A Critical Evaluation of Factors Required to Terminate the Post-Closure Monitoring Period at Solid Waste Landfills

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Policy Analysis

Critical Evaluation of Factors Required To Terminate the Postclosure Monitoring Period at Solid Waste Landfills

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Regulations governing the disposal of solid waste in landfills specify that they must be monitored for 30 years after closure unless this period is extended by the governing regulatory authority. Given the wide range of conditions under which refuse is buried, technical criteria, rather than a specific time period, are preferable for evaluation of when it is acceptable to terminate postclosure monitoring. The objectives of this paper are to identify and evaluate parameters that can be used to define the end of the postclosure monitoring period and to present a conceptual framework for an investigation of whether postclosure monitoring can be terminated at a landfill. Parameters evaluated include leachate composition and leachate and gas production. Estimates of leachate production from closed landfills are used to assess the potential environmental impacts of a hypothetical release to surface water or groundwater. The acceptability of gaseous releases should be evaluated against criteria for odors, the potential for subsurface migration, and greenhouse gas and ozone precursor emissions. The approach presented here must be tested on a site-specific basis to identify additional data requirements and regulatory activity that might be required to prepare regulators for the large number of requests to terminate postclosure monitoring expected over the next 20 years. An approach in which the frequency and extent of postclosure monitoring is reduced as warranted by site-specific data and impact analysis should provide an effective strategy to manage closed landfills.

Introduction

In 1991, the U.S. EPA promulgated regulations governing the disposal of nonhazardous wastes in landfills under Subtitle D of the Resource Conservation and Recovery Act (RCRA) (*1*). These landfills typically receive municipal solid waste (MSW), construction and demolition waste, and a wide

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variety of nonhazardous industrial wastes. MSW includes waste generated from the residential, commercial, and institutional sectors, and approximately 55% of the 220 million tons of MSW generated in 1998 were disposed of in landfills (*2*). In 1998, there were approximately 2300 permitted landfills, and landfills are projected to remain the dominant form of MSW management in the United States for the foreseeable future (*3*).

Once a landfill has reached its design capacity, a final cover must be installed. The site owner is then required to monitor and maintain a closed landfill for what is referred to as the postclosure monitoring period. Postclosure monitoring includes leachate collection and treatment, groundwater monitoring, inspection of the final cover and maintenance as required, and monitoring to ensure that methane is not migrating off-site. Regulations specify a 30-year postclosure monitoring period unless this period is extended by the regulatory agency on a site-specific basis, and private landfill owners budget funds for a 30-year postclosure period. The implication is that monitoring will be discontinued after 30 years because the landfill is stable and no longer represents a threat to the environment. However, the EPA lacks technical criteria from which to define stability. This lack of criteria will make it difficult for regulators to make decisions to either extend or reduce the postclosure monitoring period. The need for criteria on stability is further intensified by the increased interest in the operation of landfills as bioreactors, in which waste decomposition is accelerated (*4*). There are numerous advantages to bioreactor landfills, including the expectation that the landfill will reach a stable state more rapidly, thus permitting a decrease in the postclosure monitoring period (*5, 6*).

In this paper, it is proposed that technical criteria are required to provide regulators with guidance on when it is appropriate to suspend or extend postclosure monitoring. The objectives of this paper are to evaluate parameters that should be considered in the development of technical criteria on the postclosure monitoring period and to propose protocols to assess the biological and chemical stability of a landfill.

Overview of Refuse Decomposition

Cellulose and hemicellulose comprise 45−60% of the dry weight of MSW and are its major biodegradable constituents (*7*). The decomposition of these compounds to CH_4 and CO_2 in landfills is well-documented, and their decomposition contributes to long-term settlement (*6, 8, 9*). The relationship between solids decomposition, leachate composition, and gas production is well-established (*6*). In the first phase of refuse decomposition, the O_2 entrained in the refuse at burial is depleted, large quantities of CO_2 are produced, the refuse temperature increases, and the system becomes anaerobic. In phase two, there is an imbalance between the activity of the hydrolytic bacteria, which convert cellulose and hemicellulose to soluble intermediates, and that of the acetogenic and methanogenic bacteria which work together to convert these intermediates to CH_4 and CO_2. Because of this imbalance, short-chain carboxylic acids accumulate, the pH decreases, and there is little solids decomposition. During this acid phase (phase II) of decomposition, the leachate contains a high chemical oxygen demand (COD) that is attributable to carboxylic acids. Because these acids are biodegradable, the ratio of biological oxygen demand (BOD) to COD is near 1.

CH$_4$ production begins in the third phase of decomposition. This results in a decrease in the accumulation of carboxylic acids with corresponding decreases in the leachate COD and BOD and an increase in pH. During this phase, significant decomposition of cellulose and hemicellulose begins. In the fourth phase of decomposition, the accumulation of carboxylic acids is depleted and the rate of CH$_4$ production is dependent on the rate of cellulose and hemicellulose hydrolysis. In this phase, the BOD/COD is relatively low because any dissolved organic that is degradable is consumed by the indigenous microbial population. In theory, refuse will continue to decompose until little degradable material remains, at which time the landfill will become aerobic as the rate of O$_2$ infiltration exceeds its rate of depletion (10). However, the authors are not familiar with any landfills that have progressed beyond the fourth phase of decomposition.

Evaluation Framework

A characterization of leachate from decomposed refuse is presented first, followed by approaches that can be used to estimate the quantity of leachate produced from closed landfills. Data on leachate composition and quantity are then combined to illustrate methods to assess the impact of a leachate release on surface water and groundwater. Next, a model to calculate the volume of gas produced from refuse over time is presented and strategies to evaluate when gas production is no longer a threat to the environment are discussed. To evaluate leachate and gas production in the context of long-term landfill stability, the characteristics of relatively old refuse are most relevant. Thus, data from landfills that contain well-decomposed refuse, or already have had a final cover in place for several years, are emphasized throughout.

Characteristics of Leachate from Well-Decomposed Refuse

The major categories of contaminants in landfill leachate include organic matter, inorganic ions and nutrients, heavy metals, and xenobiotic organic chemicals, and reviews of leachate composition have been published (11−13). The concentrations of each category of compounds are reviewed in the following sections with a focus on concentrations typically present in well-decomposed refuse.

Dissolved Organic Matter. The three broadest measures of leachate organic content include total organic carbon (TOC), COD, and BOD. The BOD will decrease most rapidly, and its concentration can approach zero in the fourth phase of decomposition. In contrast, humic and fulvic acid concentrations increase with decomposition, so some COD and TOC will persist (12). Typical concentrations of BOD and COD in leachate from well-degraded wastes are presented in Table 1 of the Supporting Information and in the aforementioned reviews.

A commonly used indicator of the stage of waste decomposition is the BOD/COD ratio. Several researchers assign a BOD/COD value of less than 0.1 to stable leachate (8, 13, 14) which is consistent with the data in Table 1 (Supporting Information). A low BOD/COD ratio should be considered a necessary but insufficient condition to assess leachate stability. This is because leachate percolates through refuse and is collected at the bottom of a landfill. If the bottom-most layer of refuse is well-decomposed, then it will likely consume degradable organics in the leachate and leachate composition will be representative of the bottom-most layer. Thus, leachate may have a low BOD/COD, even though substantial quantities of actively decomposing refuse remain in the landfill.

Ammonia. Ammonia typically accumulates in leachate because there is no mechanism for its biodegradation under methanogenic conditions. Some have suggested that ammonia is the most significant long-term pollution problem at landfills (11, 15, 17, 18). Calculations of potential ammonia volatilization from landfills indicate that this mechanism can account for less than a 10% loss. NH$_3$−N concentrations for a number of landfills containing decomposed refuse are presented in Table 1 (Supporting Information) and concentrations of 100−1500 mg/L are common.

Heavy Metals. In a recent review of metals concentrations in leachate from 10 landfills, concentrations were typically at or below the United States drinking water standard (DWS), which is an exceedingly strict standard for leachate (10). Cd, Cr, and Zn concentrations exceeded United States DWS in 2, 2, and 1 landfill, respectively, while there were no reported excesses for Cu. Maximum reported Cd, Cr, and Zn concentrations (and their DWS) were 0.04 (0.005), 0.28 (0.01), and 5.3 (5.0) mg/L, respectively. The Pb concentration ranged from <0.005 to 0.19 mg/L (the DWS is 0), and Ni concentrations ranged from 0.0036 to 0.348 mg/L (no DWS). No metals data were available for United States landfills in which the refuse was specifically well-decomposed. However, in developing effluent guidelines for leachate treated at a landfill, the U.S. EPA elected not to include guidelines for metals as there was no evidence of problematic metals concentrations (19).

While the available data suggest that metals concentrations will not affect a determination on leachate stability in the short-term, long-term questions remain. Once refuse is completely decomposed under anaerobic conditions, it is theoretically possible that a landfill will gradually become aerobic (10). The time required to deplete all refuse organic matter has been estimated to range from a few centuries for an unsaturated, uncovered landfill to more than 500 000 years for a covered landfill fully saturated with water (20).

There are three mechanisms by which metals concentrations could increase if an anaerobic landfill were to turn aerobic: (1) oxidation of metal sulfides to metal sulfates that are generally more soluble, (2) increased complexation capacity of oxidized humic acids relative to reduced humic acids, and (3) increased metals dissolution due to a pH decrease (21, 22). A pH decrease could occur due to the increased CO$_2$ partial pressure associated with aerobic conditions and the oxidation and solubilization of reduced sulfur to sulfate. There have been a limited number of studies on metals concentrations in decomposed refuse that has become aerobic and the results are not consistent. In two studies, metals concentrations (Cd, Zn, Cr, Fe) did not increase when anaerobic refuse was subjected to aerobic conditions (23, 24). In a third study, Cd, Cr, and Zn concentrations did increase when samples of degraded waste from a landfill were aerated (22). Given the multitude of processes that can affect metals mobility in landfills including dissolution of carbonate or hydroxide precipitates, oxidation−reduction potential and its effect on both sulfides and humic acid functional groups, complexation, ion exchange, and sorption, additional experiments that are designed to specifically address the issue of long-term metals mobilization are required.

Xenobiotic Organic Compounds (XOCs). Data on the presence and concentration of XOCs in leachate have been summarized (10). The XOCs that are most frequently present in landfill leachate are aromatic hydrocarbons (benzene, toluene, ethylbenzene, and xylenes) and chlorinated solvents (e.g., trichloroethene) (11). The concentrations of specific XOCs in leachate vary between landfills, though individual XOC concentrations are generally less than 1 mg/L. There are not definitive data on XOC trends as a function of time or the state of decomposition. Although leachate data from older landfills are available, these landfills were often built prior to the enactment of RCRA and often accepted organic

solvents that are no longer permitted in landfills beyond the quantities in household waste. Thus, data on long-term trends in XOC concentrations are confounded by changes in waste composition. Theoretical considerations may be the best way to predict long-term trends in XOC concentration.

Recently, a simple box model was developed to evaluate the long-term behavior of XOCs in landfills (*25*). The model is based on equilibrium assumptions concerning phase distribution mechanisms and first-order degradation. Model results showed that volatile organic compounds (VOCs) will be released in landfill gas within a decade. For more strongly sorbing compounds, such as naphthalene, the release will take several decades.

There are several factors that could extend the life of XOCs in landfills, including the presence of wastes having high sorptive capacities. Slow desorption from such wastes may govern the release and bioavailability of XOCs (*26*, *27*). However, if a XOC is bound to another waste, then its slow release is not likely to lead to leachate concentration increases with time.

Leachate Production from Closed Landfills

The objective of this section is to develop estimates of leachate production after placement of the final cover. These estimates will then be used with leachate quality data to analyze the environmental impact of a leachate release to surface water or groundwater. Two methods to estimate leachate quantity are presented. The first uses published data on the quantity of leachate present in leachate collection and recovery systems (LCRS) and in leachate detection systems (LDS) underlying a liner system. The second method estimates leachate flow rates based on reports of liner defects and calculated leakage rates through holes in liners.

Typical Flow Rates and Liner Efficiencies. The standard liner system that meets regulatory requirements for a municipal waste landfill consists of 0.67 m of compacted clay overlain by a 1.5-mm geomembrane. A 0.33-m thick LCRS, consisting of high permeability media and perforated collection pipes, is installed above the geomembrane. Regulations require that the final cover be no more permeable than the liner system. Thus, the final cover typically includes, from the surface down, a vegetative layer, a drainage layer, a geomembrane and a clay layer. While alternative liner and cover systems exist, they are only approved when shown to provide performance equivalent to the standard liner system.

Once a final cover is in place, water that is not removed by runoff, evapotranspiration, or the drainage layer passes through the cover, percolates through the waste, and becomes leachate. A LCRS is used to remove this leachate. The liner system below the LCRS increases leachate capture by the LCRS. However, some leachate may permeate this liner and enter either the LDS, if present, or the environment. Landfills that use a LDS are referred to as double-lined landfills because there is a second liner below the LDS. Data from landfills with secondary liners are invaluable for obtaining field-scale data on the effectiveness of the LCRS.

In one study, LDS flow rates from 13 distinct cells at nine landfills that had received a final cover were reported (*28*). The landfill's primary liner contained either a geomembrane plus clay liner system (typical for Subtitle D landfills) or a geomembrane plus a geosynthetic clay plus a clay primary liner system (atypical). LDS flow rates of 5 to 210 L per hectare per day (L/(ha·day)) were reported based on 121 months of data (*28*). Additional leachate generation data was presented for landfills that have received their final cover. LCRS flow rates were reported to decrease by 75% 1 year after closure, and 90% 2−4 years after closure. Six years after closure, LCRS flow rates ranged from 5 to 1200 L/(ha·day) with a mean of 180 L/(ha·day). Nine years after closure, LCRS flow rates were negligible. These data include all of the closed landfills in the

TABLE 1. Liner Performance for New York Landfills in 1998 and 1999

	leachate collection and recovery system flow rate (L/(ha·day))	leachate detection system flow rate (L/(ha·day))	liner efficiency %[a]
	1998[b]		
mean	159.1	1.0	99.2
maximum	372.5	4.0	99.9
minimum	27.4	0.1	98.0
	1999[b]		
mean	117.4	0.8	99.1
maximum	290.8	2.6	99.9
minimum	16.3	0.1	97.1

[a] The apparent efficiency (%) is defined as (1-(LDS flow rate/LCRS flow rate)) × 100. [b] Based on 17 landfills in 1998 and 19 landfills in 1999.

study and are not specific to a certain liner type (*28*). Thus, the reported field data specify LDS flow rates of 5−210 L/(ha·day) for closed landfills and an indication that LCRS flow rates were negligible within 9 years of final cover installation, much faster than expiration of the prescribed 30-year postclosure monitoring period.

New York is one of a few states that requires double liners for MSW landfills. LCRS and LDS flow rates extracted from the annual reports of operating landfills in New York are summarized in Table 1. These data show LCRS efficiencies of greater than 99%. These data are not directly comparable to the previously described study in that none of the New York landfills yet have a final cover in place. The efficiencies presented in Table 1 are at the upper end of the range of efficiencies reported previously (*28*). This is because the previous study includes data for liner systems that are less restrictive than the applicable regulations and for systems that were installed without good quality control.

The significance of a 99% LCRS efficiency is illustrated by calculating the volume of water that would be released to a LDS, or to the environment, in the absence of a LDS. To make this calculation, assume that a landfill has a single composite liner (geomembrane + clay), a LCRS, and a final cover as previously described. In addition, assume that the cover drainage system and the LCRS both function at 99% efficiency, that 60% of annual precipitation is lost to evaporation and runoff, and that the landfill receives 101.6 cm of rainfall annually. Using these values, the volume of precipitation that would ultimately escape the cover drainage layer and the LCRS is 1.11 L/(ha·day), or 111.2 L/(ha·day), if 90% collection efficiencies are assumed. Interestingly, by assuming 60% loss to runoff and evapotranspiration and a 99% cover drainage efficiency, estimated percolation through a cover is 4.1 mm/year. This is consistent with recently reported values of 1−7 mm/year (mean 4.6) for composite covers in humid climates (*29*).

Liner Defects. A second approach to estimate leachate release is to estimate the number of liner defects and then calculate the resulting flow rate through a composite liner system. A composite liner consists of a geomembrane above a low permeability soil (clay), and defects refers to holes in a geomembrane. An estimate of three defects per hectare (standard deviation of 0.6−14.8) has been reported for landfills with good quality control during liner installation (*30*).

A methodology to calculate liner leakage rates through composite liner systems as a function of the number and shape of defects, the hydraulic head on the liner, and the saturated hydraulic conductivity of the underlying soil layer has been published (*31*). Equation 1 presents this methodology for circular holes in a geomembrane in good contact with the underlying soil.

$$Q = 0.21a^{0.1}h_w{}^{0.9}K_s{}^{0.74} \qquad (1)$$

where Q is the leakage rate (m³/s), a is the hole area (m²), h_w is the head on liner (m), and K_s is the saturated hydraulic conductivity (m/s). Good contact refers to an installation over well-prepared subgrade soil with good control over membrane wrinkling. Poor contact is a case where there is no control over membrane wrinkling and poor surface preparation of the underlying soil. For poor contact, 1.15 is substituted for 0.21 in eq 1. Equation 1 was applied to a hypothetical liner system under a 0.3 m head, which is the maximum permissible head and likely well above the actual head on a liner. The liner consisted of a geomembrane over 0.67 m of clay with a saturated hydraulic conductivity of 10^{-9} m/s. Leakage rates for 3 holes/ha were calculated to be 1.2, 6.8, 1.6, and 8.7 L/(ha·day), for 3 mm holes with good contact, 3 mm holes with poor contact, 10 mm holes with good contact, and 10 mm holes with poor contact, respectively. These calculated values are the same order of magnitude as hypothetical leachate releases calculated based on 99% cover drainage and LCRS efficiencies.

In summary, field data indicate leachate production rates of 5−210 L/(ha·day) for landfills with a final cover, as well as an observation that leachate generation rates decreased to zero nine years after final cover installation. Field data from operating landfills indicated LCRS efficiencies of 99% for liners built with good quality control, which suggests leachate release rates on the order of 1 L/(ha·day). Estimates based on the frequency of liner defects also suggest leachate release rates of less than 10 L/(ha·day). While these data are not a substitute for site-specific data, they provide an estimate of potential long-term leachate release rates.

Assessment of Potential Impacts after Termination of Postclosure Monitoring

One criterion that could be used to terminate postclosure monitoring would be a determination that a landfill is no longer a threat to the environment based on scenarios in which leachate was released. In this section, methods are presented to evaluate the impact of a hypothetical leachate release on surface water and groundwater. This is followed by an estimate of the volume of gas that would be released if a gas collection and control system were to be shut down.

Surface Water Quality Impacts of Landfill Leachate. One worst-case scenario would have leachate released directly to the environment after termination of postclosure monitoring. If a direct release of leachate to surface water does not threaten a surface water body, then the landfill can be considered stable, at least in the context of surface water. The impact analysis illustrated below focuses on dissolved organic matter and ammonia. The potential impacts associated with the release of other pollutants such as XOCs can be evaluated using the general approach illustrated for BOD and NH_3-N. In the case of a volatile chemical, rapid evaporation from surface water will be a significant fate. As discussed earlier, metals concentrations are generally less than or comparable to drinking water standards, even during the active life of a landfill. Thus, no impact assessment is proposed though work may be required on a site-specific basis if elevated metals concentrations are present.

Organic matter and ammonia both exert an oxygen demand on streams, increasing stress on aquatic life. The method presented in this section combines the Streeter Phelps model, which predicts the microbial depletion of dissolved oxygen (DO) by organic matter, with nitrification terms, to include oxygen depletion due to ammonia (eq 1 of Supporting Information) (32). The model was applied to a 40.5 ha landfill with an assumed leachate release rate of 100 L/(ha·day). Data for both the stream and leachate chemical

TABLE 2. Data Used To Calculate Dissolved Oxygen Depletion from a Leachate Release to Surface Water

	leachate	Crabtree Creek at median flow[a]	Crabtree Creek at low flow[a]
BOD (mg/L)	100	5.00	5.07
NH_3-N (mg/L)	250	0.11[b]	0.3[b]
DO (mg/L)	0	6	6
Saturation DO (mg/L)		8	8
Flow (m³/day)	4.05	87170[c]	4927[d]
Velocity (m/s)		0.3	0.03
$K_d = K_r = K_N$ (day^{-1})[e]		1.5	1.5
K_a (day^{-1})[e]		22[f]	36[f]

[a] After mixing with leachate. [b] NH_3-N plus organic N. [c] Long-term median flow (33). [d] The annual minimum 7-day consecutive low flow which, on average, will be exceeded in 9 out of 10 years (33). [e] K_d, effective deoxygenatation rate of organic carbon; K_r, overall loss rate of CBOD due to settling and oxidation of soluble BOD; K_N, rate of oxidation of NH_3 and organic nitrogen; K_a, reaeration coefficient. [f] Midpoint values for deep streams (median flow) and shallow streams (low flow) (32).
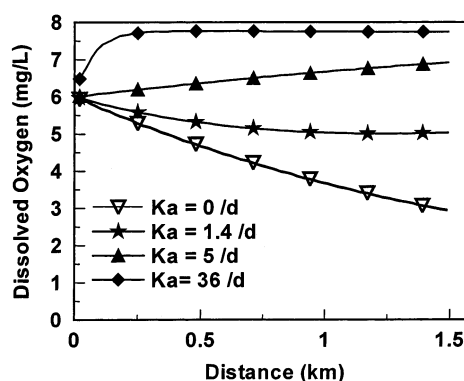

FIGURE 1. Impact of a leachate release on downstream dissolved oxygen concentrations at varying rearation constants (K_a (day^{-1})). The plot represents low-flow conditions for Crabtree Creek in North Carolina with model constants given in Table 2.

composition, as well as typical model coefficients are summarized in Table 2. Stream flow rates for median- and low-flow conditions are representative of Crabtree Creek in Raleigh, NC, which flows past a landfill that was closed in 1997. All other data are hypothetical. At median-flow conditions, the leachate discharge contributes less than 0.005% of the streamflow and has a negligible impact on the in-stream BOD and ammonia concentrations. The impact of a leachate release on a receiving stream at the low-flow condition is presented in Figure 1. As illustrated, the DO does not decrease below 5 mg/L for reaeration constants (K_a) greater than 1.4 day^{-1}. A DO of 5 mg/L is emphasized because a lower concentration is generally considered stressful to aquatic organisms (32). Reaeration is influenced by the saturation value of DO in the stream, wind speed, stream turbulence, temperature, surface films, and hydraulic structures. For Crabtree Creek, reaeration coefficients were estimated to vary between 22 and 36 day^{-1}, depending on water depth and velocity (34). Although Figure 2 extends for 1.5 km, Crabtree Creek actually discharges to a much larger river after 0.34 km at which point the effects of the leachate oxygen demand are significantly diminished.

While the model results presented in Figure 1 are intended to illustrate a methodology, site-specific data could be utilized to evaluate a landfill that has applied for a termination of postclosure monitoring. Required data would include that described in Table 2 as well as consideration of the designated use of the receiving water body which might influence acceptable water quality impacts.

A second potential impact of a leachate release on aquatic life is the ammonia concentration. Ammonia standards depend on the temperature and pH of the stream and the presence of early life stages of fish. Ambient water quality criteria for ammonia range from 0.9 to 9.6 mg of NH_3-N/L (*35*). At 0.9 mg/L, leachate with 250 mg of NH_3-N/L would require a dilution factor of 278 to meet the standard in the absence of any biodegradation, a condition met for both the median- and low-flow cases previously described.

Nutrient Loading. The objective of this section is to evaluate whether a leachate release could contribute excessive nutrients to a surface water body. Eutrophication is excessive plant and algal growth that interferes with water uses (*32*). Though eutrophication is a natural process, its rate is significantly increased by the presence of excess nutrients, especially N and P. Even after NH_3-N is oxidized to NO_3-N, the nitrogen may still cause excessive plant growth (*36*). To improve water quality for the Neuse River Basin, which includes Crabtree Creek, regulators have proposed rules that will limit discharges of N and P in the watershed.

To compare potential nutrient loads from a landfill to wastewater treatment plants, annual loads from each source have been calculated for a population of 500 000. Assuming a wastewater generation rate of 246 L/(person·day), the plant will discharge 1.23×10^5 m³/day with permissible N and P concentrations of 3.5 and 2 mg/L, respectively. This would result in N and P releases of 157 and 90 metric tons/year, respectively. To compare this release to a landfill, consider a facility with a useful life of 20 years that serves 500 000 people who send 1.8 kg/(person·day) of waste to the landfill. The surface area of this landfill will be approximately 40 ha based on a site with an average waste depth of 24.4 m, a waste density of 772 kg/m³ and 10% of the volume occupied by daily cover. If the landfill releases 100 L/(ha·day) of leachate with NH_3-N and P concentrations of 250 and 1 mg/L, respectively, then the nutrient loads would be 365 kg of N and 1.4 kg of P annually, which are well below 1% of the loads released from the wastewater treatment plant. Of course, in this example, a second landfill will be required after 20 years while the first is still releasing N and P.

The hypothetical release rates from the landfill to groundwater, expressed per unit area, are 9.13 kg of N/(ha·year) and 0.037 kg of P/(ha·year). These values are in the same range or lower than values typically given for N and P in surface water runoff; 8.4−22.3 kg of N/(ha·year) and 1.4−3.4 kg of P/(ha·year), where the range represents residential to commercial development (*37*). In addition to the potential nutrient release to groundwater, the landfill surface will also contribute surface runoff that is likely to be typical of moderate development. While the landfill cover is not likely to increase N and P concentrations above ambient levels, the cover is designed to promote runoff that could elevate N and P mass loading rates relative to forested land. In sensitive watersheds, excessive N and P mass loadings could be mitigated by the installation of stormwater control devices including ponds, wetlands, and sand filters. More generally however, releases from POTWs are much greater.

Groundwater Quality Impacts of Landfill Leachate

A second worst-case scenario would involve a leachate release to groundwater. Potential pollutants of concern include organic carbon, NH_3-N, and XOCs. As discussed earlier, metals concentrations in most leachates are low, even during the active life of a landfill.

There are a wide variety of XOCs potentially present in leachate. Most hydrophobic compounds will strongly sorb to the solid waste in the landfill and are expected to have low potential for migration in groundwater (*25*). The most common contaminants, aromatic hydrocarbons and chlorinated solvents, are moderately soluble in water and have moderate to low sorption potential, so they could migrate with the groundwater. Chlorinated solvents are often very resistant to biodegradation under aerobic conditions but can be reduced to nontoxic endproducts (ethene, ethane, carbon dioxide) in the carbon rich anaerobic environments typical of most landfills (ref *38*, Chapter 5). Thus, the XOCs of greatest concern are the aromatic hydrocarbons. While a variety of studies have shown that aromatic hydrocarbons can be biodegraded under anaerobic conditions, there is clear evidence that aromatic compounds are much more easily biodegraded under aerobic conditions and that aromatic hydrocarbon plumes can persist for years or decades if the available oxygen is depleted (ref *38*, Chapter 9).

The primary processes influencing the fate of leachate in groundwater are advection, dispersion, sorption to the aquifer matrix, and biodegradation. The impact of these processes can be evaluated using commonly available contaminant fate and transport models (*39*). Advection and dispersion can result in significant reductions in contaminant concentrations, but these processes do not result in a reduction in the total mass of a pollutant; they only distribute it over a larger volume of groundwater. Sorption slows the rate of migration but does not destroy the contaminant; so ultimately, the pollutant will breakthrough at the receiving point. As a consequence, the major focus of this analysis will be on the effects of biotransformation processes on pollutant concentration and mass flux.

Organic carbon, ammonia, and aromatic hydrocarbons are all biodegradable under aerobic conditions. There is also evidence that some portion of these pollutants can be biodegraded under anaerobic conditions using nitrate, solid-phase iron and manganese, sulfate or carbon dioxide as terminal electron acceptors. However anaerobic biodegradation rates for these pollutants are often slow, and the extent of biodegradation can vary depending on the site-specific groundwater geochemistry and contaminants present in the leachate (*11*). In addition, anaerobic degradation processes can have a variety of undesirable side effects, including production of dissolved manganese, iron, sulfide, taste, and odors. Thus, if groundwater beneath and downgradient of a landfill remains aerobic, the major pollutants of concern will be rapidly degraded. However, if the groundwater becomes anaerobic, presumably due to the release of high organic carbon leachate, pollutants or undesirable degradation products may migrate away from the landfill and adversely impact downgradient receptors.

VOCs could enter groundwater from a landfill by diffusion through the liner system or by liner leakage. Recently, a method to calculate diffusive losses of volatile compounds through a geomembrane was presented, and the analysis showed that the diffusive loss of toluene exceeded the release rate through defects based on a leachate release rate of 1 L/(ha·year) (considerably lower than the release rate assumed in the previous discussion) (*40*). The analysis also showed that long periods (30−100 years) may be required before diffusive loss rates reach steady-state. This finding suggests that long-term monitoring of the vadose zone could be required to evaluate the potential for significant diffusive releases of volatile organics, even in the absence of groundwater contamination during the postclosure monitoring period. Such monitoring would be required if risk assessments based on measured leachate concentrations indicate a potential long-term problem.

Leakage through a liner could potentially result in a single large release or several smaller releases. However, the maximum extent of dilution or aerobic biodegradation will ultimately be limited by the supply of dilution water containing DO. Because of the relatively large size of most landfills, transverse mixing of uncontaminated groundwater from outside the boundaries of the landfill will have relatively

TABLE 3. Potential Impact of Dilution and Aerobic Biodegradation on Contaminants in Leachate[a]

groundwater flux	low	medium	high
aquifer saturated thickness (m)	2	10	20
groundwater transport velocity (m/d)	0.01	0.1	0.5
aquifer porosity	0.25	0.25	0.25
landfill width (m)	637	637	637
groundwater flux under landfill (m³/day)	3.2	159.3	1592
leachate release rate (m³/day)	4.05	4.05	4.05
leachate dilution by GW (%)	44%	98%	99.75%
DO in groundwater (mg/L)	1	5	9
oxygen flux under landfill (kg/year)	1	291	5230
BOD released by landfill (kg/year)[b]	1837	1837	1837
maximum BOD oxidized based on perfect mixing (%)	0.06%	16%	285%

[a] The chemical characteristics of the leachate are presented in Table 2. [b] An equivalent BOD that includes both organic carbon and ammonia was used to calculate the total BOD release.

little impact, and the primary source of water to dilute the leachate will be groundwater migrating underneath the landfill from an upgradient recharge area. If the velocity or saturated thickness of the groundwater is high, then there will be a large supply of groundwater to dilute the leachate and supply DO.

An analysis of the potential dilution and aerobic biodegradation of leachate released from a hypothetical landfill for high, medium, and low groundwater and DO fluxes is presented in Table 3. Leachate release rates are the same as in the previous analysis of surface water impacts. The extent of dilution can vary from insignificant to very high, depending on the site-specific hydrogeologic conditions. For the medium- and low-flux cases, biodegradation will be limited by the DO supply and dilution will only distribute the mass through a larger body of water. However, for the high-flux condition, there is sufficient oxygen present (assuming perfect mixing in the aquifer) to degrade all of the BOD released from the landfill. If the leachate release rate were a factor of 10 less than the estimate used previously, then sufficient oxygen would be present to degrade all of the BOD for both the medium- and high-flux cases. However, biodegradation would still be oxygen limited for the low-flux case.

The extent of biodegradation may be further limited by mixing between a concentrated leachate source and uncontaminated groundwater from beneath the landfill. Figure 1 (Supporting Information) shows the predicted BOD mass flux versus distance downgradient from a leachate release for three conditions: (1) no biodegradation, (2) aerobic biodegradation of a single point source release of leachate, and (3) aerobic biodegradation of two smaller leachate releases. When there are two uniformly distributed sources, the plumes spread out and consume the available DO from within the landfill boundaries. However, when there is only a single point source release of leachate, mixing of the leachate with uncontaminated groundwater containing DO further limits the extent of biodegradation.

Once an anaerobic plume discharges from beneath a landfill, oxygen may begin to enter the groundwater due to diffusive mass transfer with the oxygen present in the unsaturated zone and recharge of oxygenated water. Recent modeling work has shown that diffusive mass transfer of oxygen can be very important and may limit the downgradient migration of aerobically degradable contaminants (41).

Landfill Gas

Landfill gas typically consists of 40−60% CH_4 and the remainder CO_2 plus some XOCs. The improper management of landfill gas can result in (1) nuisance odors, (2) subsurface
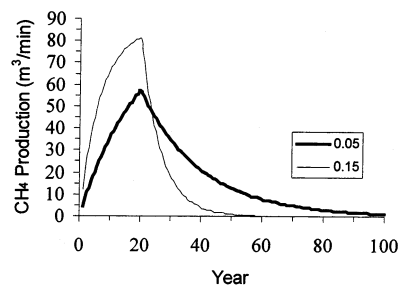
FIGURE 2. Methane production rate for a landfill that receives 286 000 metric tons/year for 20 years. With reference to eq 2, L_o = 170 L/kg, the default value used for NSPS guidelines. The decay rate (k) varies to represent the NSPS default value of 0.05 year^{-1} and enhanced decomposition in a bioreactor (k = 0.15 year^{-1}). Equation 2 was applied to the waste buried annually for 20 years, and this figure represents the total methane production for all waste present in the landfill at a given time.

migration that can result in an explosion hazard in underground structures, (3) the release of CH_4, a greenhouse gas that contributes to atmospheric climate change, and (4) the release of nonmethane organic carbon (NMOC) that contributes to ozone formation and its associated deleterious health effects. Most landfills install either passive vents in which the gas is vented to the atmosphere or active collection systems in which the collected gas is either flared or utilized as an energy source. Under the New Source Performance Standards (NSPS) and Emissions Guidelines, landfills with a capacity exceeding 2.5 million metric tons are required to collect and either flare landfill gas or use it for an energy recovery project (42). Through the postclosure monitoring period, the landfill owner is required to monitor the vadose zone to ensure that the CH_4 concentration does not exceed 25% of its lower explosive limit at the site boundary.

CH_4 production rate curves for a landfill that has received 286 000 metric tons/year for 20 years are presented in Figure 2. These curves are based on eq 2, which is the basis for the U.S. EPA Landfill Gas Model (43).

$$G = WL_o k e^{-kt} \qquad (2)$$

where G = methane production rate (m³/year), W = annual waste acceptance rate (ton/year), L_o = ultimate methane yield (m³ CH_4/ton), and k = decay rate (year^{-1}). The two curves presented in Figure 2 illustrate decomposition at typical rates in traditional landfills in nonarid regions of the United States and decomposition in landfills that are operated to accelerate decomposition. With reference to Figure 2, note that year 50 represents the 20-year life of the hypothetical landfill plus 30 years of postclosure monitoring.

For landfills with a capacity of less than 2.5 million tons, only a passive gas collection system would have been required initially, so the issue of whether a gas collection and control system can be terminated after closure is not relevant. For landfills with an active gas collection system, a finding that postclosure monitoring could be terminated must include a finding that it is safe to discontinue an active gas collection system. Whether deactivation of an active gas collection system is acceptable can be evaluated against odors, explosion hazards, and the release of greenhouse gases and NMOCs. With respect to odors, if discontinuation of an active landfill gas collection system results in problematic odors, then the gas collection system should be reactivated. This can only be judged on a site-specific basis as a function of atmospheric conditions and the proximity of people to a landfill. Decisions must also be made on what constitutes an unacceptable frequency of odor complaints. With respect to explosions, monitoring wells that are already in place should

be monitored after discontinuation of an active gas collection system. If significant CH_4 concentrations appear in the vadose zone, then the consequences of subsurface migration must be evaluated based on the site-specific geology and the proximity of underground structures. As for odors, reactivation of the gas collection system could be required. Similarly, the presence of XOCs in the vadose zone would suggest the potential for these compounds to contaminate downgradient groundwater and may warrant reactivation of a gas collection system (*44*). Vadose zone monitoring data could be required as part of an application to terminate postclosure monitoring. Finally, with respect to the release of a greenhouse gas and NMOCs, guidance must be developed on acceptable emission rates. In evaluating potential emission rates, it is important to differentiate between CH_4 production (Figure 2) and CH_4 emissions that are likely reduced by methane oxidation as gas migrates through a soil cover. Some NMOCs may also be oxidized in a soil cover (*45, 46*).

Discussion

Regulators will be facing decisions to either extend the postclosure monitoring period or allow its termination over the next 20−30 years as landfills that were closed in the 1990s approach 30 years postclosure. In future work, the conceptual approach presented here must be evaluated on a site-specific basis to determine its utility and to identify areas where further development is required. Such development could include the need for data acquisition and may identify areas where regulatory authority must be clarified through legislative or rulemaking activity.

The approach presented here is modular and flexible. For example, an alternative methodology was recently proposed for calculation of leakage through liners as a function of liner defects (*47*), and this methodology could easily be substituted for eq 1. Similarly, alternate surface water quality and landfill gas production models can be used.

Though not addressed in this paper, another aspect of landfill stability is its physical or geotechnical stability. Landfills settle as refuse decomposes, and this settlement can lead to damage to the final cover (*48*). During postclosure monitoring, these cracks are repaired, preventing excessive water infiltration. However, cracks that develop after termination of postclosure monitoring may not be repaired, and this could result in leachate production in excess of the estimates developed in the previous discussion. Thus, data to indicate that a landfill has completed its settlement may also be required to terminate the postclosure monitoring period. The need for a predictive model of waste settlement and a proposed approach has been presented (*49*).

The concepts described here have been presented to groups of regulators, landfill owners and operators, and design engineers. There is widespread agreement on the need to develop technical criteria for landfill stability (*50, 51*) though a range of views exist on how to define stability. At one extreme is the position that the landfill is only stable when its contents are no longer a threat, even under conditions of a massive release. Under these conditions, the presence of any ammonia or BOD could suggest that the landfill is not completely stable, a condition unlikely to be achieved for centuries without extensive flushing. At the other extreme is complete reliance on engineered containment systems in the form of the liner and cap. Between these two extremes is a mixed strategy, with reliance on engineered containment to minimize potential releases, an assessment of the decomposition state of the contained refuse, and evaluation of the severity of impacts of potential leachate and gas releases.

An intermediate position between monitoring landfills forever and monitoring landfills until they are determined to be completely stable would be to reduce the frequency and extent of postclosure monitoring as site data and impact analysis suggest that the landfill is no longer a threat to the environment. For example, if leachate is not accumulating in a LCRS, then it is likely that insignificant amounts of water are infiltrating through the cover. Periodic inspections of the cover, and performance of any necessary maintenance, can ensure that leachate production does not increase. Alternately, if leachate is present, then its composition and rate of accumulation can be monitored and the impact of a release analyzed. Similarly, periodic vadose zone and groundwater monitoring can be used to monitor for long-term diffusive losses of VOCs or methane. The depletion of DO in the underlying aquifer might indicate the release of leachate with high TOC and would warrant more intensive monitoring.

Acknowledgments

Supporting Information Available

Table with more detail on the composition of leachate from well-decomposed refuse, the model used to assess surface water quality impacts, and more detail on the groundwater contaminant transport modeling approach that was used to simulate the potential impacts of a leachate release to groundwater. This material is available free of charge via the Internet at http://pubs.acs.org.

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