Transport of ammonia nitrogen for groundwater pollution control in an informal low-permeability landfill site

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ABSTRACT

In informal landfills, leachate leaked into the underlying soil and groundwater has been gaining increasing attention. Recently, an informal low-permeability landfill site in northern China was investigated in detail, and it was found that the groundwater was contaminated by leachate. A groundwater flow and contamination transport model was developed using FEFLOW to forecast contaminant transport and evaluate feasible pollutant control schemes. In this model, good matches between the simulated and observed groundwater level and good matches between the simulated and observed concentration of ammonia nitrogen show that the established model can reproduce the process of groundwater movement effectively. Three kinds of schemes, including natural conditions, pollution source removal, and pump and treatment, were simulated, and the results were compared. The results showed that, under natural conditions, the pollution does not travel far horizontally and vertically. Removal of pollution sources has little effect on the concentration of ammonia nitrogen in groundwater over a period of 100 years. The pump and treatment system can effectively remove high concentrations of ammonia nitrogen in the groundwater of the landfill site, with a maximum decline of over 90%. Therefore, the pump and treatment method may be valid for short-term soil and groundwater remediation.

Key words: ammonia nitrogen, contamination transport, FEFLOW, groundwater remediation scheme, Jing-Jin-Ji region, landfill

HIGHLIGHTS

• The paper verified the reliability of numerical model with water level and quality data on a site.
• The paper explored the variations in three kinds of groundwater remediation schemes in the low-permeability landfill site.
• The paper gave the appropriate discharge rate through multiple scenarios simulation.

GRAPHICAL ABSTRACT

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INTRODUCTION

Landfills are a major waste disposal option worldwide. Formal landfills must meet strict design, operation, and closure requirements, and leachate from the landfill must be collected and treated appropriately to prevent landfill leachate from contaminating the surrounding soil and groundwater (Mukherjee et al. 2015). Informal landfills are not constructed and operated in line with applicable standards and have the characteristics of simple construction and lack of environmental protection measures (Wan et al. 2021). However, many informal landfills still exist especially in developing countries (Hong et al. 2018). Researchers have identified approximately 200 hazardous compounds, such as aromatic, halogenated, ammonium, and sulfur compounds and heavy metals, in various landfill leachates (Jensen et al. 1999; Lou et al. 2009; Gajski et al. 2012). Pollutants in landfill leachate have cumulative and harmful effects on the ecology and food chain, leading to several health issues, including carcinogenic effects, acute toxicity, and genetic toxicity in humans (Kjeldsen et al. 2002). Groundwater pollution induced by leachate leaks from informal landfills is becoming an environmental issue globally (Han et al. 2016).

In China, the annual production of municipal solid waste has reached over $2 \times 10^8$ tons, and more than 60% of this waste has been sent to landfills, causing more than 400 cities to be surrounded by domestic waste and more than $5 \times 10^4$ hm$^2$ of land to be occupied by the waste (Tan et al. 2020). In addition, thousands of informal landfills in China, either operational or closed, have increasingly caused groundwater contamination (Han et al. 2016; Cai et al. 2018). In a previous study, 95% of groundwater in the vicinity of six municipal solid waste landfills in Beijing was assessed in the bad quality category (Li et al. 2008). Gao et al. (2009) stated that 18 solid waste landfills in the Hebei Province of China cause serious daily pollution to the groundwater of the region. In recent years, the Chinese government has attempted to improve environmental problems and hence has increased investment in domestic waste treatment and informal landfill pollution control. Groundwater resource managers have been looking for ways and experimenting to control groundwater pollution in informal landfill.

Before groundwater pollution control is carried out on an actual informal landfill site, it is necessary to evaluate the effect of some widely used pollution control methods. These groundwater pollution control methods mainly include pollution source removal, pump and treatment, and permeable reactive barrier. Some studies have shown transport of the groundwater pollution induced by leachate leaks from informal landfills is an unresolved remediation challenge. Therefore, it is necessary to evaluate the effect of groundwater pollution control methods in low-permeability landfill site.

Low-permeability landfill is a landfill built in strata with hydraulic conductivities in the range of 0.001–0.1 m/d. Groundwater flows slowly in low-permeability landfills. Thus, these groundwater pollution control methods such as permeable reactive barriers, applied well in high-permeability landfills, do not take effect in low-permeability landfills. Contamination in low-permeability strata remains a significant and unresolved remediation challenge. Therefore, it is necessary to evaluate the effect of groundwater pollution control methods in low-permeability landfill site.

Ammonia nitrogen is a typical component of groundwater pollution in China. According to the Statistical Yearbook of China in 2018 (National Bureau of Statistics 2018), the combined Gross Domestic Product (GDP) in the Beijing-Tianjin-Hebei region, denoted as Jing-Jin-Ji in abbreviation short form, accounts for approximately 9.7% of China’s total GDP and is made up over 100 million people – 8% of China’s population and three times that of the Tokyo megalopolis. With the rapid development of economy and population, groundwater pollution due to high ammonia nitrogen have been paid progressively attention in Jing-Jin-Ji region in recent years. Leachate of municipal solid waste landfills, having high ammonium nitrogen content and lasting toxicological characteristics, are considered to be important sources of groundwater ammonia nitrogen contamination (Han et al. 2016). Although ammonia nitrogen does not cause cancer in humans, it may damage eyes, skin, and respiratory tissues and organs through skin contact, respiration, and other modes of exposure, affecting the normal physiological function of the human body (de la Hoz et al. 1996; Davidson et al. 2018). In addition, under appropriate conditions such as high groundwater DO and Eh, ammonia nitrogen in groundwater produces a nitrification reaction (Xin et al. 2019), and nitrite, as an intermediate, can cause cancer, teratogenesis and mutation.

The study site, a landfill, is located in the coastal economic development zone of the Jing-Jin-Ji region of China. The stratum of study site is a part of the stratum in the North China Plain. The shallow stratum is mainly quaternary stratum, and the lithology is mainly clay, silt, and silty clay, which is a typical low-permeability stratum in the Jing-Jin-Ji region. The site mainly comprises of shallow groundwater level, which greatly increases the contamination potential (Nahin et al. 2019). The landfill has caused ammonia nitrogen contamination of the surrounding soil and groundwater due to the lack of...
environmental protection measures. The objective of this study is to evaluate the effect of groundwater pollution control methods in this low-permeability landfill site. We have managed to model the transport of ammonia nitrogen in the low-permeability landfill site and used the model to investigate remediation options. The results could provide a reasonable suggestion for a groundwater remediation scheme for this type of landfill.

MATERIALS AND METHODS

Site description

The study area, which is an informal municipal solid waste landfill, is in the Jing-Jin-Ji region of China (Figure 1(a)), approximately 48 km from the coast of Bohai Bay. The landfill was used to load municipal solid waste beginning in 2013 and was decommissioned in 2015, causing accumulative amounts of solid waste of approximately 700,000 m³ and leachate of approximately 800,000 m³ calculated by the ‘infiltration coefficient method’. The landfill covered an area of ~0.16 km² and had a landfill depth of ~11 m. The terrain of the landfill site was high in the west and low in the east, with an average elevation of 0.29 m, taking mean sea level as base elevation. The study area had a warm temperate sub-humid continental monsoon climate, with an annual average temperature of 11.8 °C, an average annual precipitation of 598.5 mm, and an average annual pan evaporation of 1,142.9 mm. Groundwater in the study area is recharged mainly through precipitation infiltration and lateral inflow and discharged mainly through unconfined evaporation and lateral outflow. Because there was no obvious groundwater pumping in the study area, the change in groundwater level was relatively small over many years, and the saturated groundwater flow was almost under the equilibrium state.

Nineteen wells were drilled in the study area (Figure 1(b)) for the monitoring of water level and quality, and the groundwater sampling depths were approximately 15, 31, and 45 m. P0 is located upstream of the landfill and can be regarded as the background groundwater quality in this region. P1, P2, P3, P4, and P5 are distributed in the landfill site. P6 and P7 are located downstream of the landfill. The cross-section of AA’ (Figure 1(b)) will be used to present our simulation results.

The strata in the study area included Quaternary Holocene, Upper Pleistocene, and a part of Middle Pleistocene strata. According to geological age and genetic type, the strata in the exploration depth (80 m) were divided into 10 engineering geological layers (Figure 2). The first, second, third, and fourth continental layer were deposited in the riverbed-floodplain facies, and the fifth continental layer was deposited in the limnic facies–river floodplain facies. The first, second, and third marine layer were deposited in the littoral facies.

The aquifer system can be generalized into three types, which are the unconfined aquifer, first confined aquifer, and second confined aquifer. Each aquifer is separated by a weakly permeable layer, which is aquitard. The lithology of the unconfined aquifer with the buried depth range from 1 to 15 m is silt or silty clay. The first aquitard with the buried depth of 15–20 m consists of clay or silty clay. The lithology of the first confined aquifer with the buried depth of 20–31 m is mainly silt, silty sand, or even fine sand. The second aquitard is a silt clay or clay layer having uneven thickness. The lithology of the second confined aquifer with the buried depth of 33–45 m is mainly a silty sand and silt layer. The lowest aquitard consists mainly of silty clay. According to the regional and hydrogeological survey data, the hydraulic conductivity of the unconfined aquifer varies from $6.0 \times 10^{-4}$ to $7.1 \times 10^{-2}$ m/d, and the hydraulic conductivity of the first and second confined aquifer is in the range of $1.9 \times 10^{-3}$ to $3.4 \times 10^{-1}$ m/d (Lei 2018). The depth to groundwater in the unconfined aquifer generally varies from 1.5 to 3.1 m, and groundwater level did not change much during the period of observation from 2017 to 2018. Generally, groundwater flows from northwest to southeast (Figure 1(a)).

Soil and water sampling and measuring

Fifteen leachate samples from 15 leachate sampling points (Figure 1(b)) were collected using a water sampler. The pH tested in the field ranged from 6.6 to 7.6. The chemical oxygen demand (COD) and biological oxygen demand over 5 days (BOD₅) in all samples varied from 1,000 to 14,000 mg/L and from 100 to 1,100 mg/L, respectively. The total nitrogen and ammonia nitrogen concentrations varied from 300 to 4,500 mg/L and from 240 to 2,358 mg/L, respectively. The amounts of COD, BOD₅, total nitrogen and ammonia nitrogen were higher than limits of water pollutants discharge concentration in the Standard for Pollution Control on the Landfill Site of Municipal Solid Waste (GB 16889-2008) (State Administration for Market Regulation of the People Republic of China 2008). Lead and arsenic in the leachate were within the limits. In some samples, cadmium and mercury exceeded the limits. The maximum detection value of cadmium was 0.0208 mg/L, exceeding the limit by 1.08 times, and the maximum detection value of mercury was 0.157 mg/L, which exceeded the limit by 156 times. Chromium generally exceeded the limit of 0.1 mg/L, with a concentration of 0.191–0.944 mg/L. The overall water quality of the
The leachate was characterized by high COD, high ammonia nitrogen, low biodegradability, high chromium, and high local regional cadmium and mercury.

Groundwater samples were collected at different depths by depth-fixed sampling in eight groundwater monitoring wells (Figure 1(b)) after the wells had been adequately cleaned. The concentrations of ammonia nitrogen (NH$_4^+$), nitrite (NO$_2^-$), nitrate (NO$_3^-$), total iron (Fe), ferrous iron (Fe$^{2+}$), permanganate index, total organic carbon (TOC), and BOD$_5$ in the collected samples were analyzed in the laboratory. The pH and electrical conductivity of the groundwater were measured in the field using a multi-parameter analyzer. Figure 3 shows the values of physicochemical parameters of groundwater samples taken from the landfill at depths of 15 m in November 2017. As can be seen from Figure 3, the differences in water quality...
were significant between the samples. Notably, the sample obtained from P3 was more contaminated than other points, probably because there was a large leachate leak near P3. In general, the concentration of ammonia nitrogen, iron, and permanganate index gradually decreased with the increase of distance between groundwater monitoring points and pollution sources. Groundwater in the landfill flows from northwest to southeast. P4, P5, P6, and P7 are mainly distributed in the southeast of the landfill and P0, P1, and P2 are mainly distributed in the northwest of the landfill. The concentration of ammonia nitrogen, iron, and permanganate index in the southeast was higher than in the northwest, which may be caused by the leakage and migration of pollutants in landfill. The concentration of ammonia nitrogen at all sampling points was 0.5–6 mg/L. The concentration of nitrite was extremely low, with a maximum of 0.17 mg/L. The concentration of nitrate at all sampling points was less than 2.0 mg/L, and the pH ranged from 7.0 to 8.5. The concentrations of Fe and Fe$^{2+}$ were less than 2.0 mg/L at all sampling points, and higher than 0.3 mg/L at most sampling points. The concentration of permanganate index at all sampling points was 8.0–15.0 mg/L, and the concentrations of TOC and BOD$_5$ were mostly 7.0–9.0 mg/L. The parameter standards are referenced to the level III water quality category cited in the ‘Standard for Groundwater Quality’ (GB/T14848-2017) (State Administration for Market Regulation of the People Republic of China 2017). Compared with the standard, the concentration of these physicochemical parameters of groundwater samples in the landfill area had the characteristics of high ammonia nitrogen, iron, and permanganate index. Therefore, the landfill site has caused pollution to the surrounding groundwater.

Figure 2 | 3D geological structure.
Figure 3 | Physicochemical parameters of groundwater samples around the landfill in 2017: (a) concentration of three nitrogen; (b) concentration of two iron; and (c) other parameters.
Adsorption of ammonia nitrogen

The adsorption of ammonia nitrogen by the strata in the landfill satisfies the Langmuir isothermal adsorption relationship (Gao et al. 2008). The Langmuir equation (also known as the Langmuir isotherm) was applied (Diersch 2014) as follows:

\[ \frac{C_2}{C_1} = \frac{k_1}{1 + k_2 C_1} \]  

(1)

where \( C_2 \) is the concentration of an adsorbate in the solid phase in the state of equilibrium (unit of adsorbate mass per unit of sorbent volume), \( C_1 \) is the concentration of the adsorbate in the liquid phase in the state of equilibrium (unit of adsorbate mass per unit of solution volume), and \( k_1 \) and \( k_2 \) are Langmuir sorption coefficients.

From experimental studies in a similar nearby location (Gao et al. 2008), the Langmuir equation is \( C_2/C_1=30.3769/(1+0.0435C_1) \), \( C_2/C_1=95.0191/(1+0.042C_1) \), and \( C_2/C_1=99.028/(1+0.038C_1) \) for silt, silty clay, and silty sand, respectively. The equation in the layer 1, 2, 3, 4, 5, and 6 in Table 1 are applied by using equation for silt, silty clay, silty clay, silty sand, and silty clay, respectively. The sorption coefficient 1 \( (k_1) \) and sorption coefficient 2 \( (k_2) \) were 30.3769 and 0.0435 l/mg in the first model layer, 99.028 and 0.038 l/mg in the layers 3 and 5, and 95.0191 and 0.042 l/mg in the layers 2, 4, and 6, respectively.

Numerical model

Numerical models are important tools for predicting solute transport and assessing possible effective means for controlling and remediating groundwater pollution. The model FEFLOW (Radulescu et al. 2007; Widomski et al. 2015; Hansen et al. 2019; Hu et al. 2019; Matiatos et al. 2019; Danielescu et al. 2020) was chosen to simulate the transport of ammonia nitrogen in this study. The general formulations of the flow and mass balance equations for FEFLOW were provided by Diersch (2014). To avoid the influence of imprecisely defined boundary conditions, the model area extended outside the informal landfill site (Figure 1(a)). The model domain was about 34 km². Triangular finite-element meshes are generated by Triangle algorithm. The triangular mesh is extended to the third dimension by extruding the 2D mesh, resulting in prismatic 3D elements. To better simulate the solute transport at a finer spatial resolution, the grid cells in the landfill site were refined. The model area was discretized into 6,403 nodes and 12,618 grid cells for each model layer. There were six model layers in total (Figure 4). The total thickness of the model layers was about 100 m. The model stratification and estimation of the hydraulic conductivities for each layer are presented in Table 1.

Both the northern and southern parts of the model area are adjacent to rivers and the depth to groundwater is shallow. Therefore, the two boundaries are represented as a Dirichlet (the first-type) hydraulic-head boundary type. Groundwater flows in from the western boundary and flows out from the east boundary of the model domain. The western and eastern boundaries of the model domain are represented as a Neumann (the second-type) fluid–flux boundary condition. The flux of the western and eastern boundaries is set to \( 1 \times 10^{-5} \) m/d and \( 1 \times 10^{-4} \) m/d, respectively. The initial concentration of ammonia nitrogen in model was set as 0.5 mg/L. The pollution source (Figure 1(b)) and the western boundaries of the model area extended outside the informal landfill site were redefined. The model was run for 10 years, and the calculated hydraulic heads and solute concentrations were stored with a time interval of 365 days.

Table 1 | Model stratification and parameter estimation for the hydrogeological model

| Serial number of layer | Name                   | Range of depth below surface (m) | Lithology                  | Hydraulic conductivity (m/d) |
|------------------------|------------------------|----------------------------------|----------------------------|----------------------------|
|                        |                        |                                  |                            | \( K_{xx}/K_{yy} \) \( K_{zz} \) |
| 1                      | Unconfined aquifer     | 1–15                             | Silt, Silty clay           | \( 2.0 \times 10^{-2} \) \( 4.0 \times 10^{-3} \) |
| 2                      | The first aquitard     | 15–20                            | Silty clay, Silty sand     | \( 1.2 \times 10^{-3} \) \( 2.4 \times 10^{-4} \) |
| 3                      | The first confined aquifer | 20–30                        | Silty sand, Fine sand, Silt | \( 2.0 \times 10^{-1} \) \( 4.0 \times 10^{-2} \) |
| 4                      | The second aquitard    | 30–35                            | Silty clay                 | \( 2.0 \times 10^{-4} \) \( 4.0 \times 10^{-5} \) |
| 5                      | The second confined aquitard | 35–45                          | Silty sand, Silt           | \( 1.0 \times 10^{-2} \) \( 2.0 \times 10^{-3} \) |
| 6                      | The third aquitard     | 45–100                           | Silty clay                 | \( 2.0 \times 10^{-4} \) \( 4.0 \times 10^{-5} \) |
domain was set to 0.5 mg/L. The infiltration rate into groundwater on the top of the model was set at \(5.3 \times 10^{-7}\) m/d. Transient solute transport began in 2013. The forward Adams-Bashforth/backward trapezoid (AB/TR) method was used for simulation-time control. The trial-and-error method and the automatic parameter estimation (FePEST) method were used to obtain optimal parameter estimations. The main parameters used in model calibration included \(K_{xx}, K_{yy}, K_{zz}\), porosity, longitudinal dispersivity, transverse dispersivity, and sorption coefficient. The period for model calibration was set to 5 years. The optimal parameters for the contaminant transport modeling were found to be the following: porosity set at 0.1 for all model layers; longitudinal dispersivity of 0.5 m in layers 2 and 4, and 5 m in the other layers; and transverse dispersivity must be an order of magnitude smaller than the longitudinal dispersivity. The calibrated model was then used to predict the transport of ammonia nitrogen after long-term seepage of leachate under natural conditions and determine the distribution of the ammonia nitrogen plume after pollution source removal and pump and treatment.

RESULTS AND DISCUSSION

Model calibration

Calibration targets included the observed groundwater levels in eight wells as well as the concentration of ammonia nitrogen. The relationship between the simulated and observed water levels and solute concentration is shown in Figure 5. Notably, the simulated depth to groundwater matches well with the observed results. Additionally, the simulated concentration correlated well with the observed concentration. The observed concentration of ammonia nitrogen ranged from 0.72 to 5.32 mg/L, and the simulated concentration ranged from 0.50 to 3.59 mg/L. After model calibration, the simulated actual groundwater flow velocity at the landfill site was approximately 0.052 cm/d, and the hydraulic gradient was approximately 0.13‰, indicating that groundwater moved very slowly. The simulated and observed concentration of ammonia nitrogen in some wells had a significant difference in this study, probably because of the heterogeneity of stratigraphic parameters and influences of uneven distribution of pollution sources. The simulated ammonia nitrogen concentration in Well P2 is 3.28 mg/L, which is higher than the observed result (0.86 mg/L), probably because the low hydraulic conductivities are present near Well P2 and the average concentration of pollution sources in the model is overestimated. On the contrary, the simulated ammonia nitrogen concentration in Well P4 (1.14 mg/L) is underestimated when compared with the observed value (3.61 mg/L) probably due to the low initial concentration in the model.
Sensitivity analysis of parameters

The parameters in the model have uncertainties due to the heterogeneity of the porous medium, which will cause uncertainties in the model results. Therefore, sensitivity analyses of sorption coefficient 1 and sorption coefficient 2 were performed to test the model reliabilities. The values of sorption coefficient 1 and sorption coefficient 2 in Scenario 1, 2, 3, and 4 were changed by −20, −10, +10, and +20% for the sensitivity analysis. The sensitivity coefficient is defined as Equation (2).

\[ S = \frac{|C_i - C_0|}{C_0} \]  

where \( S \) is the sensitivity relative to the base scenario, which is dimensionless; \( C_i \) is the maximum solute concentration under scenario \( i \) (mg/L); and \( C_0 \) is the maximum solute concentration under the base scenario (mg/L).

The model was run over the duration of 100 years. The concentration of ammonia nitrogen at the OP (Figure 1(b)) at the bottom of layer 1 was selected to calculate the sensitivity of the key parameters. In the four scenarios, the sensitivity of the sorption coefficient 1 is shown in Figure 6. The sensitivity of sorption coefficient 1 under the four scenarios was very small, with a maximum value of only 0.0049. This indicates that adsorption coefficient 1 has little effect on the transport of ammonia.
Distribution of contaminant plume after long-term seepage of leachate under natural conditions

We set the section line (A–A′), as shown in Figure 1(b). Simulated pollution plumes of ammonia nitrogen under natural conditions, with and without sorption, are shown in Figure 7. The pollution plume horizontally and vertically is mainly distributed in the unconfined aquifer (Layer 1) and first aquitard (Layer 2) after 100 years of ammonia nitrogen migration, which is due to the low hydraulic conductivity of the formation, resulting in slow transport of ammonia nitrogen. By comparing Figure 7(a) and 7(b), we found that when adsorption was present, the ammonia nitrogen pollution plume had not entered the first aquitard after 100 years. Therefore, adsorption could effectively hinder the downward migration of ammonia nitrogen. It can also be seen from Figure 7(a) and 7(b) that the horizontal migration distance of ammonia nitrogen in the section line (A–A′) is approximately the same. The distance beyond the edge of the landfill that the pollution travels is zero because of the low hydraulic gradient and relatively low hydraulic conductivity. In addition, it can be concluded that adsorption has little effect on the horizontal migration of ammonia nitrogen.

Contaminant transport after pollution source removal and pump and treat

Condition after removing pollution sources

The state of the ammonia nitrogen pollution plume was simulated under conditions of pollution source removal with and without sorption, and the ammonia nitrogen concentration in landfill leachate leakage over 100 years is shown in Figure 8. Comparing Figure 8(a) with Figure 8(b), we can draw the same conclusion by comparing Figure 7(a) with Figure 7(b). This is because adsorption can effectively hinder the downward migration of ammonia nitrogen and has little effect on its horizontal migration. However, comparing with Figure 8(a) and Figure 7(a), we found that the high ammonia nitrogen concentration range (dark blue) at the top of the unconfined aquifer shown in Figure 7(a) was significantly larger than that in Figure 8(a). In addition, the maximum concentration of ammonia nitrogen in Figure 8(a) was 2,268.33 mg/L, which is 81.67 mg/L less than that shown in Figure 7(a). The maximum concentration of ammonia nitrogen at the top of the unconfined aquifer in Figure 7(b) was 2,350 mg/L, whereas in Figure 8(b), it was 2,232.32 mg/L. Therefore, we deduce that the removal of pollution sources can reduce the concentration of ammonia nitrogen in the groundwater of the landfill but only by a small amount. The reason for this result is that the transverse and vertical permeability coefficient of the landfill are so low that the migration of ammonia nitrogen in the aquifer is mainly controlled by hydrodynamic conditions.

Pump and treatment scenarios

Pump and treatment is the most widely used technology in the remediation of groundwater contaminated sites. The technology can quickly change the velocity and direction of groundwater flow and change the diffusion direction and range of groundwater pollution plume. In this study, pump and treatment is possible because the study area has aquifers. On the basis of removing the source of pollution and considering adsorption, we conducted non-point source pumping at pollution sources (Figure 1(b)), covering an area of 140,000 m². The designed discharge rates were 0, 10, 50, 100, 150, and 200 m³/d.
To reflect the change in the concentration of ammonia nitrogen at the pollution source area, an observation point (OP) in the landfill was selected (Figure 1(b)). The initial concentration of ammonia nitrogen was 2,350 mg/L at the OP of the top of the unconfined aquifer. Figure 9 shows the variation in the concentration of ammonia nitrogen at the OP of the top of the unconfined aquifer under different discharge rates over 5 years. It can be seen from Figure 9 that concentration of ammonia nitrogen changes with time under different discharge rates in the landfill, and that increasing the pumping rate removes the pollution more quickly, with 94% removed in 5 years at the maximum pumping rate modeled (200 m³/d). When the discharge rate was greater than or equal to 100 m³/d, the decline of ammonia nitrogen concentration was greater than or equal to 80%. Figure 10 shows the ammonia nitrogen concentration after 5 years of pump and treatment at 100 m³/d. In Figure 10, the high concentration of ammonia nitrogen at the top of the unconfined aquifer (2,350 mg/L) was significantly reduced. Therefore, the pump and treatment method is a good option for removing high concentrations of ammonia nitrogen in groundwater for low-permeability landfill sites.

CONCLUSION

Soil and groundwater pollution around informal landfill sites has gained increasing attention in recent years. In this study, the Jing-Jin-Ji region of China with an informal low-permeability landfill site was studied in detail. A field investigation was carried out, and it was found that there was the layered geology of alternating aquifers and aquitards at the site, and high amounts of ammonia nitrogen in the groundwater. To propose remedial solutions for this type of landfill site in the Jing-Jin-Ji region of China, we successfully created a model by FEFLOW that was applied to evaluate the transport of ammonia nitrogen under natural and different remediation options. The main conclusions are as follows.

(1) The simulated actual groundwater flow velocity at the landfill site was approximately 0.032 cm/d, and the hydraulic gradient was approximately 0.13‰, indicating that groundwater moves very slowly in low-permeability landfills.
(2) In sensitivity analysis of parameters, sorption coefficient 1 had little effect and sorption coefficient 2 had no effect on the transport of ammonia nitrogen within the fluctuation range of 20%.

(3) Under natural conditions, the pollution does not travel far – horizontally because of the low hydraulic gradient and relatively low hydraulic conductivity, and vertically because of the aquitard layer and adsorption.

**Figure 8** | Ammonia nitrogen concentration in landfill leachate leakage over 100 years in condition removing pollution sources: (a) regardless of the sorption and (b) considering the sorption. Please refer to the online version of this paper to see this figure in colour: https://doi.org/10.2166/nh.2022.089.

**Figure 9** | Change of concentration of ammonia nitrogen with time under different discharge rates at the observation point (OP) at the top of the unconfined aquifer in the landfill.
(4) Removal of pollution sources can reduce the concentration of ammonia nitrogen in the groundwater of the landfill but only by a small amount. However, pump and treatment would be very effective. The concentration of ammonia nitrogen changes with time under different discharge rates in the landfill, and that increasing the pumping rate removes the pollution more quickly, with 94% removed in 5 years at the maximum pumping rate modeled (200 m³/d).

These findings may provide a theoretical reference for managing pollution problems at many similar informal low-permeability landfill sites that exist in China and elsewhere.

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CONFLICT OF INTEREST
The authors have declared no conflict of interest.

AVAILABILITY OF DATA AND MATERIAL
The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

AUTHORS’ CONTRIBUTIONS
Juanting Niu: Conceptualization, data processing and analysis, writing the original draft, and writing, reviewing and editing. Litang Hu: Conceptualization, methodology, writing, reviewing, and editing. Menglin Zhang: Data collecting and analysing.

DATA AVAILABILITY STATEMENT
Data cannot be made publicly available; readers should contact the corresponding author for details.

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