Risk Assessment for Sustainable Waste Disposal to Landfills

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1 Introduction

In his monumental *Naturalis Historia*, published in AD 77, Pliny the Elder was already aware of the threat posed by human beings to Nature when they are guided by evil intent, such as the desire to defeat an enemy: "*As for ourselves, we envenom the point of the arrow, and we contrive to add to the destructive powers of iron itself; by the aid of poisons, we taint the waters of the stream, and we infect the various elements of Nature; indeed, the very air even, which is the main support of life, we turn into a medium for the destruction of life.*" (XVIII, 2–3).

The subjugation of Nature to human yearnings must nowadays be confronted with the need to avoid the disastrous social and economic consequences that are determined by the depletion of natural resources and the pollution of the environment.

In this context, the concept of sustainability has emerged. In the beginning, it has been regarded as the capacity of human habits and actions to be practised or maintained indefinitely, as suggested by the etymological derivation from the Latin word *sustinere*, which means "hold up, hold upright" and also "bear, undergo, endure". In this initial usage, the concept of sustainability was not characterised by any ethical implications and was strictly related to the concern for maintaining the availability of desirable materials and conditions over the long term. As a result, bad or questionable practices could be considered sustainable, such as slavery and prostitution [2].

However, when applied to human actions impacting the environment, the concept of sustainability has assumed a new and deeper meaning. In particular, the conferment of an ethical value has connoted this change. Sustainable practices have been identified through their capacity to provide for the needs of future generations as well as the present. This "paradigm shift" is represented by the definition of "sustainable

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development" provided by the UN Commission on Environment and Development (Brundtland Commission) in its Report published in 1987. The Report, titled Our Common Future, defined sustainable development as the development that "*meets the needs of the present without compromising the ability of future generations to meet their own needs*". The responsibility to future generations is the fundamental duty of environmental ethics and places its roots in the categorical imperative that was formulated by the German-American philosopher Hans Jonas: "*Act so that the effects of your action are compatible with the permanence of genuine human life*" or expressed negatively: "*Act so that the effects of your action are not destructive of the future possibility of such life*"; or simply: "*Do not compromise the conditions for an indefinite continuation of humanity on earth*"; or again turned positive: "*In your present choices, include the future wholeness of Man among the object of your will*". [13].

Applying the concept of sustainability to waste landfilling requires taking into account several issues, among which we can mention the migration of contaminants from the waste leachate to the groundwater in the aquifer beneath the landfill and the emission of greenhouse gases for the construction of the lining system.

As a result, we need to balance two conflicting needs. On the one hand, adequate groundwater protection must be guaranteed; on the other hand, the impact on the environment related to construction phases must be reduced as much as possible.

The typical approach to face the need to protect the groundwater is to increase the thickness and reduce the hydraulic conductivity of compacted clay liners. However, this approach implies high economic costs, significant consumption of natural resources (such as earth materials), and an elevated emission of greenhouse gases.

Conversely, the need to minimise the environmental impact related to barrier construction requires limiting the thickness of compacted clay liners and relying on natural attenuation processes. Unfortunately, these processes are highly uncertain and often, a suitable level of reliability in assessing lining system performance cannot be reached.

The way to guarantee the protection of the groundwater resource, avoiding recourse to oversized lining systems, is the adoption of a performance-based design approach that relies on the assessment of the risk to human health and the environment due to pollutant migration through the barrier system. In such a way, the modelling of contaminant transport that takes place through mineral and geosynthetic liners allows for a rational design of the barrier system components and the quantification of the expected performance of the whole lining system, avoiding the construction of oversized barriers on the basis of the so-called precautionary principle, which is typically invoked when a theoretical analysis is not carried out.

The responsibility to future generations, implied in the concept of sustainability, requires considering the long-term landfill post-closure conditions when the lining system's geosynthetic components have lost their functionality. Such a new perspective makes many traditional design methods inapplicable. For instance, transit-time or breakthrough design methods, which require a threshold value of the contaminant concentration to be reached at the exit of the lining system only after the end of the post-closure monitoring period (usually 30 years long), cannot be adopted, as they predict an unacceptable condition (i.e., a contaminant concentration higher than the threshold value) in the long term, without posing the problem of how the future generations will manage that theoretical threat for the groundwater quality [4, 16].

The design method should demonstrate that the risk related to contaminant migration remains less than an acceptable threshold level at a compliance point (typically represented by a well or a piezometer open in the aquifer beneath the landfill) not only during the prescribed post-closure monitoring period but also in the long term after the end of that period.

Recently, a risk assessment procedure that may be used within a design method coherent with the abovementioned purposes has been introduced in the Italian regulations for landfills. This procedure will be described and compared with other more advanced modelling approaches in the following sections.

2 Risk Assessment for Landfills in Italian Regulations

In 2020, a subsequent regulatory provision, the D.Lgs. 121/2020, has introduced in the Italian regulation for landfills, the so-called D.Lgs. 36/2003, the requirement to carry out a risk assessment analysis in the following cases:

- When local authorities must authorise particular subcategories of landfills for non-hazardous waste, such as landfills for inorganic waste with low organic or biodegradable content, landfills for primarily organic waste (to be divided into landfills considered bioreactors with biogas recovery and landfills for pre-treated organic waste) and landfills for mixed non-hazardous waste with a high content of both organic or biodegradable waste and inorganic waste, with biogas recovery.
- When requesting a derogation from the permissible concentration limit values in waste leachate, indicated in Column A of Table 1 for landfills of non-hazardous waste. This exemption cannot exceed the limit value by more than double.
- At the end of the post-closure monitoring period (whose duration is typically assumed to be 30 years after the end of the waste disposal activities).

The risk assessment consists of comparing the expected contaminant concentration in the aquifer beneath the landfill to the screening concentration shown in Column B of Table 1, which has been identified as a protective value for human health and the environment in the Italian regulation based on the “precautionary principle”, regardless of the determination of the risk related to actual exposure to the groundwater.

When a backward analysis is performed, the concentration in the groundwater is imposed to be equal to the screening value in Column B of Table 1, C_{lim}, and the concentration in the leachate, C_{l}, is calculated as follows:

$$C_{\text{l}} = \frac{C_{\text{lim}} \cdot \text{LDF}}{\text{SAM}} \tag{1}$$

Table 1 Waste contaminants in the Italian regulation for non-hazardous waste

Contaminant	Symbol	Column A Admissible concentration for non-hazardous waste, C_{adm} (mg/l)	Column B Screening concentration in the groundwater, C_{lim} (mg/l)
Arsenic	As	0.2	0.01
Barium	Ba	10	0.1
Cadmium	Cd	0.1	0.005
Total chromium	Total Cr	1	0.05
Copper	Cu	5	1
Mercury	Hg	0.02	0.001
Molybdenum	Mo	1	0.05
Nickel	Ni	1	0.02
Lead	Pb	1	0.01
Antimony	Sb	0.07	0.005
Selenium	Se	0.05	0.01
Zinc	Zn	5	3
Chlorides		2500	250
Fluorides		15	1.5
Sulfates		5000	250
Dissolved organic carbon	DOC	100	10
Total dissolved solids	TDS	10,000	500

where

$$\mathrm{SAM} = \frac{d_{\mathrm{d}}}{L_{\mathrm{GW}}} \tag{2}$$

$$\mathrm{LDF} = 1 + \frac{q_{\mathrm{GW}} \cdot \delta_{\mathrm{GW}}}{q \cdot \ell}. \tag{3}$$

In Eqs. (1)–(3), SAM is the Soil Attenuation Model parameter (≤ 1), LDF is the Leachate Dilution Factor (≥ 1), d_{d} is the depth, from the ground level, of the first liner, L_{GW} is the depth of the groundwater (i.e., the top of the aquifer for a confined aquifer or the water table for an unconfined aquifer), q_{GW} is the horizontal volumetric flux of the groundwater, δ_{GW} is the thickness of contaminant plume in the aquifer ($\delta_{\mathrm{GW}} \leq h$, where h is the thickness of the aquifer), q is the vertical volumetric flux from the landfill and ℓ is the length of the landfill in the direction of the groundwater flow. The geometrical parameters are shown in Fig. 1.

The thickness of the contaminant plume may be estimated using the following equation [3]:

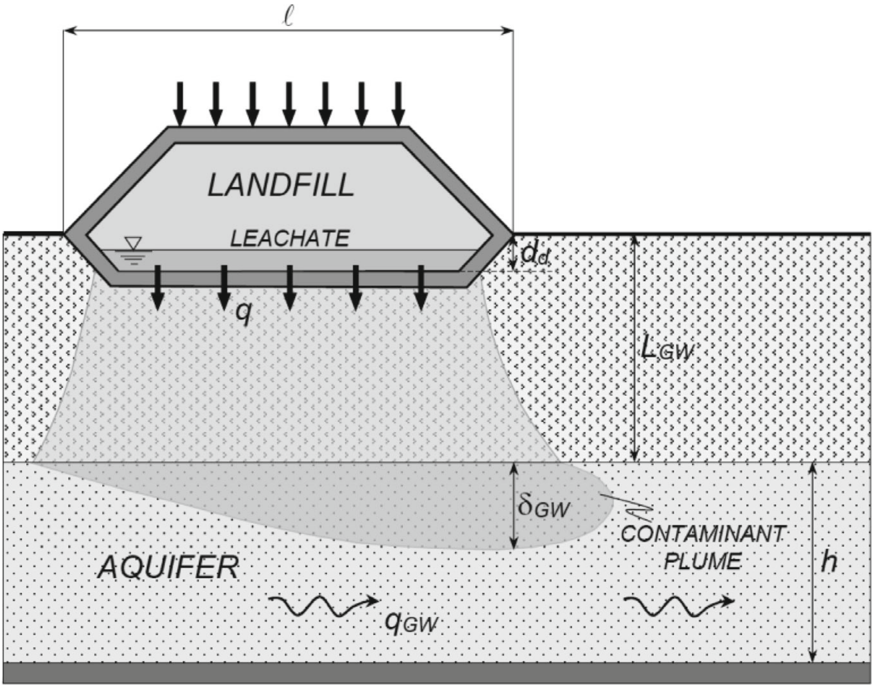

Fig. 1 Reference scheme of the landfill and the underlying aquifer scenario, which is considered in the Italian regulations for landfills

$$\delta_{\mathrm{GW}} = \sqrt{2 \cdot \alpha_{\mathrm{T}} \cdot \ell} + h\left[1 - \exp\left(-\frac{q \cdot \ell}{q_{\mathrm{GW}} \cdot h}\right)\right]. \tag{4}$$

where α_{T} is the transverse dispersivity within the aquifer. Based on the indications of ISPRA [10], $\alpha_{\mathrm{T}} = 0.005 \cdot \ell$.

The Italian regulations recommend to calculate the vertical infiltration, q, without considering the geomembrane and assuming that all the mineral layers are saturated. As a result, q can be determined using as follows:

$$q = k_{\mathrm{eq}} \frac{h_{\mathrm{p}} + L - h_{\mathrm{b}}}{L}. \tag{5}$$

where k_{eq} is the equivalent hydraulic conductivity, h_{p} is the height of the ponded leachate in the drainage layer, L is the total thickness of the artificial liners and the underlying attenuation layers that separate the landfill from the aquifer, and h_{b} is the height of the water level at the bottom of the barrier (Fig. 2).

The equivalent hydraulic conductivity, k_{eq}, in Eq. (5) is calculated as the harmonic mean of the hydraulic conductivities of individual layers:

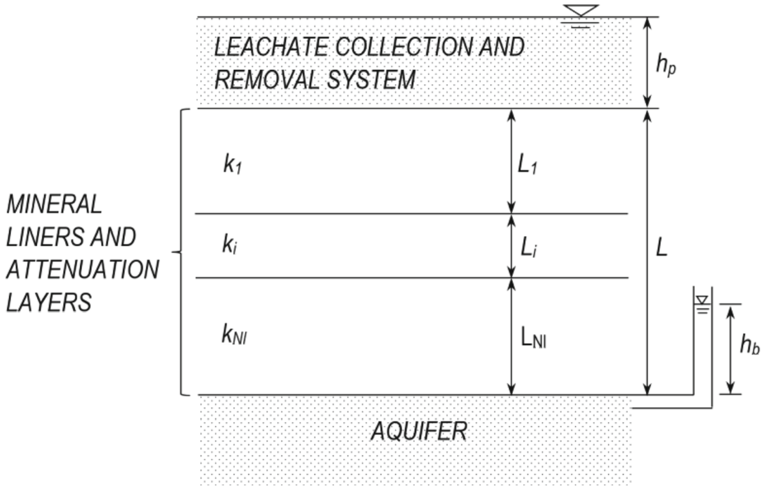

Fig. 2 Vertical profile of a landfill barrier system constituted by engineered and/or natural mineral layers

$$k_{\mathrm{eq}} = \frac{L}{\sum_{i=1}^{N_l} \frac{L_i}{k_i}}. \tag{6}$$

where L_i is the thickness of the ith layer, k_i is the hydraulic conductivity of the ith layer, and N_l is the number of mineral layers, including the natural foundation layers (also called geological barrier) that are placed between the lining system and the underlying aquifer.

The risk assessment requires calculating the acceptable contaminant concentration in the leachate waste through Eq. (1) as a function of the groundwater screening concentration and the attenuation factor, AF (≥ 1), given by the ratio between the LDF and the SAM parameter. This contaminant concentration, C_l, must be compared with the admissible concentration for the landfill given by the regulation in force (see Column A of Table 1). If C_l is lower than C_{adm}, C_l must be taken as the maximum acceptable value for the contaminant in the waste leachate. If C_l is larger than C_{adm}, then the concentration limit can be derogated, but for not more than $2 \times C_{\mathrm{adm}}$.

Although considering the geomembrane is not recommended, ISPRA [11] has suggested to calculate the leakage rate through the defect of a composite barrier using the following empirical equation derived from Giroud [9]:

$$Q = C_{\mathrm{q}} \cdot A_d^{0.1} \cdot h_p^{0.9} \cdot k_{\mathrm{eq}}^{0.74} \cdot \left[1 + 0.1\left(\frac{h_{\mathrm{p}}}{L}\right)^{0.95}\right]. \tag{7}$$

where C_{q} is a dimensionless quality coefficient of the contact between the geomembrane and the underlying mineral layer, which can be assumed equal to 0.21 for good

Table 2 Distribution of geomembrane defect features [11]

Defect type	Geomembrane defects							Area of geomembrane defects		
	Probability distribution	Frequency of defects with QC* n (number/ha)			Frequency of defects without QC* n (number/ha)			Probability distribution	Area of defects, A_d (m^2)	
Micro-holes	Triangular	0	25	25	0	750	750	Log uniform	1×10^{-8}	5×10^{-6}
Holes	Triangular	0	5	5	0	150	150	Log uniform	5×10^{-6}	1×10^{-4}
Tears	Triangular	0	0.1	2	0	0.5	10	Log uniform	1×10^{-4}	1×10^{-2}

* *QC*—quality control

contact conditions and 1.15 for poor contact conditions, and A_d is the geomembrane defect area.

The volumetric flux of water, q_d, passing through several defects of different sizes and shapes, including micro-holes, holes, and tears, can be expressed as follows:

$$q_d = n_{\text{micro - holes}} Q_{\text{micro - holes}} + n_{\text{holes}} Q_{\text{holes}} + n_{\text{tears}} Q_{\text{tears}}. \tag{8}$$

where n_j is the number of defects of the jth type per unit area (i.e., the frequency of the jth defect type), and j corresponds to micro-holes, holes, or tears, respectively. The leakage rates, Q_j, are calculated through Eq. (7). The defect frequencies and areas are obtained from Table 2.

When the geomembrane is taken into account, the volumetric flux, q, in Eqs. (3) and (4) is replaced by q_d.

3 Alternative Methods for the Landfill Risk Assessment

The following theoretical limitations characterise the theoretical approach proposed in Italian regulations (see Eq. 1).

(1) Using the *SAM* parameter is questionable as it is based on the assumption of a finite contaminant mass that is uniformly distributed between the volume given by the waste, the lining system, and the underlying natural foundation soil. However, this assumption is not coherent with the steady-state condition that can be reached by any transport model. Moreover, the multiphase partition phenomena differ significantly among the waste, the lining systems, and the natural layers separating the landfill from the aquifer, but the simple thickness proportionality expressed by Eq. 2 does not consider that change.

(2) The LDF is obtained from a contaminant mass balance within the whole aquifer volume beneath the landfill. In this way, the distribution of the contaminant concentration in the horizontal and vertical directions inside the aquifer cannot be appreciated.
(3) The volumetric flux q can be overestimated, as the unsaturated conditions that may occur in the mineral layers placed between the waste and the aquifer are not taken into account.
(4) The diffusive transport of contaminants through the artificial liner and the geological barrier is not considered, although it can be the dominant mechanism of migration for volatile organic compounds (VOCs) that are able to diffuse through polymeric geomembranes.
(5) The change in the horizontal groundwater flux, q_{GW}, beneath the landfill due to the vertical infiltration, q, is not considered.
(6) The thickness of the contaminant plume is improperly determined considering the penetration in the aquifer of the contaminant due to vertical advection. To understand this limitation, consider the case of pure advective transport in the aquifer (i.e., when $\alpha_T = 0$): the concentration inside the plume, given by Eq. 1, should be C_{lim}/SAM, which corresponds to a value of LDF = 1, while LDF is larger than 1 whenever $\delta_{GW} > 0$.

The Italian regulations allow other theoretical models to be adopted, provided that they are recognised and validated internationally. In this context, the modelling approach developed by Dominijanni and Manassero [6] and Dominijanni [7] can be considered as a possible alternative. The main hypotheses involved in such an approach are the following:

(1) The contaminant mass in the waste is infinite and the related source contaminant concentration, C_0, is constant in time.
(2) The analysis is conducted under steady-state conditions.
(3) The processes of radioactive decay and biodegradation are conservatively neglected.
(4) The vertical contaminant migration through the artificial liners and the underlying geological barrier is one-dimensional.
(5) The vertical water volumetric flux under unsaturated conditions is calculated assuming that the hydraulic conductivity, k_{uns}, is related to the suction height, ψ, through an exponential function, such as $k_{uns} = k \cdot \exp(-\alpha \cdot \psi)$, where k is the hydraulic conductivity under saturated conditions and α is the capillarity coefficient (α tends to 0 when the capillary rise tends to infinity).
(6) The only attenuation mechanisms that are taken into account are the dilution in the groundwater and the dispersion in the orthogonal direction to the groundwater flow.
(7) The horizontal and vertical components of the groundwater flux in the aquifer are obtained using the Dupuit-Forchheimer approximation, which assumes vertical equipotential lines.

If the thickness of the aquifer, h, is no more than a few metres (i.e., the aquifer may be considered "thin"), the following analytical solution can be derived [7]:

$$\mathrm{RC} = 1 - \left(\frac{\eta}{\eta + X}\right)^{\kappa} \tag{9}$$

where

$$\mathrm{RC} = \frac{C_x - C_{x0}}{C_0 - C_{x0}} \tag{10}$$

$$X = \frac{x}{\ell} \tag{11}$$

$$\eta = \frac{q_{\mathrm{GW0}} h}{a_{\mathrm{d}} q \ell} \tag{12}$$

$$\kappa = \frac{e^{P_{\mathrm{L}}}}{e^{P_{\mathrm{L}}} - 1} + \frac{(1 - a_{\mathrm{d}})\Lambda_{\mathrm{d}}}{a_{\mathrm{d}} q} \tag{13}$$

$$P_{\mathrm{L}} = \frac{q}{\Lambda} \tag{14}$$

$$\Lambda = \frac{1}{\int_0^L \frac{\mathrm{d}z}{\vartheta_{e,w} \cdot D_{\mathrm{h}}}} \tag{15}$$

$$\Lambda_d = \frac{1}{\frac{L_g}{K_g D_g} + \frac{1}{\Lambda}}. \tag{16}$$

In Eqs. (10)–(16), C_x is the contaminant concentration in the aquifer at the horizontal distance x beneath the landfill, C_{x0} is the contaminant concentration in the groundwater upstream from the landfill, q_{GW0} is the horizontal groundwater volumetric flux upstream from the landfill, P_{L} is the Peclet number of the artificial liners and the geological barrier, a_d is the fraction of the landfill area where leakage through geomembrane defects occurs, Λ is the equivalent diffusivity of the artificial liners and the geological barrier, $\vartheta_{e,w}$ is the volumetric content accessible to mobile water, D_{h} is the contaminant hydrodynamic dispersion coefficient, z is the vertical distance from the top of the liner, L_{g} is the thickness of the geomembrane, K_{g} is the partition coefficient between the geomembrane and the contaminant, and D_{g} is the contaminant diffusion coefficient through the geomembrane.

The fraction of the landfill area where leakage through geomembrane defects occurs, a_{d}, is defined as the ratio between the leakage flux passing through the geomembrane defects, q_{d}, given by Eq. (8) or the theoretical equations for defects with perfect or imperfect contact conditions reported in Dominijanni [7], and the volumetric flux, q, passing through the underlying mineral components of the artificial and the geological barriers (see Eq. 5).

The following limit conditions can be met:

(1) $a_\mathrm{d} = 0$, when the geomembrane is perfectly intact (without defects). In such case, Eq. (9) reduces to:

$$\mathrm{RC} = 1 - \exp\left(-\frac{\Lambda_\mathrm{d}\ell}{q_{\mathrm{GW0}}h}X\right). \tag{17}$$

(2) $a_\mathrm{d} = 1$, when the geomembrane is assumed to be completely degraded. In such case, if $P_\mathrm{L} > 4$, κ tends to 1 and Eq. (9) becomes:

$$\mathrm{RC} = 1 - \left(\frac{\eta}{\eta + X}\right). \tag{18}$$

When the aquifer cannot be considered "thin", the contaminant concentration distribution is obtained from the solution of the following equation [6]:

$$q_x\frac{\partial C}{\partial x} = \alpha_\mathrm{T} q_{\mathrm{GW0}}\frac{\partial^2 C}{\partial y^2} - q_y\frac{\partial C}{\partial y} \tag{19}$$

where

$$q_x = q_{\mathrm{GW0}} + \frac{a_\mathrm{d} q}{h}x \tag{20}$$

$$q_y = a_\mathrm{d} q\left(1 - \frac{y}{h}\right) \tag{21}$$

In Eq. (19), x is the horizontal distance beneath the landfill, y is the vertical distance from the top of the aquifer, q_x and q_y are the horizontal and vertical components of the groundwater volumetric flux, respectively.

As shown in Dominijanni and Manassero [6] and Dominijanni [7], Eq. (19) can be solved using the finite-difference method for both confined and unconfined aquifers. In the case of a semi-infinite (h tends to infinity) confined aquifer with $a_\mathrm{d} q/q_{\mathrm{GW0}} < 0.01$, the following analytical solution can be obtained [7]:

$$\mathrm{RC} = \mathrm{erfc}\left(\frac{Y}{2\sqrt{X}}\right) - \exp\left(\Gamma Y + \Gamma^2 X\right)\cdot \mathrm{erfc}\left(\frac{Y}{2\sqrt{X}} + \Gamma\sqrt{X}\right) \tag{22}$$

$$Y = \frac{y}{\sqrt{\alpha_\mathrm{T}\ell}} \tag{23}$$

$$\Gamma = \frac{\sqrt{\alpha_\mathrm{T}\ell}}{\alpha_\mathrm{T} q_{x0}}\left[a_\mathrm{d} q\frac{e^{P_\mathrm{L}}}{e^{P_\mathrm{L}} - 1} + (1 - a_\mathrm{d})\Lambda_\mathrm{d}\right] \tag{24}$$

The maximum concentration in the aquifer is found at $x = \ell$ for thin aquifers and at $x = \ell$, $y = 0$ for thick aquifers. In any case, a maximum local relative concentration or a suitable maximum averaged relative concentration can be found and indicated as RC_{max}. In the framework of a backward risk assessment analysis, Eq. (1) can be replaced by the following equation:

$$C_l - C_{x0} = \frac{C_{lim} - C_{x0}}{RC_{max}} \tag{25}$$

where C_{x0} represents the contaminant concentration upstream from the landfill.

4 Application Example

A numerical example is presented to show the application of a risk assessment analysis from the perspective of sustainable waste disposal in landfills. In this regard, the risk assessment will consider two scenarios:

(1) The monitoring post-closure period, which is typically assumed to end after thirty years from the termination of the waste disposal activities.
(2) A long-term condition in which the geomembrane has lost its functionality.

The artificial lining system, in accordance with Italian regulations for landfills of non-hazardous waste, includes (from the top):

- A 0.5 m-thick drainage layer;
- A 2.5 mm-thick geomembrane in HDPE;
- A 1 m-thick compacted clay liner that represents the artificial mineral barrier ($k \leq 1 \times 10^{-9}$ m/s);
- A 1 m-thick compacted clay liner that represents the artificial completion of the geological barrier ($k \leq 1 \times 10^{-9}$ m/s);

Below the lining system, a 3 m-thick layer of silt is assumed to separate the artificial lining system from a 25 m-thick confined aquifer, whose piezometric level is 1 m above the top of the aquifer ($h_b = 1$ m).

The data of the landfill site are shown in Table 3.

4.1 Monitoring Post-closure Period

The temperature at the base of the landfill is assumed to be about 20 °C; therefore, the geomembrane's service life is expected to last 120 years [14]. As a result, during the monitoring post-closure period, the migration of inorganic contaminants (e.g., Cadmium) consists only of the advective–diffusive transport through the geomembrane defects. However, organic contaminants (e.g., Benzene) can diffuse through intact geomembrane with a consequent relevant increment in the overall contaminant mass flux passing through the lining system and reaching the underlying aquifer [8].

Table 3 Fundamental data of the landfill site

Landfill	
Bottom surface of the landfill, A_l (m^2)	100,000
Depth of the landfill liner, d_d (m)	25
Landfill length in the direction of groundwater flow, ℓ (m)	200
Landfill length in the orthogonal direction to groundwater flow, W (m)	500
Depth of the top of the confined aquifer, L_{GW} (m)	30
Height of leachate in the drainage layer, h_p (m)	0.5
Aquifer	
Piezometric height of the groundwater above the top of the aquifer, h_b (m)	1
Hydraulic gradient of groundwater flux, i (−)	0.01
Hydraulic conductivity of the aquifer, k_{aq} (m/s)	8.25×10^{-5}
Horizontal volumetric flux upstream from the landfill, q_{GW} (m/s)	8.25×10^{-7}
Transverse dispersivity of the aquifer, α_T (m)	1
Aquifer thickness, h (m)	25
Lining system	
Thickness of the compacted clay liner, L_{CCL} (m)	2
Hydraulic conductivity of the compacted clay liner, k_{CCL} (m/s)	1×10^{-9}
Porosity of the compacted clay liner, n_{CCL} (−)	0.55
Tortuosity factor of the compacted clay liner, τ_{CCL} (−)	0.1
Thickness of the natural attenuation layer, L_{AL} (m)	3
Hydraulic conductivity of the natural attenuation layer, k_{AL} (m/s)	6.94×10^{-7}
Porosity of the natural attenuation layer, n_{AL} (−)	0.46
Tortuosity factor of the natural attenuation layer, τ_{AL} (−)	0.25
Geomembrane	
Thickness of the geomembrane, L_g (m)	0.0025
Area of micro-holes, $A_{micro\text{-}holes}$ (m^2)	5×10^{-6}
Frequency of micro-holes, $n_{micro\text{-}holes}$ (ha^{-1})	25
Area of holes, A_{holes} (m^2)	1×10^{-4}
Frequency of holes n_{holes} (ha^{-1})	5
Area of tears, A_{tears} (m^2)	1×10^{-2}
Frequency of tears n_{tears} (ha^{-1})	2
Partition coefficient of Benzene, K_g (−)	57
Diffusion coefficient of Benzene, D_g (m^2/s)	6×10^{-13}

The analysis is performed for both Cadmium (Cd) and Benzene to appreciate the different behaviour between inorganic and organic contaminants.

The equivalent hydraulic conductivity of the compacted clay liner, whose thickness is equal to 2 m, and the underlying 3 m-thick silty layer is equal to 2.5×10^{-9} m/s, and the corresponding volumetric flux, q, is equal to 2.25×10^{-9} m/s. However, due to the presence of the geomembrane, leakage flow occurs only in correspondence with micro-holes, holes, and tiers, determining a volumetric flux, q_{d}, equal to 2.85×10^{-10} m/s when poor contact conditions are assumed between the geomembrane and the underlying compacted clay liner. The horizontal volumetric flux of the groundwater upstream from the landfill, q_{GW}, is equal to 8.25×10^{-7} m/s. The thickness of the contaminant plume in the aquifer, given by Eq. (4), is 20.1 m and is almost coincident with the term $\sqrt{2 \cdot \alpha_{\mathrm{T}} \cdot \ell} = 20\,\mathrm{m}$.

Following the approach of Italian regulations, the same attenuation factor is found for both Cadmium and Benzene, as molecular diffusion is not considered. The SAM coefficient is 0.83, and the LDF is equal to 291. Therefore, the acceptable concentration in the leachate, C_{l}, is 291/0.83 = 350 times the screening concentration in the groundwater. For Cadmium, C_{l} is 1.75 mg/l, which is considerably higher than the admissible concentration in Column A of Table 1, which is equal to 0.1 mg/l. In the case of Benzene, the screening concentration values in Italian regulation is 0.001 mg/l and, therefore, C_{l} results to be equal to 0.350 mg/l.

A more advanced analysis should evaluate the volumetric flux through the landfill liner and the underlying natural attenuation layer, taking into account unsaturated conditions. Assuming a capillarity coefficient of 0.01 m^{-1} for the compacted clay liner and 5 m^{-1} for the underlying silt deposit, the volumetric flux, q, is reduced to 1.83×10^{-9} m/s. In this scenario, the pore water pressures in the compacted clay liner are positive up to a depth of 0.6 m from the top of the layer. The remaining 1.4 m of clay and the underlying 3 m of silt are characterised by negative pore pressures. The volumetric water content of the compacted clay liner is expected to be about the porosity also under negative pore water pressures; in the case of the silty layer, based on the retention curve, the average volumetric water content is estimated to be equal to 0.25.

However, in the presence of the geomembrane, the correction due to unsaturated conditions is expected to be minor, and the volumetric flux given by Eq. (8) can be considered reliable (i.e., $q_{\mathrm{d}} = 2.85 \times 10^{-10}$ m/s).

Including molecular diffusion as a transport mechanism requires considering the free solution diffusion coefficient for Cadmium ($D_0 = 7.17 \times 10^{-10}$ m^2/s) and Benzene ($D_0 = 1.03 \times 10^{-9}$ m^2/s), and the partition coefficient, K_{g}, and diffusion coefficient, D_{g}, for Benzene through the geomembrane.

Using the data in Table 3, Eq. (9) provides the relative concentration profiles in the aquifer beneath the landfill shown in Fig. 3. The difference in profile between Cadmium and Benzene is small as the advective–diffusive transport through the geomembrane defects is dominant over the diffusion process through the intact geomembrane.

The maximum relative concentration is found at the exit from the landfill footprint (i.e., at $x = \ell$) and is equal to 2.8×10^{-3} for Cadmium and 2.9×10^{-3} for Benzene.

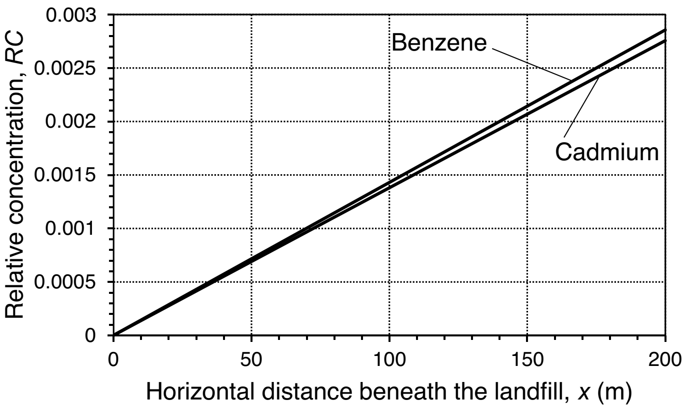

Fig. 3 Relative concentration for Cadmium and Benzene beneath the landfill

Assuming a null contaminant concentration upstream from the landfill site (i.e., $C_{x0} = 0$), the corresponding acceptable concentrations in the leachate are 363 and 350 times the screening concentration in the groundwater for Cadmium and Benzene, respectively, in good agreement with the assessment obtained following the approach of Italian regulations.

The distribution of relative concentration in the vertical direction within the aquifer can be determined through the numerical solution described in Dominijanni and Manassero [6]. The obtained values of relative concentration for Cadmium and Benzene are shown in Fig. 4.

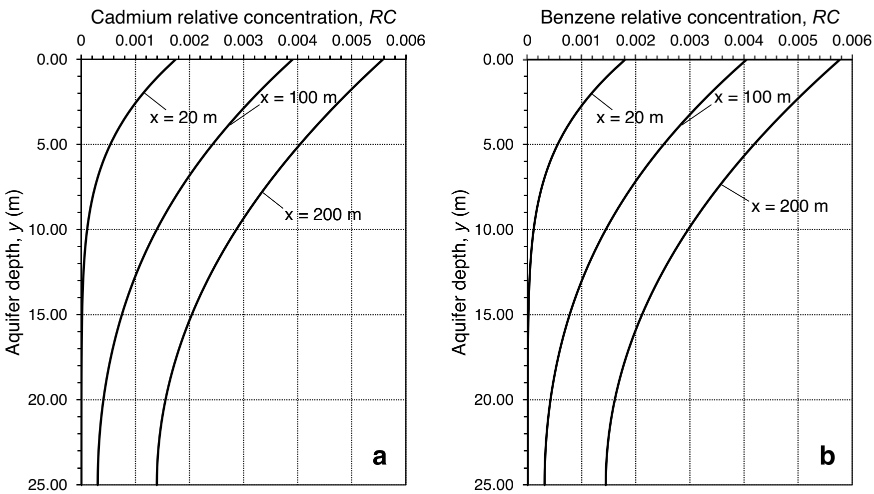

Fig. 4 Relative concentration for Cadmium **a** and Benzene **b** as a function of depth within the aquifer at different distances beneath the landfill ($x = 20$, 100, 200 m)

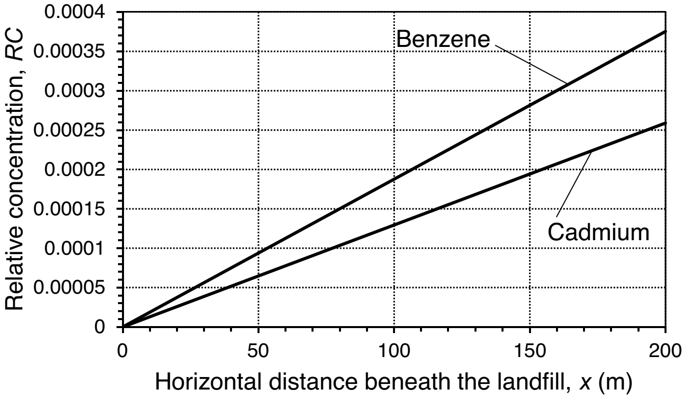

Fig. 5 Relative concentration for Cadmium and Benzene beneath the landfill in the case of a better-performing composite liner

A more significant discrepancy in the final results would be found in the case of a better-performing composite liner. Assuming the absence of micro-holes and tears and a frequency of 2.5 holes per hectare (i.e., $n_{\text{holes}} = 2.5 \times 10^{-4}\ \text{m}^{-2}$), the volumetric flux through the geomembrane defects becomes $q_{\text{d}} = 8.42 \times 10^{-11}$ m/s, and the corresponding profiles of Cadmium and Benzene are shown in Fig. 5. In this case, the relative concentration of Benzene is appreciably higher than the relative concentration of Cadmium due to the capability of Benzene to diffuse through the geomembrane. The difference in attenuation between inorganic and organic contaminants cannot be gathered from the approach of Italian regulations, which does not consider diffusion transport and, consequently, underestimates the concentration of organic contaminants.

4.2 Long-Term Condition

In the long term, when the geomembrane is degraded and has lost its functionality, the performance of the lining system is significantly reduced. Moreover, as a consequence of the decline of cover efficiency, a mound of the leachate level in the landfill is expected. The height of the leachate above the bottom liner depends on the water balance in the waste body. In this numerical example, a height of 10 m is assumed to have been found from this balance (i.e., $h_{\text{p}} = 10$ m).

Because of biodegradation, the concentration of organic contaminants is expected to drop substantially. Assuming a half-life of 25 years for Benzene in the landfill leachate, after 120 years, the concentration is decreased by a factor of 27.8, and it is not expected to be critical. Attention is therefore focused on the migration of inorganic contaminants, such as Cadmium.

Following the approach of Italian regulations, a vertical volumetric flux $q = 6.98 \times 10^{-9}$ m/s (i.e., 18.37 mm/month) is calculated. The depth of the contaminant plume is equal to 21.6 m, and LDF is equal to 13.8. Therefore, the acceptable concentration in the leachate, C_l, is 13.8/0.83 = 16.53 times the screening concentration in the groundwater. For Cadmium, C_l is 0.08 mg/l, which is lower than the admissible concentration of 0.1 mg/l in Column A of Table 1. As a consequence, the acceptable concentration at the end of the post-monitoring period must be reduced to 0.08 mg/l.

A similar result is found using Eq. (18) and determining the vertical volumetric flux under unsaturated conditions. Under the higher hydraulic gradient generated by the leachate mound, the compacted clay liner is characterised by positive pore water pressure up to a depth of 1.83 m from the top of the liner and $q = 6.464 \times 10^{-9}$ m/s. The transport of Cadmium is dominated by advection, and the concentration profile in the aquifer beneath the landfill shows a progressive increase up to a maximum relative concentration of 0.059 at $x = \ell$, as shown in Fig. 6. Assuming a null contaminant concentration upstream from the landfill site (i.e., $C_{x0} = 0$), the corresponding acceptable concentrations in the leachate is 16.95 times the screening concentration in the groundwater, in good agreement with the assessment obtained following the approach of Italian regulations.

The distribution of Cadmium relative concentration in the vertical direction within the aquifer, determined through the numerical solution described in Dominijanni and Manassero [6], is shown in Fig. 7.

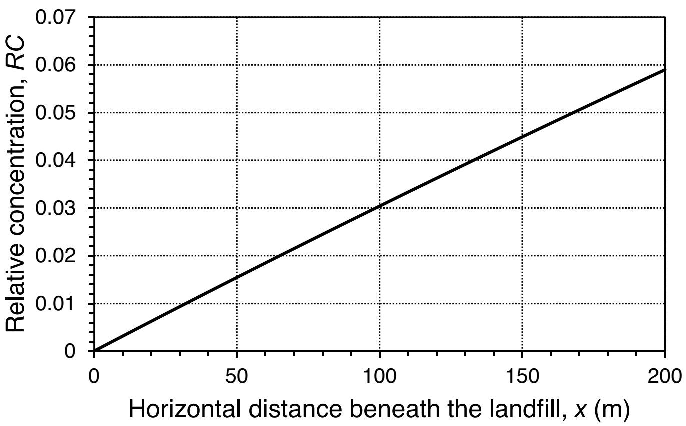

Fig. 6 Relative concentration for Cadmium beneath the landfill in the long term

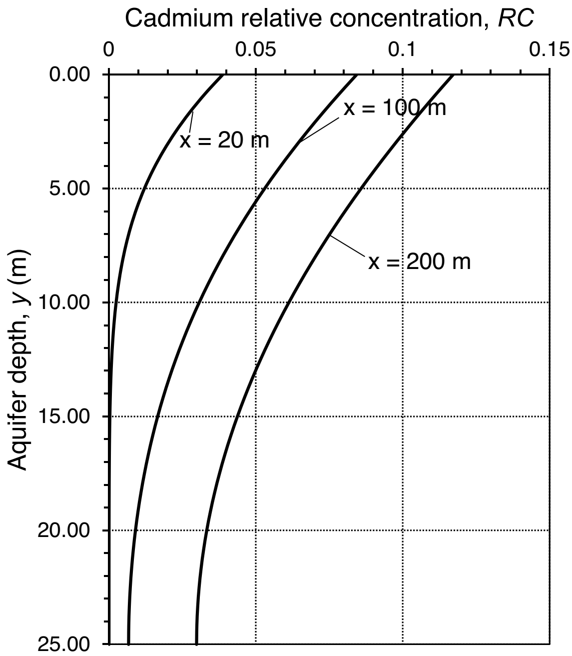

Fig. 7 Relative concentration for Cadmium in the long term as a function of depth within the aquifer at different distances beneath the landfill ($x = 20$, 100, 200 m)

5 Conclusions

The numerical examples presented in the previous section have shown that the risk assessment suggested in the Italian regulations provides results in agreement with the method developed by Dominijanni and Manassero [6] and Dominijanni [7] in the case of poorly performing composite barriers (i.e., for a high frequency of geomembrane defects including defective seams or tears) or in the absence of the geomembrane, as in the long term when the geomembrane loses its functionality due to the degradation of the polymer constituents. Under such conditions, advection is dominant over diffusion, and the acceptable leachate contaminant concentration given by Eq. (1) agrees with that obtained using the steady-state solutions described in Sect. 3. In this regard, it can be observed that the SAM coefficient appears to compensate for the overestimation of the water flow determined by assuming positive pore water pressures, which are not representative of the actual unsaturated conditions that are met in the case of natural foundation layers characterised by medium or high hydraulic conductivity.

The following additional comments may be made:

(1) The proposed steady-state solutions should not be considered as long-term, realistic simulations of contaminant migration but rather as conservative estimates of the risk related to a given contaminant concentration in the waste leachate,

in a similar way to a tier 2 analysis of the ASTM risk-based corrective action (RBCA) standard [1] for a polluted site.

(2) Beyond the proposed steady-state solutions, time-dependent analysis may be conducted to assess the risk for human health and the environment [5, 7, 14, 15]. These analyses, which could be considered a tier 3 risk assessment, allow the finite mass of the contaminant contained in the waste leachate to be taken into account. Moreover, they model attenuation mechanisms that are conservatively neglected in the steady-state solutions, such as biodegradation in the waste and the subsoil. As a result, the time-dependent analyses are expected to provide less conservative results than steady-state analyses, supported by the availability of data about the contaminant mass contained in the waste body and the rate of degradation in the leachate and soil.

(3) The analytical solution to the steady-state transport of contaminants from a landfill to the underlying aquifer can be applied not only to deterministic analyses but also to probabilistic ones. This opportunity is intriguing because many parameters of the model and its boundary conditions are characterised by random behaviour. In the context of a deterministic framework, the assignment of expected values to parameters, such as the source contaminant concentration, the hydraulic conductivity of mineral layers, the frequency, size, and shape of the geomembrane defects, and the hydrogeological properties of the aquifer, requires the designer to invoke their judgement. This critical step can be avoided by adopting a probabilistic approach, which allows us to take into account the random behaviour of the involved parameters. Moreover, one relevant advantage of the probabilistic approach is the possibility of appreciating the effect deriving from the combination of the variances of the parameters. This effect does not change the expected value of the final result but significantly affects its reliability.

The proposed risk assessment procedure is coherent with the ethical requirement of sustainability in taking care of the impact of our current activities on the quality of the environment for future generations. The *ratio* of the procedure consists of determining the contaminant concentrations in the leachate that are compatible with an acceptable risk level in the exposed groundwater resource in the long term. In this way, the potential harm for future generations caused by contaminant migration from the waste body to the underlying aquifer can be excluded.

However, a last critical consideration should be made about the threshold concentrations that are established to preserve the quality of the groundwater, especially when their values are not related to an actual risk for human health but are defined as screening values based on the "precautionary principle" considering the available knowledge about the contaminant toxicity.

It should be taken into account that the threshold concentrations we set also constitute an ethical choice. It has been estimated that the limits adopted in radiation protection in Western countries entail a cost of up to 2.5 billion dollars for each hypothetical human life saved. This cost is to be compared with the approximately

100 dollars needed to save a human life in developing countries from infectious diseases and other more widespread pathologies [12].

Climate change, the COVID-19 pandemic, and, more recently, the serious military conflicts that are bloodying the world require a reflection on the criteria we adopt for protecting human health and the environment. In fact, awareness has emerged that our health and environmental safety do not depend only on local conditions but on processes that develop on a global scale.

Therefore, we must be aware of the need to make choices that are both economically and ethically sustainable, in the face of the inequalities between the various countries of the world and the limited resources available. The sustainability perspective cannot be limited to local assessments but should take into account the global implications of our actions.

From this perspective, the progressive abandonment of the "precautionary principle", understood as a criterion of extreme caution, is desirable. The advancement of knowledge allows us to establish rational criteria for protecting human health and the environment based on specific technical and scientific knowledge.

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