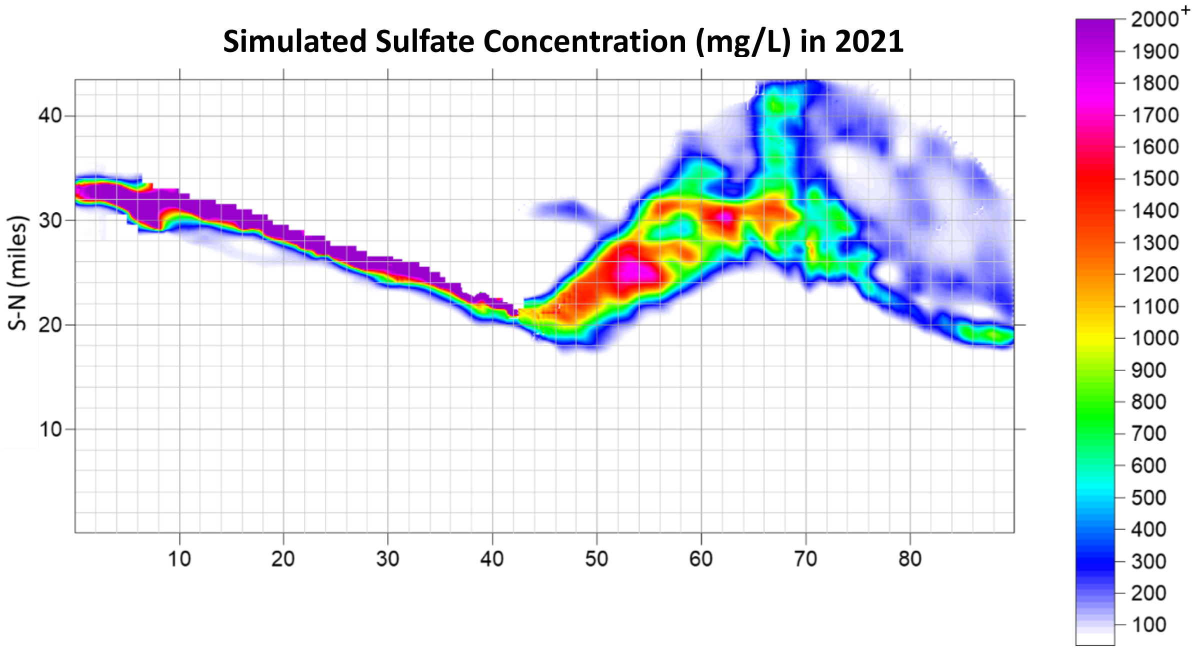
Upper Arkansas River Corridor Groundwater Quality Modeling

Funded by the

Kansas Water Office (Contract 22-111)



Gaisheng Liu, Erin Seybold, Sam Zipper, Brownie Wilson, Don Whittemore, and James Butler
Kansas Geological Survey, University of Kansas
1930 Constant Ave., Lawrence, KS 66047

Kansas Geological Survey Open-File Report 2025-19

GEOHYDROLOGY

KANSAS GEOLOGICAL SURVEY
The University of Kansas

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

Acknowledgments

The authors acknowledge the funding support of the Kansas Water Office for this project. We thank Josh Olson and Keadron Pearson of the KWO, who were the primary contacts on the contract and facilitated the project progress meetings. We also thank Xiaolong Yuan, a visiting scholar who contributed to Arkansas River streamflow and concentration data processing, and Julie Tollefson, KGS editor, who reviewed the final report.

Executive Summary

This report describes a groundwater transport model developed for the Upper Arkansas River corridor between the Kansas-Colorado state line and eastern Gray County. This model is used to simulate how sulfate and uranium contamination in the Arkansas River affects the groundwater system in the region. The model is based on a MODFLOW flow model developed for Groundwater Management District No. 3 recently by the Kansas Geological Survey (KGS). Transport simulation for a refined model in the area around the Arkansas River is performed using the software MT3DMS (Modular Three-Dimensional Transport for Multispecies Simulation).

The transport model is constructed using data gathered primarily between 2001 and 2021. The average 2000–2002 sulfate concentrations from a previous KGS project (Whittemore et al., 2023) are used to estimate initial sulfate concentrations in the model. Arkansas River flow rates are based on two U.S. Geological Survey gages that were operated during two different periods in the area, and river water concentrations are based on empirical relationships between sulfate and flow rates at different time periods. The average 2020–2022 sulfate concentrations are used to calibrate the model. The longitudinal dispersivity is selected as the calibrated parameter, and the calibrated model accurately simulates observed sulfate concentrations ($R^2 = 0.73$). Uranium concentrations are then estimated based on empirical relationships between sulfate and uranium concentrations in the aquifer.

Results from the calibrated model indicate that groundwater movement and ditch irrigation have caused significant spreading of river contaminants in the aquifer on both the north and south sides of the river. Between 2001 and 2021, the southern edge of the plume traveled about one mile to the south of the river. Sulfate plume migration has produced a noticeable impact on Garden City's public drinking supply wells in the Sand Hills area. In 2001, only two wells (G5 and G6) were inside the plume with elevated concentrations. In 2021, a third well (G1) was inside the plume with elevated concentrations, another well (G3) was on the plume edge, and all the other wells (G2, G4, and G7) were much closer to the edge of the plume than before.

The calibrated transport model is used to simulate how the contaminants in the aquifer continue to move in two different water resources management scenarios. In the status quo scenario, the historical climate, pumping, and river concentration data of 2001–2021 are repeated five times to simulate 100 future years, 2021–2121. Due to the general flow direction in the area, the plume will move farther to the southeast of the Arkansas River in this scenario. As contaminants continue to enter the aquifer through river infiltration and ditch return flow, the plume expands with more areas showing high concentrations. All of Garden City's supply wells in the Sand Hills area will have sulfate concentrations above the recommended standard of 250 mg/L sometime between 2041 and 2061.

In the no future contamination scenario, where all contaminants in the Arkansas River future flow are removed, only two of Garden City's supply wells (G2 and G4) show noticeable concentration reductions after 2070 compared to the status quo scenario, while the concentration changes in the remaining wells are insignificant. These results indicate that any contamination treatment measures performed at the river will likely take centuries to make a significant impact on the Sand Hills wells due to the large amounts of contaminants currently in the subsurface and the slow rate of groundwater lateral flow.

1. Introduction

Over the last several decades, groundwater in the Upper Arkansas River corridor has been contaminated by elevated concentrations of dissolved solids in Arkansas River water. Most of the dissolved solids originate from the soils and bedrock in eastern Colorado, where river water is diverted for irrigation and storage systems and evapotranspiration significantly increases the salinity of return flow to the river. Dissolved solids concentrations in low flows of the Arkansas River can sometimes exceed 4,000 mg/L at the Colorado-Kansas border, compared to the Environmental Protection Agency's recommended standard of 500 mg/L for public drinking water. In addition to sulfate, the uranium concentration in the river is high. The average uranium concentration of 27 samples collected during 2009–2010 is 63.5 μg/L (Whittemore et al., 2023), which is significantly higher than the maximum contaminant level of 30 μg/L for drinking water.

Contaminant inflows to Kansas tend to have the greatest concentrations when streamflow is low. Figure 1 shows the calculation of sulfate mass loads in the Arkansas River near the Colorado-Kansas border. The river flows were obtained from two U.S. Geological Survey gages that were operated during different periods on the Arkansas River near the state line. The Syracuse gage (USGS 07138000) was used prior to October 1, 1950, and the Coolidge gage (USGS 07137500) for the period afterward (fig. 1A). Based on the analyses of river water samples (Whittemore et al., 2023), three sulfate concentration–river flow relationships can be identified as described below.

For the 1981–present period (blue line in fig. 1B):

SO_4 [mg/L] = 2400 when flow rate is less than or equal to 100 ft^3/sec

SO_4 [mg/L] = $-680 \times \ln(\text{flow}) + 5520$ when flow rate is greater than 100 ft^3/sec

For the period prior to 1981 (green line in fig. 1B):

SO_4 [mg/L] = $-350 \times \ln(\text{flow}) + 3000$

The maximum sulfate concentration appears to be about 2,400 mg/L, as a further increase will lead to gypsum precipitation onto the riverbed.

Sulfate concentrations are calculated for each day between January 1, 1944, and March 31, 2021, in the Arkansas River near the state line using the concentration-flow relationships above (fig. 1C). The sulfate mass loads in the Arkansas River are then calculated by multiplying the river flow rates with sulfate concentrations. Between 1944 and 2021, the average sulfate mass load was 7 kg/sec, which means that 7 kg of sulfate was transported from Colorado into Kansas every second by the river. In terms of the average annual load, that is equivalent to 220,752 metric tons/year (metric ton simply referred to as ton hereafter). Mass loads appeared to increase after 1981 due to the different concentration-flow relationships (fig. 1B).

For the estimation of uranium concentrations in the Arkansas River, a linear relationship was identified between uranium and sulfate concentrations based on water samples analyzed between 2009 and 2022 (fig. 2A; Whittemore et al., 2023),

$U = 0.03384 \times SO_4 - 2.2$, where U is in μg/L and SO_4 is in mg/L.

Figure 2C shows the calculated uranium mass loads across the state line in the Arkansas River. Between 1944 and 2021, the average uranium mass load is 19.3 kg/day (equivalent to 7 tons/year).

To assess how the contaminated river water affects the aquifer system in the corridor, this work develops a groundwater flow and transport model. Specifically, we simulate the influence of river infiltration and ditch irrigation on groundwater sulfate and uranium concentrations. This model builds upon and refines the regional groundwater flow model the Kansas Geological Survey recently developed for Groundwater Management District #3 (GMD3) (Wilson et al., 2024). After the model is constructed and calibrated, it is used to run different future scenarios to investigate the effects of water resources management and climatic variability on the movement of sulfate and uranium in the aquifer.

2. Study Area

The study area includes portions of Hamilton, Kearney, Finney, and Gray Counties along the upper Arkansas River corridor in GMD3 (fig. 3). As with the rest of western Kansas, grassland and cropland are the two most common land-cover types under semi-arid climatic conditions. Erosion by the Arkansas River has created valley walls of moderate to steep slopes in an otherwise nearly flat plain. Crop irrigation is primarily supported by groundwater pumping from the High Plains aquifer (HPA) and an extensive ditch network with water diverted from the Arkansas River. Detailed information about the study area, including precipitation, hydrogeology, stream characteristics, water use, and ditch network and diversion rates, can be found in the GMD3 model report by the Kansas Geological Survey (Wilson et al., 2024).

The Arkansas River flow infiltration and ditch irrigation have caused significant spreading of river contaminants into the aquifer system in the upper Arkansas River corridor (fig. 3). This has produced increasingly large challenges for the drinking water supplies of the communities in the region. For example, in Lakin, a city located north of the Arkansas River, the detection of uranium levels above the MCL in the Lakin public supply from the HPA in 2007 led to the construction of a treatment plant that became operational in January 2015. In Garden City, some of the public supply wells in the Sand Hills area (south of the Arkansas River) have experienced increasing sulfate concentrations over the last two decades.

3. Groundwater Flow and Transport Model Development

Two major steps were involved in the development of the groundwater flow and transport model. First, a refined MODFLOW (Harbaugh, 2005) groundwater flow model was created for the Upper Arkansas River corridor. This model was based on an existing model developed for GMD3 by the Kansas Geological Survey (Wilson et al., 2024). The MODFLOW simulation provides information about groundwater velocity and the flow rates of contaminant sinks and sources needed for transport simulation. Second, an MT3DMS (Modular Three-Dimensional Transport for Multispecies Simulation; Zheng and Wang, 1999) model was developed for simulating sulfate transport in the aquifer system. During MT3DMS model development, the longitudinal dispersivity was treated as a calibrated parameter and the spatial distribution of sulfate concentrations in the HPA in 2020–2022 were used as calibration data targets. After the flow and transport model was calibrated, two scenarios were performed to investigate how the sulfate would continue to migrate in the aquifer system under future conditions.

3.1. MODFLOW flow model

The upper Arkansas River corridor model was cut from rows 23 to 109 and columns 1 to 180 of the regional GMD3 model constructed by the Kansas Geological Survey (fig. 4). In this way, the northern boundary of the corridor model is 5 miles north of the northernmost model cell containing an irrigation ditch, and the southern boundary is 5 miles south of the southernmost model cell containing an Arkansas River segment. The eastern boundary of the model is about 15 miles downstream of Garden City, as a major objective of the modeling effort is to investigate the distribution of the contaminant plume near the city's supply wells in the Sand Hills area. Vertically, a single model layer is used for the entire saturated zone of the alluvial aquifer and HPA; multiple layers are not considered due to the lack of detailed hydrogeological data and concentration measurements between different depths.

To improve the accuracy of transport simulations, a much finer grid mesh is used for this study compared to the GMD3 model. In the GMD3 model, each model cell is 0.5 by 0.5 mile. In this work, the cell size is refined to 0.1 by 0.1 mile, resulting in a grid of 435 rows and 900 columns (fig. 4). The black areas indicate inactive model cells, where no transportation simulations are performed due to their large distance from the Arkansas River or to their location in thin saturated areas with bedrock highs. The edge of the active model area in the HPA portion is set as specified heads with the values simulated by the regional GMD3 model. The green line represents the Arkansas River, which was simulated by the streamflow-routing package (SFR) in MODFLOW. The red points represent pumping wells, and the gray area represents the ditch canals and service areas with water diverted from the Arkansas River. Due to the limited number of concentration measurements prior to 2000, the period considered in the flow and transport model is 2001 to 2021.

All the model input data — such as recharge, pumping, stream parameters, evaporation-transpiration settings, ditch diversion rates and return flows, aquifer specific yields and hydraulic conductivities — are directly imported from the regional model. Therefore, when a model cell has pumping in the regional model, the pumping rate for the cell is evenly distributed over the corresponding 25 refined cells in the new flow and transport model. A potential future improvement would be to reproduce a new set of pumping data directly projected onto the refined model grid. Similar future improvements for the refined model grid can be made for other input data, such as stream and ditch settings, aquifer specific yields, and hydraulic conductivities.

Figure 5 shows the simulated groundwater levels by the refined MODFLOW flow model for 2001 and 2021. Comparison with the regional GMD3 model shows the simulated water levels in the study area are nearly identical between the refined and regional models. In 2001, the water-level contours are relatively smooth, and the ambient groundwater flow direction generally follows the Arkansas River. In 2021, as the effects of groundwater pumping from the HPA become more significant in the area south of the river, the water-level contours show more spatial variability indicating an increasing occurrence of pumping-induced local water-level depressions. Compared to 2001, the ambient flow direction in 2021 shifts toward the southeast and there is more groundwater moving from the north to the south of the river.

3.2. MT3DMS transport model

The MODFLOW model provides information about the rates and directions of groundwater flow and contaminant sinks and sources that are needed for contaminant transport simulation by MT3DMS. All those rates are automatically saved into a flow-transport link file by MODFLOW when a link-MT3DMS file is specified in the MODFLOW name file. There is a unique challenge in the MODFLOW-MT3DMS link file due to the way a groundwater pumping well is treated by MT3DMS. In MT3DMS, by default, a pumping well is treated as a contaminant sink and the contaminants in the pumped water are assumed to be removed immediately from the aquifer system. However, in this work, most groundwater pumping is for irrigation and the contaminants in the pumped water will eventually be transported back into the saturated zone by precipitation recharge and irrigation return flow. To address this challenge, the pumping data (except data for Garden City's public supply wells) in the flow-transport link file are changed into evaporation-transpiration rates. In MT3DMS, evaporation-transpiration is considered to remove only water but no contaminants.

In addition to the flow-transport link file, a series of additional parameters, as well as contaminant initial and boundary conditions, need to be defined for transport simulations. The transport parameters that are needed include dispersivities (both longitudinal and transverse), the molecular diffusion coefficient, and effective porosity. Dispersivities describe how far contaminants will spread out from the main plume (controlled by average groundwater flow) due to unresolved heterogeneity (e.g., local small-scale preferential flow pathways that are not simulated by the flow model). The longitudinal dispersivity (i.e., dispersivity along the flow direction) is typically the most important parameter in controlling the spatial distributions of contaminants at a given time. As a result, it is treated as a calibrated parameter during model calibration. The transverse dispersivity (dispersivity perpendicular to flow direction) is set as one-tenth of the longitudinal value as with most groundwater transport modeling studies (Zheng and Bennett, 2002). The molecular diffusion coefficient for sulfate is set to 1.3×10^{-5} cm^2/s, which is a typical value for unconsolidated aquifer settings. Compared to dispersion, the role of diffusion in groundwater contaminant spreading is much less significant. Effective porosity represents the pore space that contributes to the movement of groundwater and contaminants (i.e., advection). In this work, the specific yield values estimated during MODFLOW simulation are used to represent the effective porosity. The less mobile pore space (such as disconnected and dead-end pores) does not directly contribute to advection but provides storage space for contaminants. A total porosity (i.e., summation of mobile and immobile porosities) of 0.25 is used in the transport simulation.

The initial sulfate concentration distribution at the start of simulation (2001) is based on the field data presented in Whittemore (2000) for both the alluvial aquifer (Plate C) and HPA (Plate D). Specifically, the concentrations in the alluvial aquifer in the paleo-valley to the west of the center of township T. 25 S., R. 37 W. are based on the alluvial concentration map, while the rest of the model area is based on the HPA sulfate concentration map, despite the fact that the alluvial aquifer concentration map covers the entire length of the Arkansas River. For the boundary conditions, zero-concentration gradients are used at all four lateral boundaries, which means no contaminants cross the model boundaries by concentration gradients. If there are lateral inflows occurring at some specified-head cells, those inflows do not carry new contaminants. On the other hand, if groundwater leaves the model at some specified-head cells, those outflows will remove contaminants out of the aquifer system.

River flow infiltration and ditch irrigation using the diverted river water are the two primary sources of sulfate that contaminate the aquifer system. The calculated sulfate concentrations in fig. 1 are used for specifying river concentrations in the stream source setup in MT3DMS. Because the model is averaged over a six-month period (either April to September growing season or October to March nongrowing season), daily concentrations are averaged over each six-month stress period using daily flow rates as the weighting factors. The six-month average concentrations are also used to calculate the concentrations of irrigation return flow for both the ditch canals and service areas. Although only a portion of irrigation water is assumed to become the return flow to recharge water (Wilson et al., 2024), all the contaminants in the diverted water are assumed to enter the aquifer. This means the sulfate concentrations of the irrigation return flow are much higher than the river water concentrations, given that the ET of irrigated water does not remove sulfate from the subsurface.

The observed sulfate concentrations in 2020–2022 in fig. 3 are used as data targets for calibrating the longitudinal dispersivity. Calibration is performed using the parameter estimation program (PEST) developed by Doherty (2004). A total of 81,334 concentration data points are extracted, with a maximum of one concentration per model cell. The final calibrated value of the longitudinal dispersivity is 2,349 feet, which is 4.4 times the model cell spacing. In common transport modeling practice, a calibrated longitudinal dispersivity that is between 1 and 10 times the model cell size is considered as appropriate (Zheng and Bennett, 2002).

4. Sulfate Simulation Results

Figure 6 shows the comparison of the simulated versus observed sulfate concentrations in 2021. For the observed map, the concentrations are shown only for the HPA, and no results are plotted for the alluvial aquifer in the paleo-valley above the center of township T. 25 S., R. 37 W.; the simulated concentrations are shown for both the alluvial and HPA aquifers. The simulated sulfate concentration distribution provides a very good match to the observed values. The root mean squared difference between the observed and simulated concentrations in the HPA is 226 mg/L and the R-squared between observed and simulated values is 0.73, indicating 73% of observed spatial variation in sulfate concentration is represented by model simulation. In addition to groundwater movement, ditch irrigation has caused significant spreading of river contaminants in the aquifer on both the north and south sides of the river.

One of the key objectives of this project is to investigate how the Arkansas River contaminants affect Garden City's public water supply wells in the Sand Hills area. Figure 7 shows the simulated sulfate concentrations around the Sand Hills wellfield in 2001 and 2021. Clearly, due to the groundwater flow direction toward the southeast in the area, the sulfate plume has been moving from the river to the wellfield over time. In 2001, only wells G5 and G6 were located in the plume with elevated concentrations; well G1 was on the plume edge; and the remaining wells were all outside the plume. In 2021, well G1 was clearly inside the plume with elevated concentrations; well G3 was on the plume edge; and wells G2, G4, and G7 were much closer to the edge of the plume than before. On average, the southern edge of the plume has traveled about one mile to the south of the river from 2001 to 2021.

To further evaluate how the sulfate plume affects water quality at the Sand Hills wellfield, the simulated concentration time series are compared to observed values in all seven wells (fig. 8). Consistent with the spatial maps of simulated concentrations, wells G5 and G6 have the highest concentrations. Well G1 is in a transition zone where the concentrations were moderate with a continuous increase. The concentrations in wells G2, G3, G4, and G7 are the lowest, with G3 experiencing the most significant increase among this group. The observed values showed a relatively similar pattern to the simulated values: The water samples from wells G5 and G6 have much higher concentrations than other wells. The observed concentrations in G3 show a continuous increase from 2001 to 2021, and the increase in G2 became significant only after 2011.

In terms of the absolute magnitudes, the simulated values underpredict the observed concentration increases at the wells (fig. 8). The difference is possibly due to the differences in the pore water composition represented by the model cell versus that sampled in the field. The simulated concentrations represent average values over a model cell that contains both mobile and less mobile water in the pore space, while field sampling might represent the mobile water with higher sulfate concentrations pumped from the main plume area into the wells through more conductive flow pathways. In addition, the model-simulated concentration represents the vertical average over the entire saturated thickness, while field sampling might be more affected by the conductive layers that have higher concentrations due to preferential plume spreading.

Table 1 shows the simulated sulfate budgets from the calibrated transport model. The annual average of sulfate mass over 2001–2021 that enters the aquifer through river water infiltration is 146,525 tons. The sulfate input through ditch return flows is 65,515 tons. The sulfate leaving the model area through the specified head boundaries along the downstream (eastern) model edge is 6,742 tons. The aquifer receives a net input of 205,028 tons of sulfate per year on average during 2001–2020.

5. Sulfate Future Scenarios

Two future scenarios — status quo and no future river contamination — have been simulated to investigate future migration of sulfate in the system. In the future, more scenarios can be performed using the calibrated model to study aquifer responses to different water management and climate conditions.

5.1. Status quo scenario

The calibrated transport model is used to simulate sulfate movement 100 years into the future in the status quo scenario. The historical climate conditions of 2001 summer to 2021 spring are repeated five times to represent the conditions in 2001 summer to 2121 spring. As river flows and groundwater pumping are primarily controlled by climatic conditions, the 2001 summer to 2021 spring model input data of Arkansas River flow, groundwater pumping, and ditch diversion are also repeated five times for the status quo scenario. For the specified head boundaries, the simulated water levels in the status quo scenario in the regional GMD3 model are used. The last future year simulated in the regional model is 2083 spring. For 2083 summer to 2121 spring, the relative head change between 2063 spring and 2082 summer is repeated twice.

Preliminary model runs indicated that some dry cells occurred if pumping continued at current levels, which caused the transport model to produce spurious results in those areas. To avoid this challenge, future pumping is reduced when aquifer transmissivity (TR) is lower than 4,000 ft^2/d according to the following equation (Wilson et al., 2024),

$Q = Q_0$, if $TR \geq 4{,}000\ ft^2/d$;

$Q = Q_0 \times [1.4427*\ln(Tr)-10.9658]$, if $2000\ ft^2/d \leq TR < 4{,}000\ ft^2/d$;

$Q = 0$, if $TR < 2{,}000\ ft^2/d$.

Here Q_0 is the historic pumping and Q is reduced pumping to avoid the development of dry cells in the status quo simulation.

Figure 9 shows the simulated sulfate concentrations for four selected times in the status quo scenario. Clearly, as more sulfate mass enters the aquifer through river infiltration and ditch return flow, the plume expands with more areas showing high concentrations. Due to the general flow direction in the area, the plume moves farther to the southeast of the Arkansas River with time. Figure 10 shows the distribution of sulfate around Garden City's Sand Hills wellfield from 2021 to 2081. In 2061, all the supply wells will be inside the major plume area with elevated concentrations. Figure 11 shows the simulated sulfate concentration time series at the Sand Hills wells in the status quo scenario. Using the recommended standard of 250 mg/L for drinking water, all the wells will be above that standard sometime between 2041 and 2061. It is also interesting to notice that although well G4 shows the lowest concentrations during earlier years, it will show much higher concentrations than most other wells after 2081 due to the different travel paths of different portions of the plume. Figure 12 shows the simulated sulfate budgets. The total sulfate mass increases steadily as sulfate continues to enter the aquifer system through river infiltration and ditch return flow. The annual mass loads show a cyclic pattern over 20 years as the 2001 summer to 2021 spring river inflows and concentrations are repeated five times to represent future conditions.

5.2. No future contamination scenario

The second scenario simulated in this work assumes that there are no contaminants in the Arkansas River future flow, while all other model settings are the same as the status quo scenario. Although the total removal of all contaminants from the river is a formidable challenge, if not impossible, this scenario provides insights into how the aquifer will recover from past contaminations under the best-possible future situation.

Figure 13 shows the simulated sulfate concentrations in four selected years under the no future contamination scenario. Because groundwater flow velocity is slow and there are no significant contaminant cleanup processes, most of the contaminants remain in the aquifer throughout the future simulation. Two minor removal processes, Garden City's public drinking well pumping and the lateral outflow through the downstream specified head boundaries, do not significantly impact plume mass reduction during the 100-year timeframe simulated. In terms of concentration distribution, due to the general flow direction to the southeast, the peak concentration of the plume in the HPA area has shifted from along the river in 2021 to about 3 miles south of the river in 2081.

Figure 14 shows the simulated sulfate budgets in the no future contamination scenario. The total sulfate mass in the aquifer system is 20,708,419 tons in 2021. After 2021, sulfate mass in the aquifer system is slowly decreasing at a rate of 3,693 tons/year due to the removal by Garden City's wellfield and lateral outflow. Figure 15 compares the simulated sulfate concentrations at Garden City's drinking water wells in the Sand Hills area between the no future contamination and status quo scenarios. The impacts of removing all contaminants from river water on concentrations at the Sand Hills wells are modest in the next 100 years. Only wells G2 and G4 show noticeable concentration reductions after 2070, while the concentration changes in the remaining wells are insignificant. These results indicate that any contamination treatment measures performed at the river will likely take centuries to make a significant impact on the Sand Hills wells due to the large amount of contamination currently in the subsurface and the slow rate of groundwater lateral flow. As a result, the only way to significantly reduce the rising contamination in the Sand Hills wells is to actively treat contaminated water in the aquifer system.

6. Uranium Results

Compared to sulfate, which is largely conservative in the aquifer system, uranium can be subject to geochemical reactions such as reduction and precipitation, as well as adsorption onto the grain surfaces of the sediments. Due to the lack of information about those geochemical reaction parameters, as well as a very limited number of field measurements of uranium in groundwater samples, this work does not pursue direct model simulation of uranium. Instead, uranium is calculated based on two empirical relationships identified between sulfate and uranium (fig. 16),

$U = 0.0375 \times SO_4 + 12.5$, north of the river,

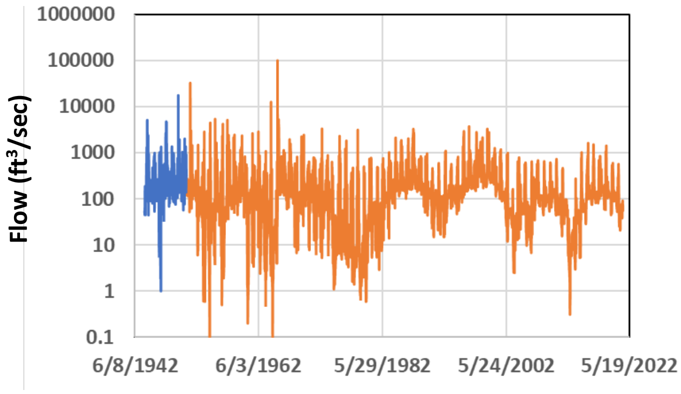
$U = 0.0065 \times SO_4 + 4.5$, south of the river,

where U is in μg/L and SO_4 is in mg/L.

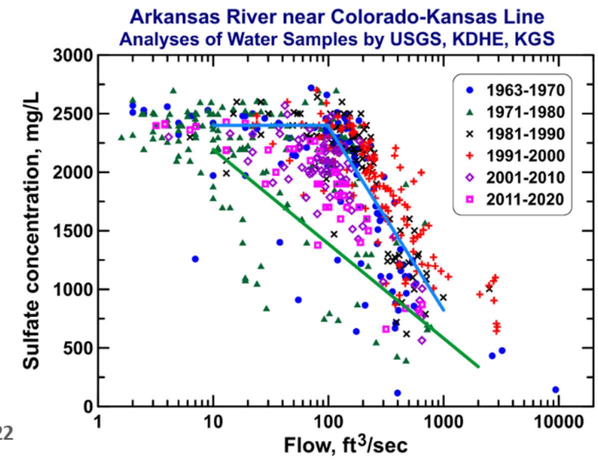
Figure 17 compares the calculated uranium concentrations to the observed values in 2021. Overall, the modeled spatial distribution is consistent with observations, as most of the high uranium concentrations are currently located along the river and in the ditch area to its north. Figures 18 and 19 show the calculated uranium concentrations at four selected years in the status quo and the no future contamination scenarios, respectively. In the status quo scenario, as more contaminants enter the aquifer system, the high uranium concentration area is expanding and slowly moving to the southeast of the river. In the no future contamination scenario, the plume is also expanding to the southeast of the river, but the areas of high concentrations (yellow, red, and purple colors) slowly decrease with time.

	Inputs (tons)	Outputs (tons)
Ark River Infiltration	146,525	
Ditch Return Flows	65,515	
Specified Head Boundaries		6,742
Aquifer Storage	205,028	

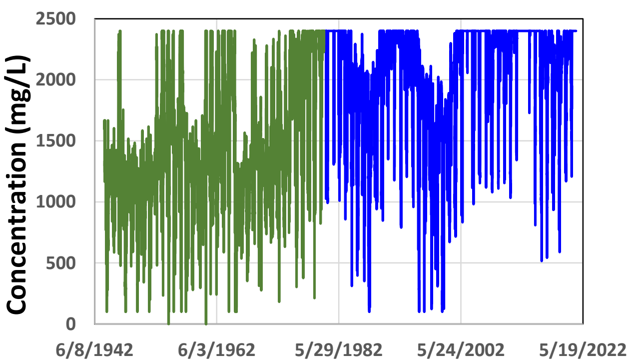
Table 1. Simulated sulfate budgets into and out of the aquifer (annual average 2001–2021). Ton = metric ton. Due to numerical solver imprecisions, the difference between total inputs and outputs (212,040 - 6,742 = tons) differs from aquifer storage (205,028 tons) by 0.1%.



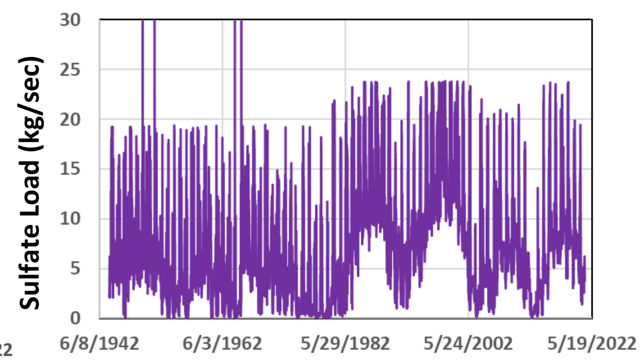
(A) River inflows near state line



(B) River sulfate concentration vs. flow

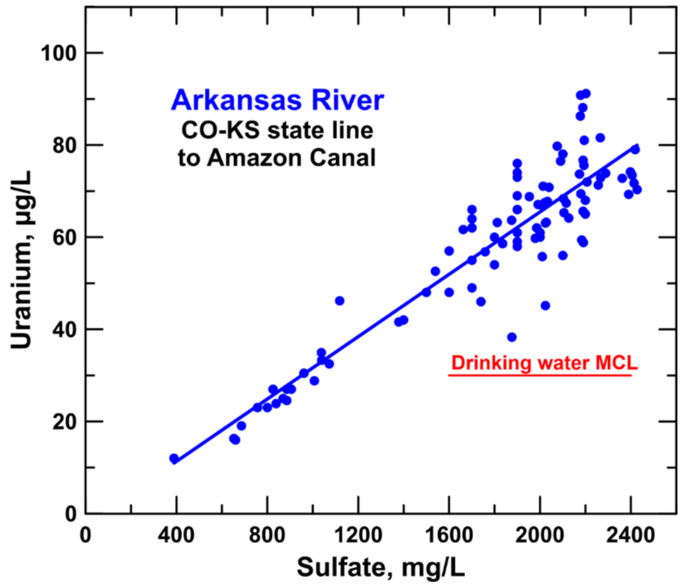


(C) Calculated sulfate concentrations

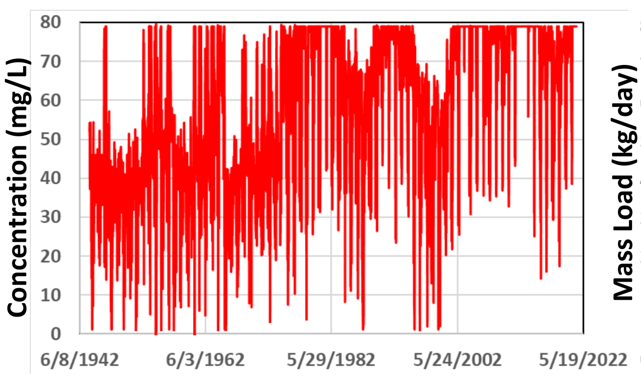


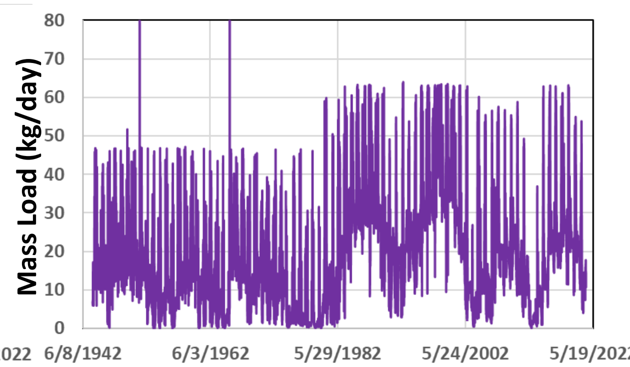
(D) Calculated sulfate mass loads

Figure 1. Arkansas river inflows, sulfate concentrations, and sulfate mass loads near the Kansas-Colorado state line. (A) The blue curve is from the Syracuse gage, and the orange curve is from the Coolidge gage. (B) The green and blue lines indicate the sulfate vs. flow relationships before and since 1981, respectively. (C) Sulfate concentrations are calculated based on the flow rates in (A) and the flow vs. concentration relationships in (B); the green curve is based on the relationship before Jan. 1, 1981, and the blue curve is based on the relationship afterward. (D) The total river sulfate mass loads are calculated by multiplying flow rates by concentrations.



(A) Uranium vs. sulfate concentrations in the Arkansas River





(B) Calculated uranium concentrations

(C) Calculated uranium mass loads

Figure 2. Calculated uranium concentrations and mass loads near the Kansas-Colorado state line. (B) Uranium concentrations are calculated based on the uranium vs. sulfate concentration linear regression relationship in (A). (C) Total river uranium mass loads are calculated by multiplying river flow rates by concentrations.

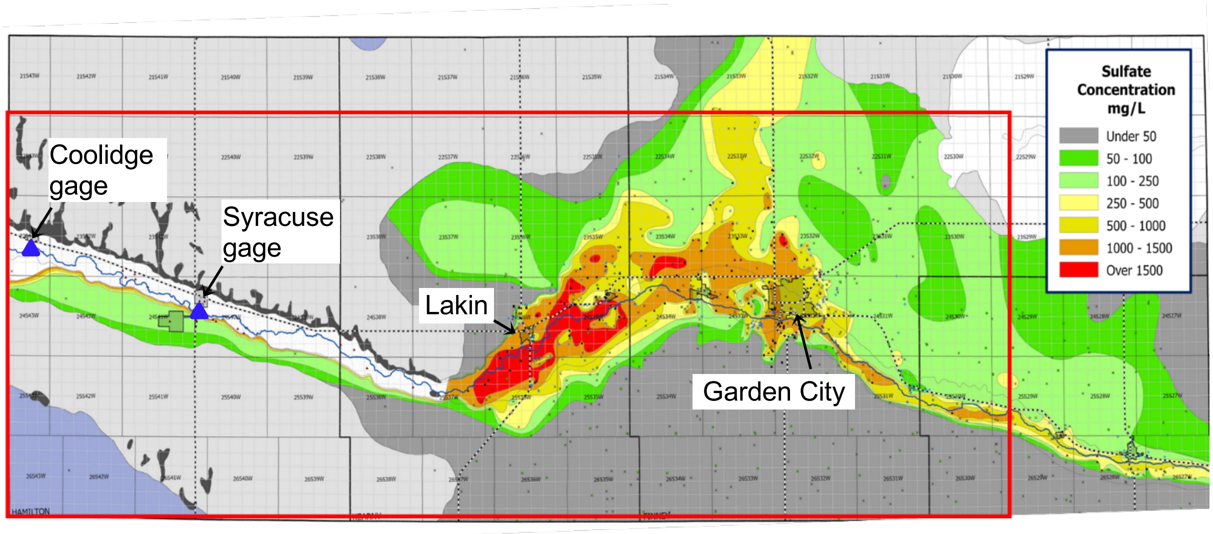


Figure 3. Map of the study area. The color contours show the average observed sulfate concentration between 2020 and 2022 in the HPA (Whittemore et al., 2023). The concentrations in the shallow Arkansas River alluvium are not plotted. The red box is the boundary of the groundwater contaminant transport model developed in this project.

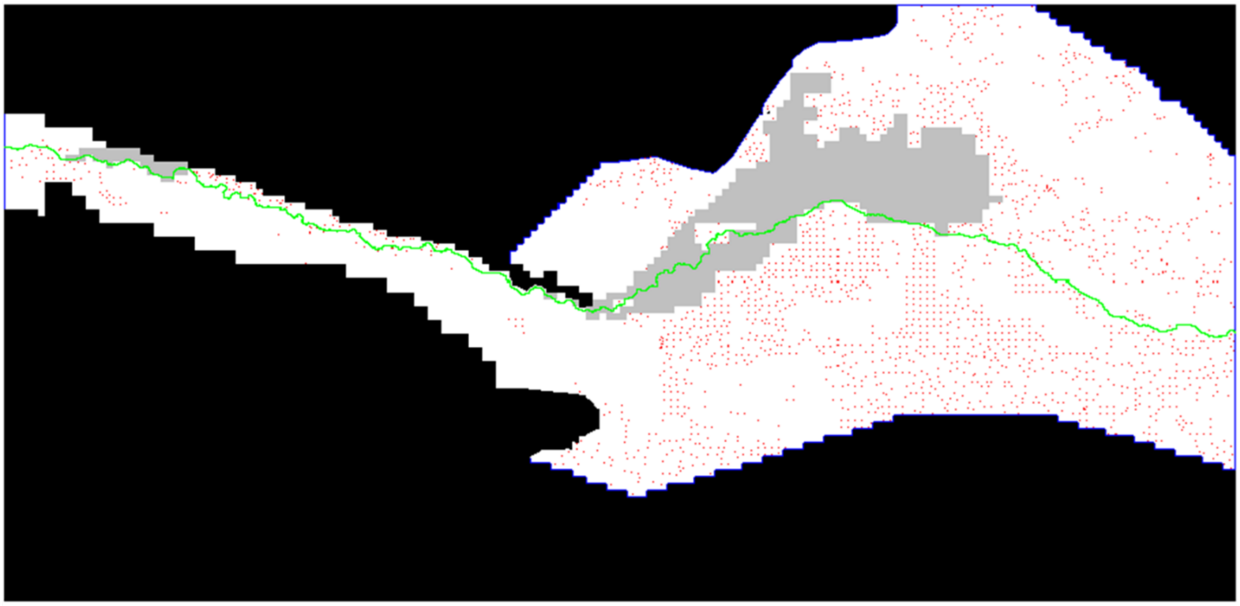
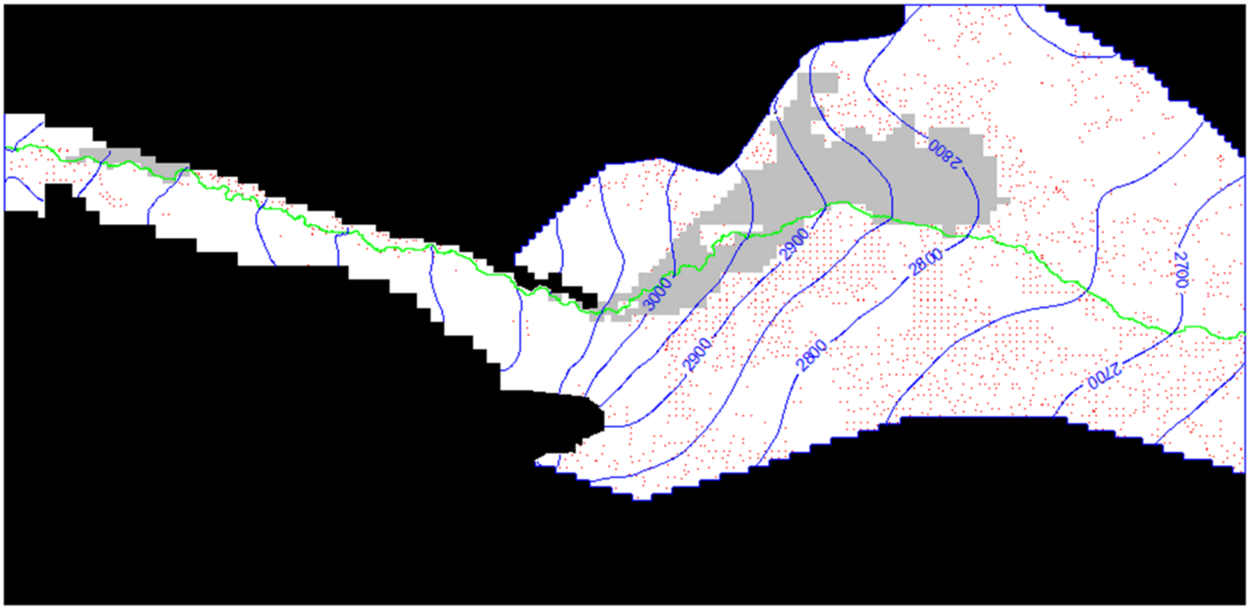
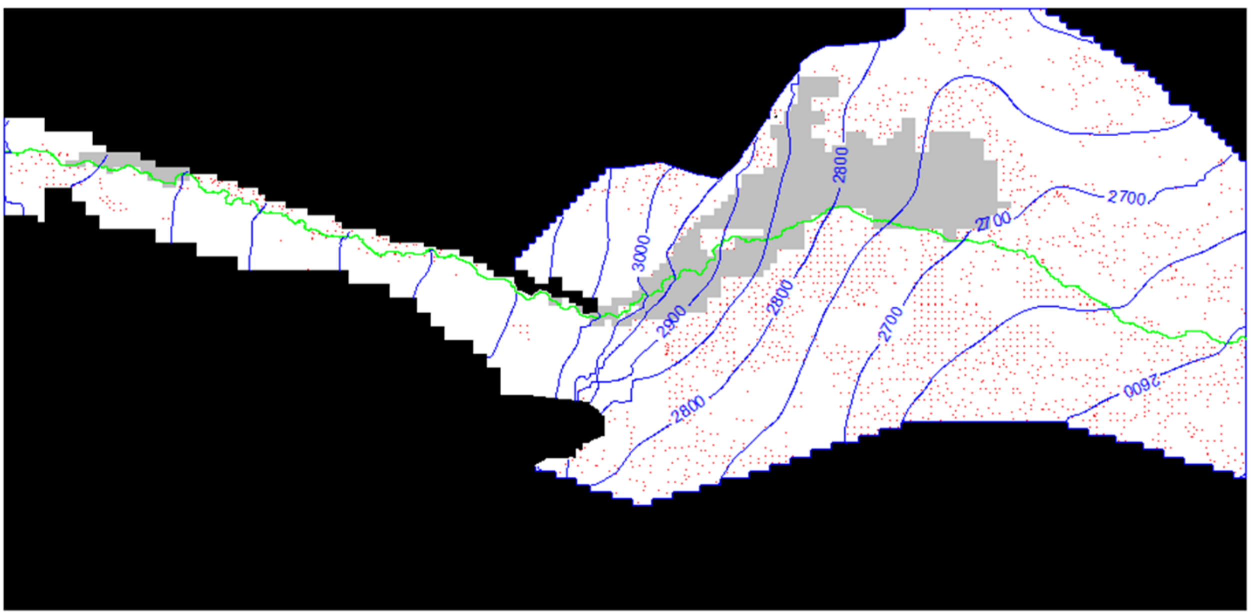


Figure 4. The upper Arkansas River groundwater contaminant transport model settings. The black area is inactive where no transport simulations are conducted due to either the large distance from the river or thin saturated thickness due to bedrock highs. The blue line represents specified heads with the values simulated by the regional GMD3 MODFLOW model. The green line represents the Arkansas River. The red points represent pumping wells, and the gray area represents the ditch canals and service areas with water diverted from the Arkansas River.



(A) Simulated groundwater levels in 2001



(B) Simulated groundwater levels in 2021

Figure 5. Simulated groundwater levels (feet above mean sea level) in (A) 2001 and (B) 2021.

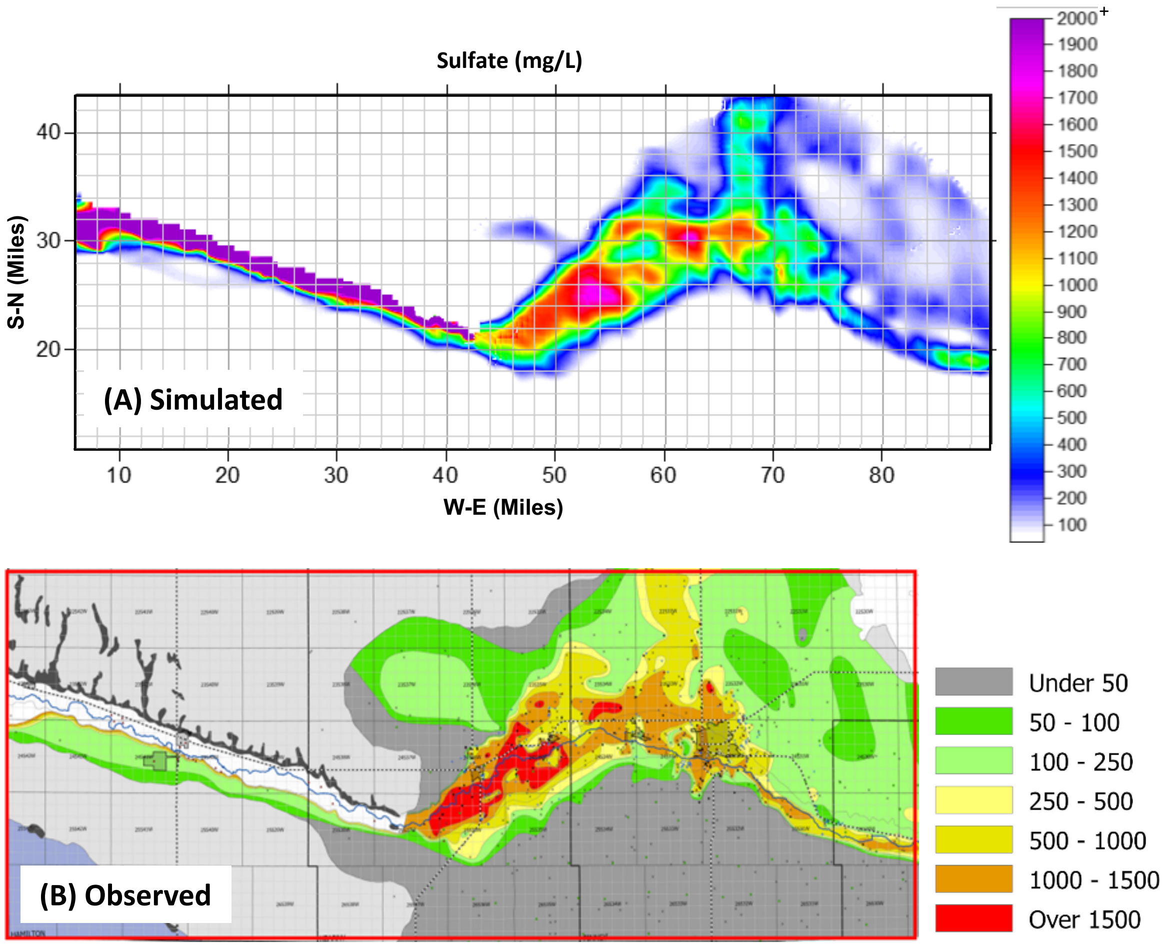


Figure 6. Simulated vs. observed sulfate concentrations in 2021. The 2000⁺ in the legend of (A) indicates the value is equal to or above 2,000 mg/L. To facilitate visual comparison, the simulated map shows the same areal extent as in the observed map by cutting away some eastern and southern model cells.

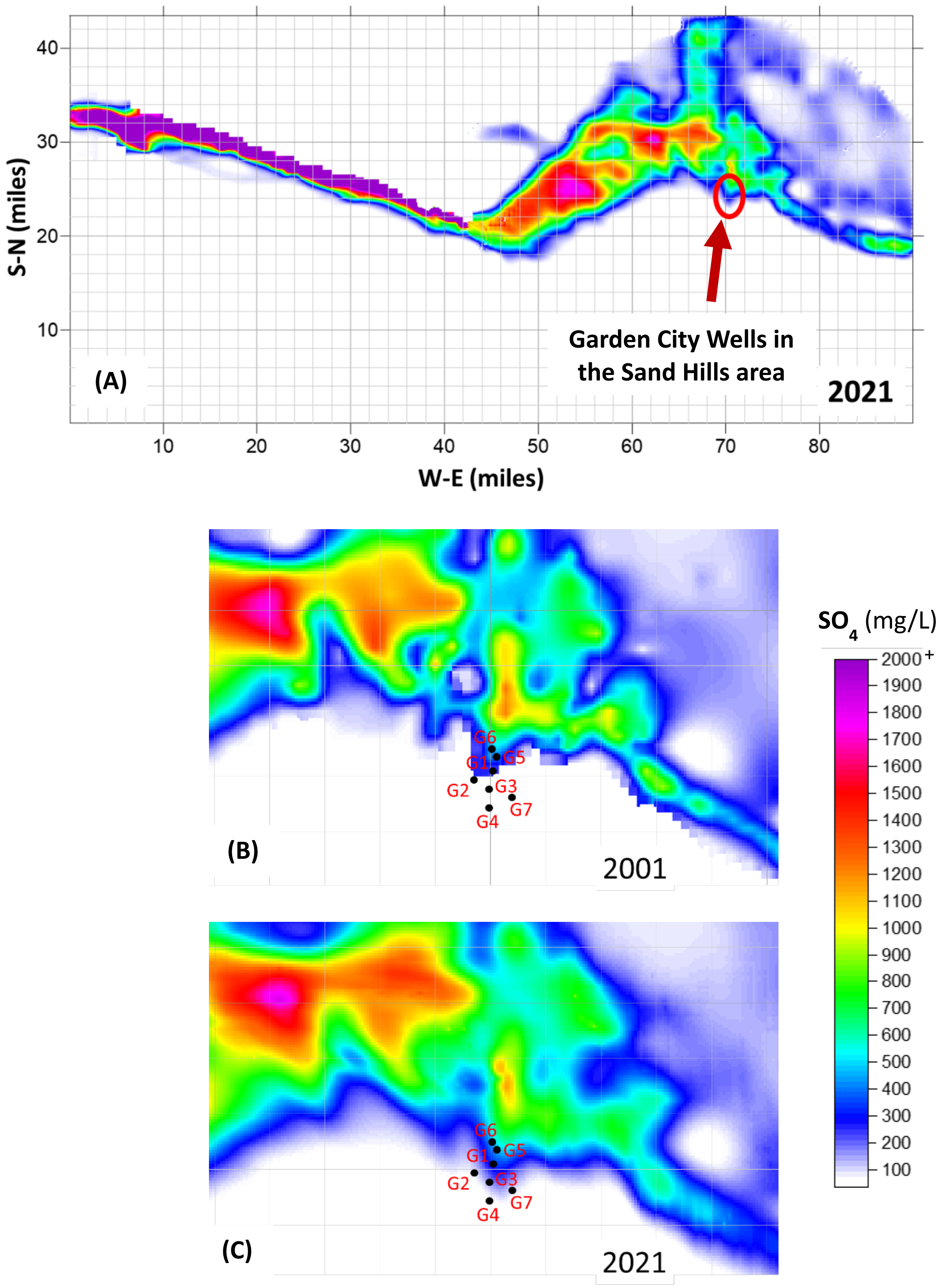


Figure 7. Simulated sulfate concentration around Garden City wells in the Sand Hills area: (A) Location of the Sand Hills wells, and simulated concentration in (B) 2001 and (C) 2021. There are seven wells, G1 to G7, on the southern edge of the plume.

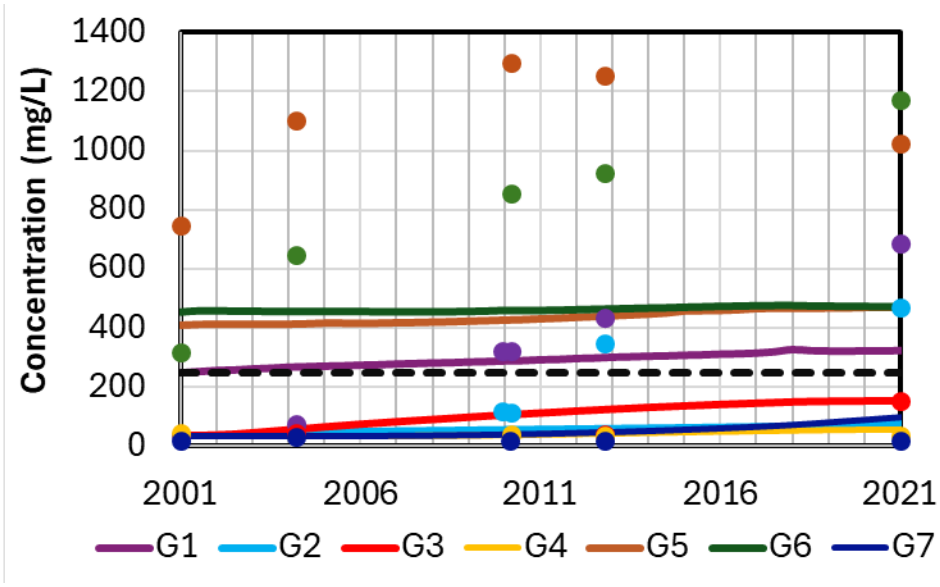


Figure 8. Comparison of simulated (curves) with observed (solid circles) sulfate concentration time series at the Sand Hills wells. Different wells are represented by different colors. Despite the differences in absolute values, both the simulated and observed values indicate relatively high concentrations in wells 5 and 6 and low concentrations in wells 3, 4, and 7. The EPA-recommended standard for sulfate in drinking water is 250 mg/L (black dashed line).

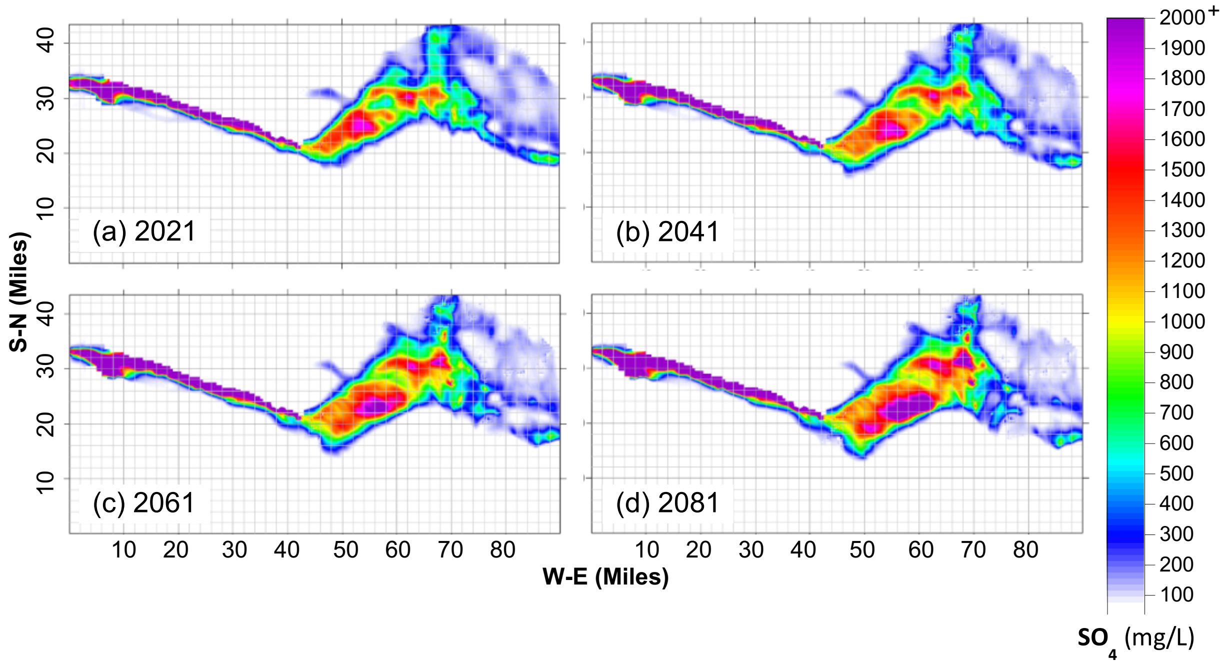


Figure 9. Simulated sulfate concentrations in the status quo scenario: (a) 2021, (b) 2041, (c) 2061, and (d) 2081.

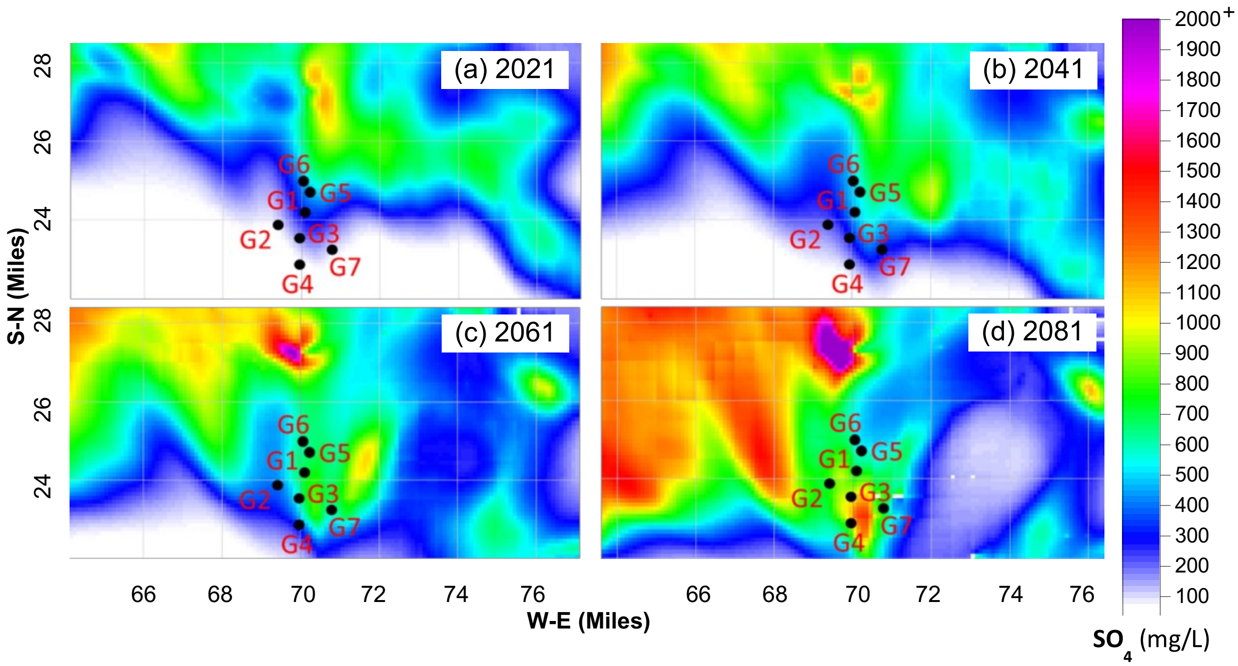


Figure 10. Status quo sulfate concentrations around Sand Hills wells: (a) 2021, (b) 2041, (c) 2061, and (d) 2081.

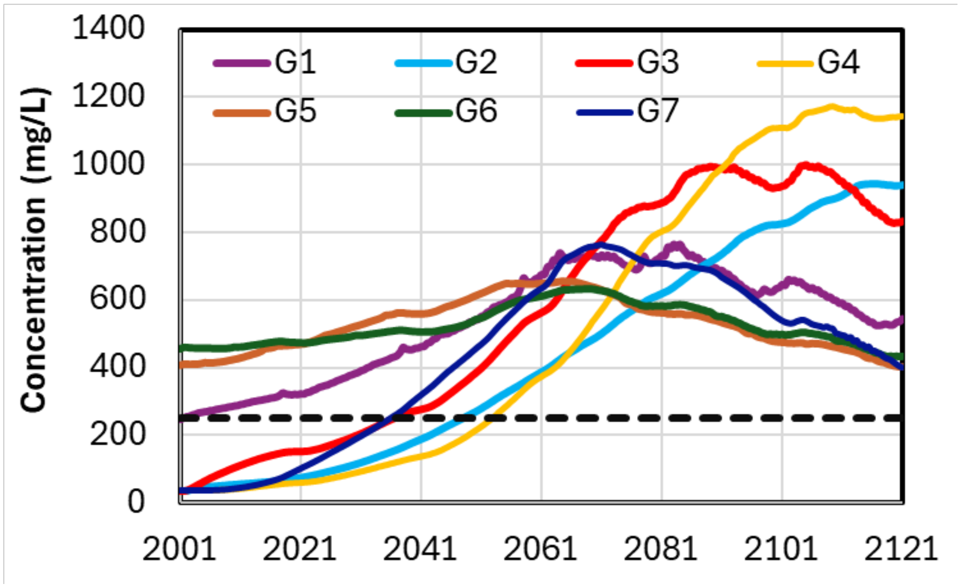


Figure 11. Status quo sulfate concentration time series at the Sand Hills wells. The EPA-recommended standard for sulfate in drinking water is 250 mg/L (black dashed line).

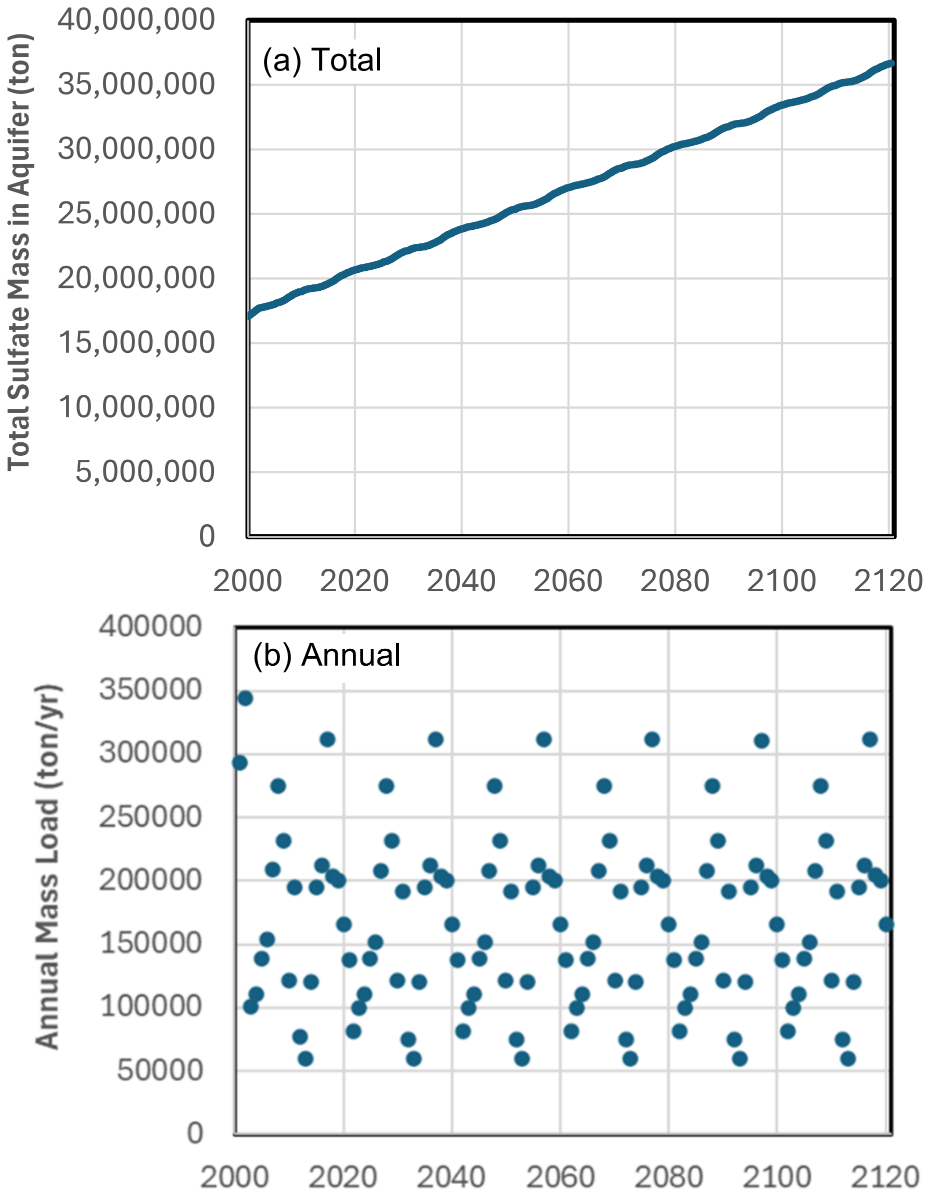


Figure 12. Simulated sulfate budgets in the status quo scenario: (a) total mass in aquifer and (b) annual mass load from Arkansas River infiltration and ditch return flow. The initial aquifer sulfate mass in 2001 is estimated to be 17,075,000 tons.

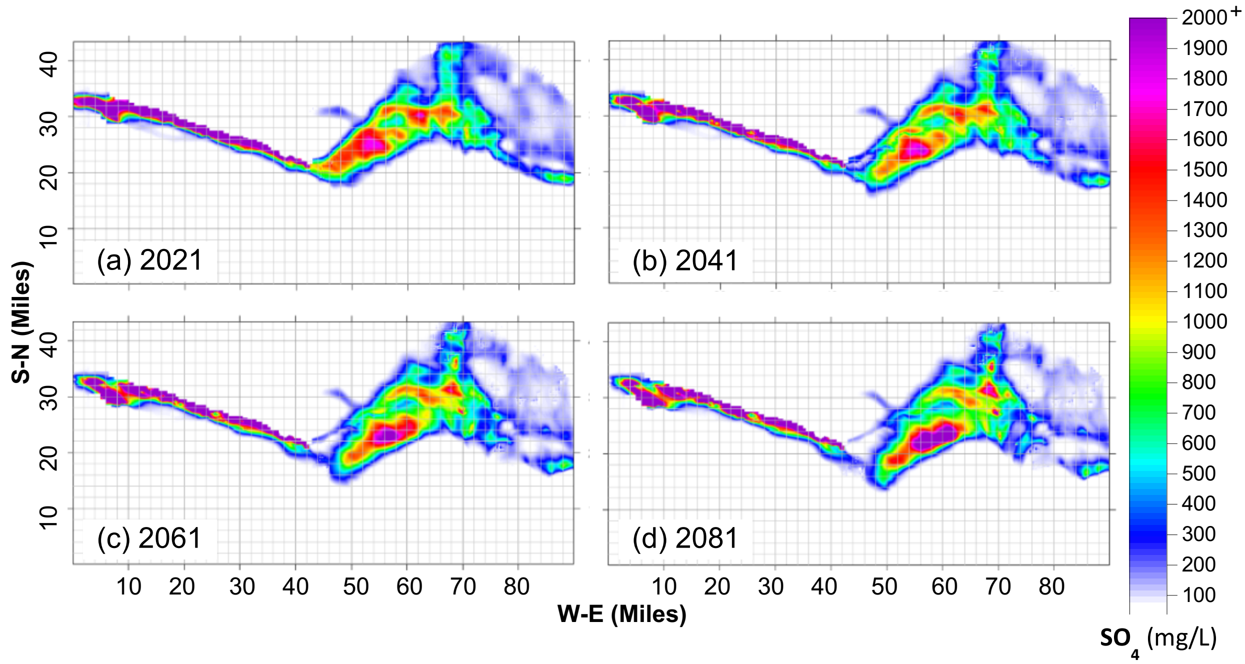


Figure 13. Simulated sulfate concentrations in the no future contamination scenario: (a) 2021, (b) 2041, (c) 2061, and (d) 2081.

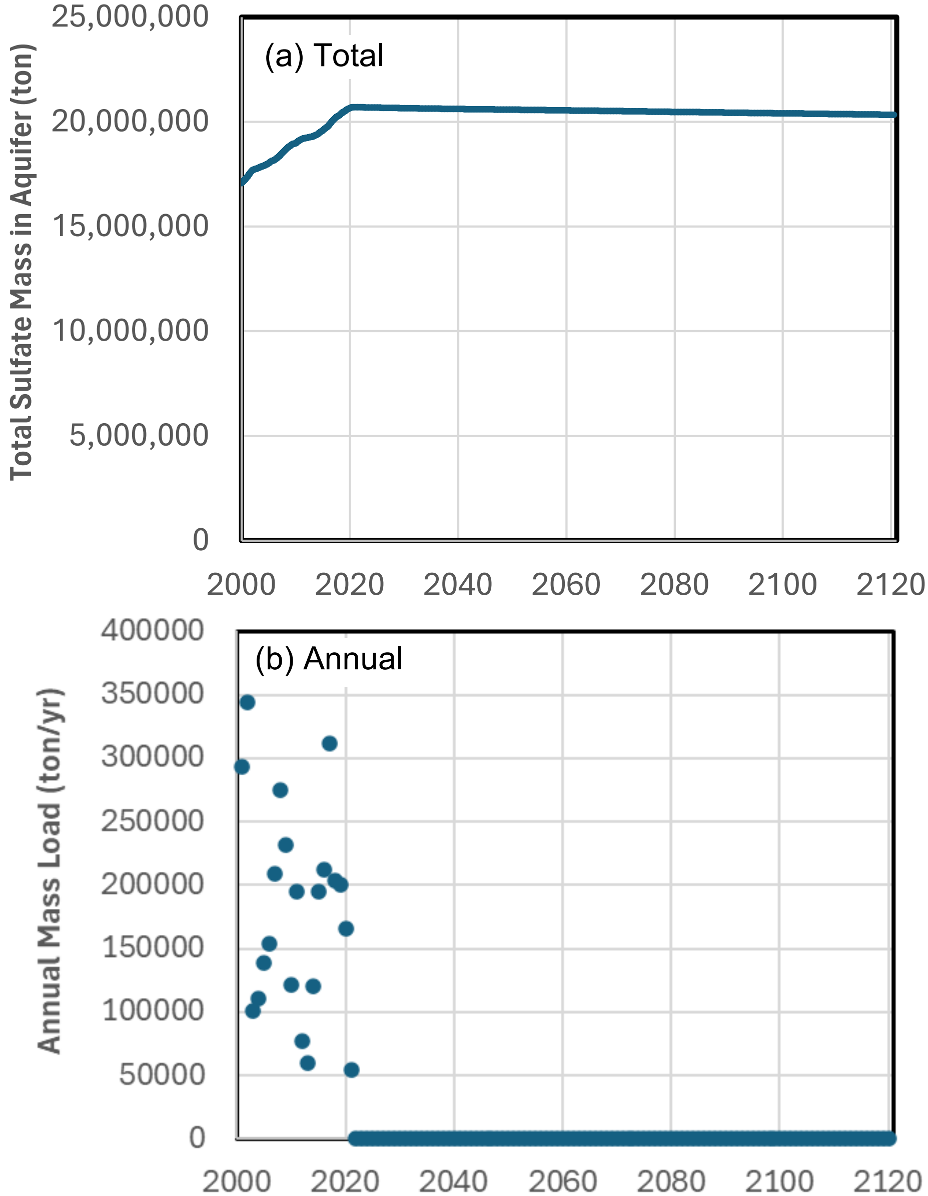


Figure 14. Simulated sulfate budgets in the no future contamination scenario: (a) total mass in aquifer and (b) annual mass load from Arkansas River infiltration and ditch return flow.

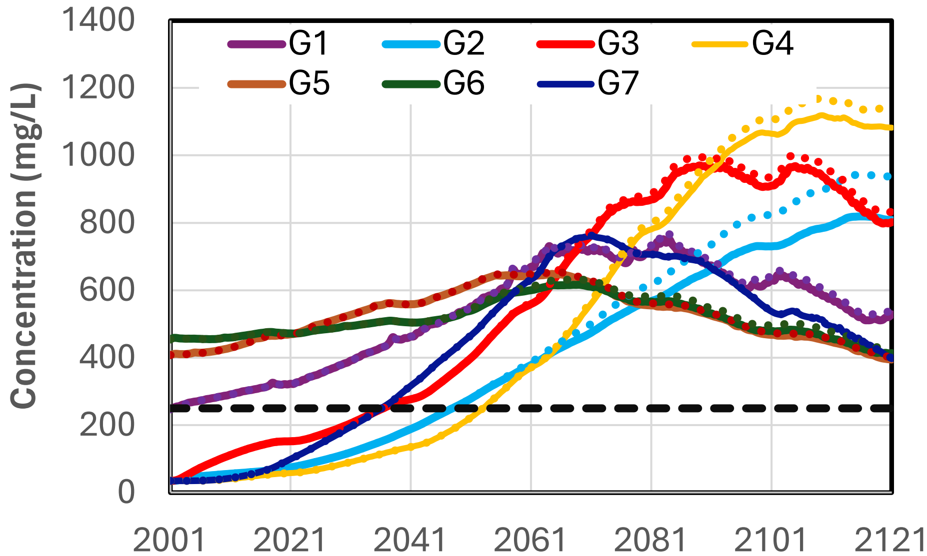


Figure 15. Comparison of sulfate concentration time series at the Sand Hills wells between the status quo (dotted lines) and no future contamination (solid lines) scenarios. The EPA-recommended standard for sulfate in drinking water is 250 mg/L (black dashed line).

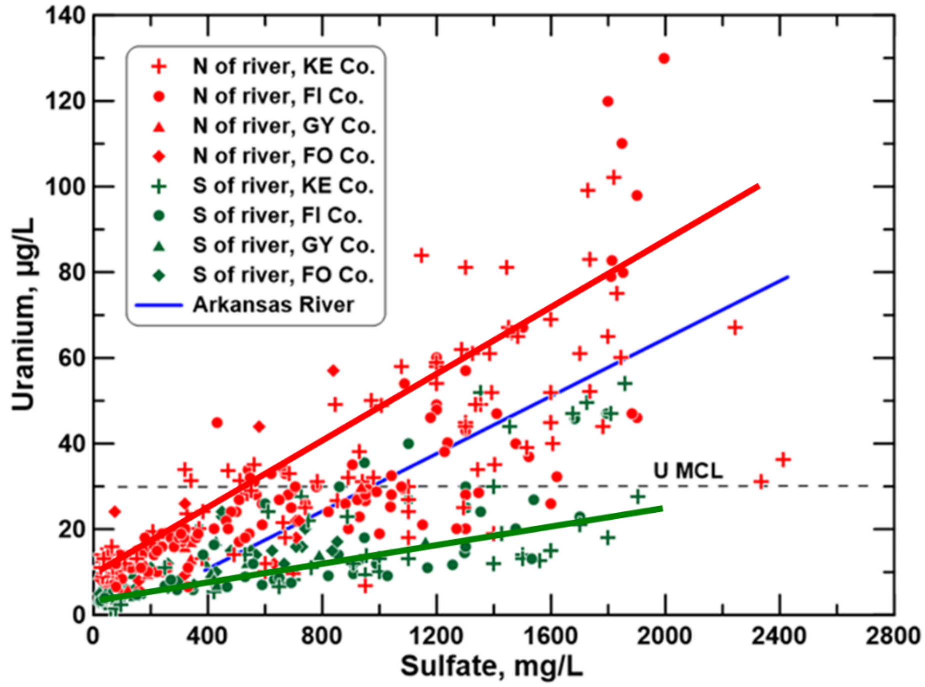


Figure 16. Relations between uranium and sulfate concentrations in groundwater samples in the Upper Arkansas River corridor (Whittemore et al., 2023). Two distinct uranium-sulfate relations were identified between the groundwater samples north (red line) versus south (green line) of the Arkansas River. The MCL for uranium in drinking water is 30 µg/L (dashed line).

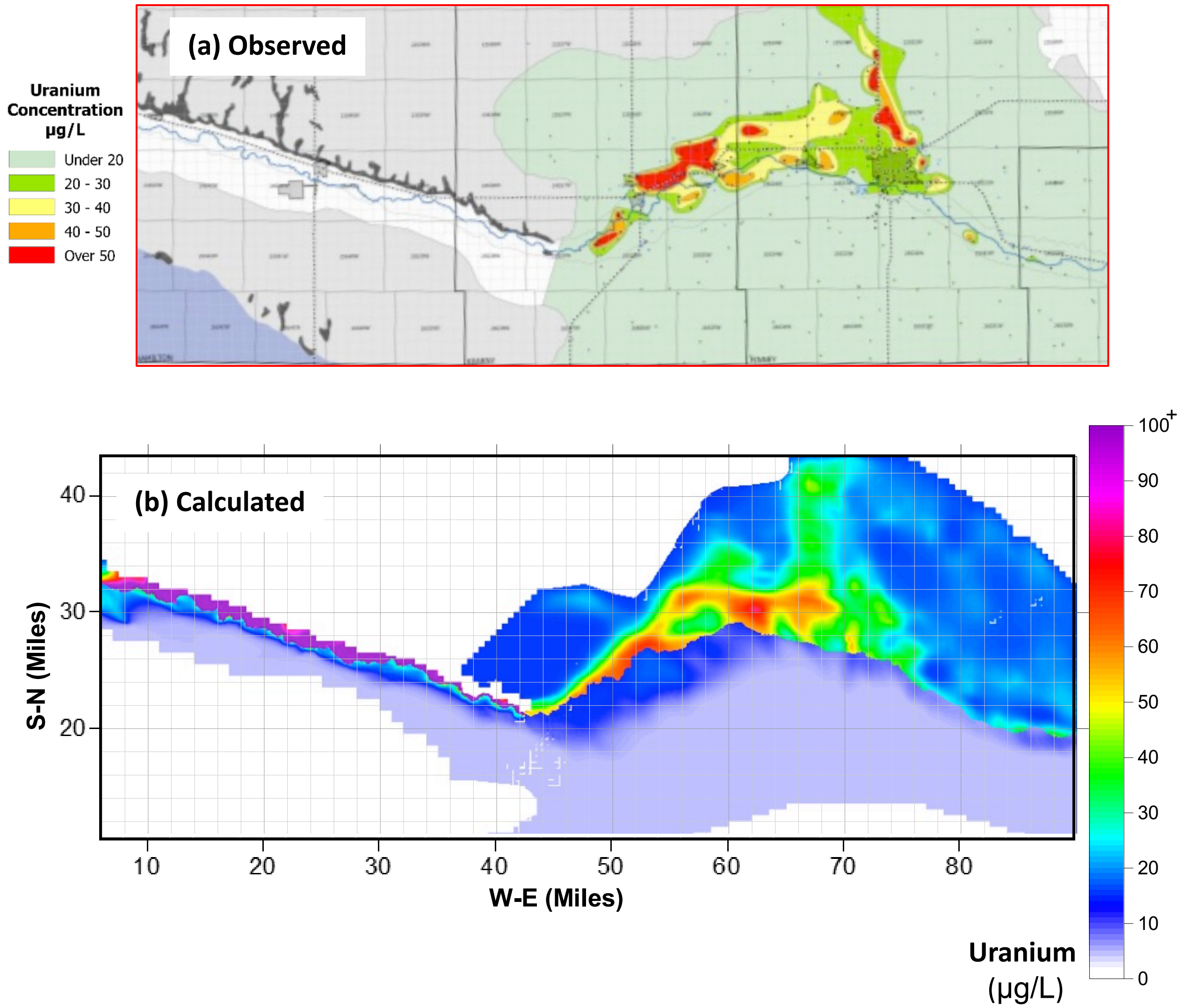


Figure 17. Comparison of observed and calculated uranium concentrations in 2021. In the observed map (A), the concentrations are plotted only for the HPA even though uranium is also high in the alluvium along the Arkansas River to the west of the center of township T. 25 S., R. 37 W. To facilitate visual comparison, the calculated uranium map (B) shows the same areal extent as in the observed map by cutting away some eastern and southern model cells.

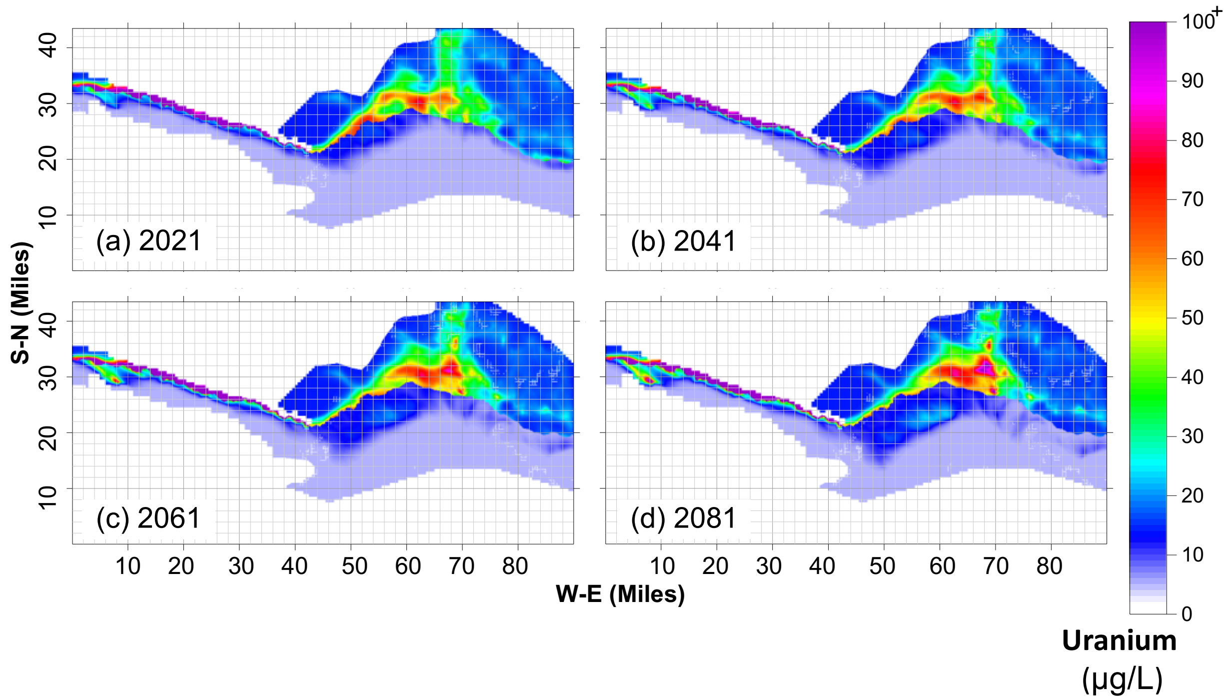


Figure 18. Calculated uranium concentrations in the status quo scenario: (a) 2021, (b) 2041, (c) 2061, and (d) 2081.

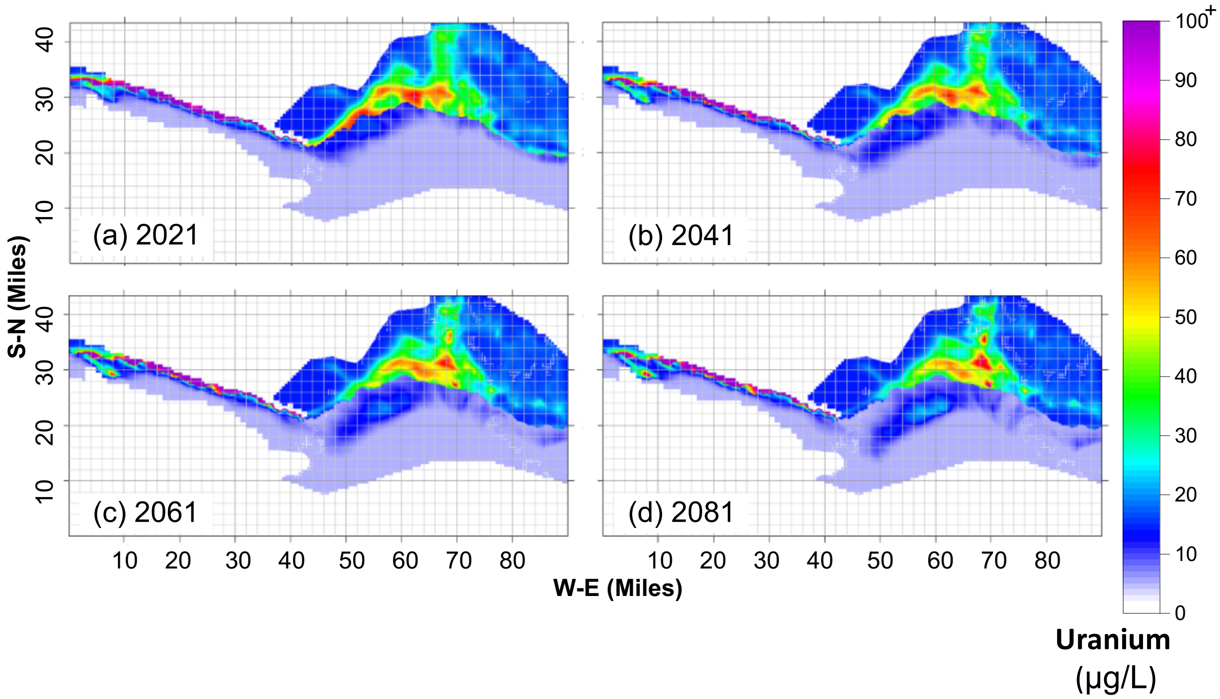


Figure 19. Calculated uranium concentrations in the no future contamination scenario: (a) 2021, (b) 2041, (c) 2061, and (d) 2081.

REFERENCES

Doherty, J., 2004, PEST—Model-Independent Parameter Estimation. User Manual, 5th Edition. Watermark Numerical Computing, 333 p.

Harbaugh, A. W., 2005, MODFLOW-2005, the U.S. Geological Survey modular ground-water model — the Ground-Water Flow Process: U.S. Geological Survey Techniques and Methods 6-A16.

Whittemore, D. O., 2000, Sulfate concentration for the High Plains aquifer in the Upper Arkansas River Corridor in southwest Kansas: Kansas Geological Survey, Open-File Report 2000-72D. http://www.kgs.ku.edu/Hydro/Publications/2000/OFR00_72/index.html

Whittemore, D. O., Seybold, E. E., Wilson, B. B., Woods, J. J., and Butler, J. J., Jr., 2023, Assessment of groundwater mineralization in the Upper Arkansas River Corridor: Kansas Geological Survey, Open-File Report 2023-21, 56 p.

Wilson, B. B., Liu, G., Bohling, G. C., Whittemore, D. O., and Butler, J. J., Jr., 2024, Update of the GMD3 groundwater flow model. High Plains Aquifer Modeling Maintenance Project: Kansas Geological Survey, Open-File Report 2024-14, 127 p.

Zheng, C., and Bennett, G.D., 2002, Applied Contaminant Transport Modeling (2nd ed.). Wiley-Interscience, New York.

Zheng, C., and Wang, P. P., 1999, MT3DMS, A modular three-dimensional multispecies transport model for simulation of advection, dispersion and chemical reactions of contaminants in groundwater systems: Vicksburg, Mississippi, Waterways Experiment Station, U.S. Army Corps of Engineers.