

MONITORING SHALLOW GROUNDWATER QUALITY IN THE SPRINGFIELD PLATEAU OF MISSOURI VIA KARST SPRINGS

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Abstract

Shallow groundwater plays an ever-important role in determining surface water quality and stream ecology. Since shallow groundwater is so readily affected by contaminants produced by human activities, its discharge to surface water may contribute to stream contamination; therefore, its monitoring is critical for protecting water resources. Seven rural karst springs from an area utilized for cattle grazing and pastureland in southwest Missouri, USA, were monitored for a year. Shallow wells in the vicinity of the sampled springs, plus two urban springs, were included to appraise the effect of land use on water quality. The study focused on $\text{NO}_3\text{-N}$ and Cl as indicators of pollution. The results show (a) a specific pattern of concentrations for each spring and the season in which the concentration of pollutants peaked, (b) the underlying Ozark Aquifer was not significantly affected by leakage from the Springfield Plateau aquifer, and (c) compared with the urban springs, the rural springs had about the same $\text{NO}_3\text{-N}$ content but less Cl . Two rural springs reached concentrations above 4 mg/L $\text{NO}_3\text{-N}$ during the spring months, a concentration that may lead to eutrophication. The area experienced an abnormally dry period during this study, for which the results represent baseline conditions and the springs' response to an expected effect (longer droughts) of climate change.

INTRODUCTION

Compared to shallow aquifers, deep aquifers are commonly preferred as a drinking water source because they commonly have a better water quality that results from being farther from surface-derived human contamination (agricultural, sewage, and industrial), a distance that also allows for natural self-cleaning processes to take place (Zheng et al., 2005; Li et al., 2021). The longer and more tortuous path of groundwater as it moves to deeper parts of the aquifer cleanses contaminants by filtration as well as by chemical and biological processes (Gorelick and Zheng, 2015; Hare et al., 2021). Thus, shallow aquifers are less frequently used for drinking purposes and, therefore, monitored less.

Recent advances in the study of near-surface water processes and their association with the decline and degradation of water resources, such as the worldwide decline in the flow of streams and springs (Weissinger et al., 2016; Ranjan and Pandey, 2019; Currell and Katz, 2022) and degraded groundwater and stream water quality (MacDonald et al., 2016; Hare et al., 2021), have renewed the attention to shallow groundwater systems and the role these systems play in surface water quality and stream ecology (Lewandowski et al., 2020; Hare et al., 2021). Degradation of the water quality of streams has manifested predominantly in regions impacted by agricultural (Boumaiza et al., 2022; Jia and Qian, 2025) or urban pollution (Popp et al., 2024).

Shallow aquifers readily reflect changes in land use and are more vulnerable to pollution (Knierim et al., 2015; Lewandowski et al., 2020; Hare et al., 2021; Li et al., 2021). For example, changing forested land to agricultural land can cause induced groundwater recharge but also an increase in salinity due to the leaching of surface salts, fertilizer, and other surface contaminants (Scanlon et al., 2007; Bouselsal and Saibi, 2022). The degradation of water quality is becoming a critical problem in many different groundwater systems worldwide, such as semi-arid areas and overexploited aquifers (Bouselsal and Saibi, 2022). Currently, there are few environments where water resources are not threatened (Currell and Katz, 2022). Karst areas are also highly vulnerable to groundwater contamination due to groundwater flowing through large fractures and cave systems without natural filtration (Kresic, 2013; Maas et al., 2019; Tobin et al., 2021). Hence, the monitoring of shallow aquifers is important for the potential input of contaminants to their receiving streams and/or deeper aquifers and for the future availability of water resources (Weissinger et al., 2016; Lewandowski et al., 2020; Ferencz et al., 2022; Currell and Katz, 2022).

In non-industrialized areas, groundwater contamination commonly comes from human activities, including sewage, septic tanks, and urban runoff. This contamination can be toxic and long-lasting when persistent chemicals such as pharmaceuticals and pesticides are present (Adamski, 1997; Pope et al., 2009; MacDonald et al., 2016). As the contaminants present in sewage and agricultural waste can be many, consisting of one or more different solutes, a particular chemical is commonly utilized as an indicator. The choice of an indicator depends on the type of pollution expected in a particular region. Chemical compounds commonly utilized as indicators of agricultural and human-related contamination include nitrate (NO_3), total dissolved solids (TDS), and chloride (Cl). These chemical compounds are good water

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quality indicators because they are present in measurable concentrations, relatively soluble, stable under oxic conditions, and are associated with pollutants such as pesticides, pharmaceuticals, and organic micropollutants (Panno and Kelly, 2004).

Nitrate is a soluble, stable component under oxidizing conditions and has been labeled an anthropogenic contaminant because of its association with agricultural and domestic wastes (Jia and Qian, 2025). The nitrogen (N) cycle is complex; in agricultural areas, N can take various forms depending on the redox conditions. Under oxidizing conditions, it is mainly present as nitrate (NO_3^-), and under reduced conditions, NO_3^- is reduced to nitrite (NO_2^-), N_2O gas (nitrous oxide), and dinitrogen gas (N_2). Sources of N include the irrigation returns of overfertilized fields (either from synthetic N-fertilizers or manure), septic tank effluents, and treated and non-treated wastewater (Boumaiza et al., 2022; Hamlin et al., 2022). Nitrate is either expressed as NO_3 or the nitrogen in nitrate ($\text{NO}_3\text{-N}$). The U.S. Environmental Protection Agency drinking water guideline for nitrate is 10 mg/L $\text{NO}_3\text{-N}$ (44.3 mg/L NO_3) (USEPA, 2012). However, recent research has found that even smaller concentrations, e.g., 4.0 mg/L $\text{NO}_3\text{-N}$ may cause eutrophication of downstream water bodies and, if chronically ingested, serious health problems, including various types of cancers, hypothyroidism, and birth defects (García-Torres et al., 2022; Hamlin et al., 2022).

The impact of shallow groundwater input on stream quality has become an increasing concern after degradation in the quality of streams in various parts of the world became evident (MacDonald et al., 2016; Hare et al., 2021; Popp et al., 2024). A thorough monitoring of shallow groundwater quality and the various pathways that groundwater takes toward its receiving stream has been increasingly studied (Cey et al., 1998; Zheng et al., 2005; Hare et al., 2021). For example, streams carrying a large load of nutrients (especially NO_3 as it is the limiting factor for growth in ocean water) can cause ocean dead zones (Tagne and Dowling, 2020). Monitoring and the implementation of strategies to reduce the level of contaminants (e.g., nutrients) is needed, even at the small watershed scale (Tagne and Dowling, 2020). Water quality of the Springfield Plateau aquifer, a shallow carbonate aquifer, is reported here based on the monitoring of several karst springs (Fig. 1).

Seven rural and two urban karst springs discharging from the Springfield Plateau aquifer of southwest Missouri (Figs. 2 and 3) were monitored in 2023. Karst springs represent the hydrological connection between groundwater and surface water, and their water quality reflects the various activities occurring at the surface (Knierim et al., 2015; Katz, 2020; Currell and Katz, 2022). The land around the studied rural springs is used for pastureland and cattle grazing; six springs are in Lawrence County and one in the rural part of Greene County (Figs. 2 and 3). Lawrence County contains the largest number of cattle in Missouri (USDA, 2025). Tall fescue grass, the common pasture grass grown to feed the cattle, has a requirement of 195 kg per ha of N-fertilizer (4–5 lbs. per 1,000 ft²) (Sleper and West, 1996). Because of the large number of cattle, the N-fertilization applied, and the abundant septic tanks, N-contamination in the unconfined Springfield Plateau aquifer was suspected, and to test this hypothesis, the water quality was monitored via springs for 1 year.

Monitoring the water quality of springs over time, especially $\text{NO}_3\text{-N}$ concentrations, will indicate how much contamination is present in the Springfield Plateau aquifer, its seasonal variation, and the difference in water quality between urban and rural springs. The results can be compared to previous regional water studies conducted in this region to determine if water quality has changed from 1993 to 1995 (Adamski, 2000) to the present.

METHODS

Study Area

The study area is located within Lawrence and Greene counties in the Springfield Plateau region of southwest Missouri, USA (Fig. 2). The Springfield Plateau consists mostly of Mississippian and Ordovician carbonate rocks with minor amounts of clastic sedimentary

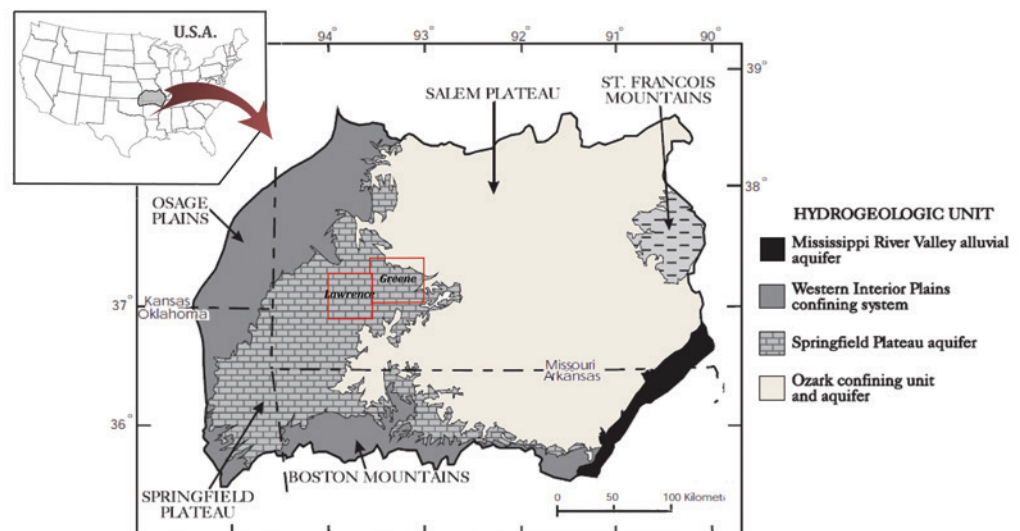


Figure 1. Hydrogeologic units and the various physiographic provinces (e.g., Springfield Plateau, Salem Plateau) that make up the Ozark Plateau physiographic province (modified from Adamski, 1997). Red boxes are Lawrence and Greene counties.

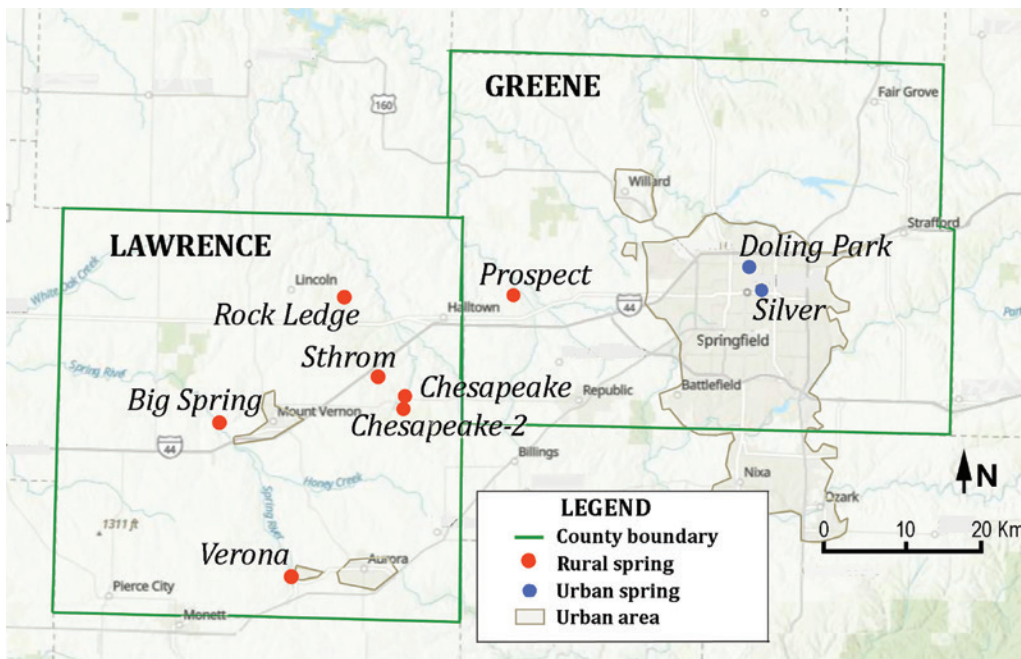


Figure 2. Location of sampled springs within Lawrence and Greene counties (Source: MODNR).



Figure 3. (a) Chesapeake Spring in Lawrence County and (b) Silver Spring in Springfield, Missouri, as examples of a rural and an urban spring, respectively. Photos by the authors.

prone to contamination because of the secondary porosity formed after dissolution enlarges the fractures in carbonate rocks, facilitating the passage of contaminants to deeper parts of the aquifer. Despite this predisposition to contamination, few water quality studies have been conducted of the Springfield Plateau aquifer, with the notable exceptions of the studies by Imes and Davis (1990), Adamski and Pugh (1996), and Adamski (2000). One possible reason for it being less studied is that the Springfield Plateau aquifer is not a major source of drinking water, as the underlying Ozark aquifer is the major drinking water source for the region and has received more attention as a result (Pope et al., 2009; Kingsbury, 2020).

In this study, the water quality of seven rural springs and two urban springs is reported, as well as their seasonal variation and their response to the abnormally dry period experienced in the region during the summer and fall of 2023. The analysis focuses on the contaminants expected from the land use as pastureland and cattle grazing, namely $\text{NO}_3\text{-N}$ and Cl concentrations, although all major ions and TDS were analyzed. The response of the springs to rain events was included by comparing water quality to the cumulative precipitation data from 3 days prior to sample collection reported by the nearest weather station (NWS, 2024).

In addition to the monitoring of the rural springs, a few private shallow (hand pump) wells located within 1 km of a studied spring were sampled once to detect possible $\text{NO}_3\text{-N}$ leakage from the Springfield Plateau aquifer. Because the wells were hand-pumped, they are likely about 10 m (25–35 feet) deep and are tapped to the Springfield Plateau aquifer or to the top of the Ozark aquifer. The locations of the sampled wells are shown in Figure 4. The two sampled urban springs, Doling Park Spring and Silver Spring, discharge from the Springfield Plateau aquifer and are used to compare between rural and urban land use effects.

units that are exposed in southwest Missouri and northwest Arkansas and extend into parts of Oklahoma and Kansas. The Mississippian-aged limestones comprise the Springfield Plateau aquifer, an unconfined aquifer with an average thickness of 30–60 m (Adamski, 2000; Clark et al., 2019). The Springfield Plateau aquifer varies in thickness, with an average of 60 m within the study area (Clark et al., 2019), and it is separated from the underlying Ozark aquifer by a thin (0–13 m) Ozark confining layer. The Ozark confining layer is absent in some areas.

The Burlington–Keokuk limestone of Mississippian age is a highly karstified rock that covers most of the study area, although the underlying Ordovician dolostone outcrops in a few locations in the northeast corner of Greene County and along the Chesapeake fault in Lawrence County (Fig. 4). This dolostone is the top of several carbonate rocks that form the Ozark aquifer.

Karstic aquifers are

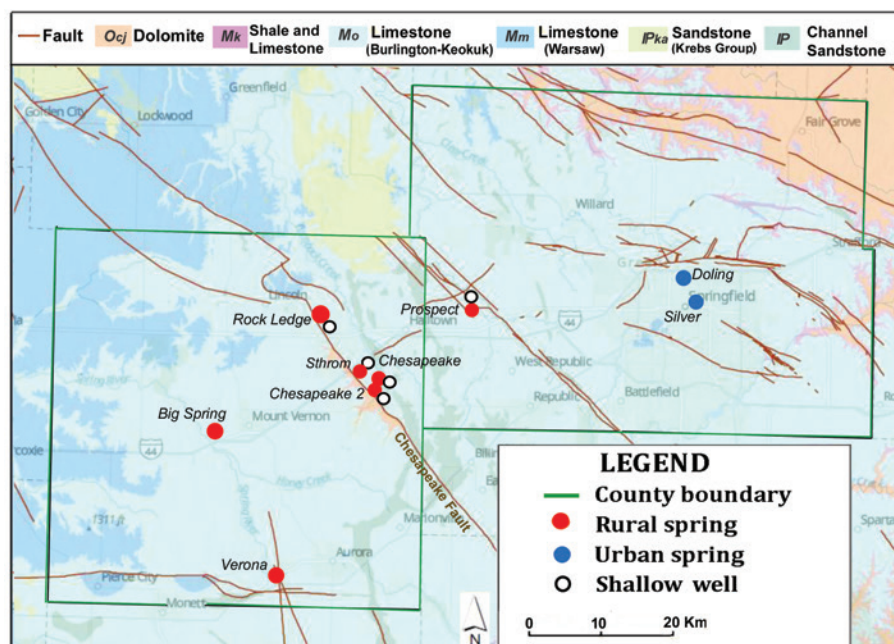


Figure 4. Geology of the study area and location of the sampled wells (Source: MODNR).

Sampling and Analyses

From the rural springs sampled, all but one had a name. The spring with no name will be referred here as Chesapeake 2 since it is near the Chesapeake Spring (see Fig. 2). The springs were sampled approximately monthly for 1 year, and their water quality analyzed according to standard guidelines and protocols (Clesceri et al., 1998). The urban springs were sampled at about the same times (within 2 days) as the rural springs. Samples were collected using acid-washed 1-L plastic bottles filled three times with samples and discarding the first two volumes of the sample. This sampling procedure was also followed for hand-pumped wells.

Samples were kept refrigerated after collection, and the pH and alkalinity were determined as soon as the samples arrived at the Water Laboratory at Missouri State University. The pH was

determined by a Fisher Scientific pH meter and the alkalinity by titration with HCl 0.01 M. $\text{NO}_3\text{-N}$ was determined by colorimetry using a Hach DR3900 Spectrophotometer and reagents for a 0.23–13.50 mg/L $\text{NO}_3\text{-N}$ concentration range. Total and calcium (Ca) hardness were determined by titration with EDTA 0.005 M, from which Ca and magnesium (Mg) concentrations were calculated. Chloride concentrations were determined by the argentometric method, and sulfate (SO_4^{2-}) by the turbidimetric method (Clesceri et al., 1998). Sodium (Na) concentrations were determined using a PerkinElmer Analyst 300 Atomic Absorption Spectrometer. Potassium (K), present in small concentrations (0.9 mg/LK; Adamski, 2000), was estimated from the Mg content after correlation with regional water quality data. Pope et al. (2009) found the closest association of K to be with Mg. Detailed analysis procedures can be found in Colmenero-Chacón (2024).

Quality Assurance/Quality Control

Sampling bottles and glassware were acid-washed and rinsed with double-distilled water. Analyses were conducted using high-grade chemical reagents, and all analytical determinations followed standard procedures (Clesceri et al., 1998). Instruments (pH meter and turbidity meter) were calibrated daily. Replicates and blanks were run approximately every eighth sample, and the methods were tested by determining the concentrations of standard solutions ($\text{NO}_3\text{-N}$ and SO_4) approximately every 15 samples. All blanks and standard solution calibrations were within the expected levels of below detection for blanks and within <1% error for standard solutions. In contrast, replicates had a small variation depending on the parameter. Chloride concentrations of replicates varied the most as the titrimetric method becomes less accurate at low concentrations. All other parameters had better replicability (<10% deviation), and the best was obtained for $\text{NO}_3\text{-N}$, with a variation among duplicates of ± 0.01 ppm. The ionic balance (EB%) had an average of 1.83% and a maximum of 10%. A few samples surpassing 10% EB were discarded from the statistical results.

RESULTS AND DISCUSSION

The precipitation in 2023 was referred to by a drought monitor (NIDIS, 2024) as “abnormally low rain,” for which the springs are considered to have reached base level at the end of the summer. Visual inspection of the spring flow during sample collection indicated a relatively unchanged flow at some springs (Big Spring, Verona, and Chesapeake), whereas other springs (Rock Ledge) dried up by late summer, or their flow was significantly diminished (Prospect, Sthrom, and Chesapeake 2). The differences in the flows were assumed to reflect the size of a spring’s draining basin, and this characteristic presumably affected the water quality of a particular spring as well.

The water quality results are listed in Table 1. The Spearman correlation coefficients between the major ions and precipitation are shown in Table 2.

Interestingly, the $\text{NO}_3\text{-N}$ concentrations did not correlate with any of the other parameters, whereas most of the other solutes (SO_4 , Cl, Na, and TDS) correlated among themselves. As expected, Ca, HCO_3 , and TDS concentrations had a strong inverse correlation with precipitation.

Table 1. Water quality data of rural springs in southwest Missouri and cumulative precipitation amounts for 3 days before sampling.

Sampling Date, 2023	pH	Alkalinity (mg/L CaCO ₃)	HCO ₃ (mg/L)	NO ₃ -N (mg/L)	Cl (mg/L)	SO ₄ (mg/L)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	TDS (mg/L)	3-Day Precip., (in)
Prospect Spring												
Mar. 12	7.56	184	112	2.63	6.4	3.3	58.4	3.4	6.1	0.4	193	0.220
Apr. 02	7.55	180	110	2.47	5.7	0.7	63.2	10.7	7.4	1.2	201	0.850
May 07	7.64	240	146	2.85	11.0	4.7	92.0	3.9	8.6	0.4	270	0.000
Aug. 26	7.82	256	156	2.63	7.8	4.6	76.0	8.8	6.4	1.0	263	0.000
Sep. 30	7.86	236	144	2.36	9.9	7.0	78.4	11.2	7.5	1.2	262	0.000
Nov. 04	7.92	224	137	2.35	6.4	3.4	74.4	16.0	7.8	1.8	249	0.000
Nov. 25	8.20	228	139	2.40	5.7	2.5	76.8	15.1	8.2	1.7	251	0.000
Rock Ledge Spring												
Apr. 02	7.33	176	107	2.46	5.0	4.9	60.8	9.2	5.7	1.0	196	0.850
May 07	7.25	212	129	2.10	7.1	7.2	66.4	1.5	6.1	0.2	221	0.000
Aug. 26	7.46	256	156	2.74	9.9	2.1	77.6	9.2	4.4	1.0	263	0.000
Sep. 30	7.39	212	129	2.18	9.2	2.0	68.0	12.2	3.9	1.3	228	0.000
Sthrom Spring												
Mar. 12	7.01	180	110	2.69	1.4	2.8	61.6	4.9	5.1	0.5	189	0.220
Apr. 02	7.01	180	110	2.28	2.1	3.4	64.0	7.3	7.9	0.8	198	0.850
May 07	7.01	180	110	2.19	5.3	3.3	67.2	4.4	6.0	0.5	199	0.000
Aug. 26	7.18	299	182	2.95	7.8	3.3	87.2	5.3	5.4	0.6	295	0.000
Sep. 30	7.38	232	142	2.71	8.5	2.0	87.2	10.2	6.8	1.1	260	0.000
Nov. 04	7.32	240	146	2.95	12.8	5.9	91.2	16.0	10.0	1.8	286	0.000
Dec. 11	7.41	244	149	2.92	12.1	5.5	97.6	10.7	10.4	1.2	289	0.000
Chesapeake Spring												
Mar. 12	6.91	176	107	3.65	<0.7	2.3	54.4	12.2	5.2	1.3	186	0.220
Apr. 02	6.97	180	110	3.40	5.4	2.0	55.2	14.6	6.1	1.6	197	0.850
May 07	6.89	150	92	3.12	2.1	2.0	50.4	11.7	5.3	1.3	167	0.000
Aug. 26	7.26	200	122	2.97	8.5	2.6	56.8	14.1	4.0	1.6	212	0.000
Chesapeake 2												
Sep. 30	7.18	188	115	2.90	5.7	0.8	53.6	19.0	3.4	2.1	201	0.000
Nov. 04	7.27	200	122	3.25	4.3	2.8	85.6	15.1	6.3	1.7	240	0.000
Dec. 11	7.34	240	146	3.30	6.4	0.9	86.4	17.5	6.2	1.9	268	0.000
Big Spring												
Mar. 12	7.14	176	107	1.73	<0.7	2.0	40.0	9.7	3.0	1.1	165	0.220
Apr. 02	7.16	152	93	1.30	2.1	1.0	40.0	13.6	4.2	1.5	156	0.850
May 07	7.15	156	95	1.05	<0.7	0.8	44.0	12.2	2.9	1.3	157	0.000
Aug. 26	7.24	214	131	1.40	<0.7	2.5	60.8	10.7	2.3	1.2	209	0.000
Sep. 30	7.24	196	120	1.29	4.3	2.9	59.2	17.5	2.0	1.9	207	0.000
Nov. 04	7.38	252	154	1.38	5.0	2.6	88.0	17.5	2.8	1.9	272	0.000
Dec. 11	7.45	252	154	1.44	3.6	0.6	80.0	24.8	2.8	2.8	268	0.000
Big Spring												
Apr. 02	6.87	156	95	4.18	0.7	5.1	60.0	6.8	6.1	0.8	179	0.850
May 07	6.93	156	95	3.88	11.4	3.0	56.0	9.7	5.7	1.1	186	0.000
Aug. 26	7.07	212	129	3.32	9.2	4.2	71.2	1.9	5.6	0.2	226	0.000

Continued

Table 1. Continued

Sep. 30	7.13	184	112	3.84	7.1	5.2	65.6	7.3	4.8	0.8	207	0.000
Nov. 04	7.07	176	107	3.49	12.8	5.9	70.4	14.1	8.0	1.6	223	0.000
Dec. 11	7.20	188	115	4.10	7.1	1.1	69.6	11.2	8.4	1.2	217	0.000
Verona Spring												
Mar. 12	6.85	108	66	5.77	<0.7	3.9	45.6	4.9	5.1	0.5	132	0.220
Apr. 02	6.85	108	66	6.09	<0.7	5.2	40.0	8.7	7.2	1.0	134	0.850
May 07	6.73	212	129	4.35	9.9	4.9	48.8	4.4	4.8	0.5	207	0.000
Aug. 26	7.01	172	105	3.42	5.0	4.0	65.6	3.9	4.2	0.4	192	0.000
Sep. 30	6.96	168	102	3.65	7.1	4.3	60.0	6.3	3.8	0.7	189	0.000
Nov. 04	7.09	164	100	2.74	7.1	4.0	63.2	23.3	6.5	2.6	208	0.000
Dec. 11	7.91	200	122	3.89	7.8	1.0	68.0	8.3	5.7	0.9	218	0.000

Table 2. Spearman correlation (ρ) among rural springs, with significant correlation (p<0.05) in bold.

	HCO ₃	NO ₃ -N	Cl	SO ₄	Ca	Mg	Na	TDS	Precip.
HCO ₃	1.00	-0.17	0.53	0.18	0.86	0.20	0.18	0.96	-0.77
NO ₃ -N		1.00	0.30	0.31	-0.06	-0.24	0.30	0.11	0.10
Cl			1.00	0.43	0.57	-0.13	0.47	0.61	-0.35
SO ₄				1.00	0.38	-0.49	0.54	0.25	-0.07
Ca					1.00	-0.01	0.47	0.93	-0.53
Mg						1.00	-0.28	0.22	-0.40
Na							1.00	0.31	0.12
TDS								1.00	-0.70
Precip.									1.00

Note: Precip. =cumulative precipitation amounts for the 3days before sampling date.

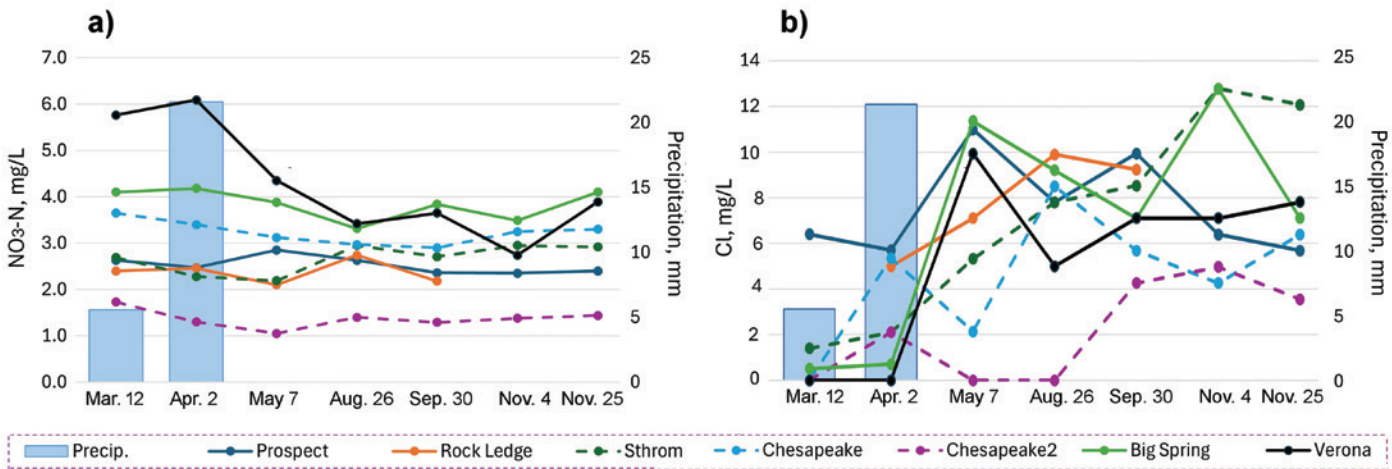


Figure 5. Concentration patterns of (a) NO₃-N and (b) Cl as a function of time for each of the seven rural springs.

The concentrations of the two parameters of interest (NO₃-N and Cl) are plotted in Figure 5 to better visualize their variation with respect to time. Figure 5 shows that each spring had its own pattern of water quality, but the response of each spring to the lack of precipitation differed; while the NO₃-N concentrations became more constant at a lower concentration, Cl concentrations increased with time.

Fescue grass is generally fertilized during the mid-spring months, which may explain that the highest NO₃-N concentrations were observed in late spring, after which the concentrations stayed at a lower level (from 1.5 mg/L NO₃-N for Chesapeake 2 Spring to 4.1 mg/L NO₃-N for Big Spring). All the rural springs had a relatively low NO₃-N concentration, which suggests that the human activities on the surface (cattle farming and pastureland) follow best practices and cause little impact on the Springfield Plateau aquifer in the vicinity of the sampled springs. The highest NO₃-N concentrations,

up to 5.9 mg/L $\text{NO}_3\text{-N}$, were detected during the late spring months at the Verona and Big springs, both located in the western part of the study area and near the headwaters of the Spring River (Fig. 2), but all other parameters were low. These high $\text{NO}_3\text{-N}$ concentrations may have resulted from the septic tank leaching fields of nearby farms, and their presence attests to the susceptibility of contamination of karst regions. The behavior of Cl was to gradually increase with time, which may be, at least in part, a concentration effect, as there was a long period of little to no precipitation.

Concentrations of $\text{NO}_3\text{-N}$ reached a peak in the spring, and an increase of Cl in the fall and early winter was also reported in another cattle-grazing area within the Springfield Plateau in the nearby Polk County, where, unlike this study, precipitation in 2022 was abundant during spring and summer (Gomez and Gutiérrez, 2022).

To visually compare the results between the rural and urban springs, chemical concentrations of four rural springs (Sthrom, Chesapeake, Big Spring, and Verona) were plotted together with the two urban springs (Doling Park and Silver, in red), as shown in Figure 6. The $\text{NO}_3\text{-N}$ concentrations of the urban springs had similar concentrations and responses to precipitation than the rural springs. In contrast, Cl concentrations of the urban springs were higher and more erratic than those of the rural springs.

The water quality of the springs within Lawrence County was compared with other studies within the Springfield Plateau aquifer, as shown in Table 3. The $\text{NO}_3\text{-N}$ concentrations showed little change from 1993 to 1995 (Adamski, 2000) to the present. In contrast, a large difference exists between $\text{NO}_3\text{-N}$ values obtained in this study and those reported for nearby unconfined carbonate aquifers of Iowa and Illinois (Table 3). The consistently low concentrations of the above chemicals within the springs of the Springfield Plateau are likely a result of rainfall comprising the recharge, a short residence time due to the large fractures in the karstified rock, and a sustainable operation of pastureland and cattle grazing activities. In contrast, increased concentrations of the same chemicals within the karst areas in Illinois and Iowa are likely the result of lateral flows contributing to aquifer recharge in addition to precipitation as well as leakage from agricultural activities that include growing a variety of row crops (Maas et al., 2019; Schilling et al., 2019).

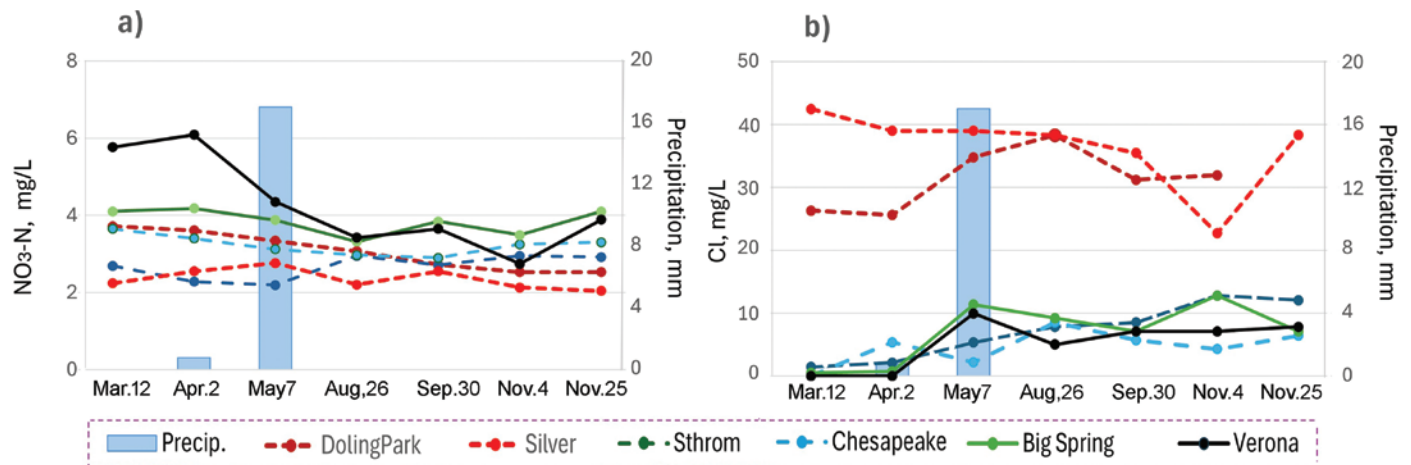


Figure 6. Concentrations of (a) $\text{NO}_3\text{-N}$ and (b) Cl for selected urban and rural springs. Urban springs (Doling Park, Silver) are shown in red.

Table 3. Summary of water quality studies on karst springs within the Springfield Plateau aquifer and nearby carbonate aquifers.

No. Springs	NO ₃ -N (mg/L)			Cl (mg/L)			References
	Min.	Ave.	Max.	Min.	Ave.	Max.	
Springfield Plateau Aquifer							
7	1.05	2.91	6.09	<0.7	5.92	12.8	This study
61 (+50 wells)	<0.05	1.90	25.00	0.9	5.90	220	Adamski (2000)
1	0.6	2.27	5.70	1.0	5.60	15.0	Knierim et al. (2015)
5	0.05	1.39	6.00	0.5	11.70	67.5	Gomez and Gutiérrez (2022)
1	2.74	4.07	4.79	n.a.	n.a.	n.a.	Peterson et al. (2002)
Other Carbonate Aquifers (Illinois, Iowa)							
5	2.92	15.53	30.10	5.4	18.63	26.7	Maas et al. (2019)
2	3.64	14.69	22.12	n.a.	n.a.	n.a.	Schilling et al. (2019)

n.a. = not available.

Table 4. Water quality of springs in the study area (average values) and its nearby hand-pumped well.

	No. samples		NO ₃ -N (mg/L)		Cl (mg/L)		TDS (mg/L)	
	Spring	Well	Spring	Well	Spring	Well	Spring	Well
Prospect	7	1	2.53	0.56	7.6	2.1	238	172
Rock Ledge	4	1	2.46	0.31	8.0	5.7	229	346
Sthrom	7	1	2.62	0.04	7.0	6.4	237	289
Chesapeake	7	1	2.90	0.92	4.0	<0.7	214	269
Chesapeake 2	7	1	2.88	0.80	4.1	0.7	208	224

Because the Ozark confining layer may be very thin and fractured, the potential leakage of contaminants from the Springfield Plateau aquifer to the underlying Ozark aquifer was tested by comparing the water quality concentration data (NO₃-N, Cl, and TDS) between each of the five springs in the study area and water collected from hand-pumped wells near (within 1 km) each spring. The shallow wells, since they are hand-pumped wells, typically draw water from 8 to 11 m (25–35 feet) deep and thus collect water from the Springfield Plateau aquifer. The wells were selected based on the availability to collect a sample, as they were private wells. The water quality results are listed in Table 4.

The concentrations, as shown in Table 4, show a sharp decrease in NO₃-N concentrations in the well water compared to that of the spring water, a less pronounced decrease in Cl concentrations, and a small increase in TDS. The TDS increase is expected because of the increased dissolution that results from a longer water–carbonate rock contact. The drought conditions during the sampling period suggest that these values are possibly the most concentrated values encountered in these springs, and that once precipitation returns to normal conditions, concentrations will decrease because of dilution.

Overall, and although the water quality of each spring was unique, all the springs complied with drinking water quality guidelines, thus there is little concern for contamination of the underlying aquifer with leakage from the Springfield Plateau aquifer. However, this conclusion may change if the Springfield Plateau aquifer becomes contaminated in the future, which is possible due to expanding housing development in the region as well as changes in precipitation due to climate change. Continued monitoring of water quality is recommended to keep both the Springfield Plateau and Ozark aquifers safe from contamination.

CONCLUSIONS

Despite the prevalence of agricultural (pastureland) land use and the large number of cattle in the study area in southwest Missouri, the water quality of the Springfield Plateau aquifer complies with drinking quality guidelines of the contaminants of interest, 10 mg/L NO₃-N and 250 mg/L Cl (USEPA, 2012). The results suggest that the number of cattle and farming activities has little to no negative impact on the water quality of the Springfield Plateau and Ozark aquifers in Lawrence County. Only two of the seven monitored springs showed NO₃-N concentrations above 4.0 mg/L, an unenforced limit to prevent eutrophication and ocean dead zones. Chloride concentrations remained low in rural springs, with values up to 12.8 mg/L, and higher but modest Cl concentrations (up to 42 mg/L) were observed in urban springs.

Despite being relatively close to each other, each spring possessed its own pattern of concentrations in response to seasonal changes in precipitation and farming activities such as fertilization application. Seasonal variations showed an increase in NO₃-N concentrations in late spring and in Cl concentrations in late fall. Oddly, NO₃-N values did not correlate with other water parameters nor with precipitation, whereas the other parameters (Ca, Mg, Na, Cl, and SO₄) were associated with each other as well as with precipitation. The low concentrations of contaminants suggest that even if the Springfield Plateau aquifer were to leak into the underlying aquifer, contamination is not a concerning issue at this time, as even though the region was experiencing a drought and water was concentrated with solutes, these remained at low concentrations. A water quality baseline for the Springfield Plateau aquifer in Lawrence County, Missouri, was obtained.

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