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Streams, sewage, and shallow groundwater: stream-aquifer interactions in the Kathmandu Valley, Nepal

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Abstract

The Kathmandu Valley in Nepal is facing a water quantity and quality crisis due to rapid urbanization and haphazard water and wastewater planning and management. Annually, groundwater extractions in the Kathmandu Valley exceed capture, resulting in groundwater table declines. Streams are often important sources of recharge to (or destination of discharges from) aquifers. However, stream-aquifer interactions in the Kathmandu Valley are poorly understood. To improve this understanding, we performed topographic surveys of water levels, and measured water quality, in streams and adjacent hand-dug wells (shallow aquifer). In pre-monsoon, 12% (2018) and 44% (2019) of wells had water levels higher than adjacent streams, indicating mostly a loss of stream water to the aquifer. However, in post-monsoon, 69% (2018) and 70% (2019) of wells had water levels higher than adjacent streams, indicating that monsoon rainfall contributes to shallow aquifer recharge which, at least temporarily, causes streams to transition from losing to gaining. Concentrations of all water quality parameters (electrical conductivity, ammonia, alkalinity, and hardness) were higher in the pre-monsoon compared to post-monsoon in both streams and wells. There was no recurring trend in water level difference longitudinally from upstream to downstream. However, water quality in streams and wells depleted from upstream to downstream. While we clearly observed seasonal refilling of the shallow aquifer, the role of the deep aquifer in seasonal storage processes deserve future research attention.

Keywords Groundwater · Kathmandu Valley · Stream-aquifer interactions · Water management · Water quality

Introduction

Stream-aquifer interactions

Water, both groundwater and surface water, is fundamental to human life (Singh et al. 2019; Mishra et al. 2016). Surface water and groundwater are interconnected to each other; the change in quantity or quality of one will inevitably affect the other (Winter et al. 1998; Fleckenstein et al. 2010). A proper understanding of groundwater and surface water interaction is crucial for effective and sustainable water resource management (Oxtobee and Novakowski 2002; Sophocleus 2002; Brenot et al. 2015). The exchange between streams and aquifers may happen in three different ways: the stream is either (1) losing—stream water infiltrates into the aquifer, (2) gaining—groundwater flows into the stream, or (3) disconnected—losing stream that is disconnected from the aquifer by an unsaturated zone (Winter et al. 1998). The interaction between streams and aquifers can vary in space

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and time. Intuitively, when a losing stream is polluted, the water quality of the stream affects the quality of the surrounding groundwater. Additional increases in groundwater pollution will likely intensify water scarcity issues (UNESCO 2012). Therefore, the impaired water quality of the Kathmandu Valley's streams (The World Bank 2013; Regmi and Mishra 2016; Dhital 2017) illustrates the vulnerability of the groundwater and the relevance of this study. Water resources and freshwater quality are fundamental for sustaining aquatic diversity and sustainable eco-development of human civilization and water resources management (Mishra et al. 2021; Kumar et al. 2020). Ultimately, knowledge of stream-aquifer interactions is a crucial component of developing effective and sustainable water management plans that integrate the issues of water quantity and quality (Brenot et al. 2015). These stream-aquifer interactions are often characterized by high spatial and temporal variability, directly impacting the water balance and stream discharge (Krause et al. 2007). The seasonal variability of these interactions is mainly due to rainfall variability throughout the year. Besides rainfall, several factors like topography, geology, local aquifer system, etc., affect these interactions (Oxtobee and Novakowski 2002).

Kathmandu Valley water situation

Surface water supplies are unpredictable, scarce, and polluted. Therefore, depending on the time of year, groundwater meets between 50 and 75% of the residential, industrial, and agricultural water demands in the Kathmandu Valley (Gautam and Prajapati 2014). Rapid urbanization, inadequate infrastructure, and changing lifestyles and socioeconomics continue to increase demand for water (Kumar et al. 2020), increase the discharge of untreated wastewater into the rivers, and reduce groundwater recharge (Shrestha et al. 2012). Understanding stream-aquifer interactions, therefore, is critical for sustainable management of both water quantity and quality.

Currently, groundwater in the Kathmandu Valley (from now on Valley) is extracted from shallow and deep aquifers, separated by interbedded clay layers with varying thicknesses (Metcalf 2000). Before the 1970s, the shallow aquifer was the only source of groundwater production. Subsequently, mechanized extraction from the deep aquifer was started by industry and the private sectors. Water in the deeper aquifer is slowly affected by several anthropogenic activities (Shrestha et al. 2012). Extraction rates from the deep aquifer have continued to increase (Shrestha et al. 2012). Since groundwater withdrawal rates are estimated to be more than discharge that can be sustainably captured (Davids and Mehl 2015), groundwater levels have been declining since the 1980s (Metcalf 2000; Pandey et al. 2010; Shrestha et al. 2012). However, the spatial distribution

of impacts between the shallow and deep aquifer is poorly understood. Anecdotal evidence of progressively more stone spouts and shallow wells going dry each year supports the conclusion that the shallow aquifer is being negatively impacted by over-extraction (Shrestha et al. 2012). In addition to over-extraction, anthropogenic activities like disposal of industrial residue, leachate, and discharge of wastewater effluent also degrade water quality (Pandey et al. 2010). Various studies have shown declines in groundwater quality over time (Khadka 1993; Jha et al. 1997; Kharel et al. 1998; Metcalf 2000; Chapagain et al. 2009, Thapa et al. 2019). Also, the shallow aquifer is contaminated by nitrates and *Escherichia coli* (*E. coli*) and the deeper aquifer by ammonia, arsenic, iron, and heavy metals (Shrestha et al. 2015, 2016a, b).

In the Valley, very little research has focused on stream-aquifer interactions. Previous studies focused on the quality of either surface water or groundwater (for example, Khadka 1993; Chettri and Smith 1995; Jha et al. 1997; ENPHO 1999; Gurung et al. 2006). Recently, Malla et al. (2015) carried out a study to understand the stream and shallow wells interrelationship along the stream corridors of Bagmati and Bishnumati rivers using stable water isotopes and physico-chemical parameters. In their study, both stream water and shallow groundwater along the Bishnumati river were heavily contaminated due to uncontrolled dumping and discharge of untreated sewage wastes. Also, they found that a substantial portion of river water was mixed with the adjacent well, which can be toxic when rivers get contaminated.

Pathak et al. (2009) assessed the vulnerability in the shallow aquifers of the Valley and developed an aquifer vulnerability map using the GIS-based DRASTIC model. The map was developed incorporating the geologic and hydrogeologic factors influencing the contamination of shallow groundwater. In the aquifer vulnerability map, a high pollution potential index represents the increased possibility of leaching of contaminants in the groundwater, and a low index indicates the groundwater that is protected from contaminants. The prepared maps are essential tools for groundwater management and associated decision-making processes.

Bajracharya et al. (2018) quantified interactions between streams and underlying aquifer(s) and their implications. Using chemical parameters and stable water isotopes, they found that interactions affecting both stream and groundwater exist near river channels, and the direction of interactions varies by location. They also found that the streams in the Valley deteriorate from upstream to downstream. The monsoon overall improves chemical ion concentrations, with values decreasing nearly by one-half compared to pre-monsoon values. The study was limited to four watersheds over 9 months, so it recommends the collection of more water quality samples from wells and streams, in addition to data on groundwater levels and adjacent surface water levels.

In this context, the understanding of the stream-aquifer interaction in terms of both quality and quantity, its spatial and temporal variation is limited. This study depicts how these changes happen especially in urbanized cities such as Kathmandu Valley.

Objectives

The aims of this study were to (1) understand stream-aquifer interactions in the Valley, (2) compare these interactions between pre- and post-monsoon periods, and (3) investigate the impact of these interactions on water quality. This research focused on answering the following questions:

- (1) What is the pre- and post-monsoon status of stream-aquifer interactions for the primary tributaries to the Bagmati River within the Valley?
- (2) How do these interactions change longitudinally from upstream to downstream?
- (3) How do pre- and post-monsoon interactions relate to the stream and groundwater quality?

Study area

The Kathmandu Valley is located between 27.537° and 27.819° N latitude and 85.1919° and 85.5272° E longitude (Fig. 3) and is surrounded by hills in the outskirts (Shrestha et al. 2016a). The Valley is elliptical in shape with a diameter of 25 km N-S and 30 km E-W (Dill et al. 2001). The average altitude of the Valley is 1350 meter above sea level (masl), however, surrounding hills reach as high as 2800 masl in elevation (Shrestha et al. 2016b). The Valley is in a semi-tropical zone, has a warm and temperate climate, and receives more than 80% of its total annual rainfall during the monsoon between June through September (Karki et al. 2017). The average annual rainfall was 1340 mm and 1500 mm in 2017 and 2018, respectively, due to high winter rainfall in 2018 (DHM 2019). The annual average rainfall varies greatly in space, from around 1200 mm on the Valley floor to as high as 2400 mm in the surrounding hills.

The Valley and its surrounding hills consist of 400 million-year-old basement rock from Precambrian to Devonian age (Shrestha et al. 2012). This layer is covered with unconsolidated to partly consolidated Pliocene and Quaternary sediments (Stocklin and Bhattarai 1977). The thickness of this unconsolidated layer ranges from 10 m at the edges of the Valley to 500 m near the center and consists of fine texture sediment in the center and coarser sediment around it (Shrestha et al. 2012). The Japan International Cooperation Agency (JICA 1990) divided the Valley into three groundwater districts (Fig. 1). The Northern Groundwater District has high recharge potential and consists of unconsolidated and highly permeable sand and gravel, forming the main

aquifer in the Valley. The upper layer of the Central Groundwater District consists of very thick stiff black clay (Kalimati clay), unconsolidated coarse sediment of low permeability coarse sediment is found under this layer. This confined aquifer is stagnant and is not directly rechargeable vertically from above because of the Kalimati clay. The Southern Groundwater District consists of thick impermeable clay, and only along the Bagmati River between Chobhar and Pharping is there an alluvial aquifer (Shrestha et al. 2012). An important implication of this division is that recharge of the deep aquifer is likely low because of the Kalimati clay layer. However, the shallow aquifer does have the potential to be recharged, which is confirmed by the annually fluctuating levels from pre- to post-monsoon. Natural recharge of the aquifer is declining due to increased sealing (hardscaping) of the surface by urbanization which prohibits rainwater infiltration (Shrestha et al. 2012). Figure 2 provides a cross-sectional view of the subsurface geology and hydrogeological system (Shrestha et al. 2012).

Methods and materials

Monitoring locations

We performed measurements in three watersheds in the pre-monsoon of 2018, and in eight watersheds in the post-monsoon of 2018 and pre- and post-monsoon of 2019 (Fig. 3). The initial pre-monsoon measurements in 2018 were performed at 16 sites from 6 to 10 April 2018 and focused on streams overlying the highly permeable Northern Groundwater District, and therefore included the Bishnumati, Dhobi Khola, and Bagmati River watersheds. Post-monsoon 2018 measurements were performed between 6 and 29 September 2018 at 35 sites. We added five other watersheds in the post-monsoon, including the Manohara, Hanumante, Godawari, Nakkhu, and Balkhu. The additional watersheds were added to improve the spatial distribution of observations and investigate the impact of different geology, hydrology, and land-use on stream-aquifer interactions. The pre- and post-monsoon measurements of 2019 were performed at the same 35 sites from the post-monsoon 2018 in April and September 2019, respectively.

Three to ten monitoring locations were chosen for each watershed for field data collection (Fig. 3). Locations were chosen based on (1) the availability of dug wells in the shallow aquifer located close (i.e., within 100 m) to the selected streams, and (2) the desire to distribute sites from upstream to downstream as much as possible. Upstream measurement locations along the Bagmati are not equidistant because BA05 (between BA04 and BA06; not shown in Fig. 3), which was measured during pre-monsoon, was inaccessible during post-monsoon. Therefore, it was removed from our

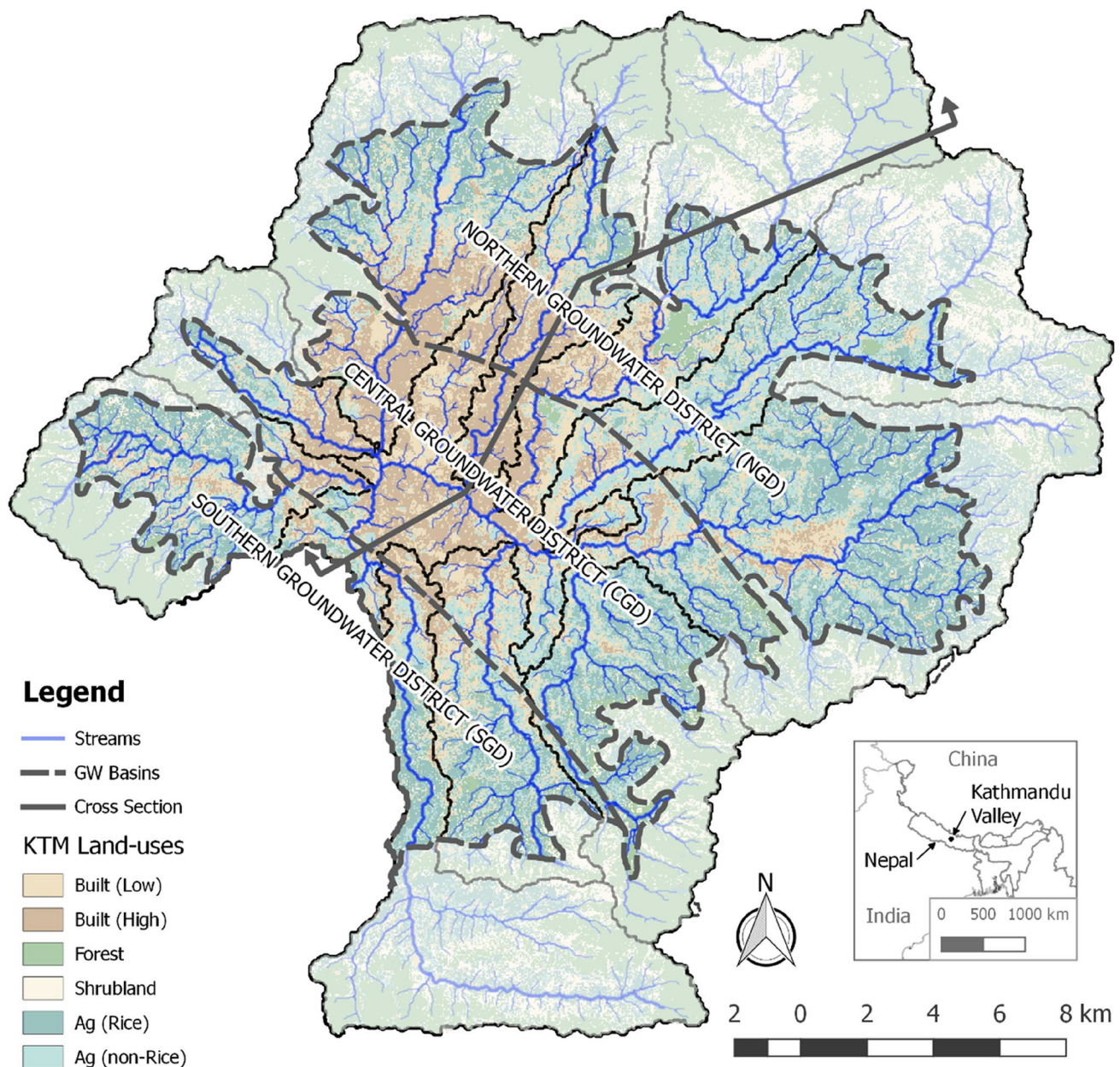


Fig. 1 Groundwater districts of the Kathmandu Valley. Edited from Groundwater management project in the Kathmandu Valley Final Report by the Japan International Cooperation Agency (JICA 1990). Retrieved from https://open_jicareport.jica.

https://open_jicareport.jica.go.jp/618/618/618_116_10869980.html October 18 2018. Copyright by JICA. Reprinted with permission. The indicated cross-sectional line is used for Fig. 2

analyses, and a new well nearby the previous one was chosen for measurements in 2019.

Data collection and analyses

This study is a part of a citizen science project called SmartPhones4Water or S4W (Davids et al. 2017, 2018, 2019). Open Data Kit (ODK) Collect, an Android smartphone application (ODK Collect; Anokwa et al. 2009),

was used to record and transmit collected data to a centralized database via Wi-Fi or cellular network. In addition to several other features, ODK Collect supports recording GPS locations, entering numerical data or text, and taking photographs of measurements in the field for data quality control. We developed a scalable ODK form that allowed multiple people to contribute quality controllable field data over the 2-year study.

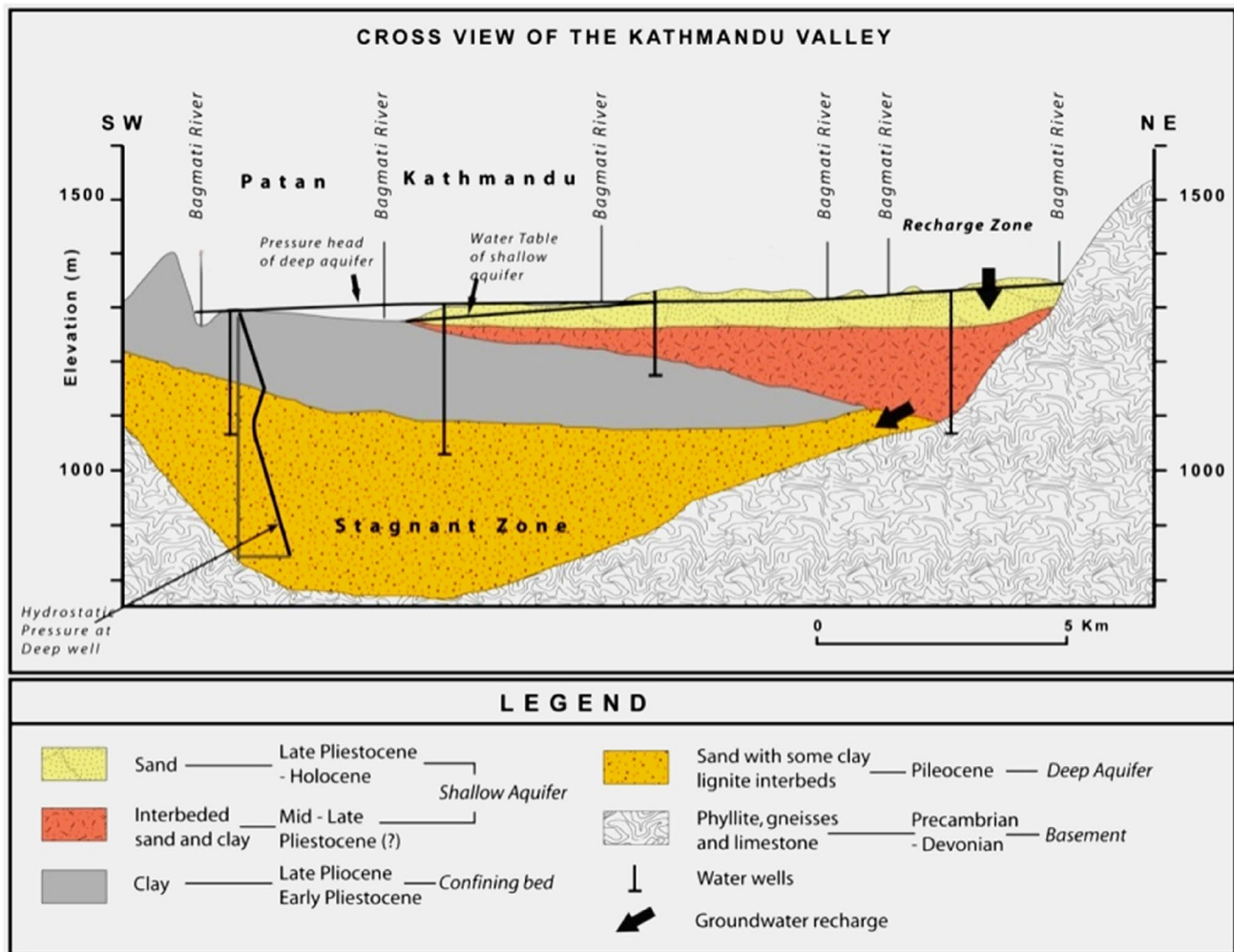


Fig. 2 Conceptual cross-section through the Kathmandu Valley Basin groundwater system. Edited from “A First Estimate of Ground Water Ages for the Deep Aquifer of the Kathmandu Basin, Nepal, Using the Radioisotope Chlorine-36” by Cresswell et al. 2001 retrieved from <https://onlinelibrary.wiley.com/doi/abs/10.1111/j.1745-6584.2001.tb02329.x>. Copyright by Cresswell et al. 2001 Reprinted with permission. Cross-section line shown in Fig. 1. Deposits within the Val-

ley contain multiple sand and gravel beds which form the principal aquifers in the northern and northeastern part of the Valley. In the central and southwestern parts of the Valley, these layers are overlain by a thick lacustrine clay layer that acts as an aquitard. The south and southeastern parts of the Valley consist of carbonate rocks which are classified as lower permeability aquifers

We developed Python scripts with Matplotlib extensions to create Figs. 6, 7, 8, 9, 10 showing water level differences, water quality parameters, and correlations using pre- and post-monsoon data. Trend lines have been made using the Numpy polyfit function (least squares polynomial fit). Box plots have been made using the pyplot Box plots function. Pearson correlation values are calculated using the Numpy correlation coefficient function *corrcoef*.

Water level measurements

All wells included in this investigation were hand-dug shallow wells with concrete ring casings. Depths of shallow wells ranged from 2.2 to 10.5 m. Well diameters ranged from

0.67 to 1.21 m. Groundwater extraction from monitoring wells was limited due to manual methods of water production with a bucket and rope. In some cases, abandoned wells nearby the streams were used for water level measurements, but an adjacent well with some amount of production was used for water quality measurements. This assisted with guaranteeing that the groundwater test was illustrative of conditions in the shallow groundwater, and not simply surface contamination introduced into the well that was not in use. We made efforts to avoid taking groundwater level measurements within a few hours of groundwater withdrawals. Because of the limited depths of these wells, the lack of penetration into potentially confined aquifer units, and the generally low production rates, we considered shallow well

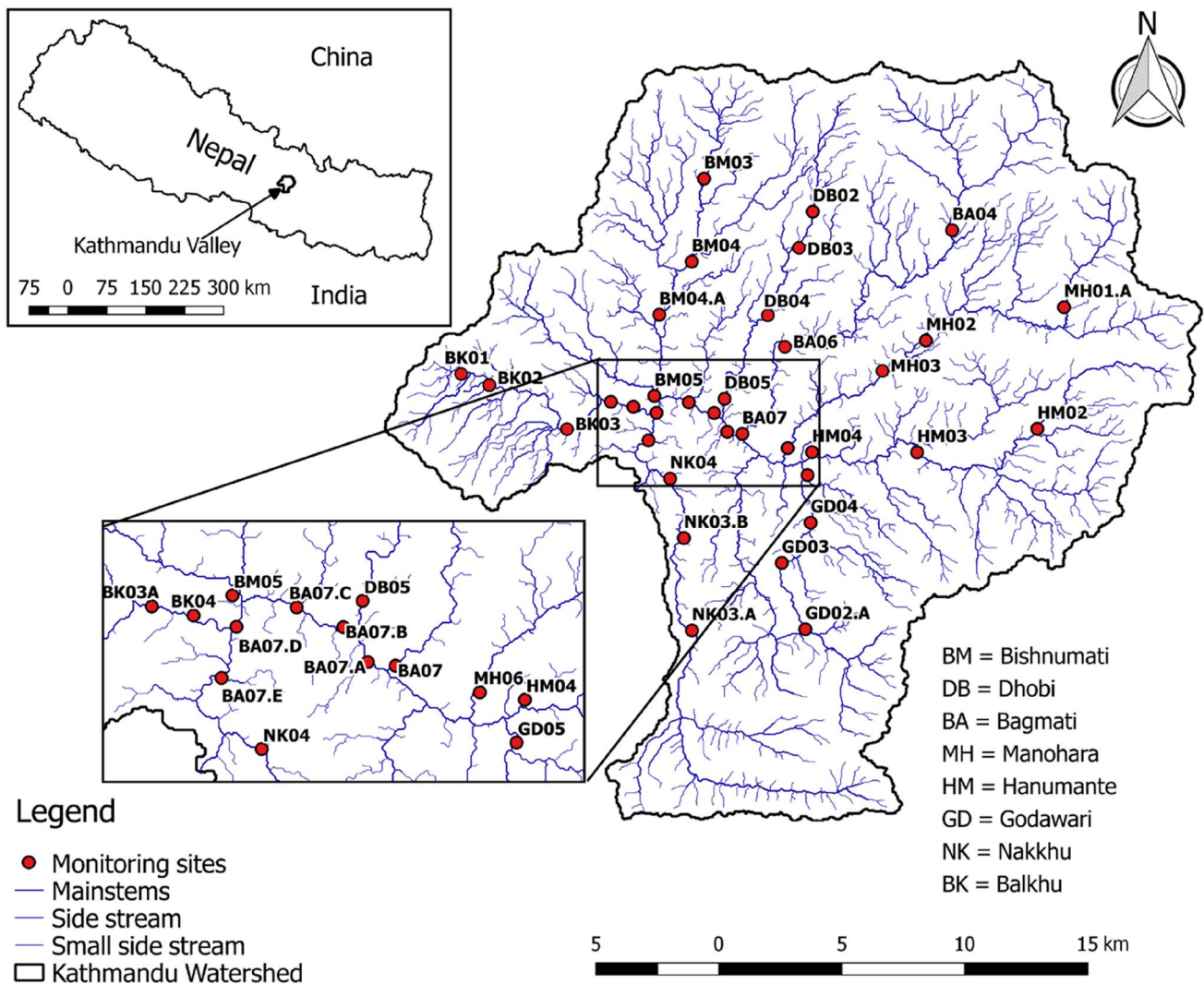


Fig. 3 Measurement locations in the Kathmandu Valley watershed with the network of the nine perennial streams used as a base map. Measurement at each site included a water level and quality measure-

ment in the stream in addition to a water level and quality measurement in an adjacent shallow (i.e., hand dug) monitoring well

levels to be representative of adjacent shallow groundwater table conditions.

To calculate stream-aquifer water level difference (Δh), topographic surveys of water levels in streams and adjacent wells were performed with a Topcon AT-B series 24× Automatic Level (Fig. 4). Topographic surveys and stream level and groundwater level measurements involved the following steps:

- (a) Selected shallow groundwater wells and stream water level measurement locations.
- (b) Identified, marked, and took pictures of reference points (RPs) on wells and benchmarks (BMs) near stream banks.
 - i. RPs were generally the top of the concrete rings used as the well casings.
 - ii. BMs were usually the top of retaining walls or the deck of bridges.
- (c) Performed a topographic survey to measure the difference in elevation between BMs and RPs (RP_BM).
 - i. In most cases, this involved a single tripod setup without any turning points.
 - ii. When sites required multiple setups, a closed loop survey was performed, and resulting errors were distributed between surveyed points.
- (d) Measured distance from BM to water surface elevation (BM_WSE).
 - i. Performed by lowering a measuring tape from the BM until it touched the water surface. By convention, these measurements were considered negative.
 - ii. Some sites were equipped with 1-m fiberglass staff gauges. In this case, water level readings were

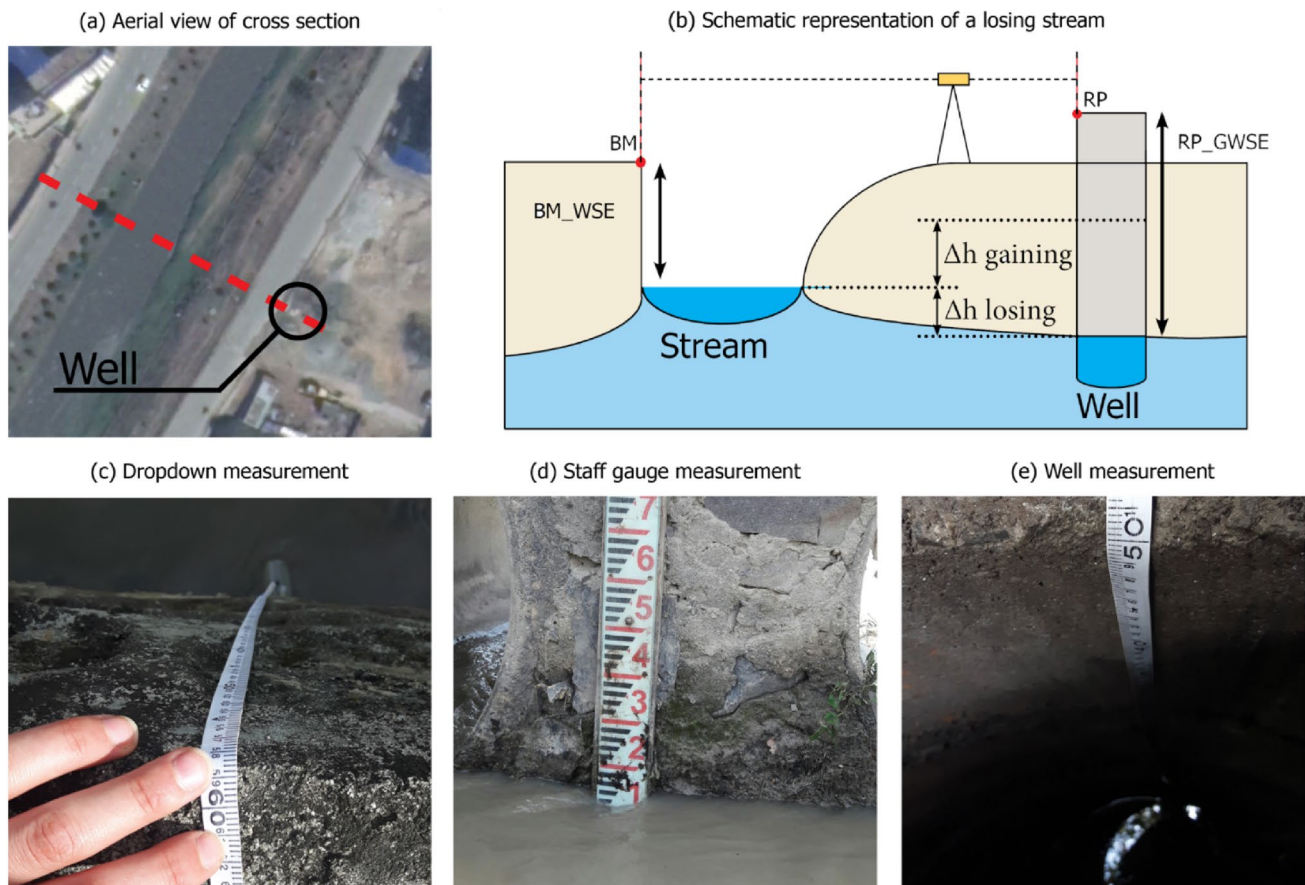


Fig. 4 Summary of stream-aquifer water level differences (Δh) measurements. Reference point (RP) and benchmark (BM) are indicated with red dots. Sub-panels include **a** aerial view of measurement site, **b** cross-sectional schematic, **c** sample measurement from BM to stream water surface elevation (WSE) as dropdown measurement,

d sample measurement from BM to stream WSE measurement with a staff gauge (note that the BM was considered the staff gauge zero mark), and **e** RP to groundwater surface elevation (GWSE) measurement in a well. Automatic level surveys were conducted to determine elevation differences between RP and BM

recorded, and the 0 mark of the staff gauge was surveyed as the BM. By convention, these measurements were considered positive.

- iii. In either case, photographs of BM to WSE measurements were taken in ODK Collect to provide quality control of data entry.
- (e) Measured distance from RP to groundwater WSE (RP_GWSE).
 - i. Performed by lowering a measuring tape from the RP until it touched the GWSE.
 - ii. Photographs of RP to GWSE measurements were taken in ODK Collect to provide quality control of data entry.
 - iii. By convention, RP to GWSE measurements were positive.

After performing the field measurements detailed above, we calculated RP to BM as:

$$RP_{BM} = RP_{Elev} - BM_{Elev} \quad (1)$$

where RP_{Elev} and BM_{Elev} are the elevations of the reference point and benchmark, respectively, from the topographic survey. We calculated the difference in stream and groundwater levels as:

$$\Delta h = RP_{BM} - BM_{WSE} - RP - GWSE \quad (2)$$

where BM_{WSE} is the distance from benchmark to water surface elevation in the stream (see step 4 above), and RP_{GWSE} is the distance from the reference point to groundwater surface elevation (see step 5 above). Using previously mentioned sign conventions, Δh was negative for losing streams and positive for gaining streams. For example, the stream in Fig. 4 is losing, so Δh would be negative.

The seasonal fluctuations of Δh of stream-well pairs of the selected watersheds are clearly shown in a watershed map with colored circles using Quantum Geographical

Information System (QGIS). Gaining stream locations are indicated with blue gradient circles, while losing stream locations are indicated with red gradient circles. Darker colors represent a larger absolute value of water level differences, either gaining or losing. We prepared such colored watershed maps for different seasons to learn about the temporal variation of stream-aquifer interactions. Pre-monsoon (blue) and post-monsoon (orange) water level differences, i.e., Δh for the selected watershed were also presented in a line graph using Python Matplotlib, as shown in Fig. 6.

Water quality measurements

We measured water quality parameters of both wells and streams to understand spatial and temporal water quality distributions better. Water quality also provided an additional and independent line of evidence for assessing stream-aquifer interactions. In both the pre- and post-monsoon of 2018 and 2019, we measured the following: electrical conductivity (EC), ammonia, phosphorus, and alkalinity. For general reference, the concentration limit set by the Government of Nepal and the World Health Organization (WHO) of each parameter is stated in Table 1. No health-based concentration limits for alkalinity and phosphorus are defined by both the Government of Nepal (GNP) and WHO and are therefore excluded from the table.

EC is an important water quality parameter because it shows a significant correlation with ten water quality parameters, including alkalinity, hardness, and chloride (Kumar and Sinha 2010). Previous research on pollution in the Kathmandu Valley also indicates EC covaries with several water quality parameters (Doorn et al. 2017; Davids et al. 2018). Phosphorus is found in natural rocks, domestic sewage, and decaying organic matter. In excess amounts, it can induce eutrophication in water bodies. Alkalinity is the water's capacity to resist changes in pH that would make the water more acidic.

We used a portable water quality test kit from the Environment and Public Health Organization (ENPHO) to measure ammonia and phosphorus. Water quality test strips from Baldwin Meadows were used to measure total alkalinity. At most sites, in-situ water quality testing was done. For

Table 1 Drinking water quality concentration limits for electrical conductivity (EC), ammonia, hardness, and chloride as set by the Government of Nepal (GNP 2008) and the World Health Organization (WHO 2017)

Parameter	Unit	GNP	WHO
Electrical conductivity	$\mu\text{S/cm}$	1500	2500
Ammonia	ppm	1.5	1.5

The units for each parameter are shown in parentheses after the name

the sites where in-situ testing was not possible, samples were taken to the S4W-Nepal office in polyethylene bottles to perform measurements later the same day. Polyethylene bottles were cleaned thoroughly before use and rinsed with sample water prior to sampling. A Greisinger GMH 3431 digital conductivity meter was used to measure in-situ EC and temperature.

Pearson's r (correlation coefficient; Rodgers and Nicewander 1988) was used to describe the strength of linear relationships between water quality parameters in streams and wells in pre- and post-monsoon season. Significance for correlations was tested with two-tailed p value hypothesis tests using an alpha level of 0.01.

Results and discussion

Results

Stream-aquifer water levels

Stream-aquifer water level differences (Δh) ranged between -4.29 m and 1.10 m in the pre-monsoon 2018, between -2.90 m and 1.28 m in the pre-monsoon 2019, between -1.34 m and 2.24 m in the post-monsoon 2018, and between -1.17 m and 3.37 m in the post-monsoon 2019 (Fig. 5). The average pre- and post-monsoon stream-aquifer Δh was -0.82 m and 0.44 m in 2018 and -0.11 m and 0.66 m in 2019, respectively. During pre-monsoon 2018, 14 out of 16 sites (88%) were losing water to the aquifer (negative Δh). In contrast, during post-monsoon 2018, only 11 out of 35 (31%) were losing. During pre-monsoon 2019, 19 out of 34 sites (56%) were losing water to the aquifer (negative Δh), and the remaining (44%) were gaining. During post-monsoon 2019, 10 out of 33 sites (30%) were losing water to the aquifer (negative Δh), and the remaining (70%) were gaining. Twelve of the fourteen sites that were losing in pre-monsoon transitioned to gaining in the post-monsoon in 2018, whereas 9 of the 19 sites that were losing in pre-monsoon transitioned to gaining in the post-monsoon in 2019. In every case, groundwater levels increased from pre-monsoon to post-monsoon (exception: MH01); the average increase from the 16 wells monitored in both seasons was 1.99 m in 2018, and the average increase from the 33 wells monitored in both seasons of 2019 was 0.99 m. There is a decrease in Δh in four sites (DB02, DB03, BA06 and MA01) from pre- to post-monsoon 2019 and an increase in Δh in two sites (DB03 and MH01) from post-monsoon 2018 to pre-monsoon 2019. This may be due to excessive groundwater extraction in the nearby groundwater wells before or during the time of measurement. There is comparatively lower Δh in most of the sites in pre-monsoon 2018 than pre-monsoon

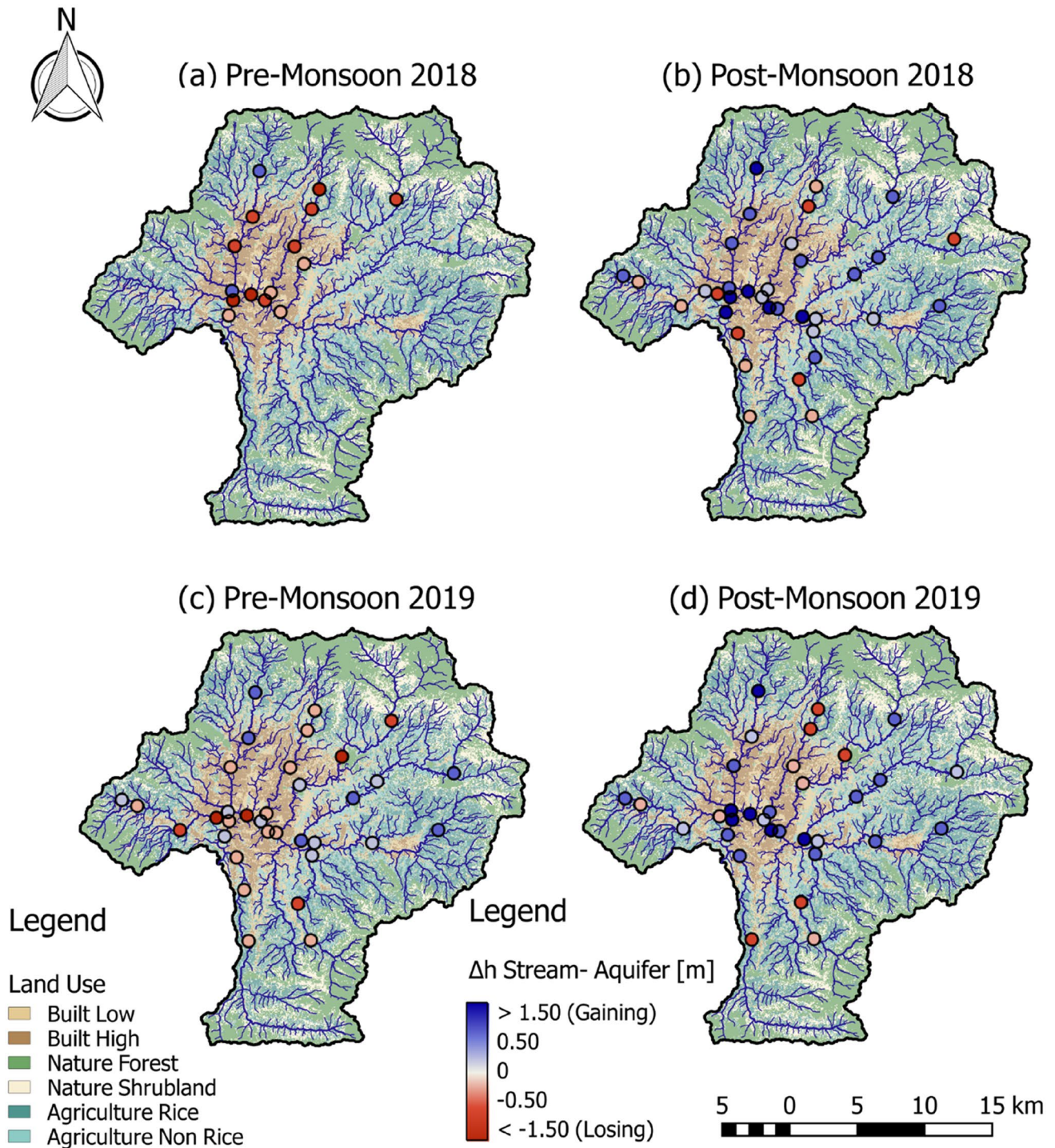


Fig. 5 Stream-aquifer water level differences in meters for **a** pre-monsoon 2018 ($n=16$), **b** post-monsoon 2018 ($n=35$), **c** pre-monsoon 2019 ($n=34$) and **d** post-monsoon 2019 ($n=33$) in the Kathmandu Valley. Land-use and stream network data are used as a base map

(Davids et al. 2018). Gaining stream locations are indicated with blue gradient circles, while losing stream locations are indicated with red gradient circles. Darker colors represent a larger absolute value of water level differences, either gaining or losing

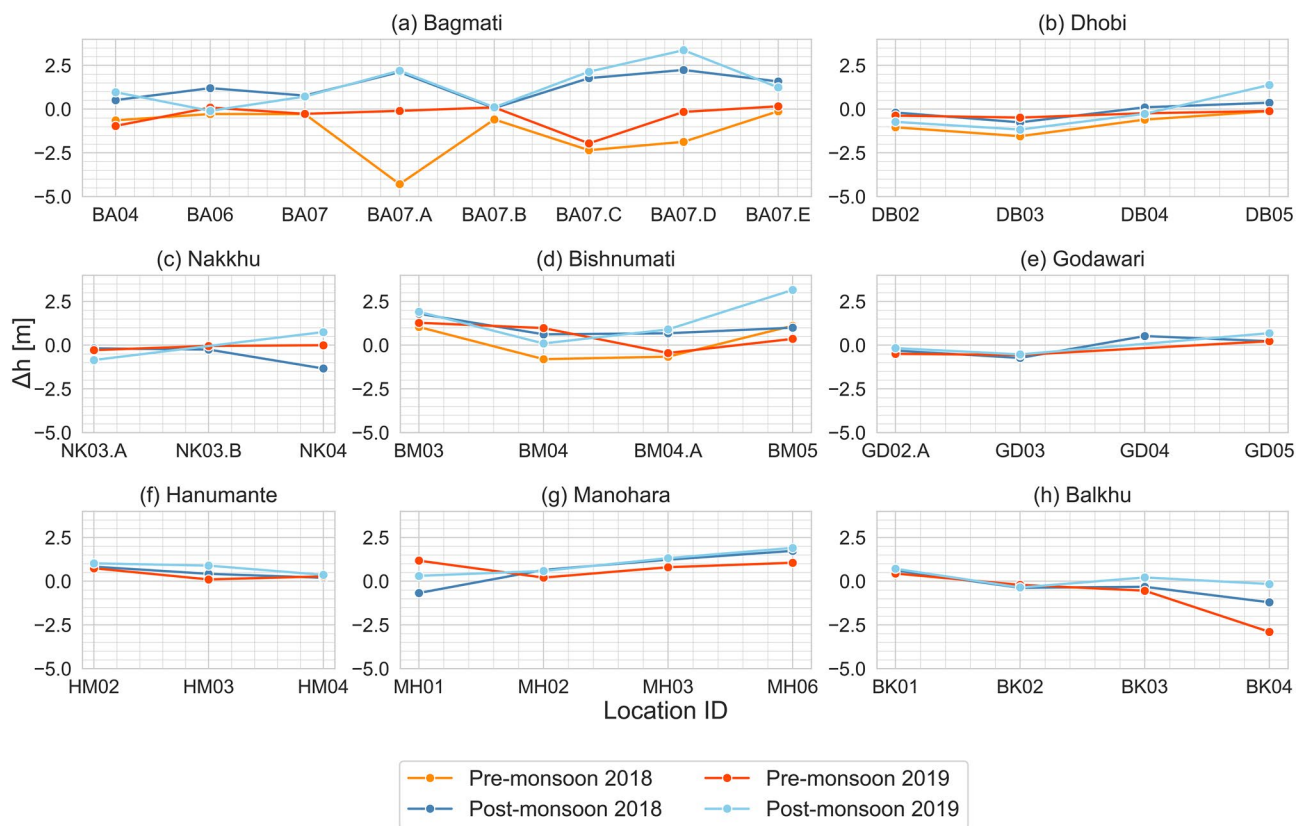


Fig. 6 Pre-monsoon (orange) and post-monsoon (blue) water level difference for the selected eight watersheds ((a) Bagmati, (b) Dhobi, (c) Nakkhu, (d) Bishnumati, (e) Godawari, (f) Hanumante, (g) Manohara, and (h) Balkhu). On horizontal axes, measurement locations are

labeled, and vertical axes show stream-aquifer water level differences. Measurements were limited to Bagmati (a), Dhobi (b) and Bishnumati (d) in pre-monsoon 2018

2019 due to the dry winter of 2018. The winter rainfall of 2019 (110 mm) was ten times more than in 2018 (10 mm).

Consistent longitudinal (upstream to downstream) trends among streams in pre- or post-monsoon were not observed (Fig. 6). Hanumante and Balkhu rivers showed decreasing trends in Δh from upstream to downstream, while Godawari, Manohara, and Dhobi Khola showed the opposite. With some exceptions, Δh is generally higher in post-monsoon than pre-monsoon. The Bagmati River showed a full transition from completely losing in pre-monsoon to completely gaining in post-monsoon (except BA05).

Regular measurements at BM05, BA07, and DB05 were performed in 2018 to improve the understanding of short-term variations and trends in stream-aquifer water level differences (Fig. 7). All sites showed linear trends in stream-aquifer water level difference (Fig. 7a), with two decreasing (BA07 and DB05) and one increasing (BM05). Groundwater level changes contribute most to the temporal variations of the water level difference, except for the second half of October 2018 for BM05 (Fig. 7b). For example, groundwater levels decreased by 0.9 m and 1.0 m, while stream

water levels decreased by 0.3 m and 0.1 m for BM07 and DB05, respectively (Fig. 7b). These measurements showed that DB05 had already transitioned from gaining to losing in early October. Extrapolation of BA07's linear trend indicated that this site most likely also transitioned from gaining to losing by the end of October. In contrast to BA07 and DB05, stream-aquifer water level differences at BM05 increased during the period of ongoing monitoring (Fig. 7a). Viewing stream and groundwater levels separately for BM05 (Fig. 7b) showed that stream levels declined as expected during the post-monsoon hydrograph recession period; however, groundwater levels unexpectedly increased, especially from the middle of October onward.

Water quality

Concentrations of all water quality parameters were higher in the pre-monsoon compared to the post-monsoon. Streams showed relatively larger differences in distributions from pre- to post-monsoon, while well differences were generally

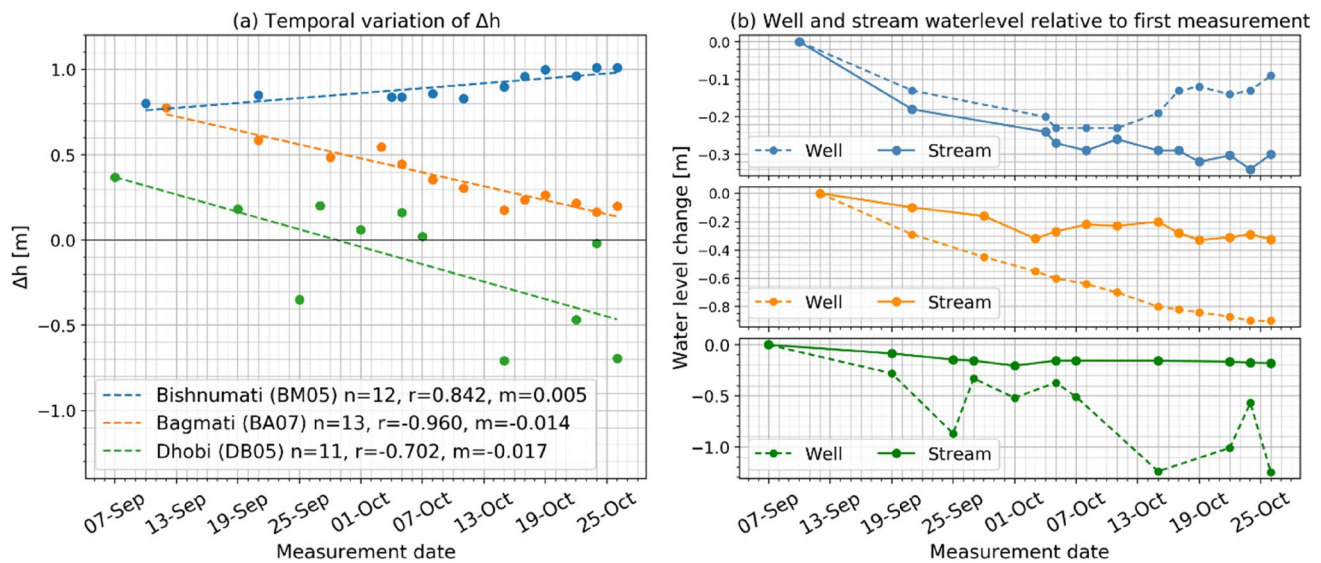


Fig. 7 Graphs showing temporal variation of the stream-aquifer water level difference (a) and water level changes for both wells and streams (b) at Dhobi (DB05; blue), Bagmati (BA07; orange), and Bishnumati (BM05; green). In the left graph (a), measurements are

indicated as points. Dashed lines represent linear trend lines, with the indicated sample sizes, slopes (m) and Pearson's r values. For the right graph, the vertical axis represents the water level difference (m) relative to each sites' initial measurement from early September

Table 2 Pearson's r (correlation coefficients) for the water quality parameters measured in streams and wells

	Stream			
	EC	Ammonia	Alkalinity	Phosphorus
Well				
EC	0.61	0.50	0.38	0.60
Ammonia	0.55	0.53	0.39	0.49
Alkalinity	0.46	0.46	0.49	0.56
Phosphorus	0.15	0.21	0.10	0.26

Values are calculated using combined pre- and post-monsoon data. Statistically significant values are shown in bold. For all correlations with alkalinity (no data available for pre-monsoon 2018): $n=107$, $p=0.01$, r critical=0.25. For correlations with EC, ammonia, and phosphorus: $n=124$, $p=0.01$, r critical=0.23

smaller. Measured EC values ranged from 32 to 1331 $\mu\text{S cm}^{-1}$, while ammonia levels ranged from 0.0 to 3.0 ppm. Alkalinity and phosphorus ranged from 0 to 240 ppm and 0 to 1 ppm, respectively.

In general, water quality deteriorated in pre- and post-monsoon from upstream to downstream for both streams and wells (specifically focused on EC). As observed in Fig. 9, pre- and post-monsoon differences in stream EC (solid lines) were larger than differences in well EC (dashed lines). In the pre-monsoon, the stream and well EC were similar. In the post-monsoon season, well EC generally exceeded stream EC at the same monitoring location. However, EC values from BM05 did not follow these general trends.

E. coli was found in all stream water samples for both pre- and post-monsoon, indicating a high likelihood of fecal contamination due to untreated waste disposal. Considering only wells with pre- and post-monsoon 2018 data ($n=16$), *E. coli* was found in 75% of wells (12 out of 16) and 63% of wells (10 out of 16) in the pre- and post-monsoon, respectively. For all post-monsoon 2018 wells, *E. coli* was present in 41% of wells (14 out of 35). In general, *E. coli* counts increased from upstream to downstream. Additional *E. coli* data are available as supplementary material.

Most parameters—13 out of 16 possible pairs—have statistically significant correlations between streams and wells (Table 2). However, phosphorus in wells appears to be uncorrelated with concentrations in streams. (For all correlations with alkalinity, there is no data available for pre-monsoon 2018): $n=107$, $p=0.01$, r critical=0.25. For correlations with EC, ammonia, and phosphorus: $n=124$, $p=0.01$, r critical=0.23).

Focusing on the diagonal of the correlation matrix (Table 2), we observed seasonal (pre- to post-monsoon) shifts in the relationships between EC in streams and wells. The linear correlations between EC were statistically significant in both pre- and post-monsoon. In the pre-monsoon, well EC was on average lower than stream EC, leading to a trend line slope of less than one ($m=0.94$). However, in the post-monsoon, well EC was on average higher than stream EC, leading to a trend line slope of greater than one ($m=1.87$). The remaining water quality parameters did not show the same seasonal shifts.

Discussion

Nature and distribution of stream-aquifer interaction

In general, streams were losing water (88% (2018) and 56% (2019)) to the shallow aquifer during pre-monsoon and were gaining water (69% (2018) and 70% (2019)) from the shallow aquifer in the post-monsoon (Figs. 5 and 6). In pre-monsoon, no recurring trend in water level difference was seen longitudinally from upstream to downstream. In post-monsoon, most losing monitoring locations were upstream, away from the valley floor and most gaining locations were downstream, in the Valley floor (Fig. 6).

Due to groundwater extraction and minimal recharge, groundwater levels in the shallow aquifer decrease in the pre-monsoon season (Brindha et al. 2014; Zhu et al. 2019). Monsoon rainfall recharges the shallow aquifer, as evidenced by increasing groundwater levels (Prajapati et al. 2021). This impact is predominantly visible on the Valley floor. This seasonal dynamic is less apparent in upstream sites, which still tend to be losing year-round, indicating a continuous recharge of the shallow (and potentially deep) aquifer(s) (Bajracharya et al. 2018; Malla et al. 2015). For example, in the Northern Groundwater District, an area of highly permeable sands and gravels (Shrestha et al. 2012), two monitoring sites (DB02 and DB03) were losing water to the aquifer in post-monsoon.

Similarly, in the Southern Groundwater District, upstream sites of Nakkhu and Godawari watersheds were losing water to the aquifer in both pre- and post-monsoon. There are relatively high permeability alluvial deposits of sand and gravel overlying lower permeability metamorphic formations along the stream corridors of the Southern Groundwater District (Shrestha et al. 2012). These narrow alluvial deposits support growing groundwater extractions for municipal, industrial, and agricultural uses, while at the same time, the surrounding watershed is undergoing rapid urbanization and hardscaping. While the Nakkhu and Godawari Rivers used to flow perennially and support populations of fish and recreational swimming, they now go dry upstream of the confluence with the Bagmati. Since shallow groundwater levels indicate that these rivers are now losing even in post-monsoon, the river now dries when runoff from the rainfall or inflows from the upper catchment cease. In contrast to Nakkhu and Godawari, all monitoring locations on the Hanumante and Manohara watersheds were gaining water from the aquifer in both seasons.

Repeated stream-aquifer water level measurements at DB05 and BA07 revealed that the transition to gaining did not persist for long (Fig. 7). DB05 already transitioned

from gaining to losing by early October. Extrapolation of the linear trend in levels for BA07 ($r = 0.96$) indicated that this site most likely also transitioned from gaining to losing by the end of October. This suggests that it may only be a short-term mounding of shallow aquifer levels along the stream alignments that seasonally recharge due to high streamflow from monsoon rains. High flows also could scour the streambed, causing potentially order of magnitude increases in streambed hydraulic conductivity. This theory could be tested by performing (1) lateral groundwater level transects running perpendicular to stream alignments and (2) precision Real-time Kinematic (RTK) GPS surveys of reference points and benchmarks. Combining these datasets will allow the construction of a three-dimensional groundwater surface model and the computation of groundwater flow directions. These data could also be used to refine a numerical groundwater flow model. Also, future stream-aquifer measurements should be repeated more frequently (e.g., weekly) after monsoon rains end to try and capture the temporal and spatial fluctuation of the transition from gaining to losing streams.

Repeated stream and groundwater level measurements at BM05 (Fig. 7) show an increase in groundwater levels in late October. Due to these unexpected results, we performed EC measurements in four wells surrounding the initial well, which all indicated much higher EC values similar to other values from the Valley floor. While we do not have sufficient information to understand the mechanisms for this discrepancy, our working hypothesis is that there must be a source of groundwater recharge other than precipitation nearby this well. Potential sources include leaky water distribution or sewage pipes. However, the reasons for the timing of these observed increases in shallow groundwater levels are unknown.

Although the methods used gave a good insight into the direction of interactions (i.e., gaining or losing), understanding their magnitude was not possible with the current methodology. Including a survey on the hydraulic conductivity (K) of the streambed and aquifer at the different monitoring locations would lead to information about the specific discharge between the stream and the aquifer. Eventually, this information would be key to setting up a water balance. However, the determination of hydraulic conductivity is generally characterized by large uncertainties, because the outcomes may vary over some order of magnitudes and are highly variable in space and time (Kalbus et al. 2006). The K value changes in time due to scour from high flows, deposition, and degree of saturation of the soil, therefore long-term monitoring would be needed (Kalbus et al. 2006). Also, the anisotropy of the soil has to be taken into account as the vertical (K_V) and horizontal (K_H) hydraulic conductivity may differ if the soil is not structure-less (Stibinger 2014). Previous research has shown that the K value in the

shallow aquifer differs from 12.5 to 44.9 m day⁻¹ in the Valley (Pandey et al. 2010). Considering the vertical component in groundwater between the stream and the aquifer, to construct a flow field map, a piezometer nest will have to be installed with two or more piezometers installed at the same location at different depths (Kalbus et al. 2006).

Streambed hydraulic conductivity (K_S), an important parameter for aspect in quantifying stream-aquifer exchanges. K_S can be estimated using two piezometers, installed above and below the semi-permeable streambed. Based on this pressure difference and the depth of the layer between the piezometers, a good estimation for the streambed hydraulic conductivity can be made. Also, a more common and practical method is the field standpipe permeameter test (Unnikrishnan and Sarda 2016).

Impact of stream-aquifer interaction in groundwater quality

Stream and shallow groundwater quality in the Valley deteriorated longitudinally from upstream to downstream (Bajracharya et al. 2018). In the pre-monsoon, most monitoring locations were losing, and observed shallow groundwater quality was similar to stream water quality. In post-monsoon, most monitoring locations had transitioned to gaining, and stream water quality was better than shallow groundwater quality (Malla et al. 2015). Intuitively, groundwater quality improvements from pre- to post-monsoon were not as large as stream water quality improvements (Figs. 8, 9, 10).

In pre-monsoon, our results suggest that polluted stream water infiltrates into the shallow aquifer. In post-monsoon, stream water quality improves more than shallow groundwater quality. During the monsoon, streamflow is diluted by relatively high-quality water, thus improving water quality (Bajracharya et al. 2018). Measurements were performed during base flow conditions (avoiding rainfall events), so increased streamflow is likely caused by increased groundwater discharge from the upper catchments, not by runoff from the urbanized Valley floor. There is a significant difference in water quality entering the Valley floor from headwater catchments with natural land uses (Davids et al. 2018). The shallow groundwater quality does not improve as quickly because the amount of groundwater in storage is high, and groundwater flow velocities are low, which leads to a slower rate of change (Bajracharya et al. 2020). It is also possible that shallow aquifer recharge in the Valley floor is of lower quality because of extensive overlying-built land-uses.

The study shows that the degree of stream-aquifer interaction and the associated water quality vary significantly between and within streams, and it is strongly influenced by several factors i.e. local climate, geological features,

land-use, etc. (Morris et al. 1997). Even though groundwater quality is a complex issue influenced by many factors, the dynamic linkage we observed between streams and shallow groundwater should not be neglected when managing water sources and wastewater in the Valley. Upstream sites tend to be losing year-round, so efforts should focus first on protecting and improving water quality in headwater catchment areas, and priority should be given to the potential recharge area in the northern groundwater district. Since Davids et al. (2018) showed that land-use is one of the main reasons for the deteriorating stream water quality longitudinally, establishing protections for natural and agricultural land uses should be a top priority for water managers. For the same reason, we suggest starting upstream and moving downstream when building sewage collection and treatment systems.

Limitations

It is crucial to mention that these results are only applicable to the shallow aquifer within the corridors nearby the streams we measured. Seasonal refilling of the shallow aquifer in these observed areas should not be misconstrued to suggest that the deeper aquifer does (or does not) undergo similar seasonal refilling. Instead, to address questions about the deep aquifer and intermediate confining beds of Kalimati clay, observations from monitoring wells penetrating these units are needed. Additionally, vertical gradients between different aquifer layers cannot be quantified with our methodology. Instead, vertical fluxes should be assessed with measurements of piezometric surfaces from carefully constructed multi-completion monitoring wells with discretely screened piezometers in the respective aquifer and aquitard zones of interest.

Despite our efforts to capture base flow conditions and avoid measurements during rainfall, an important limitation of our research methods was that the measurements represent a specific point or 'snapshot' in time. When considering water level difference measurements, vertical components of groundwater flow were not considered since this would require that a piezometer nest (or multi-completion well) would be necessary (Kalbus et al. 2006). This was well beyond the scope and budget of this investigation which leveraged existing hand dug wells. This research is limited by the availability of dug wells penetrating into the shallow aquifer. For some watersheds (e.g., Manohara), it was difficult to find suitable wells located relatively even distances (longitudinally) from each other along the streams.

Additionally, water level differences merely indicate the driving potential for groundwater flow, but an understanding of streambed hydraulic conductivities is necessary to translate potential into an actual flux. In this study, we did not measure streambed hydraulic conductivities, so it is not

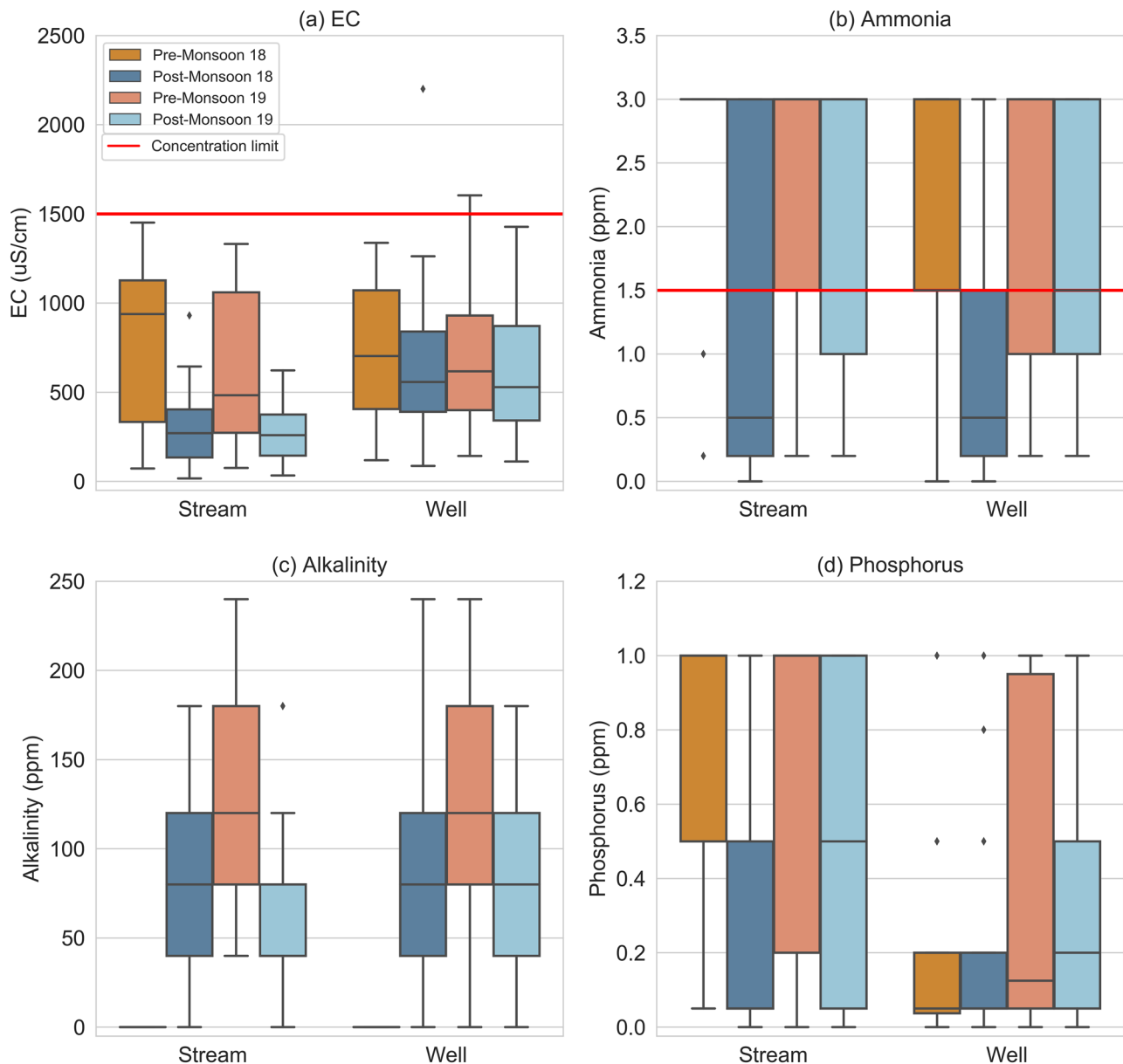


Fig. 8 Box plots showing the distribution of water quality parameter values for **(a)** EC, **(b)** Ammonia, **(c)** Alkalinity, and **(d)** Phosphorus for streams and wells for all sites in the pre- (orange) and post-monsoon (blue). For general reference, the Government of Nepal concentration limit is shown as a red line for EC, ammonia, chloride, and hardness (Government of Nepal 2005). Boxes show the interquartile range between the first and third quartiles of the dataset, while whiskers extend to show minimum and maximum values of the distribution, except for points that are determined to be “outliers,” which are

more than 1.5 times the interquartile range away from the first or third quartiles. Outliers, shown as diamond-marks, have been made partially transparent to show the presence of identical values. Alkalinity measurements were performed with Baldwin Meadow strips, where values are measured in increments of 40 ppm. These large increments provide low resolution alkalinity data and result in a more similar value between streams and groundwater, which introduce uncertainty in our analysis

possible to perform this analysis at this time. Future research should evaluate different methods for quantifying these conductivities and applying the preferred approach through our study area. Because of low flow (pre-monsoon) deposition of fine particles and algae, and high flow (monsoon) erosion and scour events, streambed conductivities are likely to vary

in time. Therefore, it is also important to repeat these measurements at least pre- and post-monsoon.

As ENPHO Water Test Kit and Baldwin Meadow strips have some limitations in measurement range and resolution, the obtained values from the analysis are an approximate estimation of drinking water quality. For a more accurate

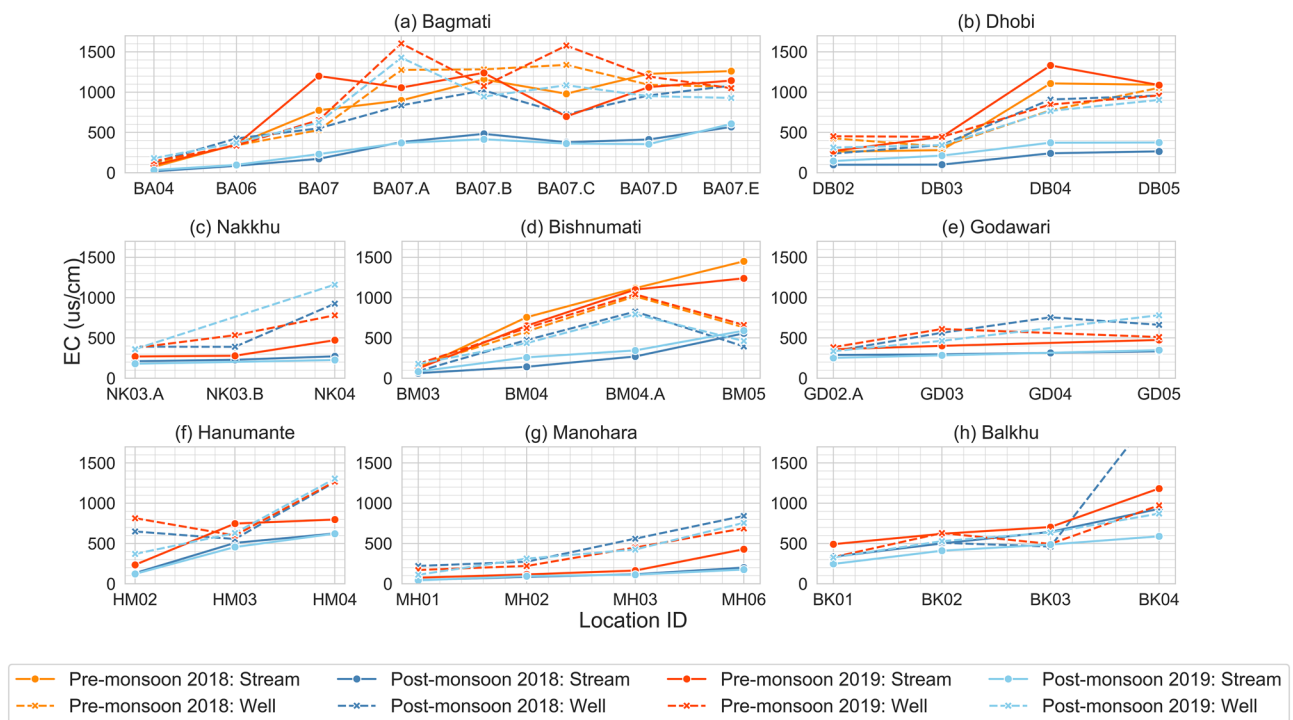


Fig. 9 Electrical conductivity (EC) of streams (solid lines) and wells (dashed lines) in pre- (orange) and post-monsoon (blue) for the selected eight watersheds ((a) Bagmati, (b) Dhobi, (c) Nakkhu, (d) Bishnumati, (e) Godawari, (f) Hanumante, (g) Manohara, and (h) Balkhu)

analysis of the drinking water quality, laboratory analysis is required.

Conclusion

While several studies have highlighted extensive overdraft in the Kathmandu Valley, our results suggest that despite increased groundwater extraction and urbanization, seasonal (monsoon) refilling of the shallow aquifer still occurs substantially within stream corridors. This seasonal refilling leads to most streams (i.e., 70% of sites in both 2018 and 2019) being gaining in the immediate post-monsoon. However, after a currently unknown period after post-monsoon, the streams we measured (i.e., 88% (2018)/ 56% (2019) of sites) transition, so that by pre-monsoon they are once again losing. Our preliminary findings from repeated measurements at two sites suggest an insight that the transition from gaining to losing after monsoon rains end happens relatively quickly, perhaps by early to mid-October.

We also find a clear connection between the water quality of streams and shallow groundwater. Concentrations of all water quality parameters (electrical conductivity, ammonia, alkalinity, and hardness) were higher in the pre-monsoon compared to post-monsoon in both streams and wells. Untreated sewage, being directly discharged into

the Valley's rivers, negatively impacts the streams themselves and the underlying shallow groundwater system. There was no recurring trend in water level difference longitudinally from upstream to downstream. However, water quality in streams and wells depleted from upstream to downstream. Unfortunately, our results also indicate that the “flushing” effect of monsoon rains that dramatically—albeit temporarily—improves stream water quality, is not as effective at “flushing” out the increasingly contaminated shallow groundwater system.

As this research only represents four “snapshots” in time, it is critical that such measurements be continued at these (and possibly other) sites on a regular basis. Only after a time series of a longer period, i.e., 5–10 years, is available will a more robust understanding of stream-aquifer interactions—and how these are changing in space and time—be possible. With this in mind, we developed the methodology to be as simple as possible.

Despite the limitation of existing data, the results of this study should help further surface water-groundwater interaction research in the Kathmandu Valley, and highlight the essential role of young researchers in generating these data. Further research should focus on conducting a detail study of subsurface lithology, chemical and isotopic analysis. Future work should also focus on assessing this method (measuring water level difference) to determine

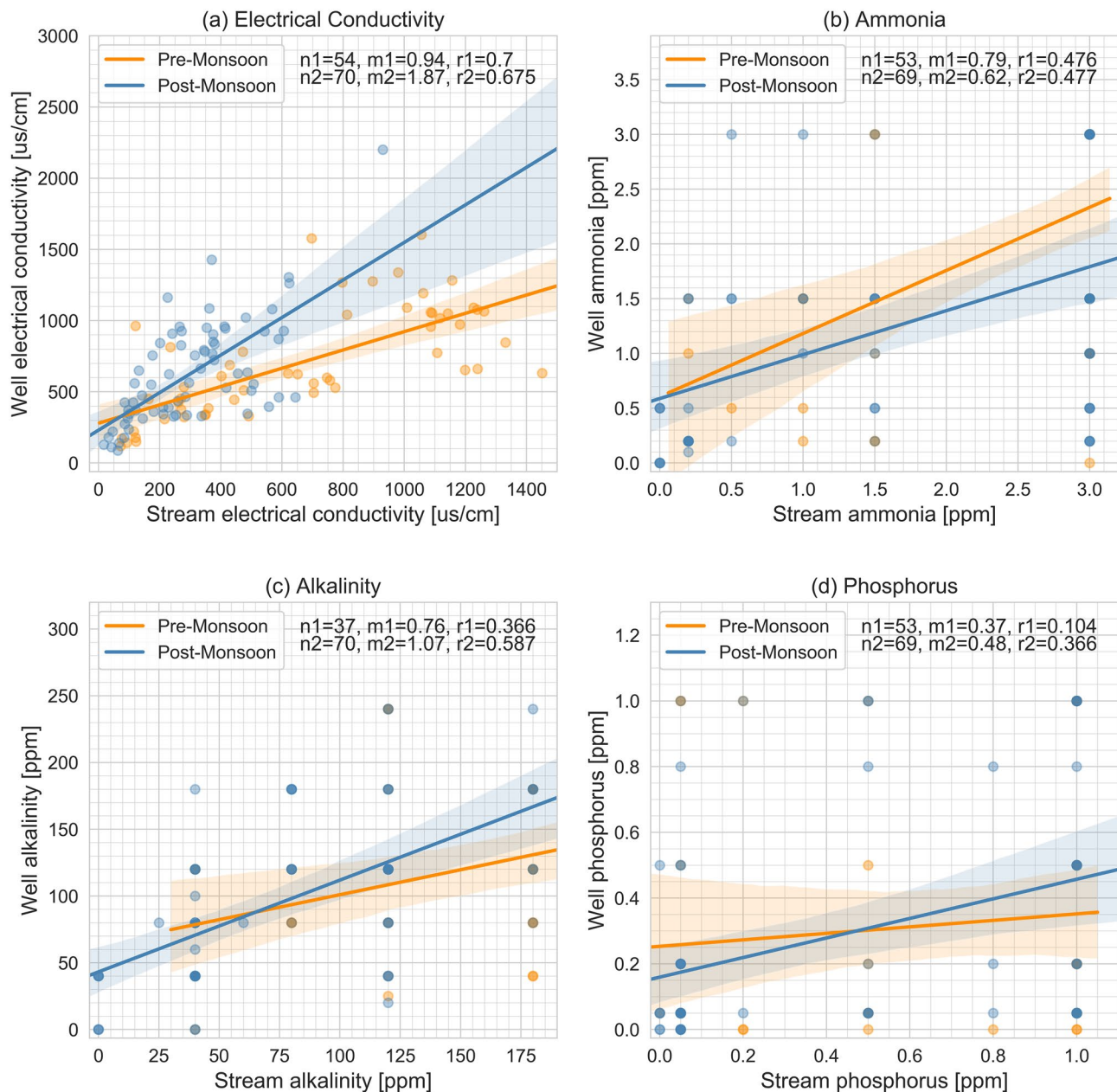


Fig. 10 Pre- and post-monsoon scatterplots of stream and well water quality results for (a) EC, (b) ammonia, (c) alkalinity, and (d) phosphorus. The water quality value of the stream and well are shown on the horizontal and the vertical axes, respectively. The number of measurements (n), correlation coefficient (r), and the slope of the trend line (m) per parameter and season are shown in the legends. Markers are partially transparent to show the presence of overlapping (identical) values. The following critical values were used: $n=16$,

critical $r=0.623$; $n=33$, critical $r=0.430$; $n=34$, critical $r=0.437$. ENPHO Water Test Kit measured ammonia and phosphorus on a scale from 0 to 3 ppm and 0 to 1 ppm, respectively. We found that ammonium concentrations at downstream sites often exceeded 3 ppm. This made it impossible to see any variation in concentration beyond the 3 ppm upper threshold, which introduced uncertainty in our correlation analysis

the direction and magnitude of the groundwater flow and therefore the stream-aquifer interactions. Computer modeling might be helpful for making detailed flow maps of the groundwater at the monitoring locations. In situ cone penetration test would be valuable to determine local soil conditions.

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Author contributions JCD and RP devised the project, the main conceptual ideas, and experimental design. Fieldwork was performed by JCD, RP, NNO, NM, KH, RB, AD, BM, ND, ABT, and SU. VP, RM, BRT, and RT contributed to the interpretation of the result. JCD, NNO, and RP prepared the manuscript with valuable input from all co-authors. All authors provided critical feedback and helped shape the research questions, methods, and manuscript.

Declarations

Conflict of interest The authors declare no conflict of interest.

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