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MULTI-COMPONENT MODELING APPROACH TO ASSESS LANDFILL LEACHATE IMPACTS ON GROUNDWATER

Landfill leachate leakage poses long-term risks to groundwater due to the persistence of both reactive and conservative pollutants. This study applies an integrated modeling framework, combining geomembrane leakage estimation, leachate quality evolution, and groundwater transport, to evaluate the impacts of varying liner and subgrade contact conditions. The results show that poor contact increases leakage and pollutant loads by more than fivefold compared to good contact, demonstrating the critical role of interface quality in containment performance. Simulations indicate that concentrations of ammoniacal nitrogen (NH_3-N) and chloride (Cl^-) decline during active and closure stages but remain environmentally significant for decades, with NH_3-N showing residual releases and Cl^- accumulating persistently as a conservative tracer. Groundwater modeling reveals strong spatial heterogeneity, with downstream locations exhibiting substantially higher breakthrough concentrations. These findings highlight the long-term vulnerability of groundwater systems to landfill leakage and underscore the need for strict liner construction quality and sustained post-closure monitoring.

1. INTRODUCTION

Landfilling remains one of the most widely adopted methods for municipal solid waste (MSW) disposal worldwide [1], and continues to play a dominant role in solid waste management [2], particularly in regions where rapid urbanization and cost constraints limit the implementation of advanced treatment technologies [3]. However, the long-term environmental sustainability of landfills is strongly challenged by leachate generation and leakage [4]. As waste undergoes biochemical degradation, large volumes of leachate enriched with organic pollutants, ammoniacal nitrogen, chloride, and other contaminants are

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produced [5]. Even with engineered liner systems, variations in the contact quality between the geomembrane and its subgrade can substantially influence leakage rates, allowing pollutants to infiltrate underlying soils and aquifers [6]. Such leakage poses persistent risks to groundwater resources, making accurate prediction and mitigation of leachate migration a critical component of modern landfill management.

Accurately quantifying leachate leakage and predicting its environmental consequences require an integrated modeling framework that couples multiple hydrological, biochemical, and transport processes [7]. Three critical components must be considered [8]. First, leakage through liners and defects must be quantified using physically based seepage and migration models. Second, the temporal evolution of leachate composition, shaped by biochemical degradation, dilution, and infiltration across different landfill stages, must be dynamically simulated. Third, the fate and transport of leaked contaminants in the subsurface environment must be assessed to determine potential risks to groundwater and surrounding ecosystems. Although many studies have investigated these processes individually, a fully coupled approach is essential for evaluating cumulative impacts under realistic landfill operational and post-closure conditions [9].

Of particular importance is the coefficient β_c, characterizing the quality of liner–subgrade contact and strongly governs both the magnitude and stabilization of leakage [10, 11]. Equally critical is understanding the temporal dynamics of leachate constituents, such as ammoniacal nitrogen (NH_3-N) and chloride (Cl^-) [12]. NH_3-N undergoes biogeochemical transformations, making it a representative indicator of reactive transport processes [13], whereas Cl^- behaves as a conservative tracer, providing insights into non-reactive transport. These parameters offer complementary perspectives for evaluating leachate migration and environmental risks [14].

To address these challenges, this study develops an integrated modeling framework that couples Giroud and Bonaparte's empirical leakage equations with a leachate concentration evolution model and a groundwater flow and solute transport model. This approach quantifies leakage rates under varying β_c values, simulates the long-term evolution of NH_3-N and Cl^- concentrations across landfill lifecycle stages, and predicts their migration through subsurface systems. The findings provide a comprehensive assessment of liner performance, leachate behavior, and groundwater contamination potential, thereby offering important insights into sustainable landfill design and management.

2. STUDY AREA AND METHODS

2.1. STUDY AREA

The study site was located in Naidong District, Tibet Autonomous Region, at the Luqiong municipal solid waste (MSW) landfill (Fig. 1). The landfill began operation in

2017, and closure is planned for 2027. The landfill body measures approximately 8 m in height and 380 m in length, consisting primarily of domestic refuse. It is situated on the margin of a slope–alluvial fan deposit within a broad alpine valley, where the terrain is relatively flat with slopes of less than 3°. Surface elevations range from 3557.8 to 3573.0 m, with a maximum local relief of 10.8 m. Two gullies, 10–30 m wide and 1.5–10.5 m deep, are located on the northeastern and southeastern margins of the landfill. Although dry during field investigations, these gullies are capable of conveying stormwater runoff to the Yarlung Tsangpo River, located approximately 1.2 km north of the site at an elevation ca. 45 m lower.

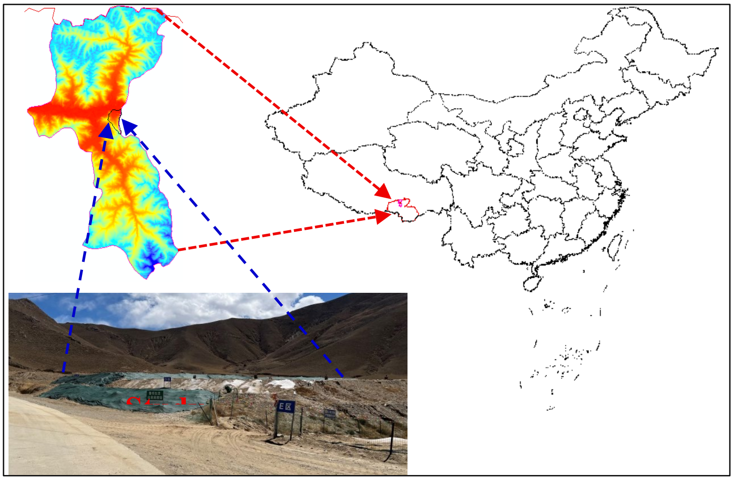

Fig. 1. Study area: the Luqiong municipal solid waste landfill)

The region has a plateau temperate semi-arid continental monsoon climate. The mean annual temperature is 8.7 °C, with extreme values ranging from −18.2 to 30.0 °C. The warmest month is June (6.6–15.8 °C), while January is the coldest (−8.6 to −0.7 °C). The mean annual sunshine duration is 2938 h, reflecting high solar radiation. Annual precipitation averages 396.9 mm, of which more than 80% falls between June and September, typically during nighttime rainfall events. By contrast, monthly precipitation from December to February is generally less than 10 mm. The mean annual potential evaporation is 2695.5 mm, and the mean relative humidity is 43%, indicating pronounced aridity. Seasonal frozen ground is present, with a standard freezing depth of about 0.6 m.

The landfill is underlain by Quaternary Holocene deposits, including eolian fine sand (Q^{4eol}), slope–alluvial silty sand, and silty gravel (Q^{4dl+pl}). These materials are generally loose to slightly compacted, poorly sorted, and highly permeable. Hydraulic conductivity ranges from 5.18 to 10.25 m/d, favoring rapid infiltration and potential solute migration. Groundwater occurs primarily as phreatic water within the Quaternary deposits, recharged by precipitation and snowmelt, and discharges toward the Yarlung Tsangpo valley. No groundwater table was encountered in boreholes during field surveys, suggesting a relatively deep groundwater level. The combination of permeable strata, surface runoff pathways, and proximity to major rivers indicates a high potential for leachate migration, underscoring the importance of predictive modeling.

2.2. METHODS

Leakage rate for HDPE geomembrane in landfills. Leakage through defects in high-density polyethylene (HDPE) geomembranes is a critical concern in the environmental performance of landfill liner systems. Although HDPE geomembranes are widely used due to their low permeability, chemical resistance, and long service life, imperfections such as punctures, tears, seam defects, and stress cracking may occur during installation or operation [15]. These defects can allow leachate to migrate into the underlying layers, potentially contaminating the surrounding soil and groundwater. The magnitude of leakage depends on factors including defect size and shape, the hydraulic head acting on the geomembrane, the nature of the underlying material (e.g., compacted clay or a geosynthetic clay liner), and the presence of cushioning geotextiles [16].

To quantitatively evaluate this phenomenon, Giroud and Bonaparte proposed empirical equations for estimating the leakage rate through localized defects in geomembranes. For a geomembrane in direct contact with a clay liner, the leakage rate can be expressed as follows [17]:

$$q = \beta_{\mathrm{c}} \left(1 + 0.1 \left(\frac{h_w}{L_s} \right)^{0.95} \right) \alpha^{0.1} h_w^{0.9} k_s^{0.74} N \tag{1}$$

where q is the leakage rate of leachate, m/d, α is the area of the defects in the HDPE, m^2, k_s and L_s are the hydraulic conductivity, m/s, and thickness of the compacted clay liner, m, respectively. h_{w} is the leachate depth on the HDPE, m, N is the hole density in the HDPE, 1/ha, β_{c} is the coefficient with values 0.21 (good contact) and 1.15 (poor contact) between HDPE and the compacted clay liner. For good contact, the HDPE geomembrane and the compacted clay liner or subgrade form a tight interface with minimal void space. In contrast, poor contact is characterized by local gaps, wrinkles, and voids between the geomembrane and the underlying clay. Intermediate contact represents conditions that fall between these two extremes.

Leachate concentration over time. The temporal variation of leachate concentration is a key indicator for evaluating the long-term environmental impact of landfill operations. Leachate composition changes over time as a result of complex biochemical, physical, and chemical processes within the waste mass, including organic matter degradation, nitrification–denitrification, mineral dissolution, and adsorption–desorption [18]. In the early acidic phase, leachate typically exhibits high concentrations of organic pollutants and ammoniacal nitrogen, followed by a gradual decline during the methanogenic phase as microbial activity stabilizes and pollutant generation rates decrease [19]. However, the rate and pattern of concentration change are strongly influenced by factors such as waste composition, moisture content, infiltration rates, temperature, and operational practices. Quantifying the temporal evolution of leachate quality provides important insights into landfill stabilization processes. It also supports the design of leachate treatment systems, improves the prediction of pollutant loads, and strengthens the assessment of potential risks to groundwater and surface water.

In this study, the temporal variation of leachate concentration was simulated using the mathematical model previously developed and fully described in our earlier work [20]. This model conceptualizes leachate as a mixture of two representative components, including a readily leachable fraction and an organic-associated fraction, both of which decay exponentially with time as a function of the cumulative liquid-to-solid ratio (*L*/*S*). Importantly, the framework explicitly accounts for the long-term attenuation of source strength after landfill closure, and is therefore applicable to both the active landfilling stage and the closed-landfill stage. The governing equation is expressed as follows [20]:

$$C_t = C_{0,\,\text{leach}} \exp\left(-\frac{C_{0,\,\text{leach}}}{m_{0,\,\text{leach}}} \Delta\left(\frac{L}{S}\right) h_0 t \right) + C_{0,\,\text{org}} \exp\left(-\frac{c_{0,\,\text{org}}}{m_{0,\,\text{org}}} \Delta\left(\frac{L}{S}\right) h_0 t \right) \tag{2}$$

where, C_t is the specific pollutant concentration in leachate at time t, mg/dm^3, $C_{0,\,\text{leach}}$ is the substance concentration readily leachable after the intensive reactor phase, mg/dm^3, $m_{0,\,\text{leach}}$ is the leachable fraction of mobile substances, mg/kg, $C_{0,\,\text{org}}$ is the substance concentration made available for leaching via continuing degradation process directly after the intensive reactor phase, mg/dm^3, $m_{0,\,\text{org}}$ is the fraction of mobile substances still being made available via organic degradation, mg/kg. $\Delta\left(\frac{L}{S}\right)$ is the change in the deposited waste ratio per year, dm^3/(kg· year), h_0 is the highly heterogeneous water flow factor, t is the time after the intensive reactor phase, year.

Groundwater contaminant transport model. A two-dimensional steady-state groundwater flow model was adopted to represent the dominant lateral groundwater flow and contaminant migration at the site scale. This simplification is appropriate because the aquifer exhibits a relatively uniform thickness and the regional hydraulic gradient is primarily horizontal, whereas vertical head variations are comparatively limited. Moreover, the objective of this study is to quantify long-term down-gradient transport trends rather than local-scale vertical heterogeneity. The steady-state groundwater flow field was simulated based on Darcy's law and the groundwater flow continuity equation, and for a saturated, homogeneous, and isotropic aquifer, the governing equation is expressed as follows [21]:

$$W=\frac{\partial}{\partial x}\left(K_x\frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_y\frac{\partial h}{\partial y}\right) \tag{3}$$

where h is the hydraulic head, m, K_x, K_y are the hydraulic conductivities in the x, y directions, m/s, and W is the volumetric flux per unit volume representing sources or sinks, 1/s.

Boundary conditions were specified according to hydrogeological conditions: constant-head boundaries were assigned to the river and recharge zones, while no-flow boundaries were applied along impermeable geological contacts. Hydraulic conductivity values were determined from field pumping tests, and recharge rates were estimated from precipitation and infiltration data. The steady-state flow equation was solved numerically until the residual error met the model convergence criteria.

The transport of pollutants in groundwater was modeled using the advection–dispersion–reaction equation under saturated flow conditions [22]:

$$\frac{\partial(\theta C)}{\partial t}=\nabla(\theta D\nabla C)-\nabla(qC)-\theta\lambda C+S \tag{4}$$

where θ is the porosity, C is the solute concentration, mg/dm^3, D is the hydrodynamic dispersion tensor, m^2/s, q is the Darcy velocity vector, m/s, obtained from the groundwater flow solution, λ is the first-order decay coefficient, 1/s, and S is the external source/sink term, mg/(dm$^3\cdot$s). The dispersion tensor was defined as follows [23]:

$$D_{i,j}=\alpha_T|V|\delta_{i,j}+(\alpha_L-\alpha_T)\frac{v_i v_j}{|V|}+D_m\delta_{i,j} \tag{5}$$

where α_L and α_T are the longitudinal and transverse dispersivities, m, D_m is the molecular diffusion coefficient, m^2/s, v_i *and* v_j are the seepage velocity vectors of longitudinal and transverse directions, m/s, $\delta_{i,j}$ is the Kronecker delta, which ensures that the isotropic components contribute only to the diagonal terms of the dispersion tensor.

Initial conditions for solute concentration were set based on field sampling data, and source term concentrations were defined according to leachate leakage rate calculations from the landfill liner system. The transport equation was solved using a finite-difference scheme, with the groundwater flow field from Section 2.1 serving as input for advective transport.

3. RESULTS AND DISCUSSION

3.1. LEACHATE LEAKAGE THROUGH GEOMEMBRANE DEFECTS

Figure 2 shows the leachate leakage rates under three liner–subgrade contact conditions, represented by β_c values of 0.21, 0.68, and 1.13, over the entire landfill life cycle.

For interpretation, the timeline was divided into three operational stages: landfill stage (1–120 months), covering stage (121–180 months), and completely closed stage (> 180 months).

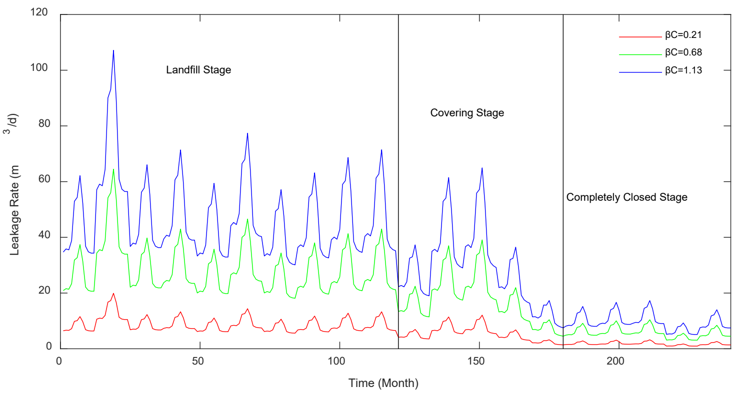

Fig. 2. Leakage rate during different stages of landfill operation under various β_c values

During the landfill stage (1–120 months), all scenarios exhibited a clear upward trend in leakage rates, driven by high leachate generation due to precipitation infiltration, waste decomposition, and elevated hydraulic heads above the liner. Under good contact ($\beta_c = 0.21$), leakage increased gradually from 6.66 m^3/d to around 15–18 m^3/d, reflecting effective attenuation of flow through intimate geomembrane–clay contact. In contrast, the intermediate contact ($\beta_c = 0.68$) and poor-contact ($\beta_c = 1.13$) conditions showed markedly higher initial leakage (21.57 m^3/d and 35.84 m^3/d, respectively) and steeper growth, indicating more rapid bypass of the clay liner through defect zones.

In the covering stage (121–180 months), the placement of final cover systems reduced infiltration rates, resulting in a slowdown in leakage growth for all scenarios. Nevertheless, absolute leakage rates remained strongly dependent on β_c, with poor-contact conditions sustaining more than double the leakage of good-contact conditions at the same time. The reduced growth rates during this stage suggest that the effectiveness of the final cover in limiting infiltration begins to offset the impact of defect enlargement and liner aging.

During the completely closed stage (more than 180 months), leakage rates approached a quasi-steady state. Under $\beta_c = 0.21$, values stabilized at approximately 1.78 m^3/d, while $\beta_c = 0.68$ and $\beta_c = 1.13$ reached around 5.76 m^3/d and 9.58 m^3/d, respectively. The persistence of high leakage in poor-contact conditions highlights that even with minimal infiltration, contact quality remains a decisive factor controlling long-term leakage potential.

Simulation results demonstrate that leakage rates are highly sensitive to both the defect characteristics of the HDPE geomembrane and the contact quality with the underlying subgrade. Under good contact conditions ($\beta_c = 0.21$), leakage rates remained relatively low and stabilized quickly, reflecting effective attenuation by intimate liner–subgrade interaction. In contrast, intermediate ($\beta_c = 0.68$) and poor ($\beta_c = 1.13$) contact conditions resulted in markedly higher leakage rates, with prolonged stabilization times. These findings underscore the importance of achieving close geomembrane–subgrade contact during installation, as imperfections such as wrinkles or voids can significantly increase leachate percolation.

The results align with previous studies [24, 25], which highlight β_c as a critical control parameter. Importantly, even small differences in contact quality produced leakage variations of one order of magnitude, suggesting that field-scale heterogeneity in liner construction could critically affect long-term containment performance.

3.2. THE CONCENTRATION OF LEACHATE LEAKAGE

Figure 3 illustrates the temporal evolution of NH_3-N and Cl^- concentrations in landfill leachate across three operational stages: landfilling stage, covering stage, and completely closed stage.

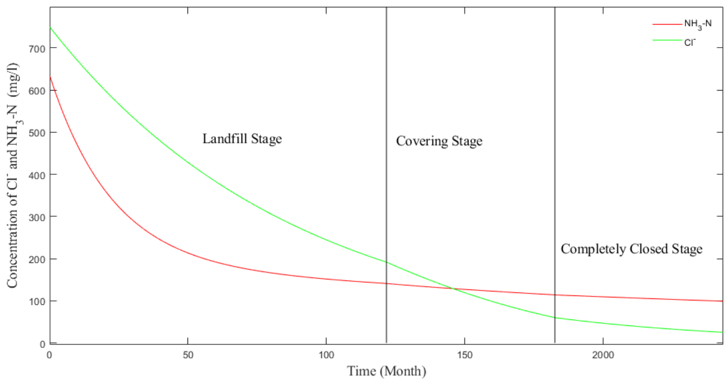

Fig. 3. Concentration of Cl^- and NH_3-N during different landfill operation stages

During the landfilling stage, the concentrations of NH_3-N and Cl^- showed a pronounced overall decreasing trend, reflecting the progressive attenuation of solute release and transport as the landfill system evolved. Specifically, NH_3-N declined from 633.64 mg/dm^3 at month one to 141.89 mg/dm^3 at month 120, corresponding to a total reduction of 491.75 mg/dm^3 and an average attenuation rate of approximately 3.03 mg/(dm^3·month). Similarly, Cl^- decreased from 749.44 mg/dm^3 to 195.34 mg/ dm^3, with a cumulative reduction of 554.10 mg/dm^3 and an average attenuation rate of approximately 4.5 mg/(dm^3·month). Overall, both solutes experienced substantial concentration decay during the landfilling stage, while the attenuation of Cl^- was relatively faster than that of NH_3-N, suggesting stronger dilution and/or transport-controlled reduction for conservative ions in the early landfill development period.

In the covering stage, the concentrations of both NH_3-N and Cl^- exhibited a clear and nearly linear declining trend, indicating that the covering measure effectively reduced the continuous solute input from the source zone to the aquifer. Specifically, NH_3-N decreased from 141.89 mg/dm^3 at month 120 to 115.06 mg/dm^3 at month 180, corresponding to a total reduction of 26.83 mg/dm^3 and an average attenuation rate of approximately 0.444 mg/(dm^3·month). In comparison, Cl^- showed a much larger decline, decreasing from 195.34 mg/dm^3 to 63.80 mg/dm^3 over the same period, with a total decrease of 131.54 mg/dm^3 and an average attenuation rate of approximately 2.214 mg/(dm^3·month). Overall, Cl^- attenuated more rapidly than NH_3-N during the covering stage, suggesting a stronger reduction in conservative solute transport under the imposed covering conditions.

In the completely closed stage, concentrations of both NH_3-N and Cl^- approached near-steady-state levels, with minor fluctuations likely driven by seasonal variations in infiltration and temperature. The persistence of relatively high NH_3-N concentrations indicates that nitrogen release from slowly degrading organic matter continues for decades after closure, posing long-term risks to groundwater. The near-conservative behavior of Cl^- makes it a reliable tracer for assessing leachate migration and for validating transport model predictions in the post-closure phase.

Figure 4a presents the NH_3-N pollution loads under three liner–subgrade contact conditions, represented by β_c values of 0.21 (good contact), 0.68 (intermediate contact), and 1.13 (poor contact), across the landfill lifecycle. The timeline is divided into three operational stages: the landfilling stage (1–120 months), covering stage (121–180 months), and completely closed stage (> 180 months).

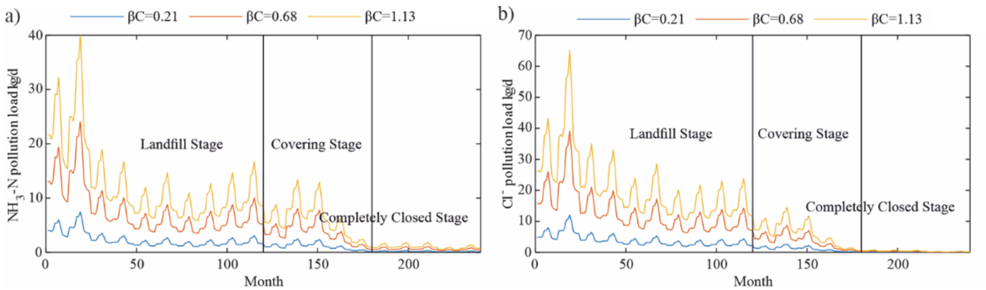

Fig. 4. Pollutant loads of NH_3-N (a) and Cl^- (b)
during various landfill operational stages under different β_c values

Landfilling stage (1–120 months). During active operation, all scenarios showed a consistent decreasing trend in NH_3-N loads, despite sustained high leachate generation driven by precipitation infiltration, waste decomposition, and elevated hydraulic heads above the liner. The stage-average loads were 2.388 kg/day for β_c = 0.21, 7.734 kg/day for β_c = 0.68, and 12.852 kg/day for β_c = 1.13. Relative to the good contact scenario, intermediate contact and poor contact conditions produced mean loads approximately 3.24 and 5.38 times higher, respectively. Linear trend analysis showed slopes of –0.024, –0.078, and –0.129 kg/day per month, with all scenarios exhibiting a similar relative decline (ca. 60.3%) from the beginning to the end of the stage. These results indicate that while pollutant loads naturally decline during active filling, poor contact significantly amplifies the absolute magnitude of emissions.

Covering stage (121–180 months). Following the installation of the final cover system, infiltration rates decreased markedly, leading to a sharper reduction in NH_3-N loads across all scenarios. The mean loads during this stage were 1.089, 3.526, and 5.860 kg/day for $\beta_c = 0.21$, 0.68, and 1.13, respectively. Slopes were –0.021, –0.067, and –0.112 kg/day per month, with relative reductions of approximately 83.8% in each scenario over the stage. Although the decline rates accelerated compared to the waste-filling period, the proportional differences between contact conditions remained stable, with intermediate and poor contact still producing ca. 3.24 and 5.38 times higher loads than good contact. This suggests that while cover systems effectively reduce overall loads, initial liner–subgrade contact quality continues to dominate the magnitude of post-closure pollutant emissions.

Completely closed stage (more than 180 months). In the long-term post-closure stage, NH_3-N loads approached quasi-steady-state levels, with mean values of 0.191, 0.618, and 1.027 kg/d for $\beta_c = 0.21$, 0.68, and 1.13, respectively. Decline rates were minimal (–0.001, –0.005, and –0.008 kg/day per month), and relative changes over the period were approximately –20.0% for all scenarios. Despite low absolute magnitudes, the proportional differences between scenarios persisted, indicating that the effects of linear–subgrade contact quality are long-lived and continue to influence residual NH_3-N emissions decades after closure.

Figure 4b presents the chloride (Cl^-) pollution loads under three liner-subgrade contact conditions, represented by β_c values of 0.21 (good contact), 0.68 (intermediate contact), and 1.13 (poor contact), across the landfill lifecycle.

Landfilling stage (1–120 months). During the active landfilling stage, Cl^- loads were relatively high in all scenarios and showed a consistent decline over time. Stage-average loads were 3.987, 12.910, and 21.453 kg/day for $\beta_c = 0.21$, 0.68, and 1.13, respectively. Relative to the good-contact case, intermediate and poor-contact scenarios produced mean loads approximately 3.24 and 5.38 times higher. Linear trends revealed slopes of –0.037, –0.119, and –0.198 kg/day per month, with a similar relative reduction (ca. 54.6%) observed across all scenarios from the beginning to the end of the stage. The proportional scaling between β_c and Cl^- loads remained consistent, highlighting the importance of liner-subgrade contact quality in controlling contaminant fluxes during high-generation periods.

Covering stage (121–180 months). Following the installation of the final cover system, infiltration rates decreased substantially, resulting in sharper declines in Cl^- loads. Mean values dropped to 1.087, 3.520, and 5.850 kg/day for $\beta_c = 0.21$, 0.68, and 1.13, respectively. The corresponding slopes were –0.031, –0.100, and –0.166 kg/day per month, and the relative reduction over the stage reached approximately 93.6% in all cases, significantly higher than in the waste-filling period. Despite the absolute load reductions, proportional differences between scenarios persisted, with intermediate and poor contact conditions still producing ca. 3.24 and 5.38 times higher loads than the good-contact case. This persistence underscores that initial contact quality exerts a lasting influence even after substantial hydrological improvements from closure measures.

Completely closed stage (>180 months). In the long-term post-closure phase, Cl^- loads reached their lowest levels, with mean values of 0.074, 0.238, and 0.396 kg/day for $\beta_c = 0.21$, 0.68, and 1.13, respectively. Decline rates were minimal (–0.001, –0.004, and –0.007 kg/day per month), and relative changes over the period were approximately –60.8% for all scenarios. Although absolute load magnitudes were small, the proportional differences between scenarios were maintained, indicating that the effects of liner–subgrade contact quality persist for decades after landfill closure.

As the landfill transitioned to the methanogenic phase, both NH_3-N and Cl^- showed gradual declines. NH_3-N decreased due to ammonification stabilization and subsequent biogeochemical transformation, while Cl^- decreased more steadily, primarily due to dilution and flushing rather than degradation. The contrasting behaviors confirm their complementary roles: NH_3-N reflects reactive processes, while Cl^- provides a conservative benchmark for assessing leachate migration.

3.3. CONTAMINANT SOLUTE TRANSPORT

The simulated NH_3-N concentrations in groundwater over 240 months reveal clear spatiotemporal variations as shown in Fig. 5. Within the evaluation range, the overall maximum concentration of NH_3-N reached 16.20 mg/dm^3 on month 120, indicating a mid-term peak after nearly 10 years of simulation (Fig. 5a). Spatially, point B exhibited a relatively low maximum value of 2.69 mg/dm^3 (month 8.9), while point A experienced a significantly higher peak of 16.20 mg/dm^3 (month 120). These results demonstrate that hydrogeological conditions strongly influence pollutant accumulation, with point A showing a much higher vulnerability to NH_3-N contamination compared to point B.

The temporal evolution of NH_3-N concentration also indicates distinct stages. In the early phase (< 1000 days), all monitoring values remained very low (< 1.0 mg/dm^3), suggesting that initial pollutant migration was strongly retarded by adsorption and dispersion processes. Point B reached its maximum much earlier than point A, implying a more rapid but limited breakthrough due to its hydrogeological setting, likely characterized by preferential pathways with smaller storage capacity. In contrast, point B showed a delayed but much higher accumulation, reflecting a gradual buildup of NH_3-N in downstream zones with larger retention or reduced attenuation capacity.

From a long-term perspective, the difference between overall maximum (shown in Fig. 5a 16.20 mg/dm^3) and the maxima at points A and B highlights that local peak concentrations may either underestimate or overestimate the risk if evaluated in isolation. This indicates the necessity of integrating both global maximum values and site-specific monitoring to capture the full spectrum of groundwater contamination risks. Moreover, the fact that the mean NH_3-N concentration across the evaluation period remained relatively moderate (8.89 mg/dm^3) suggests that occasional extreme peaks, rather than long-term averages, govern the environmental and health risk potential.

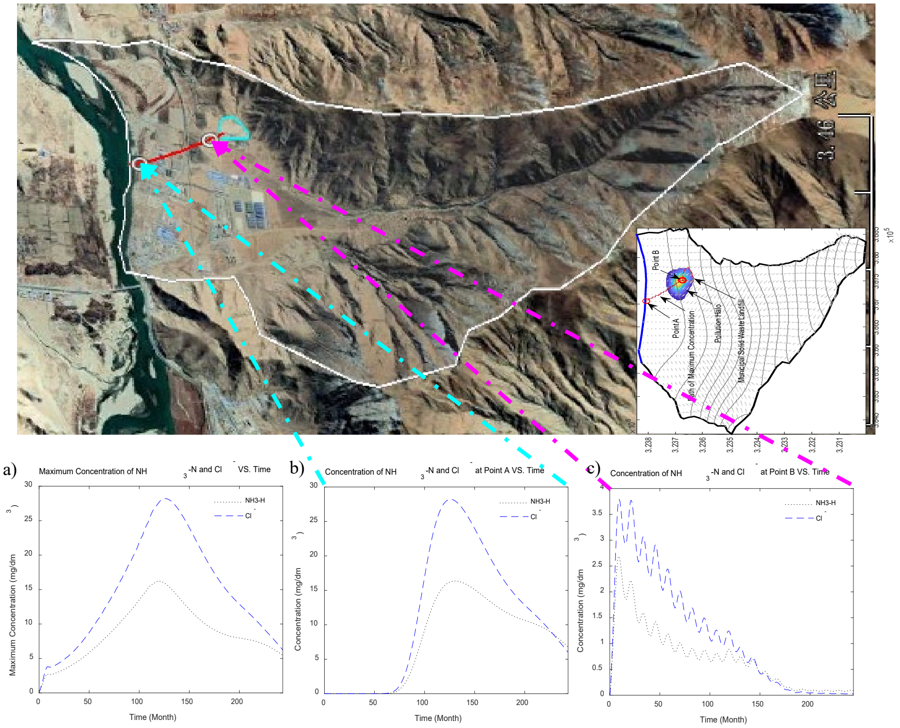

Fig. 5. Concentrations of Cl$^-$ and NH_3-N (β_c = 0.21) (more details provided in the text)

The simulated chloride (Cl$^-$) concentrations in groundwater exhibited significant temporal and spatial variability over the 240 month evaluation period. The overall maximum concentration within the evaluation domain reached 28.21 mg/dm^3 on 126 month, nearly at the mid-point of the simulation, indicating a pronounced long-term accumulation effect. At point B, the maximum concentration was only 3.79 mg/dm^3 (month 9.3), while point A recorded a peak identical to the global maximum (28.21 mg/dm^3, month 126). This disparity underscores the spatial heterogeneity of solute migration and suggests that point B is located in a more vulnerable hydrogeological zone with higher solute accumulation potential.

The temporal evolution reveals distinct migration and breakthrough characteristics. Point B reached its peak concentration early (within the first year), but the value remained relatively low, reflecting limited accumulation and possibly stronger dilution or attenuation capacity. In contrast, point A showed a delayed but much higher breakthrough, with

Cl^- concentration gradually building up over decades before reaching the overall maximum. This pattern indicates that downstream or low-permeability zones tend to accumulate chloride over extended periods, posing a higher long-term risk compared to upstream sites with quicker but weaker responses.

Statistical analysis also highlights the difference between averages and extremes. While the mean chloride concentration across the evaluation domain was 14.83 mg/dm^3, close to half of the maximum, the standard deviation (7.73 mg/dm^3) and extreme peaks imply substantial temporal fluctuations. Such results suggest that risk evaluation based solely on average concentrations may underestimate the severity of contamination events. Therefore, considering both peak values and temporal dynamics is essential for accurately assessing long-term groundwater salinization risks under pollutant loading scenarios.

3.4. UNCERTAINTY ANALYSIS

Despite the integrated modeling framework developed in this study, several sources of uncertainty may influence the accuracy of the simulated leachate leakage, concentration evolution, and groundwater contaminant transport. These uncertainties arise from model parameterization, conceptual assumptions, and hydrogeological heterogeneity, and should be considered when interpreting the results.

Uncertainty in leachate concentration evolution parameters (C_0 and m_0). The parameters governing the temporal evolution of leachate quality, namely the initial leachable concentration C_0 and the leachable mass fraction m_0, are subject to considerable variability due to differences in waste composition, moisture content, degradation pathways, and local climatic conditions. Although literature-based values were adopted, actual landfill behavior may deviate from assumed parameter ranges, particularly during the transition from acidogenic to methanogenic phases. These uncertainties can influence both the timing and magnitude of NH_3-N and Cl^- release. Sensitivity analyses in previous studies [8] have shown that small changes in C_0 and m_0 can result in substantial variations in predicted pollutant loads, suggesting that site-specific monitoring data would further enhance model reliability.

Hydrogeological heterogeneity and its impact on groundwater flow. The groundwater flow model assumes representative, spatially averaged hydraulic conductivity values for the Quaternary aquifer. However, field investigations indicate that the subsurface materials, ranging from eolian sand to silty gravel, exhibit significant vertical and lateral variability. Such heterogeneity can create preferential flow pathways or low-permeability zones, influencing plume migration speed and direction. The contrast in breakthrough behavior between points A and B highlights this uncertainty: small-scale heterogeneity not captured in the model may amplify or attenuate solute transport in ways not fully reflected in the numerical simulations. Incorporating stochastic or geostatistical approaches could further quantify these uncertainties.

Uncertainty in liner–subgrade contact quality (β_c). Although this study evaluated a range of β_c values representing good, intermediate, and poor contact, real-world field conditions often vary spatially across the liner surface. Local wrinkles, voids, and subgrade imperfections can produce β_c distributions rather than discrete values. As leakage responds nonlinearly to contact quality, even small deviations in β_c may cause disproportionately large variations in leakage rates. Hence, the modeled leakage scenarios represent idealized endmembers rather than spatially continuous field conditions.

Simplifying assumptions in transport processes. The solute transport model employs a first-order decay coefficient for NH_3-N and assumes conservative behavior for Cl^-. In reality, reactions such as sorption, ammonification, volatilization, and microbial transformations may exhibit non-linear or time-dependent behavior. Similarly, dispersivity values may increase with travel distance, an effect not explicitly captured in the present model. While the adopted assumptions follow standard practice, they introduce uncertainty in predicting long-term migration patterns.

Overall, while the integrated model provides a robust framework for evaluating landfill leachate impacts, the uncertainties associated with parameter variability, hydrogeological heterogeneity, and simplifying assumptions may influence quantitative predictions. These uncertainties do not undermine the qualitative trends identified but highlight the need for site-specific calibration, long-term monitoring, and further refinement using stochastic or probabilistic approaches in future studies.

3.5. REAL-WORLD IMPLICATIONS FOR GROUNDWATER RISK MANAGEMENT AND ANTI-SEEPAGE CONSTRUCTION

The simulation results demonstrate that even small variations in liner–subgrade contact quality can lead to several-fold increases in leakage and pollutant loads. These differences translate into pronounced spatial heterogeneity in groundwater impacts, as evidenced by the contrasting breakthrough concentrations at points A and B. This underscores that groundwater contamination from landfills is inherently non-uniform and is strongly influenced by localized hydrogeological conditions and long-term patterns of pollutant accumulation.

In practical landfill operations, this means that groundwater risk assessments cannot rely solely on average concentrations or isolated monitoring wells. Monitoring networks should be designed to capture downstream plume migration pathways, long-term accumulation of conservative ions (e.g., Cl^-), and delayed release of reactive nitrogen species (NH_3-N). The modeled persistence of contaminants over several decades further highlights the need for extended post-closure monitoring, particularly in areas underlain by deep aquifers or hydraulically connected alluvial systems. Regulators should therefore adopt risk-based monitoring durations rather than fixed-year requirements, ensuring that monitoring continues until pollutant fluxes stabilize at environmentally acceptable levels.

Moreover, the results indicate that peak concentrations, rather than statistical averages, are more likely to govern ecological and human health risks. This finding has direct implications for groundwater standards, suggesting that assessments should consider worst-case plume scenarios, utilize conservative tracers as early-warning indicators, and incorporate predictive modeling during closure planning to better anticipate long-term risk trajectories.

The strong dependency of leakage on β_c indicates that achieving high-quality contact between the geomembrane and the subgrade is among the most effective measures for reducing long-term groundwater risks. Poor contact conditions, resulting from inadequate subgrade preparation, geomembrane wrinkles, or improper installation, can increase leakage rates by more than fivefold. Therefore, anti-seepage performance must be evaluated not only through material specifications but also through construction quality and field verification.

Engineering standards should be strengthened to require proper subgrade compaction, controlled geomembrane deployment, wrinkle minimization, and rigorous quality assurance/quality control protocols during installation. Field-scale verification of contact quality, such as electrical leak detection, dielectric integrity surveys, or vacuum box testing, should be mandated, as visual inspection alone is insufficient.

Because β_c may vary spatially across a landfill base, design guidelines should incorporate sensitivity analyses and adopt conservative β_c values or composite liner systems (e.g., geomembrane + geosynthetic clay liner in hydrogeologically vulnerable zones. For older or legacy landfills, retrofit strategies such as stormwater diversion, leachate recirculation control, and installation of engineered covers can significantly reduce hydraulic loading and thereby limit long-term leakage.

At the policy level, permitting frameworks should integrate model-based leakage predictions and adopt performance-based criteria that explicitly consider contact quality. Regulations should emphasize installation integrity and post-construction certification rather than only mandating material properties. Long-term management policies should encourage periodic liner performance assessments, promote the use of rapid defect detection technologies, and require closure plans to include provisions for extended groundwater monitoring and timely remediation if contaminant plumes are detected.

4. CONCLUSION

This study developed an integrated modeling framework that couples geomembrane leakage estimation, leachate concentration evolution, and groundwater contaminant transport to assess landfill leachate impacts under varying liner–subgrade contact conditions.

The results indicate that leakage magnitude and stabilization are highly dependent on contact quality (β_c), with poor contact producing leakage and pollutant loads more than five times higher than those under good contact. Closure measures effectively reduced infiltration and loads, yet the influence of β_c persisted well into the post-closure period.

The simulations demonstrate that both ammoniacal nitrogen (NH_3-N) and chloride (Cl^-) exhibit gradual concentration declines from the active landfilling phase through the post-closure period, yet can persist at non-negligible levels for decades after closure. Although NH_3-N undergoes biochemical attenuation, its incomplete removal results in sustained long-term residual risks. In contrast, Cl^- behaves as a conservative tracer and therefore provides a robust indicator of persistent solute storage, cumulative accumulation, and delayed groundwater breakthrough. Overall, the results highlight that the long-term effectiveness of landfill containment and the magnitude of groundwater impacts are strongly controlled by local hydrogeological conditions, including aquifer permeability, hydraulic gradients, and preferential flow pathways.

Overall, the findings underscore three key points: (i) liner–subgrade contact quality exerts a decisive and lasting influence on leachate leakage and contaminant transport, (ii) both peak and mean concentrations must be considered in groundwater risk assessments, as extreme values govern long-term risks, and (iii) extended post-closure monitoring is essential due to delayed contaminant migration. From an engineering perspective, strict quality control to ensure good liner–subgrade contact during construction remains the most effective strategy for minimizing long-term groundwater contamination risks from landfill leachate.

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