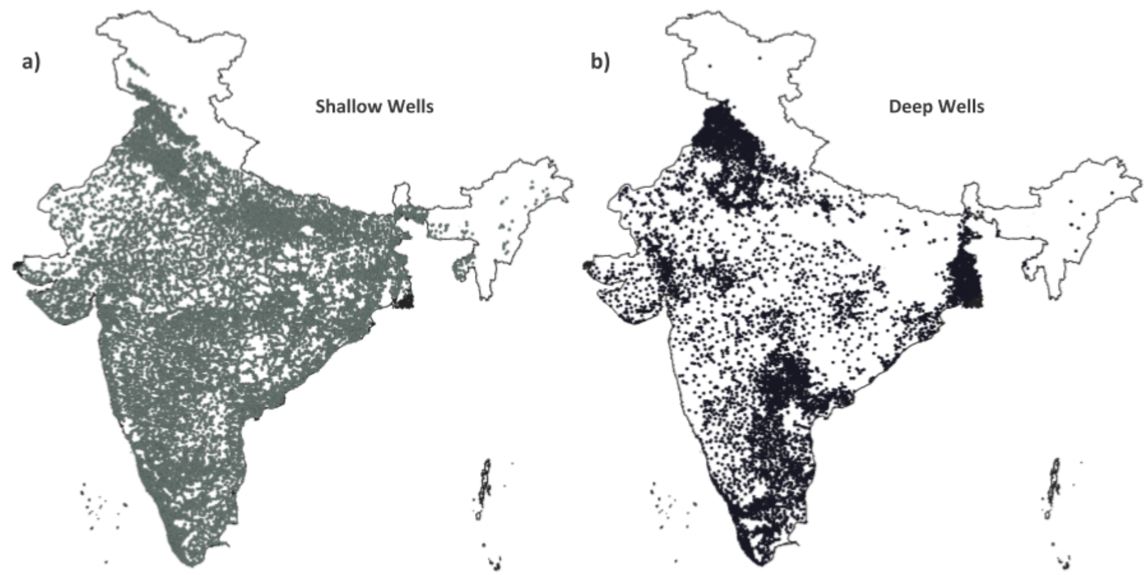


Figure B.5: a) Location of all the shallow monitoring wells in the Central Groundwater Board (CGWB) database, b) Location of all the deep monitoring wells in the CGWB database. Figures show good coverage across India for both shallow and deep monitoring wells, with the shallow well network being more extensive

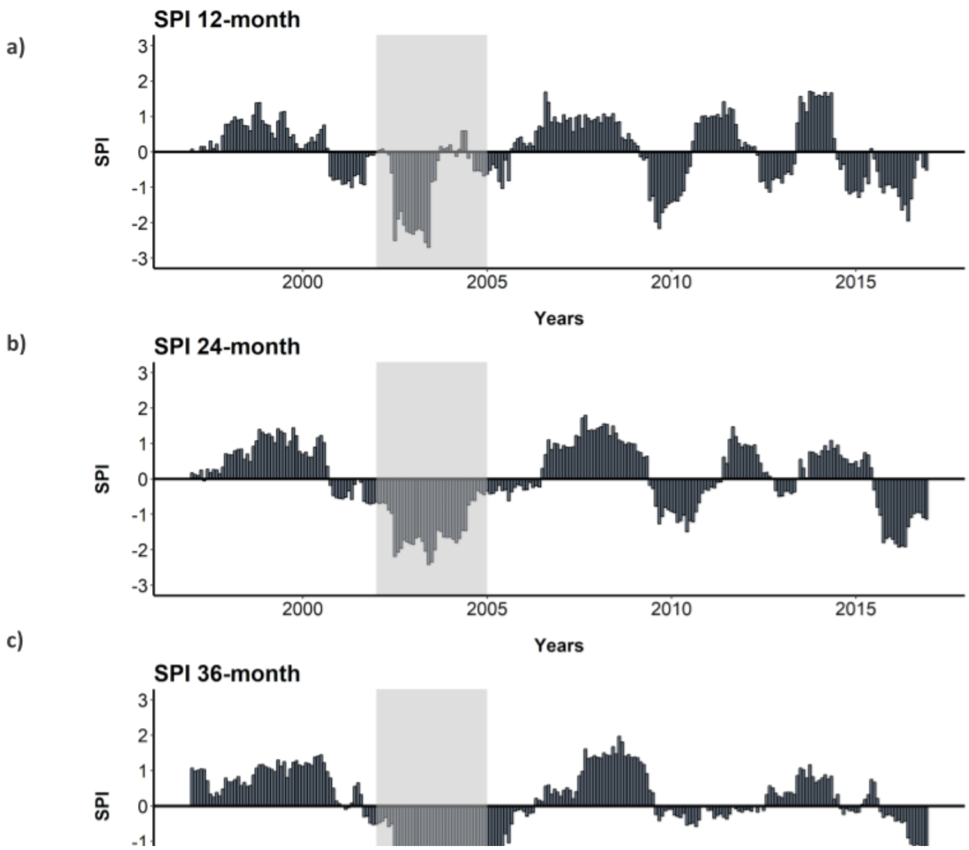


Figure B.6: Standardized Precipitation Index (SPI) time series for India between 1996 and 2016. a) 12-month SPI, b) 24-month SPI and c) 36-month SPI. The shaded area represents the period between 2002-2004 and corresponds to a severe drought in large parts of the country.

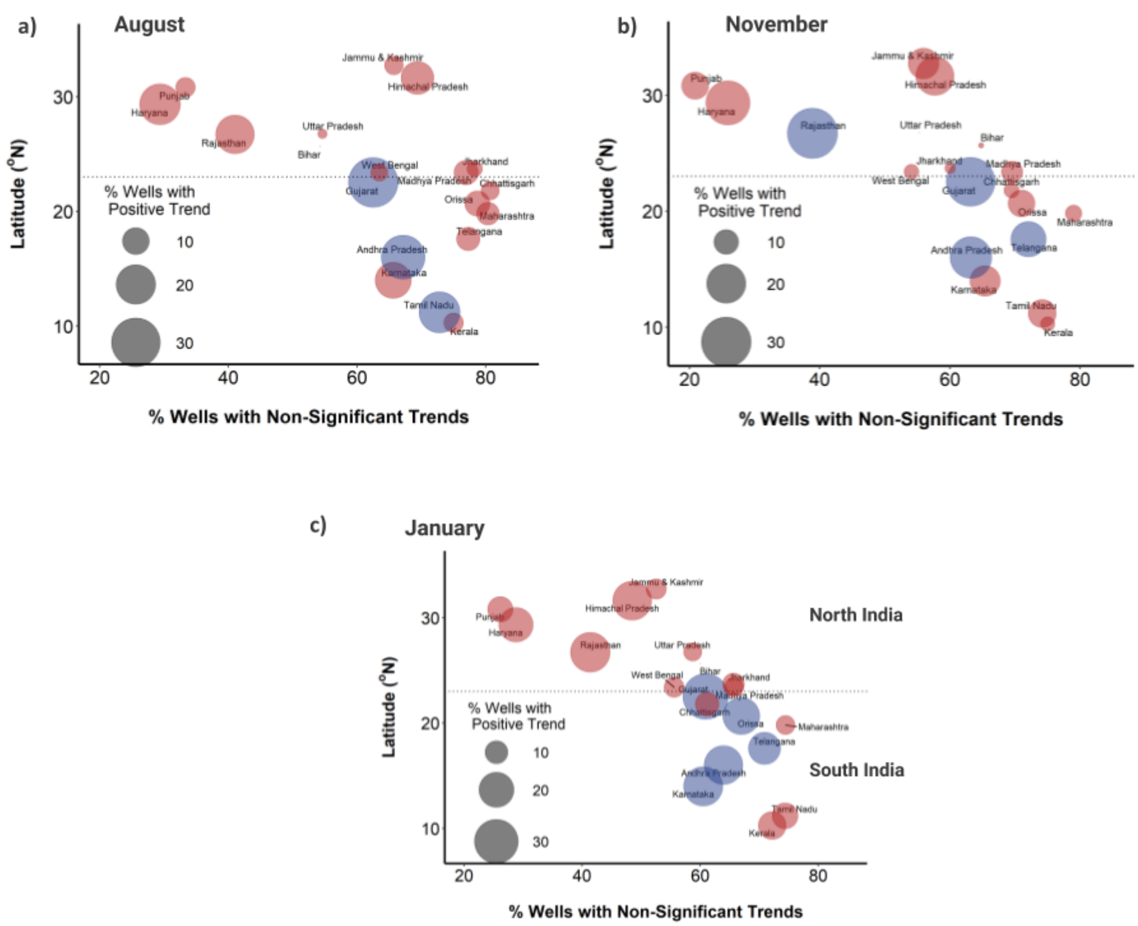


Figure B.7: Percentage of wells with non-significant trends ($p-value < 0.1$) in a) August, b) November and c) January. Each circle represents a State and the size of the circle represents the percentage of wells with ($p - value < 0.1$) positive trends. Red circles represents States with greater negative trends than positive, while blue circles represents states with more positive trends. The results are consistent with the May trends discussed in the main text. Results show that a majority of States have a large percentage of wells (>50%) have non-significant long-term trends.

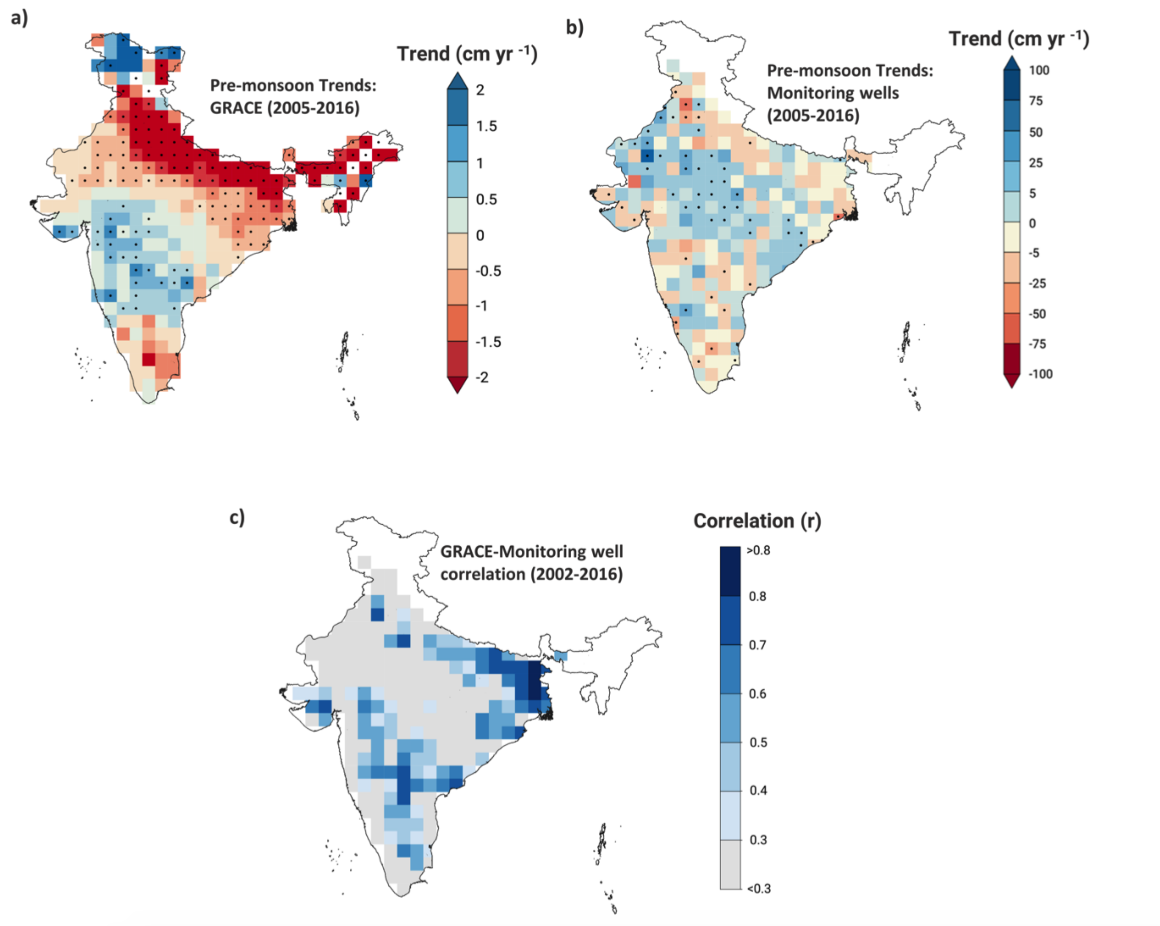


Figure B.8: Monthly trends ($cm\,yr^{-}1$) in groundwater from (a) GRACE anomaly, and (b) monitoring wells for the pre-monsoon season (May) between 2005-2016. Dots represents areas where the trends are significant ($p - value < 0.1$). Please note that the grid cell-level trends using monitoring well data was calculated after taking the median water level of all the monitoring wells falling within the grid cell. Results show that large regions in SI have non-significant trends. (c) Map of correlation coefficient between GRACE anomaly and monitoring well time series between 2002-2016. Results show that the correlation is stronger in Southern and Eastern India, and lower in North-West and Central India.

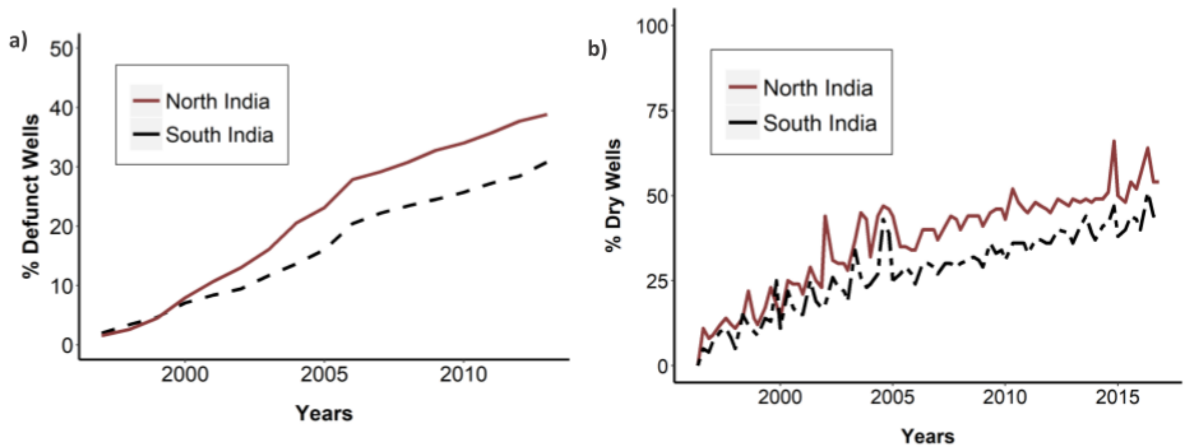


Figure B.9: a) Percentage (cumulative) of defunct wells in South and North India b) Percentage of dry wells in South and North India. Results highlight that both dry and defunct wells have been monotonically increasing in India, indicative of groundwater stress. Please note that there was no data recorded in August 2012 in the entire database, so that point was removed from the %dry and %defunct well calculations.

Table B.1: Description of the different data sources used to assess groundwater stress in South India

Data Source	Data Type	Scale	Time Period	Study	Data Source
GRACE Satellite	Hydrological	Regional	2002-2016	Asoka et al. (2017); Bhanja et al. (2017); Panda and Wahr (2016); Current Study	NASA/Jet Propulsion Lab
Monitoring Wells	Hydrological	Local	2002-2016	Asoka et al. (2017); Bhanja et al. (2017); Current Study	Government Agencies (e.g. CGWB)
Newspaper Articles	Social/Media	Local	1998-2017	Current Study	Factiva Database
Agricultural Census	Social Survey	Local	1996-2011	Current Study	Government Agencies; ICRISAT

Appendix C

Supplementary Material for Chapter 5

Table C.1: Crop parameters used in the analysis

Crop (mm)	L_{ini} (days)	L_{dev} (days)	L_{mid} (days)	L_{late} (days)	$K_{c,ini}$	$K_{c,dev}$	$K_{c,mid}$	$K_{c,late}$
Paddy	30	40	45	20	1.0	1.15	1.2	0.9
Sorghum	20	35	40	30	0.3	1.0	1.1	0.55

Table C.2: Tank equation based on field observation by Steiff (2016)

Volume-Stage	$Stage = (\frac{V_{tank}}{22914})^{(1/1.9461)}$
Stage-Area	$Area = (42942 * Stage)^{1.0993}$
Sluice-Stage	$Sluice(m^3) = \begin{cases} 12.6; & 0 < Stage < 0.785m \\ (Stage - 0.785) * 5.1903 * 86.4; & 0.785m < Stage < 1.185m \\ (Stage - 0.785) * 5.1903 * 86.4 + ; & Stage > 1.185m \\ \quad (Stage - 1.185) * 9.6768 * 86.4 & \end{cases}$

Table C.3: Precipitation thresholds used to determine antecedent moisture conditions

Antecedent moisture conditions	**Sum of precipitation over previous 5 days**	
	Monsoon Season	Dry Season
AMC_I	<13 mm	<36 mm
AMC_{II}	13-28 mm	36-53 mm
AMC_{III}	>28 mm	>53 mm

Table C.4: Equation used to estimate runoff using the SCS-Curve Number approach

Runoff (mm)	$I(t) = \frac{(P-0.2S)^2}{(P+0.8S)}$; where P is rainfall (mm)
Maximum Storage of Watershed (mm)	$S = (\frac{25400}{CN}) - 254$; where CN is the curve number
AMC_I	$CN_I = \frac{CN_{II}}{2.281-0.01281CN_{II}}$
AMC_{II}	$CN_{III} = \frac{CN_{II}}{0.427-0.00573CN_{II}}$

Table C.5: Criteria used to estimate the baseline Groundwater Spread Area (GSA) parameter in modelled system. For a GSA of 3 surface command area units (surface command area for modelled system is equal to 27 ha), we found maximum NSE-coefficient and minimal differences between modelled and estimated (based on field data) tank outflow volumes. Note, that for GSA >4 surface command area units the tank behaviour remained unchanged for the single season with field measured tank data.

Groundwater Spread Area	NSE Co-efficient	Percent difference between measured and modelled tank outflow volume		
(in units of surface command area)		ET	GW Exchange	Sluice
1	0.35	-18.8	68.49	-42.73
2	0.57	-6.69	38.3	-23.0
3	0.9	19.6	2.7	8.1
4	0.9	33.01	-10.62	19.33
5	0.9	33.01	-10.62	19.33
6	0.9	33.01	-10.62	19.33
7	0.9	33.01	-10.62	19.33
8	0.9	33.01	-10.62	19.33

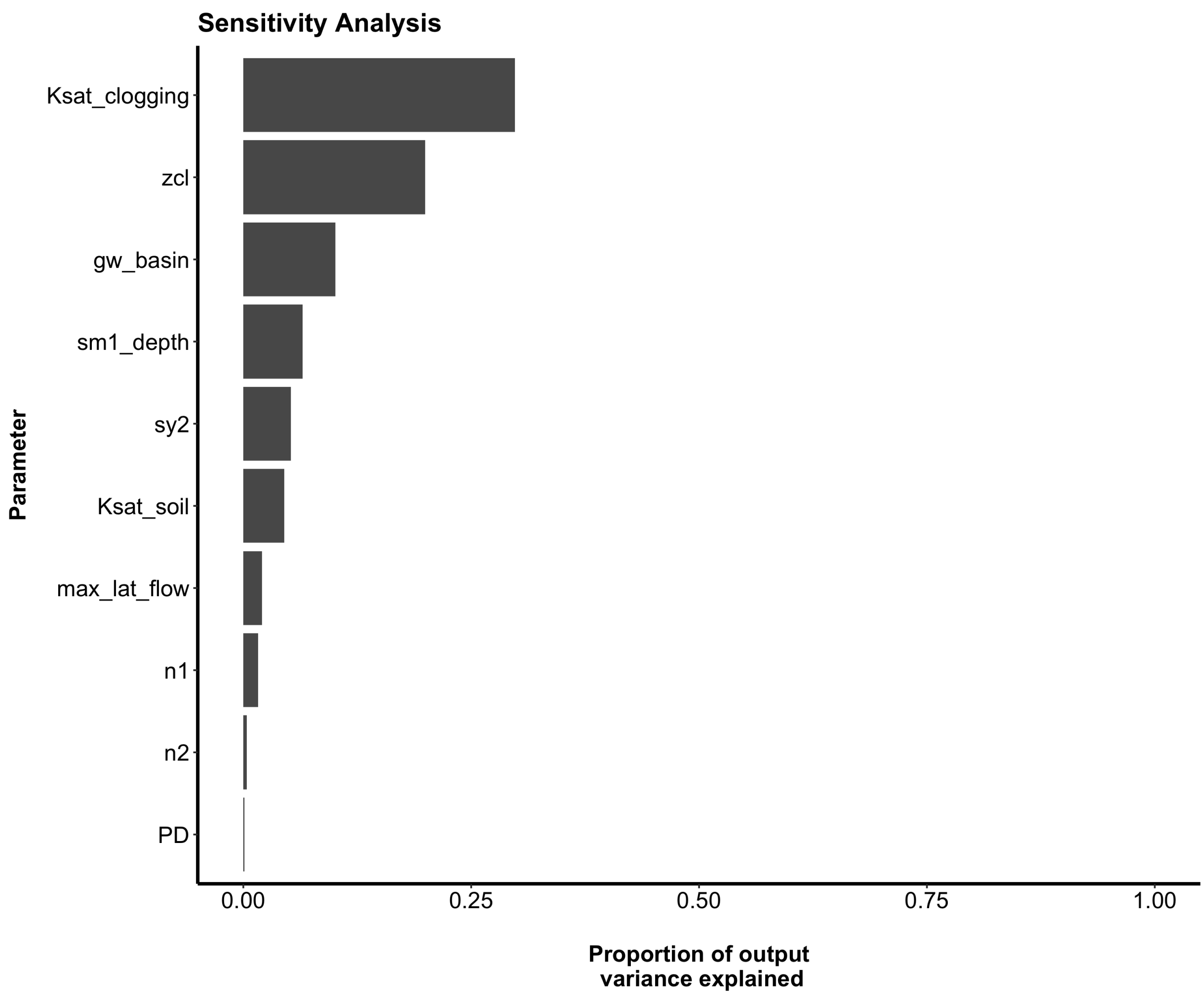


Figure C.1: Results from a partial sensitivity analysis using the FAST algorithm (Reusser et al., 2011). Plots highlight the extent of partial variance accounted for by each parameter. The plot shows the parameters exerting the greatest influence on the ratio of sluice and groundwater outflow from the tank. Results were obtained by running the model over a 20-yr period for 403 parameter sets.

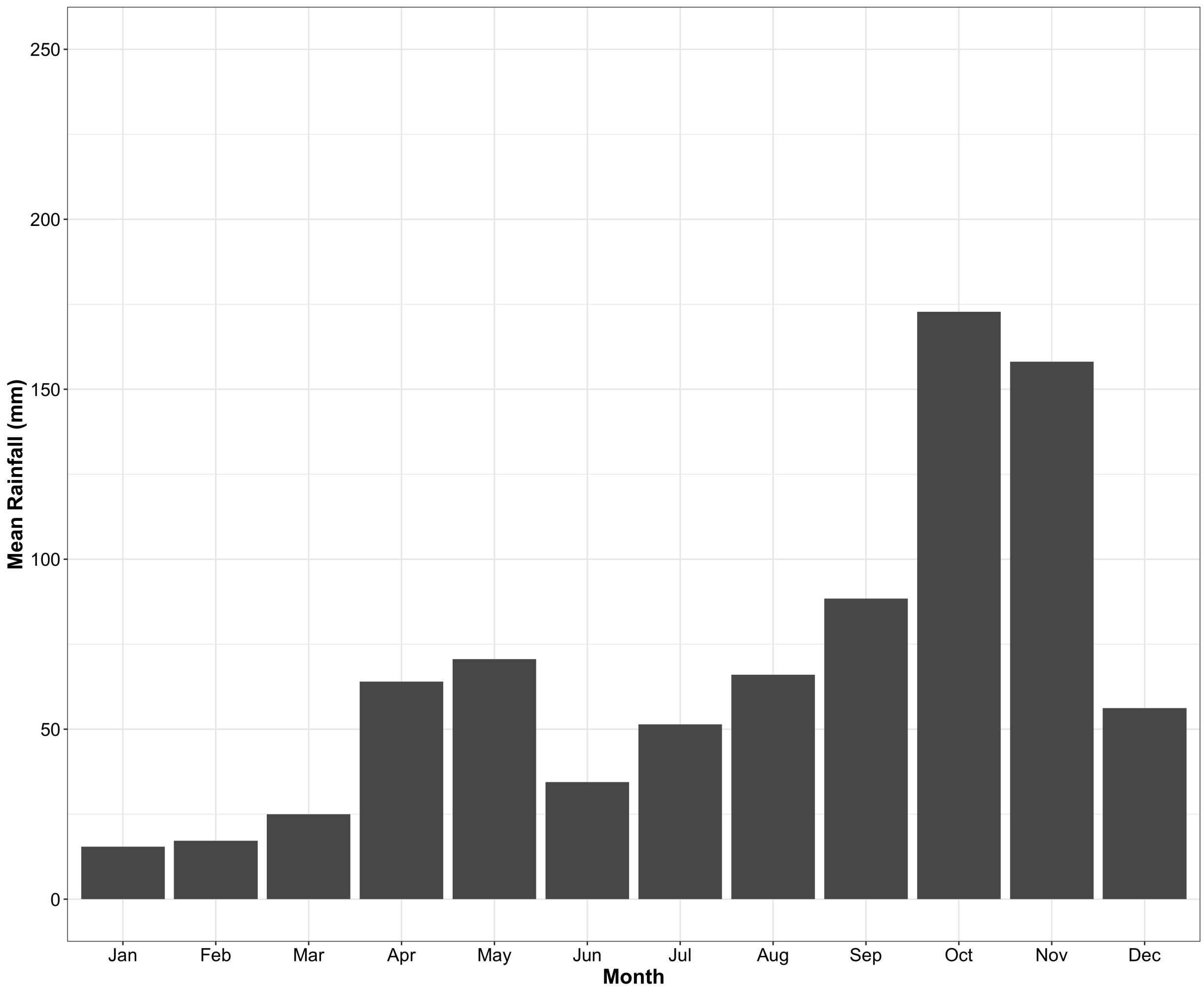


Figure C.2: Mean monthly precipitation amounts (in mm) for the 54-yr forcing data used in the analysis

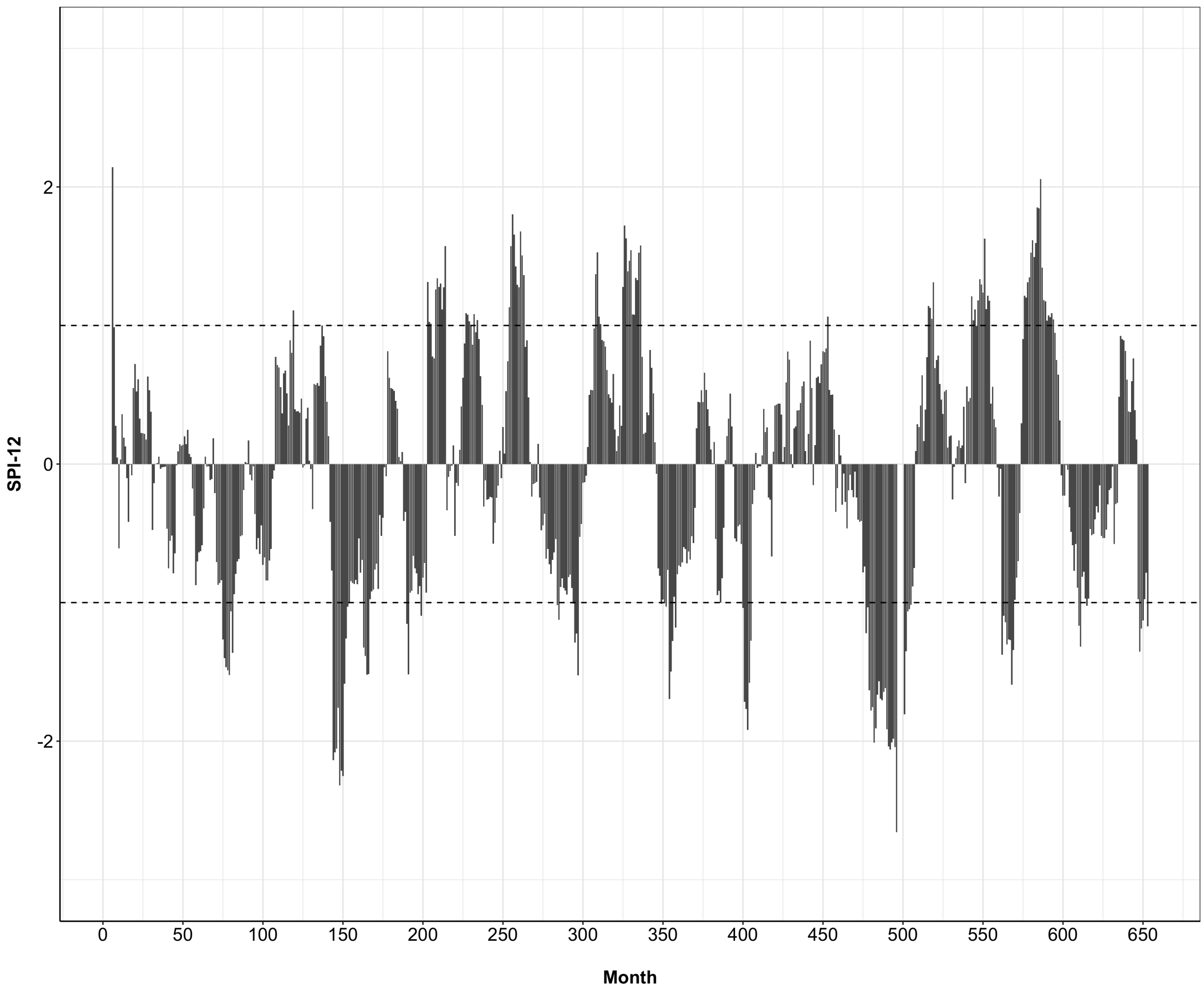


Figure C.3: 12-month Standardized Precipitation Index (SPI) time series for the 54-yr forcing data used in the analysis. SPI-12 values less than -1 represent drought periods

Table C.6: Pairwise comparisons using the Mann-Whitney U Test to assess difference in *monsoon* season %ET requirements met over the simulation period (54-years). Cells show the statistical significance of test results for each combination of model configuration (represented by rows and columns). $*$ represents results where the distribution means are significantly different ($p < 0.05$), while X represents results where distribution means are not statistically significant ($p > 0.05$).

	w/Tank Storage as a function of GSA (units: Ratio of GSA to Surface Command Area)									
	No Tank Storage	$\frac{GSA}{CommandArea} = 1$	2 CA	3 CA	4 CA	5 CA	6 CA	7 CA	8 CA	9 CA
1	*	NA	NA	NA	NA	NA	NA	NA	NA	NA
2	*	X	NA	NA	NA	NA	NA	NA	NA	NA
3	*	X	X	NA	NA	NA	NA	NA	NA	NA
4	*	X	X	X	NA	NA	NA	NA	NA	NA
5	*	X	X	X	X	NA	NA	NA	NA	NA
6	X	X	X	X	X	X	NA	NA	NA	NA
7	X	X	X	X	X	X	X	NA	NA	NA
8	X	X	X	X	X	X	X	X	NA	NA
9	X	X	X	X	X	X	X	X	X	NA
10	X	*	X	X	X	X	X	X	X	X

Table C.7: Pairwise comparisons using the Mann-Whitney U Test to assess difference in *Dry* season %ET requirements met over the simulation period (54-years). Cells show the statistical significance of test results for each combination of model configuration (represented by rows and columns). $*$ represents results where the distribution means are significantly different ($p < 0.05$), while X represents results where distribution means are not statistically significant ($p0.05$).

	w/Tank Storage as a function of GSA (units: Ratio of GSA to Surface Command Area)									
	No Tank Storage	$\frac{GSA}{CommandArea} = 1$	2 CA	3 CA	4 CA	5 CA	6 CA	7 CA	8 CA	9 CA
1	*	NA	NA	NA	NA	NA	NA	NA	NA	NA
2	*	X	NA	NA	NA	NA	NA	NA	NA	NA
3	*	X	X	NA	NA	NA	NA	NA	NA	NA
4	*	X	X	X	NA	NA	NA	NA	NA	NA
5	*	X	X	X	X	NA	NA	NA	NA	NA
6	*	*	X	X	X	X	NA	NA	NA	NA
7	*	*	*	X	X	X	X	NA	NA	NA
8	X	*	*	X	X	X	X	X	NA	NA
9	X	*	*	*	X	X	X	X	X	NA
10	X	*	*	*	X	X	X	X	X	X