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Technical Considerations for Evaluating the Environmental Emissions from RCRA Subtitle D Landfills Beyond the 30-Year Post-Closure Care Period

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Technical Considerations for Evaluating the Environmental Emissions from RCRA Subtitle D Landfills Beyond the 30-Year Post-Closure Care Period

by

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Notice/Disclaimer

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Foreword

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Acronyms and Abbreviations

AP-42	Compilation of Air Pollutant Emission Factors
App I	40 CFR 258 Appendix I
App II	40 CFR 258 Appendix II
ASTSWMO	Association of State and Territorial Solid Waste Management Officials
BLS	US Bureau of Labor Statistics
BOD	Biochemical Oxygen Demand
CDD	Construction and Demolition Debris
CFR	Code of Federal Regulations
CH_4	Methane
cm	Centimeter
COD	Chemical Oxygen Demand
COPC	Contaminant of Potential Concern
C.U.	Color Units
EPA	United States Environmental Protection Agency
EPACMTP	EPA's Composite Model for Leachate Migration with Transformation Products
EREF	Environmental Research and Education Foundation
ERG	Eastern Research Group, Inc.
FAC	Florida Administrative Code
FDEP	Florida Department of Environmental Protection
ft	Foot
ft^3	Cubic Feet
GPAD	Gallons Per Acre Per Day
GCCS	Gas Collection and Control System
GCL	Geosynthetic Clay Liner
GHG	Greenhouse Gas
GM	Geomembrane
HAP	Hazardous Air Pollutant
HDPE	High-Density Polyethylene

HELP	Hydrologic Evaluation of Landfill Performance
HHE	Human Health and the Environment
IDNR	Iowa Department of Natural Resources
ITRC	Interstate Technology and Regulatory Council
IWCS	Innovative Waste Consulting Services, LLC
IWEM	Industrial Waste Management Evaluation Model
KDHE	Kansas Department of Health and Environment
k	Landfill Gas Decay Rate (year $^{-1}$)
k_l	Leachate Collection Decay Rate (year $^{-1}$)
kg	Kilogram
L	Liter
lb	Pound
LCRS	Leachate Collection and Removal System
LCS	Leachate Collection System
LDS	Leachate Detection System
LFG	Landfill Gas
LFGTE	Landfill Gas-to-Electricity or Energy
LLDPE	Linear Low-Density Polyethylene
m^2	Square Meter
m^3	Cubic Meter
µg	Microgram
MCL	Maximum Contaminant Level
MDEP	Massachusetts Department of Environmental Protection
MDL	Method Detection Limit
mg	Milligram
Mg	Megagram
mm	Millimeter
MSW	Municipal Solid Waste
MSWLF	Municipal Solid Waste Landfill
NESHAP	National Emission Standards for Hazardous Air Pollutants
NCSU	North Carolina State University

NMOC	Nonmethane Organic Compound
NOAA	National Oceanic and Atmospheric Administration
NRMRL	National Risk Management Research Laboratory
NSPS	New Source Performance Standards
NTU	Nephelometric Turbidity Unit
OH EPA	Ohio Environmental Protection Agency
ORD	Office of Research and Development
PCB	Polychlorinated Biphenyls
PCC	Post-Closure Care
PE	Polyethylene
ppmv	Parts Per Million by Volume
PVC	Polyvinyl Chloride
RCRA	Resource Conservation and Recovery Act
SCFM	Standard Cubic Feet per Minute
SEAs	State Environmental Agencies
SEM	Surface Emissions Monitoring
SMCL	Secondary Maximum Contaminant Level
SSE	Sum of Squared Errors
SST	Total Sum of Squares
STP	Standard Temperature and Pressure
S.U.	Standard Units
TDS	Total Dissolved Solids
TOC	Total Organic Carbon
UCL	Upper Confidence Limit
UDSHW	Utah Division of Solid and Hazardous Waste
US	United States
USGS	United States Geological Survey
VDEQ	Virginia Department of Environmental Quality
VLDPE	Very Low-Density Polyethylene
VTANR	Vermont Agency of Natural Resources
WADOC	Washington Department of Ecology

WDEQ	Wyoming Department of Environmental Quality
WWTP	Wastewater Treatment Plant

Executive Summary

Title 40 of the Code of Federal Regulations, 40 CFR Part 258 (herein referred to as *§258*) includes standards for the design, operation, closure, post-closure care (PCC), monitoring and other requirements for municipal solid waste landfills (MSWLFs or Subtitle D landfills) under the authority of Subtitle D of the Resource Conservation and Recovery Act (RCRA). These regulations require the owner or operator of each MSWLF unit to conduct PCC for 30 years unless an extended or reduced period is demonstrated to be necessary or sufficient, respectively, to protect human health and the environment (HHE). In addition to §258, the owners/operators of MSWLF units may be required to meet additional regulations during the PCC period pertaining to the management of landfill gas (LFG). These additional regulations include the New Source Performance Standards (NSPS) for MSWLFs and the National Emission Standards for Hazardous Air Pollutants (NESHAP). In general, MSWLFs required to meet these regulations must construct, operate, and monitor an active LFG collection and control system (GCCS). The GCCS system must be operated for at least 15 years (or sooner due to lack of adequate LFG flow per Subpart XXX) and until the annual nonmethane organic compounds (NMOCs) emissions are less than 50 Mg/year (or 34 Mg/year if required to meet the requirements of 40 CFR 60 Subpart XXX).

RCRA Subtitle D PCC requirements (including the 30-year period) were promulgated in 1991 and 1992, and it is expected that several MSWLFs will reach or surpass a PCC period of 30 years in the next 5-10 years. Unlike NSPS thresholds for terminating GCCS, RCRA Subtitle D regulations do not specify numeric criteria for determining the PCC extensions or termination. This determination is dependent on the current and potential HHE impacts of MSWLFs. The availability of various site-specific data (e.g., in-place waste characteristics, LFG and leachate collection rate and quality, groundwater quality, surface and subsurface gas emissions, closure cap performance, landfill settlement) is vital for assessing HHE impacts of MSWLFs. This report presents an assessment of the nature and prevalence of available data that can be used for evaluating the HHE impacts of closed MSWLFs based on a review of data from nine closed MSWLFs (or MSWLF with closed cells) located in different regions of the United States. It also presents data gaps, approaches to identify the contaminants of potential concern and technical approaches to estimate emission rates of these contaminants, and operating and monitoring considerations for robust evaluation and mitigation of long-term HHE impacts of MSWLFs.

The analyses provided in this report are intended to be relevant to state environmental agencies, MSWLF owners/operators, community decision-makers, and other stakeholders interested in understanding approaches for assessing the HHE impacts of modification or termination of MSWLF PCC. Nine sites that contained at least one MSW cell that has been closed for at least five years, and has environmental monitoring records (e.g., groundwater monitoring, LFG collection rate and quality, leachate collection rate and quality) and located in different climate zones of the US were selected for detailed evaluation. The study cell(s) footprint ranged from 6 to 69 acres and contained 0.63 to 4.40 million metric tons of MSW, respectively.

The available actual and estimated cost data of different PCC activities were analyzed to develop an understanding of the financial impacts associated with PCC. Study-cell specific PCC cost data

were available only for three sites; the average annual PCC cost for the cells at these sites was $5,300 per acre (in 2017 dollars). The average annual PCC cost for the remaining sites, for which only site-wide cost data were available, was estimated to be $6,450 per acre (in 2017 dollars).

Landfill gas collection rate and methane content data were available for review from eight of the nine sites. As expected, the methane collection rates from the closed sites exhibited a declining trend, which is potentially attributed to the first-order decay kinetics of the anaerobic waste decomposition process. Site-specific decay rates were estimated by best-fitting the first-order waste decomposition rate equation, which is typically used for estimating the methane and NMOCs generation rate from landfills to the measured methane collection rates. The annual NMOCs collection rates – calculated based on the site-specific decay rate and methane collection potential estimates and waste placement data – are estimated to decline below 34 Mg/year for seven of the study sites within 30 years of closure. These findings may not be applicable to other MSWLFs in the US as the study sites are smaller than approximately 75% of MSWLFs in the United States.

In order to estimate GCCS operation timeframes for typical size MSWLFs, NMOCs and methane collection rates were modeled for MSWLFs containing approximately 3.35 (small MSWLF), 7.85 (medium MSWLF), and 19.1 (large MSWLF) million metric tons of MSW, which correspond to the 25^{th}, 50^{th}, and 75^{th} percentile capacity of MSWLFs in the United States, respectively. Decay rates ranging from 0.02 to 0.22 year $^{-1}$ were used to represent conditions from arid weather (slow waste decomposition) conditions to bioreactor operation (fast waste decomposition). A methane generation potential of 100 m^3 per Mg waste, NMOCs content of 4,000 ppmv (as hexane), and a gas collection efficiency of 100% was assumed for this analysis. The results suggest that annual NMOCs collection rates for small MSWLFs located in arid to moderate precipitation areas are not likely to decline below 50 Mg per year within 30 years after closure. NMOCs collection rates for medium and large MSWLFs are not expected to decrease below 50 Mg per year within 30 years after closure irrespective of location. The operation of MSWLFs to promote more rapid waste stabilization (e.g., bioreactor landfill operation) has the potential to significantly reduce the timeframes needed for annual NMOCs collection rates to decline below the NSPS threshold of 50 Mg/year and required GCCS operating timeframe.

The analysis also suggests that the mass-based remaining methane (Mg methane) and NMOCs generation potential (Mg NMOCs) are more appropriate indicators of the HHE impacts than the percent remaining methane generation potential, which are currently used by some state as a PCC period evaluation criterion. The results also suggest that a potential increase in the decay rate can increase the annual NMOCs collection rates above the NSPS threshold of 50 Mg per year after GCCS operation termination if the in-place waste has considerable remaining NMOCs generation potential; the decay rate was assumed to increase by 100% ten years after GCCS termination for this analysis. The final cover, therefore, should be rigorously maintained even after GCCS termination until the NMOCs generation potential and the leaching potential of the in-place waste has declined to levels that are unlikely to pose a risk to HHE. In addition, landfill owners and regulators should also continue surface and subsurface emissions and odor monitoring to proactively identify signs of an increase in LFG generation rate after GCCS operation termination and have provisions in place to resume GCCS operation, if needed, to control these issues.

Leachate collection records were reviewed and analyzed to identify approaches for estimating post-closure leachate collection rates at MSWLFs. In general, except for those landfill cells that recirculated leachate after closure, the most recently-measured leachate collection rate from all of the study cells (as of the time of this study) was less than 100 gallons per acre per day (GPAD). None of the landfill cells examined exhibited trends indicative of having reached a steady-state leachate collection/generation rate. Several sites exhibited a general declining trend in leachate collection rates after closure. The hydraulic efficiency of the primary liner was estimated for four of the study cells equipped with a double bottom liner system based on the available leachate collection system (LCS) and leak detection system (LDS) flow rates. The sum of leachate flow rates from the LCS and LDS was assumed to represent the leachate generation rate for this analysis. The primary liner efficiency was estimated to range from 96.8% to 99.6%. Substantial differences in several leachate indicator parameters (chloride, trace organics, total organic carbon) among LCS, LDS, and groundwater suggest groundwater intrusion might be a significant source of liquids collected from the LDS.

Three modeling methodologies were evaluated to estimate PCC leachate collection rates from each of the cells of the study sites: first-order decay modeling, unsaturated flow modeling, and the Hydrologic Evaluation of Landfill Performance (HELP) model. All three approaches provided relatively similar approximations to measured leachate collection rates for a majority of the study cells.

The comprehensiveness of available leachate quality data at the study sites was analyzed for the parameters specified in the federal regulations for groundwater monitoring at Subtitle D landfills (Appendix I and Appendix II of §258). In the current study, data for a total of 272 constituents were evaluated to assess the leachate quality at eight of the study sites. The selected constituents either have a primary or secondary maximum contaminant level (MCL or SMCL), or are listed in 40 CFR 258 Appendix I or II, or were used by EPA (2017a) for leachate quality evaluation of Subtitle C landfills. Only three sites reported leachate constituent data for every Appendix I parameter, and only one of the study sites reported at least one measurement for all but three of the Appendix II parameters. The leachate characteristic data for less than half of the Appendix I parameters were available for two study sites. More than half of the study sites reported leachate constituent concentration data for ten or fewer Appendix II parameters (excluding Appendix I parameters). The available data suggest that apart from the lack of the data for a large number of groundwater monitoring parameters, the small number of measurements available at the study sites may limit a reliable HHE impact assessment for several constituents.

A screening analysis was conducted to identify the contaminants frequently measured in leachate above respective risk-based protection standards after closure to identify the contaminants of potential concern. Parameters never measured above their risk-based standards are not expected to present a risk to HHE in the future. The federal primary and secondary drinking water standards were used as the thresholds for this evaluation. Fifteen out of 68 primary/secondary drinking water parameters monitored at least once were measured at concentrations above the respective method detection limit (MDL) in more than 50% of the samples. Six of these parameters (i.e., arsenic, total dissolved solids (TDS), iron, manganese, chloride, and color) were measured above their MCL/SMCL in more than 94% of the samples. Among all of the constituents with MCL, arsenic

and turbidity were the only primary MCL parameters that consistently exceeded the MCL. Arsenic exhibited a declining and an increasing trend at four and three study cells, respectively. The most recent arsenic measurements at three of the sites, which exhibited a declining trend, were above the MCL. A majority of organic compounds measured at the sites were undetected in most of the samples. The analysis presented in the report is, by no means, represents a comprehensive evaluation as it was limited to the constituents that were measured at the study sites and excluded various emerging contaminants (e.g., PFAS) that may have HHE impacts.

It should be noted that the leachate quality is typically reflective of the decomposition status of the bottom-most waste layer and does not necessarily represent the degree of stabilization of the entire landfill. A well-decomposed waste layer above the LCS may attenuate the concentration of parameters such as biological oxygen demand (BOD) and chemical oxygen demand (COD) that are commonly used to assess leachate and waste stability. Moreover, a relatively lower concentration of a large number of contaminants than the respective MCL/SMCL may also be due to partial stabilization/mineralization of landfilled waste because of the lack of exposure to adequate moisture. After the LCS operation termination, leachate might accumulate within the landfill and eventually discharge into the environment as leakage through liner defects and/or side slope seeps resulting from the leachate build-up. The stabilization/mineralization of the unstable waste constituents and ensuing elevated contaminants concentrations in leachate after LCS operation termination due to potential moisture intrusion into the landfill may pose a risk to HHE. The impacts of any future moisture intrusion on potential leachate emissions can be evaluated and mitigated by actively recirculating leachate or adding other liquids sources (e.g., stormwater) while the site is actively monitored and maintained by owners/operators and regulators. The liquids addition, however, has design, operating, and monitoring challenges such as unavailability of moisture source especially in arid areas, the complexity of adding liquids to achieve uniform moisture distribution in the landfill, flooding of gas collection devices, and a need to collect and manage excess leachate at the end of bioreactor operation.

Groundwater quality data were reviewed and analyzed to identify impacts to groundwater for three sets of parameters: those with MCL, those with SMCL, and for some parameters without MCL or SMCL. Occasional MCL exceedances were observed following cell closure. Three sites exhibited recurring groundwater exceedances. Due to observed exceedances above the respective MCL at several upgradient wells, the recurring exceedances at two of these sites could not be conclusively attributed to the lined cells. Monitoring data from the third site indicates that the elevated levels of arsenic observed at downgradient wells may be a result of subsurface geochemical changes below the liner system. The laboratory detection limit was greater than the respective MCLs for several measurements (e.g., several organics at one site, arsenic and thallium measurements at another site). Only one site with silver and zinc (only Appendix II parameters with SMCL) measurement data had a single silver exceedance. As of the time of this evaluation, all three of the sites under assessment monitoring and/or corrective action had instances where parameters without an MCL (vanadium, 1,1-dichloroethane, and 1,4-dioxane) contributed to groundwater impacts; these parameters had a state-specified risk-based standard.

A comprehensive review of the data collected from the study sites suggests several data gaps or data quality issues that could hinder a robust and quantitative HHE impact assessment of PCC

reduction or termination at the study sites. The ability for the owner and/or operator of an MSWLF unit to demonstrate the protection of HHE following completion of PCC is dependent on having quality environmental monitoring data specific to the MSWLF unit of interest. MSWLFs owners/operators should consider collecting the following data that are important for assessing HHE impacts:

1. **Waste Tonnage and Composition** – Waste-specific (e.g., MSW, industrial waste, construction and demolition debris) disposal tonnages were available for several study sites. Detailed waste characterization data (e.g., plastics, paper, food waste, household hazardous waste etc.) estimated based on composition studies were available only for one site. Both composition and tonnage information, especially of industrial waste, are valuable for HHE impact assessment.
2. **Leachate Quality Data** – Although RCRA Subtitle D regulations do not require routine leachate chemical characterization, leachate quality data are essential for the HHE impact evaluation. MSWLF owners/operators should consider harmonizing the list of monitoring parameters for groundwater and leachate for a comprehensive HHE impact assessment. Leachate quality should be monitored for the required groundwater monitoring parameters, constituents that occur at elevated levels in leachate (e.g., chloride, ammonia, total dissolved solids), and those that are commonly used for assessing the waste stability (e.g., biochemical oxygen demand and chemical oxygen demand). Furthermore, monitoring of emerging contaminants such as per- and polyfluoroalkyl substances and pharmaceuticals, which have been reported to be present in leachate, should be considered for comprehensive HHE impact assessment. The laboratory reporting limits of the monitored parameters should be equal to or lower than the respective groundwater protection standard.
3. **Groundwater Monitoring System** – A periodic review of changes such as surrounding land use/zoning changes that can impact the groundwater flux and flow direction should be considered while assessing the long-term impacts of modifying or terminating PCC. Monitoring of groundwater quality with respect to leachate indicator parameters such as chloride, ammonia, BOD, and COD should be considered.
4. **Settlement** – Differential settlement of the landfill surface represents one of the more probable risks to the integrity of the final cover. The compromises in the final cover system might result in moisture intrusion, which could subsequently increase gas and leachate generation rates and the fugitive gas emissions. Routine settlement monitoring data can be used to estimate the future settlement rate. In addition, settlement data, when used in conjunction with a temporal analysis of LFG collection and leachate quality, can provide an indication of waste stabilization status. Settlement data were not available for several study sites. It should be noted that landfill owners routinely conduct topographic surveys during landfill operating life to assess airspace usage and availability. Continuation of these surveys after closure would provide valuable data for evaluating the waste stabilization rate and the magnitude of total and differential settlement.
5. **Monitoring Records** – Some monitoring data (e.g., perimeter probes monitoring for tracking subsurface gas migration and surface emissions monitoring for identifying

fugitive LFG emissions), which are typically required to be routinely monitored for MSWLFs, were not available for review/analysis for several study sites. MSWLFs owners/operators and regulators should consider implementing documentation systems for cataloging monitoring data for prompt retrieval and analysis.

1. Introduction

1.1. Background

The disposal in landfills has been the predominant method of managing municipal solid waste (MSW) generated in the United States (US), and approximately 139 million tons, or 52.1% of all MSW, were landfilled in 2017 (EPA 2019a). Landfills receiving MSW are required to comply with federal, state, and local, if applicable, regulations. The requirements found under Title 40 of the Code of Federal Regulations (CFR), Part 258 (herein referred to as *§258*): Criteria for Municipal Solid Waste Landfills (referred herein to as MSWLFs), specify the performance-based design, operating, monitoring, closure and post-closure care (PCC) criteria for these landfills. The PCC criteria require the operation, maintenance, and monitoring of MSWLFs for 30 years after closure or as necessary to ensure that the MSWLFs do not pose a threat to human health and the environment (HHE). The federal PCC criteria allow the reduction (or extension) of the 30-year PCC period if the modified period is sufficient (or necessary) for protecting HHE. However, currently, there is no federal guidance or specific direction on approaches that an MSWLF owner/operator can use for making a demonstration supporting the reduction or termination of PCC activities.

Many MSWLFs have closed since the promulgation of §258 in the early 1990s and are quickly approaching the end of the 30-year PCC period. Figure 1-1 shows the number of MSWLFs closed and the corresponding 30-year PCC timeline based on the closure date reported to the US Environmental Protection Agency (EPA) (EPA 2019b) under the federal mandatory greenhouse gas (GHG) reporting program, as described in Title 40 of the CFR Part 98 (40 CFR 98).

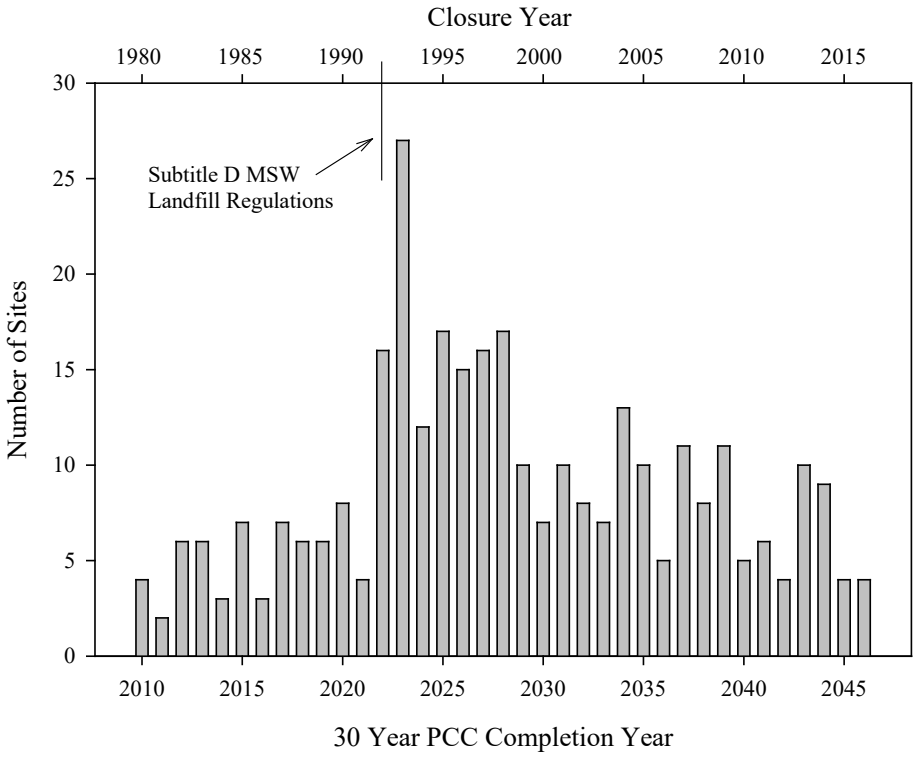


Figure 1-1. Number of Closed MSWLFs by Reported Closure Year and the Projected 30-Year PCC Completion Year based on the EPA Greenhouse Gas Reporting Program Database (EPA 2019b)

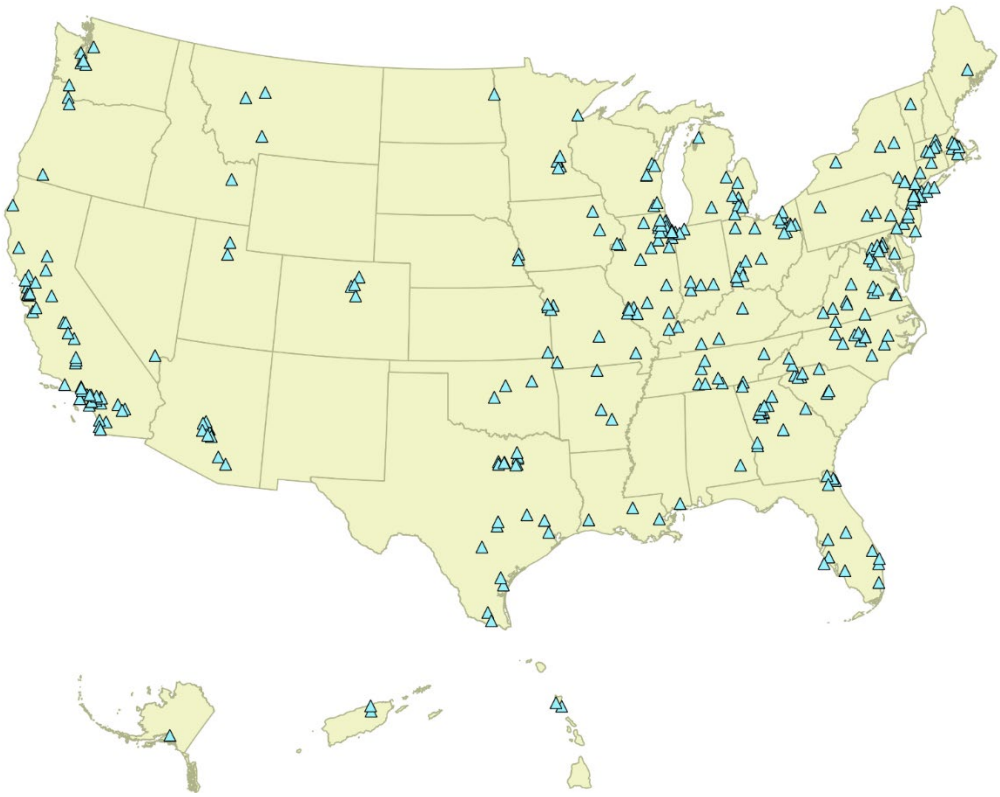


Figure 1-2. Locations of All Closed MSWLFs Included in the EPA Greenhouse Gas Reporting Program Database (EPA 2019b)

There are 330 MSWLFs that are listed as closed in the EPA Greenhouse Gas Reporting Program database (EPA, 2019b) as of 2018 – the locations of these closed MSWLFs are presented in Figure 1-2. As a point of comparison, there are approximately 980 operating sites included in the EPA greenhouse gas database. It should be noted that landfills presented in Figures 1-1 and 1-2 are only a subset of the total number of closed MSWLFs in the US as only MSWLFs that emit GHG above a regulations-specified amount are required to report data to the GHG reporting program. The number of operating and closed landfills in the US is greater than the numbers presented above.

The Environmental Research and Education Foundation and the Interstate Technology and Regulatory Council have published a performance-based methodology to assess HHE impacts of closed MSWLFs (EREF 2006, ITRC 2006). In addition, a few state environmental agencies (SEAs) have developed criteria for evaluating the PCC period of closed MSWLFs. The Association of State and Territorial Solid Waste Management Officials (ASTSWMO) conducted a survey in 2011 to gather information on states' policies and/or regulations with respect to post-closure requirements beyond 30 years for Subtitle C and D landfills (ASTSWMO 2013). The survey showed considerable variation among the SEAs regarding the approaches, processes, and procedures that could be used to adjust the PCC period. ASTSWMO recommended that EPA develop guidance on the review and consideration for adjusting the PCC period for MSWLFs. EPA (2017a) recently published a guidance document to evaluate the performance of Subtitle C landfills under PCC. This report presents the application of various approaches and criteria for evaluating long-term environmental emissions potential of closed Subtitle D landfills.

1.2. Objectives and Scope

The primary objectives of this study were to (1) assess the nature and prevalence of the design, operation, and monitoring data available for MSWLFs that can be used for HHE impacts evaluation, (2) present application of approaches that site owners/operators and engineers can use to evaluate monitoring data to identify contaminants of potential concern (COPCs) and develop potential emission rates of these contaminants to the atmosphere and groundwater/surface water, (3) identify data gaps, and (4) present operating and monitoring considerations for MSWLFs owners to evaluate and mitigate HHE impacts of MSWLFs. To provide real-world context, data from nine MSWLF sites scattered across the US were compiled and evaluated to assess data availability, demonstrate the use of the analysis approaches, and identify typical data gaps. Specifically, this report discusses an evaluation of environmental monitoring data related to leachate quantity and quality, landfill gas (LFG) subsurface migration, LFG surface emissions, LFG quantity and quality, groundwater monitoring data, and the final cover system settlement and maintenance that can be used to assess the HHE impacts of MSWLFs. The PCC cost data for the study sites were also reviewed to understand the cost associated with PCC activities and PCC extension beyond the 30-year period. Several of the approaches proposed by EREF (2006), ITRC (2006), and EPA (2017a) were used for analyzing data from these study sites.

The EPA ORD collected, reviewed, and analyzed environmental monitoring data from nine MSWLF sites with closed cells. Sites that were selected had at least one closed cell with a containment system (liner and cap) that included a geomembrane, had/have an active gas collection system, and had monitoring data/records readily available. Initially, EPA attempted to identify sites with cells that had been in PCC for a minimum of 10 years. However, due to the scarcity of closed MSWLFs with all the monitoring data listed above, sites closed before 2010 (with at least five years of PCC data) were also considered for the study.

It should be noted that PCC, as discussed with respect to cells located at the nine MSWLF sites selected for this study, refers to the period after the cell had been capped (with a geomembrane) and no longer received waste. The final cover of the study cells may not necessarily meet the regulatory definition of "closed" (e.g., one site has an exposed geomembrane cap, which only meets the state's definition of an intermediate closure system); the cover systems evaluated in this study may be permitted as a final or intermediate cover system.

This study primarily focuses on the requirements of 40 CFR Part 258: Criteria for MSWLFs. Additional federal regulations that MSWLFs may be required to comply with are the Standards of Performance for MSWLFs (i.e., 40 CFR Part 60 Subpart WWW and XXX) and the National Emission Standards for Hazardous Air Pollutants (NESHAP) (i.e., 40 CFR Part 63, Subpart AAAA). These requirements are discussed in more detail in the next chapter of this report. It should be noted that MSWLFs operation and monitoring is regulated by the states (with EPA-approved regulatory programs) requirements, which may be more stringent than the federal regulations.

The data evaluation and analysis approaches presented in this report are expected to be useful to all stakeholders, including site owners, operators, regulators, and engineers associated with MSWLFs design, permitting, operation, monitoring, closure, and PCC activities. The evaluations presented in the report have limitations and do not represent a comprehensive HHE risk assessment. The stakeholders should critically evaluate the appropriateness of the assumptions and limitations of the analysis presented in the report before electing to use these for their sites.

Furthermore, the application of approaches for evaluating the study sites is imperfect due to the gaps in LFG, leachate, and groundwater data available for the study sites. This report is not meant to serve as a standalone manual for conducting a comprehensive assessment of HHE risks associated with closed RCRA Subtitle D landfills for evaluating appropriate PCC duration.

1.3. Report Organization

This report is organized into ten chapters. Chapter 1 introduces the objectives and scope of the study and describes the organization of this report. Chapter 2 describes the federal regulations pertaining to PCC of MSWLFs and summarizes several sets of state and industry guidance for evaluating and modifying PCC at MSWLFs. The study sites' selection criteria, the types of data collected, and a summary of the key design and operating features of each of the cell(s) at the nine MSWLFs chosen for this study are presented in Chapter 3. Chapter 4 summarize the cost of different PCC activities at the study sites. Chapter 5 presents a detailed review of site LFG data and the methane monitoring probes surrounding the sites, and example approaches to estimate long-term gas emissions potential of a site at any given point of time. Chapter 6 summarizes temporal trends in leachate quantity and quality from the sites and describes approaches to estimate long-term leachate generation rate and to identify the COPCs with respect to HHE. Groundwater monitoring data from the sites are reviewed in Chapter 7. Chapter 7 also presents approaches to assess sources of impacts to groundwater quality around the MSWLF unit. Chapter 8 presents an evaluation of settlement and surface emissions monitoring (SEM) data. A summary of the key findings of this study and operating and monitoring considerations for MSWLF's stakeholders are provided in Chapter 9, along with recommendations for PCC data collection. A list of the data and information sources used in the development of this report is included in Chapter 10.

2. Background

The EPA promulgated minimum national criteria under Subtitle D of the Resource Conservation and Recovery Act (RCRA) for MSWLFs codified under §258 (referred herein to as *§258* or *Subtitle D* or *MSWLF regulations*). MSWLF units that ceased receiving waste by October 9, 1991, were exempted from these criteria. The MSWLFs that received flood-related waste or received less than 100 tons of waste per day were also exempted from all these criteria except the final cover requirements if they stopped accepting waste on or before April 9, 1994 (or October 9, 1994, under certain conditions). The regulations also conditionally exempt another subset of MSWLFs from some of the requirements. For example, MSWLFs accepting 20 tons of waste per day or less (based on annual average) are conditionally exempt from the design and groundwater monitoring criteria. Because these MSWLFs are exempted from requirements such as liner design, groundwater monitoring, and PCC, these MSWLFs were not considered for review in this study.

This chapter summarizes the federal requirements promulgated in §258 for the design, operation, monitoring, closure, and PCC of MSWLFs. For a detailed description of Subtitle D regulations, readers should refer to the CFR, which is electronically available at https://www.ecfr.gov. The requirement not specific to PCC are also presented as many of these must be met throughout the PCC period, and a number of states require compliance with these requirements as a starting point to evaluate an adjustment to the frequency of PCC activities.

2.1. Landfill Operation and Design

Subpart C of MSWLF regulations (§258.20-§258.29) lists the operational criteria of an MSWLF unit. The following is a summary of the operational criteria:

1. The concentration of methane gas generated at the facility shall not exceed 25% of the lower explosive limit for methane in the facility structures (excluding gas collection and control system (GCCS) components) and shall not exceed the lower explosive limit at the facility boundary. The criteria require routine monitoring to ensure compliance.
2. Landfill owners or operators shall meet the applicable standards developed under a State Implementation Plan pursuant to section 110 of the Clean Air Act. The open burning of waste (with limited exceptions) is prohibited.
3. Landfill owners or operators shall have a run-on and run-off control system to manage stormwater run-off resulting from a 24-hour 25-year storm event and shall prevent the discharge of pollutants into waters of the US.
4. Bulk or noncontainerized liquid waste may not be placed in the MSWLF unless it is household waste (other than septic waste), or it is leachate or gas condensate from the MSWLF and the MSWLF has a composite liner and leachate collection system.
5. All the records pertaining to design, operation, inspection, training, notification, PCC plan, financial assurance, etc. must be retained near the facility or as approved by the state authority.

MSWLF design criteria are listed in Subpart D of §258. A new MSWLF or lateral expansion of an MSWLF shall have a composite liner with a leachate collection system that can maintain less than a 30-cm depth of leachate over the liner. The composite liner shall contain a minimum 30-mil thick flexible membrane liner (60-mil thick if high-density polyethylene) overlain by a two-foot thick layer of compacted soil with hydraulic conductivity no more than $1x10^{-7}$ cm/sec. The federal regulations do not specifically require a leak detection system for MSWLFs. Alternative

liner designs, which ensure that the concentration of specific parameters would not exceed their maximum contaminant level (MCL) in the uppermost aquifer at the point of compliance, are allowed. The parameters and their MCL are presented in Table K-1 of Attachment K.

2.2. Monitoring

Subpart E of §258 provides requirements for groundwater quality monitoring at MSWLFs. A groundwater monitoring system shall be installed with a sufficient number of wells at appropriate locations and depths to evaluate the impact of the MSWLF unit on the uppermost aquifer. Groundwater monitoring is performed under a detection monitoring (§258.54) or assessment monitoring (§258.55) program. The detection monitoring shall be performed at all the groundwater monitoring wells for the parameters listed in §258 Appendix I (hereafter referred to as 'App I'). App I has a total of 62 parameters consisting of 15 inorganics and 47 organic parameters. The regulation allows the states to modify this list of parameters. Unless a demonstration is made showing the need for an alternative monitoring frequency, detection monitoring is required at least semi-annually during the MSWLF's active life and the PCC period.

Assessment monitoring is required when a statistically significant increase over background concentrations has been detected for any parameter listed in App I or a state-approved alternative list. The groundwater samples must be analyzed for all the constituents listed in Appendix II of §258 (hereafter referred to as 'App II') within 90-days of starting an assessment monitoring program and annually thereafter. The regulations allow the states to modify the list of assessment monitoring parameters. App II has a total of 215 parameters; all App I parameters are also listed in App II. App I and II parameters are listed in Table J-2 of Attachment J.

Within 90 days of finding that a constituent listed in App II is exceeding the groundwater protection standard as defined under §258.55 (h) or (i), an assessment of corrective measures and remedial action is required. If all App II parameters are measured at or below background values for two consecutive sampling events, the owner or operator may return to detection monitoring.

Additionally, §258.23 requires the quarterly monitoring of methane concentrations in site structures and at the property boundary. The methane gas shall not accumulate at levels equal to or more than 25% of the lower explosive limit in site structures and shall not exceed the lower explosive limit at the property boundary. Subsurface methane monitoring locations must be selected around the periphery of the site at appropriately selected depths based on the site hydrogeologic and hydraulic conditions. The landfill owners shall implement a remediation plan for the methane gas releases if methane is detected above the levels mentioned above.

2.3. Closure and Post-Closure Care

Closure and PCC criteria and requirements are listed in Subpart F of §258. Based on the closure criteria as listed in §258.60, MSWLFs are required to be capped with a final cover system designed to minimize infiltration and erosion. The final cover shall have permeability less than or equal to the permeability of any bottom liner system or natural subsoil present, or a permeability no more than $1x10^{-5}$ cm/sec, whichever is less. The final cover design should include an earthen material infiltration layer with a minimum 18-inch thickness, and a 6-inch earthen material layer capable of sustaining native vegetative growth. The Director of an approved State may approve an alternate final cover design that achieves an equivalent reduction in infiltration and equivalent protection from wind and water erosion. The regulations require the preparation of a closure plan that lists the steps necessary to close all MSWLF units at any point during their active life.

Unless the Director of an approved State accepts an alternative PCC period (based on a demonstration by the owner or operator of an MSWLF unit that the alternative PCC period is sufficiently protective of HHE), §258.61 requires the MSWLFs owner or operator to perform PCC for 30 years following the closure. Specifically, the PCC of each MSWLF unit must be performed to:

1. Maintain the integrity and effectiveness of the final cover. This includes repairing the cover as needed to address the effects of settlement, subsidence, erosion, or other events and prevent the final cover system erosion and damage from stormwater run-on and run-off;
2. Maintain and operate the leachate collection system in accordance with §258.40. Leachate management can be discontinued if it is demonstrated that the leachate no longer poses a threat to HHE;
3. Monitor groundwater in accordance with §258 Subpart E and maintain the groundwater monitoring system;
4. Maintain and operate the explosive gas monitoring system in accordance with §258.23.

In addition to these operation and monitoring requirements, the landfill owners may be required to implement an active gas collection and control system per the New Source Performance Standards (NSPS). The NSPS operations and monitoring requirements are presented in Section 2.5. The owner or operator must prepare a PCC plan and place it in the operating record no later than the initial receipt of waste (or October 9, 1993 – whichever was later). At a minimum, the PCC plan must include:

1. A description of monitoring and maintenance activities and the associated frequency for each MSWLF unit;
2. Contact information of the person or office to contact during PCC period;
3. A description of the planned use of the property during PCC.

Following the completion of the PCC period, the State Director shall be notified by the owner or operator that a certification (signed by an independent professional engineer or approved by the director of an approved state) verifying that the PCC has been completed has been placed in the operating record.

2.4. Financial Assurance

Subpart G of §258 provides requirements for the financial assurance for closure, post-closure care, and corrective action, if applicable. The owner or operator is required to have a detailed written estimate (in current dollars) of the cost of hiring a third party to close the largest area of all MSWLF units ever requiring a final cover (as required under §258.60) at any time during the active life in accordance with the closure plan (§258.61).

The owner or operator is also required to have a detailed written estimate, in current dollars, of the cost of hiring a third party to conduct PCC for the MSWLF unit in compliance with the post-closure plan developed under §258.61 of this part. The PCC estimate used to demonstrate financial assurance must account for the total costs of conducting PCC, including annual and periodic cost, as described in the post-closure plan over the entire PCC period.

The cost estimates must be based on the most expensive closure and PCC costs and must be annually adjusted for inflation. The estimated costs must be increased if the changes in closure or post-closure care plan increase the maximum costs. The owner or operator of each MSWLF unit must establish financial assurance for closure and PCC and must provide continuous coverage

until released from financial assurance requirements. The regulation also specifies the allowable mechanisms such as trust fund, surety bond, insurance for demonstrating financial assurance.

2.5. NSPS and NESHAP Regulations

NSPS for MSWLFs are applicable to all MSWLFs that commenced construction, reconstruction, or modification on or after May 30, 1991. The NSPS are promulgated under 40 CFR §60, Subpart WWW. MSWLFs that commenced construction, reconstruction, or modification after July 17, 2014 must also meet the additional standards promulgated under 40 CFR §60, Subpart XXX. In general, the NSPS describe the following:

- Whether an MSWLF is required to install a GCCS and conduct SEM
- Design of the GCCS and the parts of the MSWLF that the GCCS needs to collect gas from
- Standards for operating and monitoring the GCCS
- Standards for conducting SEM
- Timeline of GCCS operation termination
- Approved test methods for meeting numerical standards
- Provisions and deadlines to remain in compliance if standards are not met
- Reporting and recordkeeping requirements

Under 40 CFR §60.752 (b)(2)(v), the NSPS includes specific provisions for when the GCCS can be capped, removed, or decommissioned. For this to occur, all the following conditions must be met:

- The MSWLF must be closed.
- The GCCS must have been in operation for at least 15 years (or shown that it would be unable to operate in 15 years due to declining gas flow if meeting the requirements of Subpart XXX).
- The non-methane organic compound (NMOC) emission rate must be less than 50 megagrams per year (or 34 megagrams per year if meeting the requirements of Subpart XXX) on three successive test dates. Tests entail collecting and analyzing samples of LFG to determine the concentration of NMOCs.

NESHAP for MSWLFs are promulgated under 40 CFR §63, Subpart AAAA, and are applicable to MSWLFs that accepted waste after November 8, 1987, or that have additional capacity to accept waste and meet any one of the following:

- Are considered a major source or are collocated with a major source of hazardous air pollutants (HAPs)
- Are required to follow NSPS
- Are operated as or including a bioreactor that has a design capacity greater or equal to 2.5 million megagrams and 2.5 million cubic meters of waste, and that was not permanently closed as of January 16, 2003

As generally described in §63, a major source is a stationary source that annually emits ten or more tons of any individual HAP or 25 or more tons of any combination of HAPs. A full list of the 187

HAPs is included in Section 112(b) of the Clean Air Act. A bioreactor landfill is defined (§63.1990) as "a MSWLF where any liquid other than leachate (leachate includes landfill gas condensate) is added in a controlled fashion into the waste mass (often in combination with recirculating leachate) to reach a minimum average moisture content of at least 40 percent by weight to accelerate or enhance the anaerobic (without oxygen) biodegradation of the waste."

The MSWLFs that must meet the requirements of NSPS would typically also need to meet the requirements of Subpart AAAA. However, Subpart AAAA also requires that MSWLFs operated as or including a bioreactor with a design capacity greater or equal to 2.5 million megagrams and 2.5 million cubic meters of waste (and which were not permanently closed as of January 16, 2003) to also comply with NSPS, irrespective of NMOCs emissions.

Besides requiring MSWLFs operated as or including a bioreactor to comply with NSPS, Subpart AAAA also requires the development of a Startup, Shutdown, and Malfunction Plan for the gas control system for all MSWLFs that must meet the requirements of NSPS.

As described in §63.1992(b), MSW bioreactor landfills do not need to comply with Subpart AAAA if they are a closed landfill, have permanently ceased adding liquids to the bioreactor and have not added liquids to the bioreactor for at least one year. The owners or operators of MSWLFs are not required to comply with the requirements of Subpart AAAA once they are no longer required to apply controls as specified in 40 CFR §60.752(b)(2)(v) of Subpart WWW (§63.1950).

2.6. Existing Guidance on PCC Period Evaluation

2.6.1. Overview

PCC at MSWLF sites is conducted according to the general conditions specified in §258.61, which include operating, maintenance and monitoring requirements for the final cover, leachate collection, groundwater monitoring, and gas monitoring systems. The regulatory default PCC period is 30 years. However, once the MSWLF owner or operator demonstrates that the site no longer poses a significant threat to HHE, the owner or operator can present a request for a reduction in the frequency of PCC monitoring and maintenance activities, or for early PCC termination, to an approved State Director. The 30-year period included in §258 was adopted from the Subtitle C PCC requirement that was based on an EPA estimate that "…it might take as long as 30 years for material leaching from hazardous wastes to migrate to groundwater…" (ITRC 2006). EPA (2017a) recently published a guidance document to evaluate the performance of Subtitle C landfills under PCC.

While the federal regulations allow the reduction (or extension) of the PCC period if the owner or operator can demonstrate the protection of HHE, detailed processes and specific criteria/provisions that can be used to make such demonstrations are not provided in the federal regulations. The Association of State and Territorial Solid Waste Management Officials (ASTSWMO) conducted a survey in 2011 to gather information on states' policies and/or regulations with respect to post-closure requirements beyond 30 years for Subtitle C and D landfills (ASTSWMO 2013). The survey showed considerable variation among the SEAs regarding the approaches, processes, and procedures that could be used to adjust the PCC period. ASTSWMO recommended that EPA develop guidance on the review and consideration for adjusting the PCC period for MSWLFs. The ASTSWMO survey data suggested a lack of experience pertaining to adjusting or concluding the PCC period; only three out of 26 SEAs that responded to the survey had an MSWLF that had been in PCC for at least 30 years. Only fiveSEAs reported having established specific and two SEAS

reported having mandatory criteria in the state regulations that may be used for extending and reducing the PCC period, respectively. Nine additional states reported publishing specific PCC period evaluation criteria as guidance, policy, or other types of documents.

The Environmental Research and Education Foundation (EREF 2006) and the Interstate Technology and Regulatory Council (ITRC 2006) have proposed an iterative and modular performance-based methodology (also referred to as functional stability approach) to evaluate closed MSWLFs. The ITRC (2006) approach appears to be similar to that of EREF (2006). The approach entails a modular assessment of the leachate collection and control system, the GCCS, the groundwater monitoring system, and the cap system; and the associated pathways that these systems can impact HHE. The impacts to HHE associated with a less-frequent maintenance and monitoring program (or even termination) of each of these four systems are sequentially evaluated to assess if the proposed modifications provide sufficient protection to HHE. Following the implementation of the change(s), the approach calls for confirmatory and surveillance monitoring to verify that such modifications do not result in an inadvertent risk to HHE. This approach of incremental reduction in PCC monitoring program does not rely on a complete stabilization of waste and assume that some de minimis level of control such as maintenance of the final cover would continue beyond the PCC to manage HHE impacts at the point of exposure.

The complete stabilization approach (also referred to as organic stability approach) of long-term landfill management, on the other hand, necessitates monitoring of the landfill until it is completely stable with respect to chemical, biological, and physical characteristics containment system (Morris and Barlaz 2011). Once a landfill is completely stable, a failure of the containment system would not result in HHE impacts. Based on an analysis of the monitoring data from a closed MSWLF in NY, O'Donnell et al. (2018) reported that the organic stability approach would require longer LFG and leachate management period than the functional stability approach. Morris and Barlaz (2011) indicated that this approach might not be practical for managing closed landfills due to factors such as the lack of a mechanism for ammonia transformation in the anaerobic environment of the landfill and the potential presence of trace organic compounds.

Published guidelines for the termination of PCC at MSWLFs for eight states were publicly available as of 2018. These state guidance documents are summarized in the following sections, arranged by the environmental containment systems most commonly evaluated during PCC, to provide a sense of the type and specificity of the available guidelines. In general, these guidelines provide criteria pertaining to monitoring data, duration and frequency of data collection, and trends needed for PCC reduction or termination application.

2.6.2. Leachate Management

Five of the eight state guidance documents identified have criteria for evaluating leachate quantity (FDEP 2016; VDEQ 2007; UDSHW 2012; WADOC 2011; WDEQ 2000). Three of these specify no leachate generation or leachate generation at a historically low rate as a criterion for PCC termination (FDEP 2016; UDSHW 2012; WDEQ 2000). One state's guidance advises that leachate should not be produced for five years prior to petitioning to end PCC (WDEQ 2000). Most guidance documents suggest the review of historical leachate collection records for proposing the termination of PCC. Three state guidance documents suggest the review of biological and chemical oxygen demand in the leachate (FDEP 2016; IDNR 2016; KDHE 2014). Two of these state documents also recommend the review of ammonia and total suspended solids leachate concentrations (IDNR 2016; KDHE 2014).

The duration of leachate quality records recommended for inclusion in the application to terminate PCC varies among the guidance documents. Three of the eight states do not specify a leachate monitoring duration in their PCC termination guidance (VDEQ 2007; WADOC 2011; WDEQ 2000). The state with the most extensive leachate quality records review recommends the review of three consecutive five-year demonstration periods that show key parameters of leachate quality to be in equilibrium or decreasing (KDHE 2014). One state suggests ten years (three years of data before closure and seven years of data after closure) of annual sampling records before PCC termination (FDEP 2016). Two state guidance documents recommend presenting at least five years of monitoring data (IDNR 2016; UDSHW 2012), and one of these documents also recommends an additional five years of monitoring records if the leachate collection system is decommissioned before terminating PCC (IDNR 2016). Two state guidance documents do not provide a specific leachate quality record duration (VDEQ 2007; WDEQ 2000).

2.6.3. Landfill Gas Management

Similar to the guidance on leachate monitoring, the duration and frequency of landfill gas measurement suggested for the PCC termination petition varies among the state guidance documents. Three of the eight guidance documents reviewed list the gas monitoring timelines the same as for leachate characterization (FDEP 2016; IDNR 2016; KDHE 2014). One state guidance document recommends the review of five years of gas monitoring data (VTANR 2013), while another suggests eight consecutive sampling events (WADOC 2011). One state's guidance provides the option of including either twelve consecutive months or three consecutive years of quarterly methane monitoring data in the petition for PCC termination (WDEQ 2000). Two states do not specify gas monitoring data periods in their PCC termination guidance (VDEQ 2007; UDSHW 2012).

Seven of the eight state guidance documents reviewed suggest that the MSWLF operator provides evidence that the landfill gas generation rate has either stabilized or shown a declining trend in methane production (FDEP 2016; IDNR 2016; KDHE 2014; UDSHW 2012; VTANR 2013; WADOC 2011; WDEQ 2000). One of these state guidance documents specifically suggests achieving methane production that is below 10% of the peak rate flow rate or achieving remaining methane generation potential of less than 10% of the total generation potential before considering changes to the active LFG system (FDEP 2016). Five documents also mention that methane concentrations in structures built on the landfill site be below 25% of the lower explosive limit of methane to avoid explosion and toxicity risks (VDEQ 2007; UDSHW 2012; VTANR 2013; WADOC 2011; WDEQ 2000).

2.6.4. Groundwater Monitoring

MSWLF operators attempting to reduce the frequency of or termination of PCC should show a history of compliance with groundwater quality standards when petitioning to the approved State Director. In the collected guidance documents for eight states, three states recommend that the operator provides five years of groundwater monitoring records after closure (FDEP 2016; IDNR 2016; VTANR 2013), and one additional state recommends a three-year monitoring period (VDEQ 2007). The remaining four states do not specify a timeline, though one state's guidance suggests the MSWLF operator can terminate groundwater monitoring once records of environmental monitoring and control systems have demonstrated the facility closure is protective of HHE (WDEQ 2000). All state guidance reviewed suggests no significant risk to HHE should be present at the site, and parameters of concern must be below existing standards. One state guidance

document specifies that concentrations should be less than one half the state's groundwater protection standards (FDEP 2016). Four other state guidance documents recommend that a decreasing trend in parameter concentrations should be observed (IDNR 2016; VDEQ 2007; VTANR 2013; WADOC 2011). Based on a national survey, ASTSWMO (2013) reported that 25 states (out of 27 states responded to survey) do not allow termination of PCC if corrective action is still necessary at the site.

2.6.5. Final Cover Management and Maintenance

The final cover integrity and performance would be dependent on the magnitude of its settlement and differential settlement. Excessive differential settlement may result in final cover irregularities that can enhance its erosion and/or cause stormwater ponding, which would increase the potential for moisture percolation into the landfill. Of the eight state guidance documents describing the content of applications for reducing or terminating PCC, one state recommends including ten years of settlement monitoring data (FDEP 2016). Three other states suggest collecting and analyzing settlement data for at least five years (IDNR 2016; UDSHW 2012; WDEQ 2000). Four state guidance documents specify a qualitative criterion of very low or negligible final cover settlement rate for PCC termination (IDNR 2016; UDSHW 2012; VTANR 2013; WADOC 2011). FDEP (2016) recommends achieving an annual settlement rate of less than 5% of the total post-closure settlement before reducing or eliminating the final cover maintenance. Additional final cover considerations include qualitative assessments of vegetation and cover membrane integrity (FDEP 2016; WDEQ 2000).

3. Closed Landfill Case Studies

3.1. Study Sites Selection Process

The following data sources were used to identify closed MSWLF:

1. Greenhouse gas reporting database. The website https://www.epa.gov/ghgreporting lists all the MSWLF sites in the US that reported GHG estimates to EPA. Based on the closure date listed in this database, approximately 300 MSWLFs were closed from 1993 through 2013.
2. State environmental agencies (SEAs) websites. Several SEAs maintain a list of closed MSWLF sites. For example, the Florida Department of Environmental Protection (FDEP 2018) and the Massachusetts Department of Environmental Protection (MDEP 2018) provide a list of all the active and inactive MSWLF sites along with details such as contact information, amount of waste deposited, and closure year.
3. EPA regional offices, facility engineer contacts, and institutional knowledge of project team members.

The scope of the study was limited to the evaluation of the data from only nine closed sites due to time constraints. The following criteria were used for selecting nine sites (referred herein to as the study sites) with closed MSWLF cells from over 300 closed MSWLFs identified from the sources listed above. The site search was ended once nine sites meeting the following criteria were identified:

1. **Active GCCS** – Only sites with active GCCS were selected for the study.
2. **Leachate collection and quality data** – Sites with routinely-tracked leachate collection rates and chemical characterization data were selected.
3. **Geographic location** – Due to variation in the average annual precipitation, temperature, and evapotranspiration rates, the geographic location of a landfill is expected to have a significant impact on leachate and gas generation rates. A 2017 EPA study of Subtitle C landfills assessed sites located in four geographical regions of the US: Northeast, Northwest, Southeast, and Southwest (EPA 2017a). An attempt was made to select MSWLF sites from each of these four regions in the same proportion as the regional population to achieve an approximate regional representation.
4. **Liner Type** –An evaluation of the long-term leachate generation rate was one of the primary objectives of this study. The leachate collection rate from the sites with an impervious liner component are expected to be more representative of the actual leachate generation rate than the sites without a geomembrane liner. Therefore, only sites/cells with a bottom liner configuration that included a geomembrane were selected.
5. **PCC Data Duration** – Initially, sites closed before 2005 were targeted for selecting sites with at least ten years of PCC data as long-term data are vital for meaningful analysis of trends over time. However, due to scarcity of closed MSWLFs meeting all of the above criteria, sites closed before 2010 (with at least five years of PCC data) were also considered for the study.

3.2. Data Collected

For each of the nine study sites selected for detailed evaluation, information was gathered on the design of the bottom liner and final (or intermediate) cover system of the cell(s) of interest. The data pertaining to the hydrogeography and topography of the site were also collected. These data were typically available through site construction permit applications, site environmental impact assessments, and construction quality assurance documentation.

Table 3-1 presents a summary of the environmental monitoring and operational data that were collected and reviewed for each site. An "X" indicates that a substantially complete dataset was obtained, a "P" indicates that the dataset was extremely limited or missing key pieces of information, and an "N/A" indicates that the dataset is not applicable to the site. It should be noted that MSWLFs commonly accept non-MSW materials (beyond commercial or residential MSW) that may pose a different level of HHE risk than MSW (e.g., yard waste, some industrial wastes, land clearing debris, waste-to-energy or coal ash). An understanding of the amounts and nature of waste materials such as industrial waste and contaminated soils deposited in the landfill is valuable for HHE impact assessment. Waste-specific (e.g., MSW, industrial waste, construction and demolition debris) disposal tonnages were available for several study sites. However, detailed waste characterization data (e.g., plastics, paper, food waste, household hazardous waste etc.) estimated based on composition studies were available only for one site (Table 3-1).

The cell-specific data were analyzed where available. The data from all the cells were used for the analysis presented in the report for Site H as all the cells at the site met the liner criteria listed in Section 3.1. For Site C, the leachate collection rate and quality for Cells C1 and C2 were individually available for analysis, whereas only cumulative LFG flow rate and composition data were available from these cells. The leachate flow rate and composition were for individual cells C1 and C3 were analyzed. Only the collective LFG data for these were analyzed. Similarly, leachate and LFG data from the entire site, including the cells that did not meet the liner criterion, were analyzed for Site E as the study-cell-specific data were not available. For Site G, leachate collection rate and post-closure cost data were available and analyzed for individual study cells (G3 and G3). However, only cumulative LFG flow rate data were available from several cells (including G3 and G4) and used for the analysis for Site G. Table 3-2 summarizes the data available/analyzed for each site. More details about the available data and corresponding cells are presented in the report chapters and Attachments A-I.

Table 3-1. Environmental Monitoring and Operational Data Collected for Each Study Site

Data Type		Site								
Category	**Subcategory**	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**
Landfill Gas	Landfill Gas Flow	X	X	X	X	X	X	X	X	No
	Landfill Gas Composition	X	X	X	X	X	X	X	P	No
	Surface Emissions Monitoring	No	No	P	X	X	X	No	X	X
	Methane Monitoring Probes	X	X	No	X	X	X	No	X	X
Leachate	Collection Rate	X	X	X	X	X	X	X	X	X
	Quality	X	X	X	X	X	X	P	X	X
	Recirculation Quantity	X	N/A	X	N/A	X	N/A	N/A	N/A	No
Groundwater	Monitoring Wells Locations and Groundwater Quality Data	X	X	X	X	X	X	X	X	X
Final Cover	Topographic Survey or Settlement Data	No	P	X	No	P	P	No	No	X
	Maintenance Issues	P	P	P	X	X	X	No	No	X
Operations	Disposal Quantity	X	P	X	X	X	X	P	X	X
	Waste Composition	No	No	No	X	No	No	No	No	No
Post-Closure Care	Actual Costs	No	X	No	X	X	No	No	No	No

Notes:

X- a substantially complete dataset was obtained;
P- dataset is extremely limited or missing key pieces of information;
N/A- dataset is not applicable to the site.
No – dataset not available

Table 3-2. Environmental Monitoring and Operational Data Analyzed for Each Study Site

Data Type		Site								
Category	**Subcategory**	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**
Landfill Gas	Landfill Gas Flow	S	S	C	S	S	S	S	C	S
	Landfill Gas Composition	S	S	C	S	S	S	S	C	S
	Surface Emissions Monitoring	No	No	C	S	S	C	No	C	C
	Methane Monitoring Probes	S	S	No	S	S	S	No	C	C
Leachate	Collection Rate	C	S	C	C	S	C	C	C	C
	Quality	C	S	C	C	S	C	S	C	C
	Recirculation Quantity	C	N/A	C	N/A	S	N/A	N/A	N/A	No
Groundwater	Monitoring Wells Locations and Groundwater Quality Data	S	S	C	C	S	C	C	C	C
Final Cover	Topographic Survey or Settlement Data	No	P	C	No	P	P	No	No	C
	Maintenance Issues	P	P	C	C	P	C	No	No	C
Operations	Disposal Quantity	C	C	C	C	C	C	C	C	C
	Waste Composition	No	No	No	C	C	No	No	No	No
Post-Closure Care	Costs	S	S	S	S	S	S	C	C	C

Notes:

S- sitewide/multiple cell data were analyzed as the study cell-specific data were not available;
C- study cell-specific data were available and analyzed;
P- dataset is extremely limited or missing key pieces of information;
N/A- dataset is not applicable to the site.
No – dataset not available

3.4. Descriptions of Selected Sites

The area of the cell(s) selected from each site ranged from 6–69 acres, and five of the sites had standalone lined cells (i.e., the lined cell(s) was not piggybacked over an unlined cell(s)). Additional information on the capacity, types of waste accepted (i.e., in addition to MSW), design of the final cover and bottom liner system, annual precipitation, and annual rainfall for each of the cell(s) is provided in Table 3-3. Five of the study sites are publicly owned and operated, and the rest four are privately owned and operated.

Material-specific disposal amounts were available for several sites (e.g., Site G, E, and I). The materials listed in the Table 3-3 are major waste categories that were accepted at each of the study sites as explicitly described in the site documents reviewed – they are not intended to be an exhaustive list of the materials received at each site. The "composite liner" description, unless otherwise noted, refers to a 60-mil high-density polyethylene (HDPE) geomembrane (GM) in direct contact with an underlying natural or geosynthetic clay layer.

Annual precipitation data was derived from the closest weather station to each site – the values represent an average of annual station precipitation data collected from 1981 to 2010 (NOAA 2017 http://bit.ly/1LFSEqM). On average, each site was approximately 5 miles from the closest weather station; Site A was the furthest at approximately 12 miles from the nearest station.

Table 3-4 presents an operational summary for the studied cell(s) at each of the sites. At the time of the development of this report, two sites (i.e., Site F and Site I) have been in PCC for 20 years. All of the sites have or have had a GCCS and, except for two of the sites (i.e., Site A and Site G), GCCS was operating at the time of this study. It should be noted that the actual start date for gas collection at Site G is unknown – the start year of 2008 listed in Table 3-1 represents the commencement of the landfill gas-to-electricity (LFGTE) system at this site. Four of the sites have had LFGTE projects, and these projects are still ongoing at two of these sites. A fifth site (i.e., Site D) is beneficially using landfill gas in a direct thermal application as a process fuel at a nearby chemical and pharmaceutical facility.

Three of the sites reported recirculating leachate with the annual average rate ranging from 6 to 1,009 GPAD (annual leachate recirculation volumes were divided by 365 and the corresponding cell area). Two of the three sites conducted leachate recirculation during the PCC period. For sites with two studied cells that recorded cell-specific leachate data (i.e., Site C and Site G), the range of values presented in the table represents the minimum and maximum values for both cells. Additional details on leachate collection rate trends are discussed in Section 6.

Each study site's construction details including timeframe, liner construction, site's hydrogeology, the final cover system details, waste placement and composition, and monitoring details including, groundwater monitoring, leachate collection rate and quality monitoring, landfill gas monitoring, the final cover maintenance, landfill settlement, and PCC cost were summarized. Individual site-specific summaries of all the studied cells at each site are presented in Attachments A to I.

Table 3-3. Design Summary for the Studied Cell(s) at Each Site

Site	A	B	C	D	E	F	G	H	I
US Region	Southeast	Northwest	Northeast	Southeast	Northeast	Southwest	Northeast	Northeast	Northeast
Ownership	Public	Private	Public	Public	Public	Private	Public	Private	Private
Area of Study Cell(s) (acres)	28	20	40.8	69	6	38	15	60	51
Standalone Lined Cell(s)	No	No	Yes	Yes	No	Yes	No	Yes	Yes
Waste Quantity in Study Cell(s) (million MT)	1.20	0.88	1.40	4.40	1.38	0.79	0.63	3.46	3.48
Site Capacity (million MT)	2.05	3.27	Not available	5.56	3.97	1.85	1.64	3.46	Not available
Additional Wastes Accepted	Additional waste info not available	Industrial, CDD, coal ash, sewage sludge, asbestos	Additional waste info not available	Industrial, CDD, land clearing debris	Pulp and paper mill waste, industrial, sludges, utility ash, foundry waste, treated contaminated soil	Additional waste info not available	Petroleum-contaminated soil, wastewater treatment sludge, whey, ash	Sludge, CDD, non-hazardous industrial waste, ash, asbestos	Alternative daily cover, special wastes, contaminated soil, CDD, sludge, asbestos, ash
Study Cell(s) Final Cover (layers are listed from the bottom to the top)	40-mil LLDPE geomembrane (GM) with 18-inch granular drainage and 6-inch topsoil layers	60-mil HDPE GM with geocomposite drainage, 18-inch soil, and 6-inch topsoil layers	Exposed 35-mil scrim-reinforced polypropylene GM	40-mil LLDPE GM with geocomposite drainage, 18-inch protective soil, and 6-inch vegetative soil layers	40-mil VLDPE GM with 12-inch sand drainage, 18-inch soil, and 6-inch topsoil layers	40-mil HDPE GM with geonet drainage (side slopes). 40-mil HDPE GM with GCL (top deck); GM covered with 12-inch vegetative soil layers	Cell G3: 40-mil VLDPE GM overlain by geogrid, 24-inch protective soil, and 6-inch topsoil layers Cell G4: GCL, 40-mil VLDPE, geotextile, 24-inch protective soil, and 6-inch topsoil layers	30-mil PVC/40-mil HDPE GM with geonet drainage/geocomposite, geotextile, and 24-inch vegetative layers	40-mil PE GM with geonet drainage, geotextile, 18-inch granular cover, and 4-inch vegetative layers
Study Cell(s) Bottom Liner	Composite with 1 ft of 10^{-7} cm/s clay	Composite with 2 ft of 10^{-7} cm/s clay	Double 30-mil PVC (primary and secondary) with 12–18-inch secondary drainage layer	Composite with 2 ft of 10^{-7} cm/s clay	Composite with 4 ft of 10^{-7} cm/s clay	Phase I: 60-mil HDPE GM Phase II: Composite with 1 ft of 10^{-6} cm/s clay Phase III: Composite with 2 ft of 10^{-7} cm/s clay	Cell G3 (part): 36-mil hypalon GM with secondary collection layer over 2 ft of 10^{-7} cm/s clay of secondary liner Cell G4: Double composite with primary GM underlain by an intermediate barrier layer of 6 inches of 10^{-7} cm/s clay overlaying 1 ft of 10^{-5} cm/s clay over geotextile over geonet over secondary geomembrane over 2 ft of 10^{-7} cm/s clay	Double liner system with geonet between the primary and secondary GMs. Cells 1-7 secondary GM underlain with 6 inches of 10^{-5} cm/s (max) soil overlying 5-ft compacted base. Cell 8 secondary GM underlain by GCL.	Double composite liner with primary 60-mil HDPE underlain by GCL and secondary GM underlain by clay subbase. Primary and secondary drainage layer consists of a sand drainage layer, geotextile, and a geonet from top to bottom.
Average Annual Precipitation (inches)	51	56	47	46	30	22	46	47	51
Approximate Groundwater Depth (ft)	25–75	140–260	1–8	7–40	5–40	12–26	12–65	2–13	0–55

Table 3-4. Operational Summary for the Studied Cell(s) at Each Site

Site	A	B	C	D	E	F	G	H	I
Study Cell(s) Start–Closure Year	1988–1998	1996–2004	1984–1998	1996–2009	1997–2004	1989–1997	1987–2000	1991–2008	1990–1997
Years since Closure (as of 2016)	18	12	18	7	12	19	16-23	8	19
Gas Collection Start–Stop Year	1999–2011	1991–N/A	1994–N/A	1999–N/A	1991–N/A	1999–N/A	2008[a]–2012	1996–N/A	1994–N/A
Landfill Gas (or Methane) Collection Rate and Composition Data Available	1999-2011	2007-2016	1995-2015	2009-2016	1997-2016	1999-2015	2008-2012	2005-2017	Not Available
Observed Methane Collection Rate (standard cubic feet per minute)	12–924	40–302	268–798	515–1333	63–924	36–367	0–104	328–1218	Not Available
Landfill Gas-to-Electricity Project Start–Stop Year	2002–2005	N/A	2007–N/A	N/A	2009–N/A	N/A	2008-Unknown	N/A	N/A
Leachate Recirculation Start–Stop Year	2003–N/A	N/A	1986–1994	N/A	2011–N/A	N/A	N/A	N/A	N/A
Post-Closure Leachate Collection Rate Range (gallons per acre per day)	320–1009	49–180	25–183	3–21	118–427	14–64	38–2,070	78–302	56–122
Leachate Recirculation Rate Range (gallons per acre per day)	89–1009	N/A	6–245	N/A	0–183	N/A	N/A	N/A	N/A

Notes:

N/A- dataset is not applicable to the site.

[a] Start date of GCCS not known. LFGTE (Landfill Gas-to-Electricity) project commenced in 2008.

3.5. Data Presentation

Throughout this report, the pertinent data are summarized using box-and-whisker plots. This type of plot provides a visual portrayal of the statistical distribution of the data. Figure 3-1 presents a definition sketch of the box-and-whisker plot. The top, middle, and bottom of the box represent the 75th, 50th (i.e., the median), and 25th percentiles, respectively. The lines that extend upward and downward (whiskers) from the box represent the 90th and 10th percentiles, respectively. The values less than the 10th percentile or more than the 90th percentile are presented individually outside the whiskers. These plots were prepared using Sigmaplot 11 (Systat Software, Inc) or Excel (Microsoft Corporation) software.

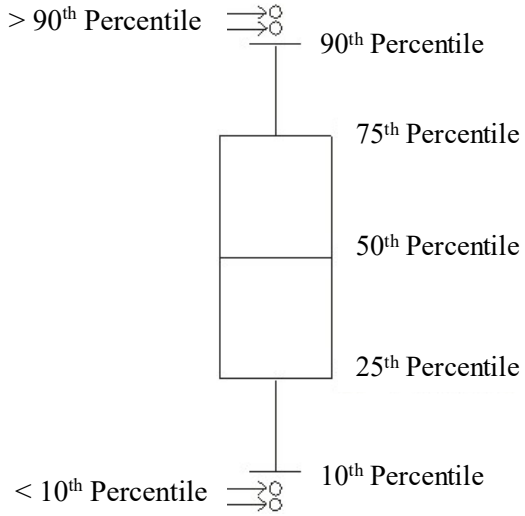


Figure 3-1. Box-and-Whisker Plot Definition Sketch

4. Post-Closure Care Cost

4.1. Overview

The termination or extension of the PCC period may have considerable financial implications for the site owners. An understanding of the cost for various operations, maintenance, and monitoring activities helps in the identification of the major cost centers. The site owners may consider prioritizing the assessment of HHE impacts associated with the termination of cost-intensive activities. As a means of assessing the financial impact of MSWLF maintenance and monitoring beyond the 30-year PCC period, available PCC cost data for all the nine study sites were evaluated. PCC costs were organized into the following six major cost categories:

(1) leachate management including monitoring and treatment,
(2) GCCS management,
(3) final cover maintenance including revegetation, mowing, and cover soil and runoff control device maintenance,
(4) groundwater and subsurface gas monitoring and maintenance,
(5) administrative expenditures (e.g., permitting), and
(6) other expenditures (e.g., utilities, surface water monitoring and maintenance, site security).

Available PCC cost data had limitations such as unavailability of actual PCC cost data for each year since the closure, and lack of cost data exclusive to the studied cell(s). Furthermore, as discussed later in this section, the available data suggest that the cost varied over a wide range due to several factors, including inconsistencies in cost categorization, mixed availability of actual cost and cost estimates, and the necessity for occasional capital-intensive system upgrades. Because of these limitations, landfill owners and engineers should consider tracking and using the site-specific cost data for evaluating the financial impacts of PCC activities instead of using the data presented in the report as proxies.

A summary of the availability of PCC cost data for the study sites is presented in Table 4-1. Among the nine study sites, the actual costs associated with PCC activities were available only for three sites (Sites B, D, and E) for a limited number of years. For the other sites, the available PCC cost data were estimated values. For Sites H and I, the available PCC cost data were specific to the study cell(s). Site G provided estimated PCC costs for the entire site, and the estimated cost for individual cells (G3 and G4) calculated based on the design capacity of each cell and total sitewide PCC cost. Site A estimated PCC cost data for was available for years 2, 6, and 13 after closure; however, each of these year's cost represented different waste footprint area of the site (28, 69, and 84 acres, respectively), including the study cell(s). For other sites, the available PCC cost data was associated with the maintenance of other site features (e.g., groundwater monitoring, subsurface gas monitoring, cover maintenance) that may not necessarily be exclusive to the study cell(s).

Table 4-1. Summary of Availability of PCC Cost Data from the Nine Study Sites

Site	A	B	C	D	E	F	G		H	I[2]
							Cell G3	Cell G4		
Closure Year	1998	2004	1998	2009	2004	1997	1993	2000	2008	1997
Data Available for Years after Closure	2, 6, 15	8-11	14	3-7	0-11	2, 4, 6, 15, 18	8, 12, 15-17	1, 5, 8-10	0-30[1]	0-1, 3, 5-12, 14, 16
Actual Post-Closure Care Cost		✓		✓	✓					
Exclusive to Study Cell(s)							✓	✓	✓	✓
Representing Area (Acre)	28, 69, 84[3]	65	127	110.5	61	105	7.1	7.9	60	51

Notes:

(1) Site H data included a cost estimate of 31 years (closure year and 30 years after closure).

(2) For Site I, the cost data were included only for the years in which new estimates were available.

(3) For Site A, PCC cost of year 2, 6, and 15 after closure represented the waste footprint of 28, 69, and 84 acres, respectively.

Figure 4-1 shows the distribution of available annual PCC costs per unit acre waste footprint for each site. For consistency, the available PCC cost at each site was adjusted to 2017 dollars based on consumer price index values (BLS 2018). Figure 4-1 shows only one data point for Site C and Site H. As shown in Table 4-1, Site C has only one year of data available. Site H provided a summary of total projected expenditures for 31 years of PCC (including the closing year and 30 PCC years); the total PCC cost was divided by 31 to obtain the average annual PCC cost.

Annual PCC costs for the cells studied at Sites G, H, and I varied over a wide range from approximately $1,200 to $11,000 per acre, with an average of the cost of approximately $5,300 per acre. For the other sites that included PCC cost for the entire site, including the study cell(s), the annual PCC costs ranged between approximately $1,550 to $37,000 per acre, with an average cost of approximately $6,450 per acre and a median cost of $3,900 per acre. This wide range is due to many factors, including inconsistencies in cost categorization, mixed availability of actual cost and cost estimates, and the necessity for occasional capital-intensive system upgrades. For example, this range includes one year of high cost for Site E for constructing a sewer connection to pump leachate directly to the local wastewater treatment plant (WWTP) (i.e., Year 2 of PCC). As a point of comparison, Morris and Barlaz (2011) estimated annual PCC cost for a hypothetical MSWLF in PCC years 10 to 30 to be approximately $2,000 per acre based on 2009 dollars, which corresponds to approximately $2,300 per acre in 2017 dollars.

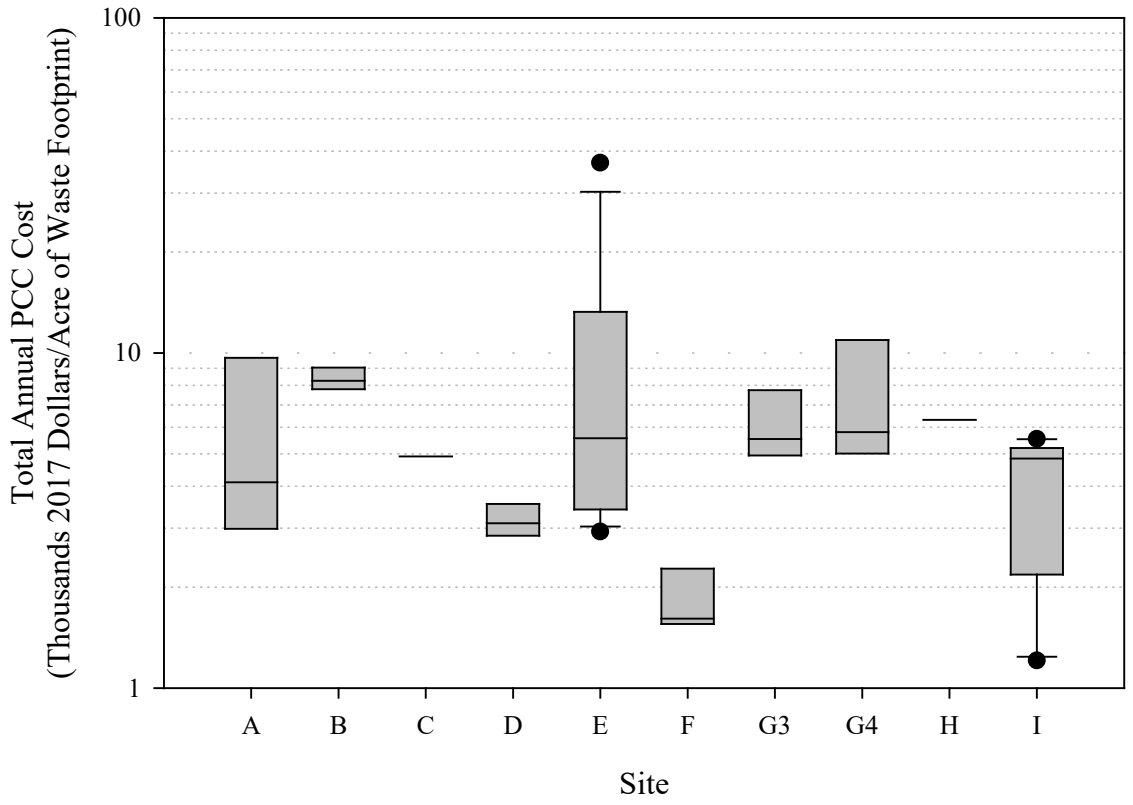


Figure 4-1. Distribution of Total Annual PCC Cost at Each Study Site

Figure 4-2 presents the average percent contribution of different cost categories for the cell(s) at each of the study sites based on the available data. The wide variation in the cost and relative fraction of different activities among the sites is due to inconsistent cost accounting practices used for tracking the cost. For example, a couple of sites (e.g., Sites D and E) tracked leachate monitoring and management cost separately, whereas others likely included it in the total leachate management cost. GCCS management costs were not available for Sites C and G. Also, Site G documents presenting PCC cost data did not have an 'other' cost category. Cover maintenance cost was not separately available for Site H, and administrative support costs were not separately available for Site E and are not included in Figure 4-2.

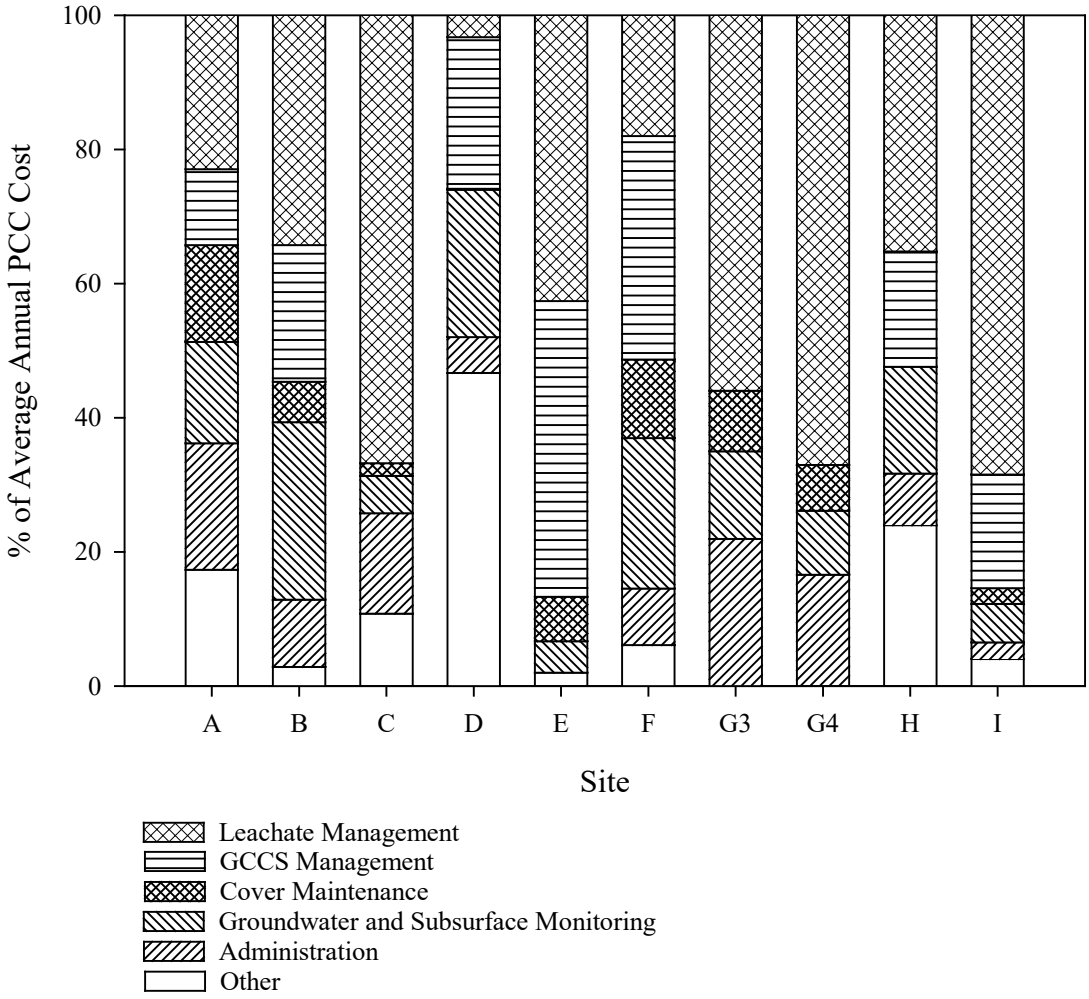


Figure 4-2. Average Annual Cost Distribution at Each Study Site/Cell After Closure

Overall, leachate management costs during PCC ranged between approximately 3% and 68% of the total average annual PCC cost among the studied sites. Relative to other PCC activities, leachate management appeared to be one of the most cost-intensive activities for all of the study sites except for Sites D, E, and F. Although Sites C and G showed leachate management costs in the range of 56% to 67% of total annual average PCC cost, the absence of GCCS management cost data for these sites potentially inflated the relative fraction of other cost categories including leachate management. In addition, the leachate management cost for both Sites C and G is overestimated as these include the cost of management of leachate from the entire site and not from the study cells.

The leachate management cost at Sites A, H, and I, which treated/pre-treated leachate onsite, constituted approximately 23%, 35%, and 68% of the overall cost, respectively. Site I had an on-site leachate pretreatment plant, and the pretreated leachate is discharged to a local sewer system connected to a nearby WWTP. Site H has an on-site leachate treatment plant that discharges its effluent to a nearby surface water body. Site A treated a fraction of the leachate and recirculated the rest into the landfill. Treated leachate was spray irrigated over an area of the top deck of the Site A study cell.

Site D appears to have a substantially lower average annual PCC costs for leachate management, possibly due to relatively lower leachate collection rates at Site D compared to the other sites, as discussed in Chapter 6. Site D started direct discharge to a WWTP starting a year before closure, the PCC leachate management cost included in this analysis does not include the construction cost of connection to the sewer system, which was completed before closure. Site E constructed a connection to the local sewer system to allow direct discharge of leachate to a WWTP in PCC year 2. The construction cost is included in the leachate management cost, which is a reason for the high leachate management cost at Site E as compared to other sites.

Based on the available data of seven sites, the GCCS management cost ranged from approximately 11% to 44% of the total cost of PCC among the sites. GCCS management represented the highest PCC cost at two sites and the second-highest cost at three sites. As discussed before, GCCS management cost data for Sites C and G were not available. The GCCS management expenditures of Sites B, D, E, and F included the cost associated with GCCS management for the entire site. GCCS management cost data for Site A included the cost associated with the study cell and an unlined cell and that for Site I was exclusive to the closed studied cell at the site.

The reported groundwater and subsurface gas monitoring cost ranged between approximately 5% to 26% of the total average annual PCC cost for the sites where the PCC cost was reported for the entire site. For Sites G, H, and I (for which cell-specific PCC cost data were available), the groundwater and subsurface gas monitoring cost ranged from approximately 6% to 16% of the total average annual PCC cost. The average annual administrative costs and other expenditures among the sites for which the PCC costs were reported for the entire site ranged between approximately 5% to 19% and 2% to 47%, respectively. Annual administrative costs for Site E were not available. For Sites G, H, and I for which cell-specific PCC cost data were available, the average annual administrative and other cost varied in the range of approximately 3% to 22% and 4% to 24%, respectively. PCC costs associated with the 'Other' category for Site G were not available.

Annual final cover maintenance costs ranged between approximately 0.2% to 14% of the total average annual PCC cost for sites where the PCC costs were reported for the entire site. For Sites G and I (for which cell-specific cap maintenance PCC cost data were available), the annual closure cap maintenance expenditures ranged between approximately 2% to 9%. As discussed above, Site H cover maintenance costs were not available.

4.2. Leachate Management Cost

Leachate management cost includes the cost associated with leachate collection system operation and maintenance such as cleaning (when exclusively available), leachate hauling for off-site treatment (if applicable), treatment (on-site and/or off-site), and leachate sampling and analysis (when exclusively available). As discussed earlier, the categories used for leachate management costs were not consistent among the sites and were not necessarily consistent for a site among the years of available cost data.

Post-closure annual leachate management cost was evaluated for six study sites (excluding Site A and C) based on the available data. All the collected leachate was recirculated for a majority of duration after closure at Site A, and the annual sitewide leachate collection rates were not available for Site C for the years leachate management cost was available. As leachate management cost is expected to depend on the leachate collection rate, the annual leachate management costs were

normalized with the annual leachate collection rate for each site for a consistent comparison among the sites. Figure 4-3 shows the distribution of the annual leachate management cost per gallon of leachate collected at each site since closure. The leachate management cost ranged between approximately $5.5 to $219 per 1,000 gallons of leachate collected. On an area scale, annual leachate management cost among the sites (except Sites A and C) ranged from $23 to $25,162 per acre per year, with a median of approximately $2,300 per acre per year. The median annual leachate management cost varied over an order of magnitude among the sites. The annual cost varied over an order of magnitude for Sites D and E. The data suggest that a significant variation in the leachate management cost can occur over time due to various factors including implementation of a capital-intensive project(s), e.g., construction of a sewer connection for pumping leachate to the local WWTP at Site E. In addition, the cost is expected to decline over time as the leachate collection rate decreases.

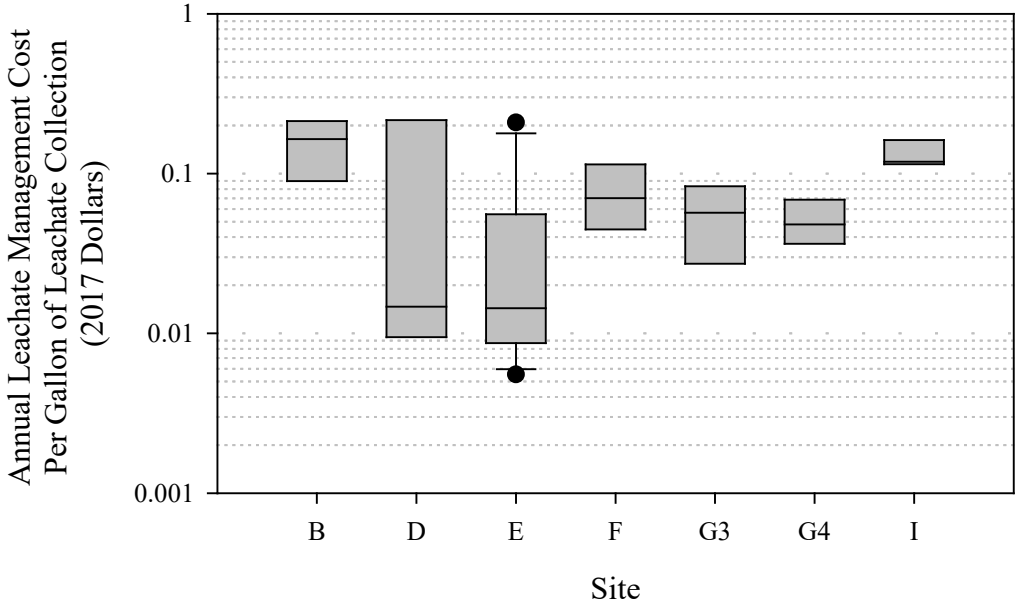


Figure 4-3. Distribution of Leachate Management Cost at Six Study Sites after Closure (Site G Has Two Study Cells)

4.3. GCCS Management Cost

The GCCS management cost among the sites may vary based on factors such as the type of GCCS devices, the number of gas extraction wells, the presence/absence of a landfill gas-to-electricity (LFGTE) project, landfill capacity, the age of waste, etc. Figure 4-4 shows the distribution of annual GCCS management costs per million cubic feet of LFG collected after closure at six of the study sites (excluding Sites C, G, and I). Although GCCS management cost data for PCC Year 15 for Site A, PCC Year 0 and 1 for Site E, PCC Year 15 and 18 for Site F, and several PCC years (as shown in Table 4-1) for Site I were available, these were not included in the data presented in Figure 4-4 due to a lack of the LFG collection rates for these years. As discussed previously, the GCCS management costs at Sites B, D, E, and F were for the entire site and not just for the study cell(s). The annual GCCS management cost among the sites ranged from approximately $26 to $1,270 per million cubic foot of annual LFG collected. As shown in Figure 4-4, GCCS management costs for Site B and E were considerably high compared to the other sites.

Site E had a considerably higher GCCS management cost for PCC years 2 to 5 (i.e., with an average of approximately $449,000 per year) as compared to other PCC years (0 to 1 and 6 to 11) (i.e., with an average of approximately $155,000 per year). The higher cost appears to be related to the expansion of the site's GCCS for LFGTE system installation. Site B GCCS management cost includes the cost of a contractor to operate and maintain the GCCS, which probably is the reason for the elevated cost for this site. The annual GCCS management cost ranged from approximately $2,700 to $593,000 among the study sites (excluding Sites C, G, and I), which is equivalent to $96 to $9,731 per acre per year among the study sites (median of approximately $1,770 per acre per year).

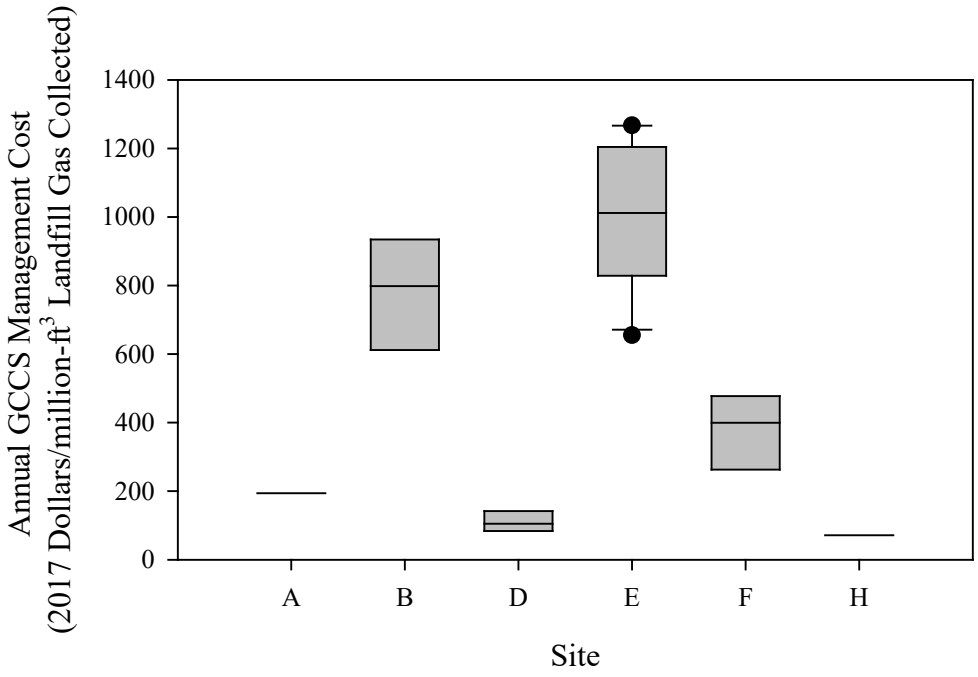


Figure 4-4. Distribution of Annual GCCS Management Cost at Seven Study Sites After Closure

Sites E and I, which had PCC data available for the most number of years (12 and 13 years, respectively), the average annual GCCS management cost was approximately $253,000 and $35,000, respectively. Both the sites (E and I) have an LFGTE system. The commencement of the LFGTE system at Site E substantially reduced the PCC cost associated with power purchase at the site. The average electricity cost before LFGTE startup at the site was approximately $24,000 (average of years 0 to 4 into PCC), which reduced to an average of approximately $1,800 (average of years 5 to 11 into PCC) after the implementation of the LFGTE system.

4.4. Monitoring Cost

The monitoring activities during PCC involve groundwater quality, subsurface gas migration, surface emissions, the final cover integrity, and settlement monitoring. As discussed earlier, leachate and GCCS monitoring costs are included in the leachate and GCCS management cost categories, respectively. The specific PCC cost distribution for each type of monitoring activity was not available. In the current analysis, the available cost associated with groundwater and subsurface gas monitoring were evaluated as one category.

Groundwater and subsurface gas monitoring costs among the sites are expected to vary depending on the number of monitoring locations, frequency of monitoring, and the number of groundwater quality parameters being monitored. Figure 4-5 presents a distribution of annual groundwater and subsurface gas monitoring costs during the PCC period for the study sites. The annual groundwater and subsurface gas monitoring cost among the sites ranged from approximately $2,400 to $169,600, which is equivalent to $40 to $2,600 per acre per year among the study sites (median of approximately $490 per acre per year). It should be noted that the cost presented in Figure 4-5 for the study cells G3 and G4 are specific to individual cells and are associated with the entire site for the other sites. This probably is one of the reasons for the lower cost for G3 and G4 compare to those for the other sites.

Site B had substantially higher monitoring costs as compared to other sites. The average annual monitoring cost at Site B was approximately $144,000, whereas the average of the remaining sites was approximately $27,000. The elevated monitoring costs at Site B might be associated with groundwater impacts observed at the site with respect to the state groundwater quality standards. The Subtitle D regulations (§258.55) and equivalent state regulations require enhanced groundwater monitoring (more parameters and more frequent analysis) in the event of observed groundwater impacts. The site may need to assess and implement corrective measures depending on the nature of impacts. The implementation of enhanced monitoring and corrective measures at Site B likely resulted in increased monitoring costs.

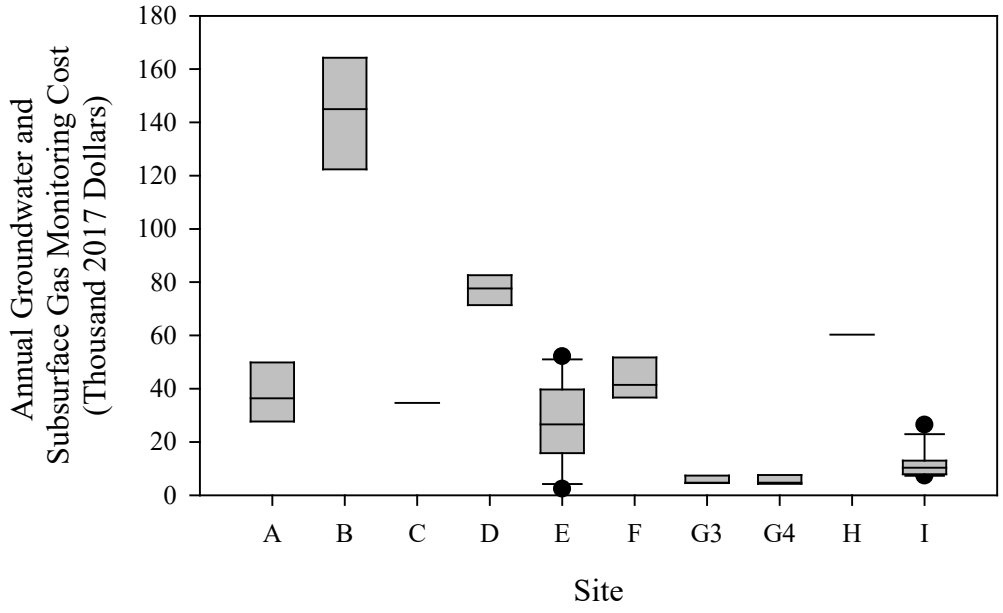


Figure 4-5. Distribution of Annual Groundwater and Subsurface Gas Monitoring Cost After Closure at Each Study Sites

4.5. Final cover Maintenance Cost

The final cover maintenance costs are generally associated with revegetation, mowing, grading to accommodate differential settlement, geomembrane repair, and/or stormwater control device maintenance. Figure 4-6 shows the distribution of annual the final cover maintenance costs per area of waste footprint during PCC at eight of the study sites (Site G had two cells, and data for Site H were not available). Data for the total surface area of the final cover were not available;

therefore, the cover maintenance cost was normalized based on the waste footprint area. Sites G and I reported study cell-specific annual final cover maintenance costs, which ranged from approximately $56 to $595 per acre. For the other sites that had sitewide cover maintenance cost data available, the cover maintenance cost varied in the range of approximately $24 to $1,450 per acre, excluding one year (year 1 after closure) of cover maintenance cost of approximately $3,400 per acre at Site E.

Site E presumably reported cap maintenance cost in a category listed as land surface care cost. The annual land surface care cost for Site E during year 1 after closure ($206,000) was substantially higher than the other years (varied between $2,400 to $83,300). The reasons for high land surface care cost for year 1 after closure appears to be related to regrading (clay spreading). Site C, which had only one year (i.e., Year 14) of PCC costs available, reported an annual site-wide final cover maintenance cost of approximately $92 per acre.

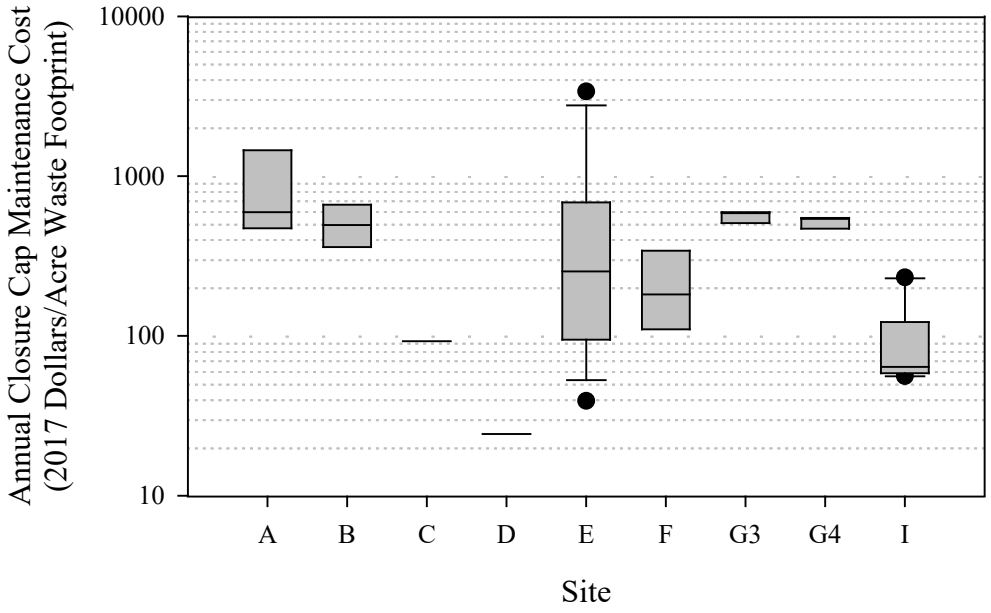


Figure 4-6. Distribution of Annual Final Cover Maintenance Cost During PCC at the Eight Study Sites (Site G has two study cells)

The most common final cover maintenance activities among the sites (excluding Site C) were mowing, regrading, and revegetation of the cover and stormwater swales following erosion or differential settlement. At Site B, erosion in the stormwater control devices and sloughing of cover soil was observed during PCC, which was repaired by the placement of riprap in and around the stormwater control features and by installing trenches embedded in the final cover soil and sloped downward towards a stormwater control ditch.

Cover repairs/maintenance were also performed when SEM exceedances were observed. For example, at Site E, cover soil was excavated and backfilled with compacted clay or bentonite at the location of observed SEM exceedances, and at Site H, additional soil cover was placed. Site I owners/operators reported damage to the geosynthetic cap due to differential settlement. The repair included replacing the boots around the gas wells and exposing and repairing the geosynthetic cap.

As discussed above, Site C has an exposed geomembrane cap, although the cap had been repaired after closure, the details of these repairs were not available.

The final cover maintenance cost is expected to be dependent on the magnitude of differential settlement, which is expected to decline over time. Figure 4-7 shows a distribution of the final cover maintenance costs and the annual volume loss observed at Site I during PCC. The annual volume loss represents the yearly change in in-place waste volumes, which were using the annual topographic data and approximate landfill bottom. During the initial closure years (Years 1 to 5), greater settlement and corresponding higher cap maintenance costs were observed for Site I. The annual settlement volume and the annual final cover maintenance costs decreased substantially following Year 5 of PCC. A decrease in differential settlement potentially reduces the need for regrading and revegetation, thereby reducing the cover maintenance cost. As presented in Table 4-1, for Site I, annual PCC cost estimates were available from a specific PCC year up to 30 years into PCC and in the current analysis, PCC cost data were included only for the years in which new estimates were available. New estimates of cost for PCC years 2, 4, 13, and 15 were not available.

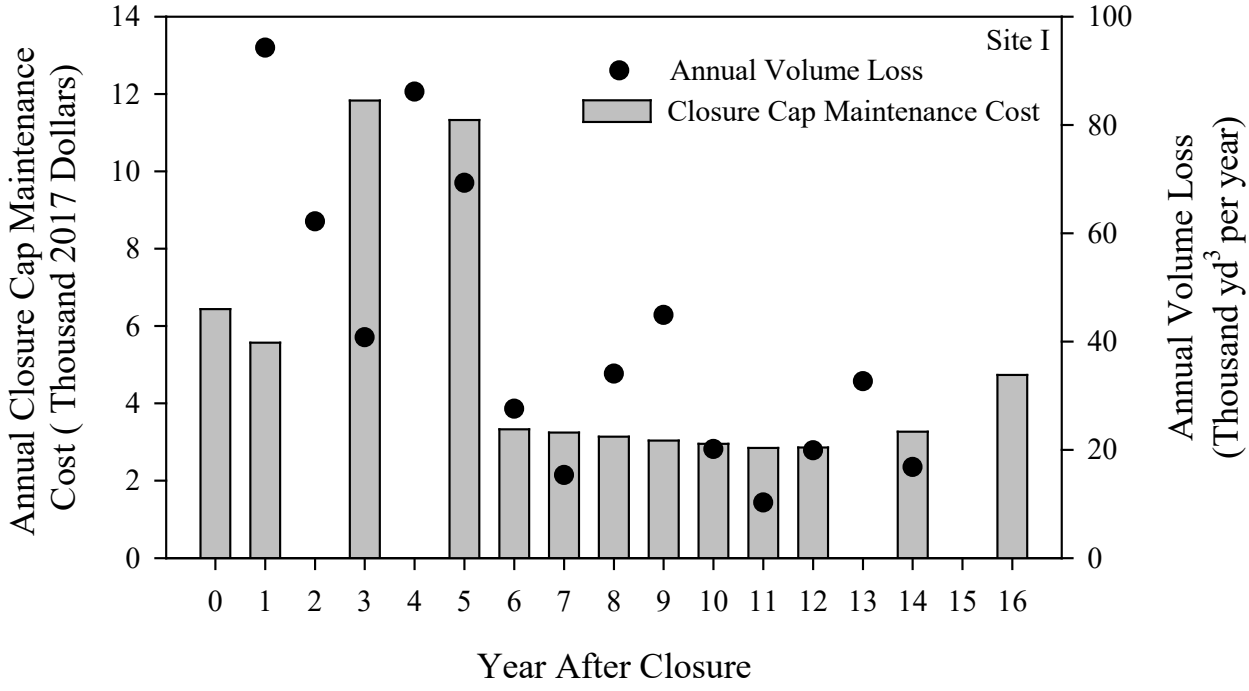


Figure 4-7. Annual Final Cover Maintenance Cost and Annual Volume Loss at Site I

4.6. Summary

The available PCC cost data from the nine study sites were analyzed to evaluate the financial impact of different MSWLF PCC activities. The available data were organized into the following six major cost categories: (1) leachate management, (2) GCCS management, (3) final cover maintenance, (4) groundwater and subsurface gas monitoring and maintenance, (5) engineering support and administration, and (6) other miscellaneous expenditures.

Limited PCC cost data were available for the study sites. Actual PCC cost data were available only for three sites (Sites B, D, and E); only estimated cost data were available for the other locations. Additionally, only three sites (Sites G, H, and I) provided cost data exclusive to the study cells.

For the other sites, the available cost data included the cost of maintaining other site cells as well. All the available PCC cost data were adjusted to 2017 dollars based on consumer price index values. For the sites with available study cell-specific data, the annual PCC cost varied from approximately $1,200 to $11,000 per acre of waste footprint, with an average of $5,300 per acre. For the sites where PCC cost data represented the entire site, annual PCC cost ranged between $1,550 to $37,000 per acre with an average of $6,450 per acre. Inconsistencies in cost categories used for tracking PCC cost, and the necessity for occasional system upgrades (e.g., GCCS expansion at Site E, and construction of a sewer connection for pumping leachate to the local WWTP) appears to be one of the primary reasons for such a wide variation in annual PCC cost at the study sites.

Leachate management costs include those expenses associated with leachate collection, hauling (as applicable), treatment (on-site and/or off-site), sampling and analysis (when exclusively available), and leachate collection system maintenance (when exclusively available). The average annual leachate management cost after closure ranged from approximately 3% to 68% among all of the sites, and it represented the greatest cost at six of the nine sites. The leachate management cost among sites with available cost and leachate collection rate data ranged from $5.5 to $219 per 1,000 gallons of leachate collected with an average of $79 per 1,000 gallons of leachate collected. This is equivalent to $23 to $25,162 per acre per year, with a median of approximately $2,300 per acre per year.

Among the seven sites for which GCCS management cost data were available, GCCS management cost ranged from approximately 11% to 44% of the annual average PCC cost. The annual GCCS management cost at these seven sites ranged from approximately $2,700 to $593,000 per year. LFGTE systems were implemented at two of the study sites (Sites E and I). Installation of an LFGTE project substantially reduced the cost associated with power purchase at Site E from an annual average of $24,000 before to $1,800 after the commencement of the LFGTE project.

The average annual groundwater and subsurface gas monitoring cost varied between approximately 5% to 26% of the total average annual PCC cost. The annual groundwater and subsurface gas monitoring cost among the sites ranged from approximately $2,400 to $169,600, which is equivalent to $40 to $2,600 per acre per year among the study sites (median of approximately $490 per acre per year). The specific PCC cost distribution for each type of monitoring activity was not available.

Annual final cover maintenance cost among the sites (except Site H) ranged between approximately 0.2% to 14% of the total average annual cost. Two sites (Sites G and I) for which study cell-specific cover maintenance cost data were available resulted in annual cover maintenance costs ranged from approximately $56 to $595 per acre of the waste footprint. For the remaining sites, the annual cover maintenance cost ranged from approximately $24 per acre to $3,400 per acre. The final cover maintenance cost at Site I was observed to decline with the amount of differential settlement, which was observed to reduce at Site I over time.

5. Landfill Gas

5.1. Overview

As discussed previously, LFG, if not controlled, is one of the primary sources of potential impact to HHE. RCRA Subtitle D regulations require the operation and maintenance of the gas monitoring system to ensure that the concentration of the methane generated by the facility does not exceed 25 percent of the lower explosive limit for methane in facility structures and the lower explosivity limit for methane at the facility property boundary (§258.23). The landfill owner/operator is required to implement a remediation plan for controlling the methane gas releases if methane gas levels exceeding these limits are detected in the facility structures or at the property boundary. The first objective of this chapter is to assess the frequency of methane detection above these thresholds in the facility structure and property boundary at the study sites.

Objective 1. Assess the frequency of subsurface methane migration to the facility structures and property boundary of the study sites.

Although the Subtitle D regulations for MSWLFs do not specifically require installation and operation of a GCCS, NSPS require construction and operation of a GCCS at MSWLFs that generate more than 50 Mg (or 34 Mg if regulated under Subpart XXX) of NMOCs annually. NSPS regulations (Subparts WWW and XXX) specify numerical thresholds for GCCS implementation and termination timeline at MSWLFs. NSPS (§60.752 (b)(2)(v)) allows for the removal or decommissioning of a GCCS at MSWLFs after closure as long as (a) the GCCS has been in operation for at least 15 years (or sooner due to lack of adequate LFG flow per Subpart XXX), and (b) the annual NMOCs generation rate is less than 50 Mg per year (or 34 Mg per year if meeting the requirements of Subpart XXX). NSPS specifies using the measured LFG collection rate and NMOCs concentration to estimate the annual NMOCs rate for comparison to these thresholds for sites with active GCCS. The NMOCs rates calculated using the measured LFG collection rates are, therefore, referred herein as the NMOC collection rate.

The blower/flare system operating constraints are additional considerations that should be taken into account for GCCS termination as NSPS Subpart XXX allows GCCS termination sooner than 15 years due to a lack of adequate flow rate. The blower/flare system is typically designed to handle the estimated peak LFG collection rate. The lower end of the LFG flow rate for the safe operation of flare ranges from 5 to 10% of the design (or peak) flow rate. The blower/flare system may need to be replaced or retrofitted to combust LFG below these flow rates if the NMOCs collection rate corresponding to these LFG flow rates is greater than 50 or 34 Mg/year. An understanding of these durations would allow assessment of whether the blower/flare system would need to be retrofitted or replaced before NSPS Subpart WWW or XXX allows its termination or decommissioning. The purpose of the second objective of this chapter, presented below, is to evaluate (a) whether the annual NMOCs collection rate from closed MSWLFs would be less than 50 (or 34) Mg within the minimum required GCCS operating period of 15 years or within 30 years after closure, and (b) whether the LFG flow rate would decline below 5% or 10% of the peak LFG flow rates before achieving annual NMOCs collection rates of 50 (or 34) Mg. The study sites sizes, however, are not representative of the size of approximately 75% of MSWLFs in the US as six of the study sites contain less than 4 million metric tons of waste. This analysis was performed for the study sites as well as hypothetical landfills that are representative of typical landfills in the US.

Objective 2. Assess the timeframe needed for the annual NMOCs collection rate from study sites to drop below the NSPS threshold of 50 (or 34) Mg per year and the timeframe for the LFG flow rate to decline below 5% and 10% of the peak flow rates.

Although NSPS allows termination of GCCS operation upon demonstration that annual NMOCs collection rate is below 50 (or 34) Mg, there is a possibility of a spike in the annual NMOCs generation rate and the associated potential collection rate in the future above the NSPS thresholds. The NMOCs generation rate is dictated by the LFG generation rate, which is expected to substantially increase with moisture intrusion if the in-place waste has substantial remaining LFG generation potential at the time of GCCS operation termination. The purpose of the third objective, presented as follows, is to address the question of whether the closed MSWLFs can generate NMOCs above the NSPS threshold of 50 and 34 Mg/year after termination of GCCS operation.

Objective 3. Assess whether MSWLFs would have sufficient NMOCs generation potential to generate 50 (or 34) Mg of NMOCs annually after the termination of GCCS operation.

In addition to the NSPS, there may be state-specific required/suggested thresholds that may impact GCCS termination. For example, the FDEP (FDEP 2016) suggests achieving a methane production rate that is below 10% of the peak rate or achieving remaining methane generation potential of less than 10% of the total generation potential before considering changes to the active LFG system. The goal of the fourth objective is to estimate the remaining methane generation potential of the in-place waste and the associated timeframes to achieve the targeted reduction of methane generation potential.

Objective 4. Assess the remaining methane generation potential at 30 years after closure and assess the timeframes needed for the remaining methane potential to drop below 25% and 10% of the total generation potential.

Robust analysis methods and approaches are critical for reliably estimating the remaining methane and NMOCs generation/collection potential. An additional objective of this chapter is to present and discuss the approaches that can be used for estimating the remaining methane and NMOCs generation potential and timeframes to achieve annual NMOCs collection rate of 50 (or 34) Mg for a closed MSWLF. The analysis presented in the chapter should not be considered as a comprehensive evaluation of HHE impacts with respect to LFG due to various assumptions and limitations. These assumptions and limitations are presented along with the analysis. These limitations are also summarized in the last section of this chapter.

5.2. Methodology

5.2.1. Data Sources

The analysis presented in this Chapter is based on the following two data sources: the first is the nine case-study sites, and the second is the Greenhouse Gas Reporting Database (https://www3.epa.gov/enviro/).

1) Case-Study sites

Perimeter monitoring probe data for methane were available for review from seven of the nine study sites. Some carbon dioxide and oxygen data were also available for all sites except Site H. A summary of the number of probes surrounding the study cells, as well as the total number of data points analyzed from these probes for each of the sites are presented in Table 5-1. Facility

structures methane concentration data were only available from three sites (Sites B, F, and H). The data from all the probes at the study sites were analyzed for all the sites, except for Site G and I. Probes data were not available for Site G and study cell-specific probe data were available for Site I. A total of 7,598 methane monitoring probe readings were compiled from the sites with available data; approximately 56% of these readings are for Sites E and I.

The presence of an active GCCS and the availability of LFG flow rate and composition data were part of the site selection criteria. Therefore, the LFG flow rate and methane content data were available for all the study sites/cells. Table 5-1 summarizes the key attributes relevant to each site's GCCS. LFG flow rate and methane content data were available for all of the sites except Site I; although the sitewide LFG flow rate data were available for Site I, the study cell-specific LFG flow rate and composition data were not available in data sources reviewed for the closed study cell. More details about GCCS and historical LFG/methane collection rate trends are presented in the individual site descriptions included in Attachments A-I.

2) **Greenhouse Gas Reporting Database**

As shown in Table 5-1, at least three of the study sites are relatively small (i.e., contain less than 2.5 million MT of waste) and were not regulated under the NSPS rules. Therefore, the results of some of the analysis may not be representative of typical MSWLFs in the US. Data from the GHG database were primarily used to identify the representative size of MSWLFs in the US.

The capacity of the MSWLFs, reported as part of the annual GHG reports, were downloaded from the Envirofacts database (EPA 2019b). This database contains data from all the MSWLFs that are required to report annual methane generation and emissions amounts per federal regulations (40 CFR 98 Subpart HH) (referred hereinto to as *GHG reporting regulations*); the GHG reporting regulations require the MSWLFs owners to report annual methane generation and emissions that accepted waste on or after January 1, 1980, and are estimated to generate more than 25,000 metric tons CO_2 equivalent GHGs. Data obtained from the database were used to estimate:

1) Disposal capacity of MSWLFs that are representative of the 25^{th}, 50^{th}, and 75^{th} percentile of the capacity of MSWLFs in the US; the MSWLF containing in-place waste amounts corresponding to 25^{th}, 50^{th}, and 75^{th} percentile MSWLF capacity in the US is referred herein to as small, medium, and large MSWLF, respectively.
2) Operating lifespan for each MSWLF included in the GHG reporting regulations database was calculated based on the first year of waste acceptance and the actual or estimated closure year included in the GHG database. The median lifespan of MSWLFs with capacities ranging from 15^{th} to 35^{th} percentile capacity was used as the representative lifespan of the small MSWLF. Similarly, the median lifespan of MSWLFs with capacity in the 40^{th}-60^{th} and 65^{th}-85^{th} percentile capacity ranges were used as the representative lifespans of medium and large MSWLFs, respectively.

Table 5-1. Key Attributes Relevant to the GCCS of Each Study Site

Data Type	Site	A	B	C	D	E	F	G	H	I
Timeline and In-place Waste Amount	Study Cell(s) Start–Closure Year	1988–1998	1996–2004	1984–1998	1996–2009	1997–2004	1989–1997	1987–2000	1991–2008	1990–1997
	Years after Closure (as of 2016)	18	12	18	7	12	19	16-23	8	19
	Total Sitewide In-Place Waste Amount (Million Metric Tons)	2.05	3.27	X	5.56	3.97	1.85	1.64	3.46	X
Subsurface Monitoring Probes Data	Years (after Closure) of Available Probe Data	2007, 2009, 2011, 2015	2005-2015	X	2010-2015	2004-2016	1997, 2000, 2002-2011	X	2008-2016	1997-2009, 2011, 2013-2015
	Number of Probes	54	23	X	26	19	13	X	10	11
	Total Number of Data Points Available	194	689	X	678	2,206	926	X	870	2,035
Landfill Gas Collection and Control System	Operating Period[a]	1999–2011	1991–N/A	1994–N/A	1999–N/A	1991–N/A	1999–N/A	2008[b]–2012	1996–N/A	1994–N/A
	Methane Collection Rate Years Modeled for Site-Specific Decay Rate Estimation	1999–2011	2010–2016	1998–2002	2009–2016	2005–2016	2000–2009	2008–2012	2007–2017	X
	Methane Collection Rate (standard cubic feet per minute)	12–924	40–302	268–798	515–1333	63–924	36–367	0–104	328–1218	X
	Landfill Gas-to-Electricity Project Start–Stop Year	2002–2005	N/A	2007–N/A	N/A	2009–N/A	N/A	2008–N/A	N/A	N/A

N/A- Not applicable to the site.
X – data not available
[a] N/A- GCCS was operating at the time of this study
[b] Start date of GCCS not known. LFGTE (landfill gas-to-electricity) project commenced in 2008.

5.2.2. Methane Collection Rate for the Study Sites

The blower/flare systems at MSWLFs are typically equipped with a flow meter and a datalogger to record the LFG flow rate several times per hour (e.g., once every 15 mins). LFG composition is typically analyzed using a handheld monitor. The frequency of the available LFG flow and composition readings varied from once per year to multiple times per day. For Site G, the methane flow rates were available. For four of the sites (Sites A, B, C, and F), an associated LFG

composition concentration was available for each LFG flow measurement; the methane flow rate was calculated by multiplying the LFG flow rate with the methane content for these sites. The methane flow rate was calculated by using Eqn. 5-1.

$$Methane\ flow\ (scfm) = Total\ LFG\ flow\ (scfm) \times Methane\ Fraction \qquad \text{(Eqn. 5-1)}$$

For Site E, the reported methane mass (in kilograms) was converted to the volumetric methane flow rate at standard temperature and pressure using Eqn. 5-2. NSPS (§60.2) specifies a temperature of 68 °F and 101.3 kilopascals as standard conditions. The methane density at standard conditions was calculated using the ideal gas law and GHG reporting regulations (§98.343) specified methane density of 0.0423 lb methane per cubic feet at 60 °F and 1 atm (101.3 kilopascals).

$$Methane\ flow\ (scfm) = Methane\ mass\ (\frac{kg}{min}) \times 2.2\frac{lb}{kg} \times \frac{1\ cubic\ feet}{0.0417\ lb\ methane\ @\ 68\ °F\ and\ 1\ atm} \qquad \text{(Eqn. 5-2)}$$

For Site H, LFG flow rate data were available on an approximately monthly basis, but only annual LFG composition data were available. The monthly LFG flow rates were multiplied by the associated annual methane composition value to estimate the monthly methane flow (Eqn. 5-1). For Site D, the average of the methane content data available was used for the analysis presented in this Chapter; weekly to monthly gas composition data were available only for a limited period (April 2015 through December 2016) for this site. The average of the available methane concentration measurements was multiplied by all the LFG flow values to estimate methane flow rates for Site D.

5.2.3. Methane and NMOCs Collection Rate Estimation

The estimation of LFG, methane, and NMOCs generation rate using the EPA's LandGEM model is standard practice for estimating the LFG rates for GCCS design and the annual NMOCs generation rate for compliance purposes (EPA, 2005). The LandGEM model uses the first-order decay model, as presented in Eqn. 5-3.

$$Q_{CH4}(t) = \sum_{i=1}^{n} M_i L_o k\, e^{-k(t-t_i)} \qquad \text{(Eqn. 5-3)}$$

Where,
$Q_{CH4}(t)$ = modeled methane generation rate in m^3 CH_4 $year^{-1}$ at time t.
M_i = waste placed in year i in metric tons,
L_o = the waste methane generation potential in m^3 CH_4 per metric ton of waste placed in the landfill,
k = decay rate on a per-year basis,
n= lifespan of landfill
t_i = placement time, in years, of M_i.
t= time, in years

LandGEM estimates the LFG generation rate based on the specified methane content of LFG. It further estimates the NMOCs generation rate using the LFG generation rate and NMOCs content of LFG. LandGEM allows the selection of one of several default values or user-specified values for L_o, k, and NMOCs content for modeling LFG and NMOCs generation rate.

NSPS allows the use of the measured LFG collection rate to estimate the annual NMOCs generation/emission rate for the sites with active GCCS for the purpose of determining when the system can be removed. As all of the sites have or had a GCCS, the measured collection rates were used to estimate the site-specific decay rate and the methane potential using inverse first-order decay modeling. The site-specific methane collection potential and decay rates were subsequently used in LandGEM model to estimate the future LFG and NMOCs collection rates. A description of the methodology to estimate the methane collection potential and decay rate for the study sites is presented in Section 5.2.4. As the methane potential estimated using this approach is based on the methane collection rate and not the generation rate, it is referred herein to as the methane collection potential (L_c).

Table 5-2 list the values of modeling parameters used for the estimate of the future methane and NMOCs collection rates. The annual disposal amounts were available for all of the sites and were used for LandGEM modeling. The site-specific estimates of the methane collection potential and decay rates were used for the modeling.

NSPS allows the use of a site-specific measurement, if available, for estimating the annual NMOCs emission rate and comparison to the NSPS threshold (50 (or 34) Mg per year). Due to a lack of the site-specific NMOCs measurements, the NSPS-default of 4,000 parts per million volume as hexane was used. The results of LandGEM modeling were used to estimate the post-closure period for the annual NMOCs collection rate to decline below the NSPS thresholds of 50 (or 34) Mg per year for the study sites.

Table 5-2. Modeling Parameters for the Estimation of the Future Methane and NMOCs Collection Rate for the Study Sites

Parameter	Value Used
Waste Mass*	Annual site-specific disposal mass (presented in Attachments A through I)
Decay Rate, k (year $^{-1}$)	Site-specific estimate for the study sites
Methane Collection Potential, L_c (m^3 methane per MT waste)	Site-specific estimate for the study sites
Nonmethane Organic Compounds (NMOCs) (ppmv)	4000

* The waste placement data only for the study cells C1 and C2 were used as these data were not available for the other cells at Site C. Sites G and C were actively accepting waste at the time of this study. Site G was assumed to close in 2016 for the analysis.

As discussed later, the study sites sizes are not representative of the size of approximately 75% of MSWLFs in the US as six of the study sites contain less than 4 million metric tons of waste. In addition to the study sites, the methane and NMOCs collection rates were also estimated for the hypothetical small, medium, and large MSWLF representative of the 25th, 50th, and 75th percentile capacity of the landfills in the GHG reporting database, respectively, as described in Section 5.2.1. In order to estimate the impact of decay rate on the methane and NMOCs collection rate trend for these landfills, five scenarios each with a different k value were simulated using the first-order decay model for each of the three examples MSWLFs. The decay rates used were 0.02, 0.038, 0.057, 0.17, and 0.22 year $^{-1}$. The first three decay rate values selected for this analysis are the GHG Reporting Regulations-specified decay rate for the precipitation zones with an annual rainfall of less than 20 inches (arid area), 20-40 inches (moderate precipitation zone), and more than 40 inches (wet zone). The decay rates of 0.17 and 0.22 year $^{-1}$ are the median and average,

respectively, of the decay rates reported for bioreactor landfills (EPA 2006, Yazdani et al. 2006, Kim and Townsend 2012, Wang et al. 2013).

As mentioned earlier, NSPS allows the use of the measured LFG collection rate for the sites with active GCCS to estimate the annual NMOCs generation/emission rate for the purpose of determining when the system can be removed. NSPS also explicitly allows the use of AP-42-suggested L_o for estimating the LFG collection rate for GCCS design. The AP-42 suggested value of 100 m^3 CH_4 per Mg was used for the first-order decay modeling to estimate the future annual methane and NMOCs generation rates for these hypothetical MSWLFs analyses. It should be noted that LandGEM model (Eqn 5-3) calculates the LFG generation rate and not the collection rate. An LFG collection efficiency of 100% was assumed for these example sites to calculate the LFG collection rate from the estimated generation rate for a conservative analysis. In other words, the collection rates were assumed to be equal to the generation rate.

A spreadsheet-based first-order decay model was developed and used for estimating the methane collection rate of these hypothetical cases. LandGEM was not used for these cases as the estimated annual NMOCs collection rate for some of the cases did not decline below 50 Mg/year within the 150-year duration (since the landfill start) used for LandGEM model. The annual NMOCs collection rate was calculated from the estimated methane collection rate using the following equation (adapted from the equation used by NSPS Subparts WWW and XXX); methane was assumed to constitute 50% (by volume) of LFG:

$$M_{NMOC} = 2 \times Q_{CH4} \times C_{NMOC} \times 3.6 \times 10^{-9} \quad \text{(Eqn. 5-4)}$$

where:
M_{NMOC} = Annual NMOCs collection rate (Mg $year^{-1}$)
Q_{CH4} = CH_4 collection rate (m^3 $year^{-1}$); and
C_{NMOC} = Concentration of NMOCs in LFG (ppmv) (4,000 ppmv as hexane used)
3.6×10^{-9} = Conversion factor (m^3 hexane to Mg hexane)
2 = factor for calculating LFG flow rate using methane flow rate (volume LFG/volume CH_4)

5.2.4. Site-Specific Decay Rate and Methane Collection Potential Estimation

A site-specific k and methane potential values can be estimated by conducting an inverse first-order decay modeling (also referred herein as regression analysis) to best-fit the rate modeled using the first-order decay model to the available methane collection rate. Given the availability of long-term historical methane collection data from eight of the study sites, a regression analysis was conducted to estimate the site-specific methane potential and decay rate for each site. As the methane potential estimated using this approach is based on the methane collection rate and not the generation rate, it is referred herein to as the methane collection potential (L_c). Because LandGEM does not include a regression analysis feature, a first-order decay model (Eqn. 5-3) was developed for each site to estimate the monthly methane collection rate for the period for which the measured methane flow data were available. Monthly disposal amounts were estimated by dividing the annual disposal amounts by 12 for the sites for which monthly disposal data were not available.

The sum of squared errors (SSE) presented in Eqn. 5-5 is a measure of the relative goodness of fit. The Microsoft® Excel function "Solver" was used to minimize the SSE by changing k and L_c values. The initial values to initiate the Solver function and constraints used for the regression analysis are presented in Table 5-3. For most sites, all three model simulations (corresponding to three different initial values for L_c) provided the same best-fit L_c and k. The best-fit L_c and k that

resulted in the lowest SSE were selected from the regression analyses corresponding to the three initial values of L_c (20, 100 and 230 m^3/Mg) for each site if the three model runs provided varying L_c and k. Site-specific L_c and k estimates are presented in Section 5.3.3.

$$SSE = \sum_j^n \left(Q_{modeled,j} - Q_{measured,j}\right)^2 \quad \text{(Eqn. 5-5)}$$

Where,

$Q_{modeled,j}$ = the modeled monthly methane flow rate for month j,
$Q_{measured,j}$ = the monthly methane flow rate calculated using the reported flow rate and methane content for month j, and
n = number of months that monthly recovered methane quantity data are reported for the site.

Table 5-3. Initial Values and Constraints for Parameters used for Modeling

Parameter	Initial Value	Lower–Upper constraints
Decay Rate, k (year $^{-1}$)	0.05	0.001–2.2 (Faour et al. 2007)
Methane Collection Potential, L_c (m^3 methane per MT waste)	20, 100, 230	20–230 (Krause et al. 2016)

Furthermore, the coefficient of correlation (r^2 value) (Eqn. 5-6) was calculated using the SSE, and the total sum of squares (SST) was calculated as presented in equation (Eqn. 5-7). The SST is a measure of the total variation of the site methane flow rates with respect to the average of the monthly/annual recovered methane flow rates (i.e., $Q_{measured,avg}$) available for the site.

$$r^2 = 1 - \frac{SSE}{SST} \quad \text{(Eqn. 5-6)}$$

Where,

$$SST = \sum_j^n \left(Q_{measured,avg} - Q_{measured,j}\right)^2 \quad \text{(Eqn. 5-7)}$$

All available methane collection data were included in this analysis for Sites A and D. As described in Attachments B, C, E, F, G, and H, LFG data for only selected periods for these sites were used for regression analysis. The methane collection data only for the period where the data appeared to follow a first-order decay declining trend, were used for regression analysis. Several factors such as the expansion of GCCS to new areas (e.g., Sites C and F), the progressive decommissioning of wells (e.g., Site G), or aggressive wellfield adjustment during the GCCS start-up phase (e.g., Site B) result in deviation of the measured collection rate from the first-order decay trend. The data, which were indicative of major changes to the GCCS, were excluded from the regression analysis. It should be noted the GCCS changes only impacts the LFG collection rate and not the LFG generation rate. For the years for which the measured/calculated methane collection rates were not available, the modeled data points were not included in the SSE calculation.

5.2.5. Potential of Elevated NMOCs Generation Rate after GCCS Termination

The potential for a spike in the annual NMOCs generation rate after the termination of GCCS operation was evaluated for a small MSWLF in the moderate precipitation zone (20-40 inches annual precipitation) and a large MSWLF in the arid area (less than 20 inches annual precipitation). The objective was to assess whether a closed MSWLFs can generate NMOCs above the NSPS threshold of 50 and 34 Mg per year after termination of GCCS operation. A decay rate of 0.02 and 0.038 year $^{-1}$ was used for arid and moderate precipitation zones, respectively, as discussed in Section 5.2.3. It was assumed that the decay rate would increase by 100% to 0.04 and 0.076 year $^{-1}$ for arid and moderate precipitation zones, respectively, ten years after GCCS decommissioning

(i.e., ten years after the annual NMOCs collection rate decline below 50 Mg and the resultant GCCS operation termination). The decay rate can potentially increase due to moisture intrusion into the landfill through a compromised final cover (assuming PCC is terminated and the final cover is not monitored and maintained at this point). Because LandGEM does not allow the specification of a time-varying decay rate, a spreadsheet-based first-order decay model was developed and used for this analysis.

5.2.6. Remaining Methane and NMOCs Generation Potential

The annual methane generation rate from LandGEM or the developed first-order decay model spreadsheet were aggregated to calculate the cumulative amount of methane generated over time. The remaining methane generation potential was calculated by deducting the cumulative methane generated from the total methane generation potential as depicted by the following equation (Eqn. 5-8):

$$V_{CH4\ remaining,j} = \sum_{i=1}^{n} M_i L_o - \sum_{t=0}^{t_j} Q_{CH4,t} \quad \text{(Eqn. 5-8)}$$

Where,

$V_{CH4\ remaining,j}$= Total remaining methane generation potential (m^3 CH_4) at the end of year t_j

$Q_{CH4,t}$ = modeled annual methane generation rate in m^3 CH_4 $year^{-1}$ during year t.

M_i = waste placed in year i in metric tons,

L_o = the waste methane generation potential in m^3 CH_4 per metric ton of waste placed in the landfill,

n= lifespan of the landfill

t, t_j= time, in years

The remaining methane generation potential per unit in-place waste was calculated by dividing the total remaining methane generation potential by the in-place waste mass as depicted by the following equation (Eqn. 5-9):

$$L_{CH4\ remaining,j} = \frac{V_{CH4\ remaining,j}}{\sum_{i=1}^{n} M_i} \quad \text{(Eqn. 5-9)}$$

Where,

$L_{CH4\ remaining,j}$= Remaining methane generation potential per unit in-place waste (m^3 CH_4 per MT waste) at the end of year t_j

The percent remaining methane generation was calculated by dividing the remaining methane generation potential with the total methane generation potential as depicted by the following equation (Eqn. 5-10):

$$P_{CH4\ remaining,j} = \frac{V_{CH4\ remaining,j}}{\sum_{i=1}^{n} M_i L_o} \times 100 \quad \text{(Eqn. 5-10)}$$

Where,

$P_{CH4\ remaining,j}$= Percent remaining methane generation potential (%) at the end of year t_j

The annual NMOCs generation rate from LandGEM model or the developed first-order decay model spreadsheet were aggregated to calculate the cumulative amount of NMOCs generated over time. The remaining NMOCs generation potential was calculated by deducting the cumulative NMOCs generated from the total NMOCs generation potential as depicted by the following equation (Eqn. 5-11):

$$M_{NMOCs\ remaining,j} = 2 \times C_{NMOC} \times 3.6 \times 10^{-9} \times \sum_{i=1}^{n} M_i L_o - \sum_{t=0}^{t_j} M_{NMOCs,t} \qquad \text{(Eqn. 5-11)}$$

Where,

$M_{NMOCs\ remaining,j}$= Total remaining NMOCs generation potential (Mg as hexane) at the end of year t_j

$M_{NMOCs,t}$ = modeled annual NMOCs generation rate in Mg year $^{-1}$ during year t, and other terms as defined above.

The site-specific L_c estimates were used as a proxy for L_o for estimating the percent and mass-based remaining methane and NMOCs generation potential. It should be noted that the percent remaining methane and NMOCs generation potential would be the same irrespective of whether L_o or L_c value used for the estimation. The use of L_c instead of L_o would, however, result in an underestimation of the mass-based remaining generation potential.

5.3. Results and Discussion

5.3.1. Subsurface Methane Monitoring Data

The regulations in §258 require the installation of subsurface methane monitoring probes around the periphery of the site and the quarterly monitoring of these probes to identify the subsurface migration of LFG. Site operators are also required to monitor the methane concentration in on-site structures to verify that LFG is not migrating to the facility property boundary above the lower explosivity limit and accumulating in the structures beyond 25% of the lower explosive limit (i.e., 1.25% by volume in air). All the available perimeter methane monitoring probe data were analyzed.

A summary of the number of probes surrounding the study cells, as well as the total number of data points analyzed from these probes for each of the sites is presented in Table 5-4. Methane was detected in 542 instances (i.e., approximately 7.1%) out of these 7,598 readings and exceeded the lower explosive limit of methane (i.e., 5% methane by volume in air) in 103 instances (i.e., approximately 1.4% of all readings included in the evaluation). More than 55% of the detected methane measurements occurred at Site E.

Table 5-4 summarizes the number of exceedances where the methane concentration was measured above the lower explosive limit of 5% as a function of years since closure. The available data suggest that the subsurface methane exceedances occurred relatively infrequently over the periods with data available for review. As presented in Table 5-4, only 103 methane exceedances were observed at five of the seven sites. Almost 80% of these methane concentrations above the lower explosivity limit of methane occurred at Site E, and 10% of exceedances occurred at Site B. Both these sites contained either unlined cells or cells lined with a compacted clay liner. Ten probes at Site E exhibited methane measurements greater than its LEL at least once; the latest exceedance was observed in 2011. Methane concentration was above its LEL for 80 measurements at ten probes. Sixty-six of these 80 methane measurements greater than the LEL were observed at three probes located near the cells lined with a compacted clay liner. The cell lined with geomembrane is not likely to cause the elevated methane concentrations observed at these probes. At Site B, ten of 12 measurements above the LEL of methane occurred at a single probe located near the area where the lined cell(s) adjoin the unlined cell, and the exceedances are likely caused by the unlined cell.

All of the observed exceedances were recorded within the first five years after closure for all the sites except for Site E. Relatively infrequent methane detection/exceedances at probes in the

vicinity of the study cells are like due to active gas collection and a geomembrane bottom liner system. The GCCS at Site A was terminated during the 13th year since closure. The available perimeter probe monitoring data do not show any methane exceedance since the termination of GCCS for Site A.

The structure methane concentration data (2,105 measurements) were also available for Sites B, F, and H. The methane monitoring data (70 measurements) were available for 15 structures at Site B for year 1 and 2 since the closure. Methane was detected in two measurements at Site B. All methane measurements were lower than 25% of its LEL. The maximum detected methane concentration was 0.1% (volume basis).

Methane monitoring data were available for 41 facility structures at Site F for years 3 and 5 through 14 since the closure. A total of 1,155 measurements were available. Five of the measurements exhibited (at four structures) methane concentration greater than 25% of its LEL. The maximum methane concentration at these locations was 8.5%. These structures are located near the unlined cells of the Site F; therefore, the study cell was not likely the cause of these exceedances.

Based on 880 measurements at ten facility structures at Site H, methane was detected at a pumphouse on eight sampling events during years 1 and 2 since closure. None of the methane measurements in structures at Site H were above 25% of its LEL– the maximum methane concentration observed was 1% (volume basis).

Table 5-4. Number of Annual Methane Monitoring Probe Methane Exceedances at the Site as a Function of Time Since Closure

Site	A	B	D	E	F	H	I	Years Since Closure
Years of Available Probe Data	2007, 2009, 2011, 2015	2005-2015	2010-2015	2004-2016	1997, 2000, 2002-2011	2008-2016	1997-2009, 2011, 2013-2015	
Number of Probes	54	23	26	19	13	10	11	
Total Number of Data Points Available	194	689	678	2,206	926	870	2,035	
Closure Year	1998	2004	2009	2004	1997	2008	1997	
Exceedances per Year	-	-	-	6	3	0	4	0
	-	2	0	16	-	0	0	1
	-	4	1	23	-	0	0	2
	-	4	1	17	0	0	0	3
	-	2	0	5	-	0	0	4
	-	0	2	5	0	0	0	5
	-	0	0	7	0	0	0	6
	-	0		1	0	0	0	7
	-	0		0	0	0	0	8
	0	0		0	0		0	9
	-	0		0	0		0	10
	0	0		0	0		0	11
	-			0	0		0	12
	0				0		-	13
	-				0		0	14
	-				-		-	15
	-				-		0	16
	0				-		0	17
					-		0	18

"-" indicates that probe or facility structure methane data was not available for the site for that year. Shaded cells indicate future years.

5.3.2. Methane Collection Rate Trends

Figure 5-1 presents the temporal variation of the methane flow rate for the study sites after closure. The methane flow rate for all of the sites shows a declining trend after closure. An increase in the LFG flow rate at Sites C and F is associated with the expansion of the GCCS to collect LFG from new active cells (at Site C) and old cells (at Site F). The GCCS expansion results in an increase in the LFG collection rate but does not impact the LFG generation rate. GCCS operation at Site A was terminated 12 years after closure due to an inadequate LFG flow rate. It should be noted that due to a lack of study-cell specific LFG data, data from several or all the closed cells at the site were used for the analysis presented in the report.

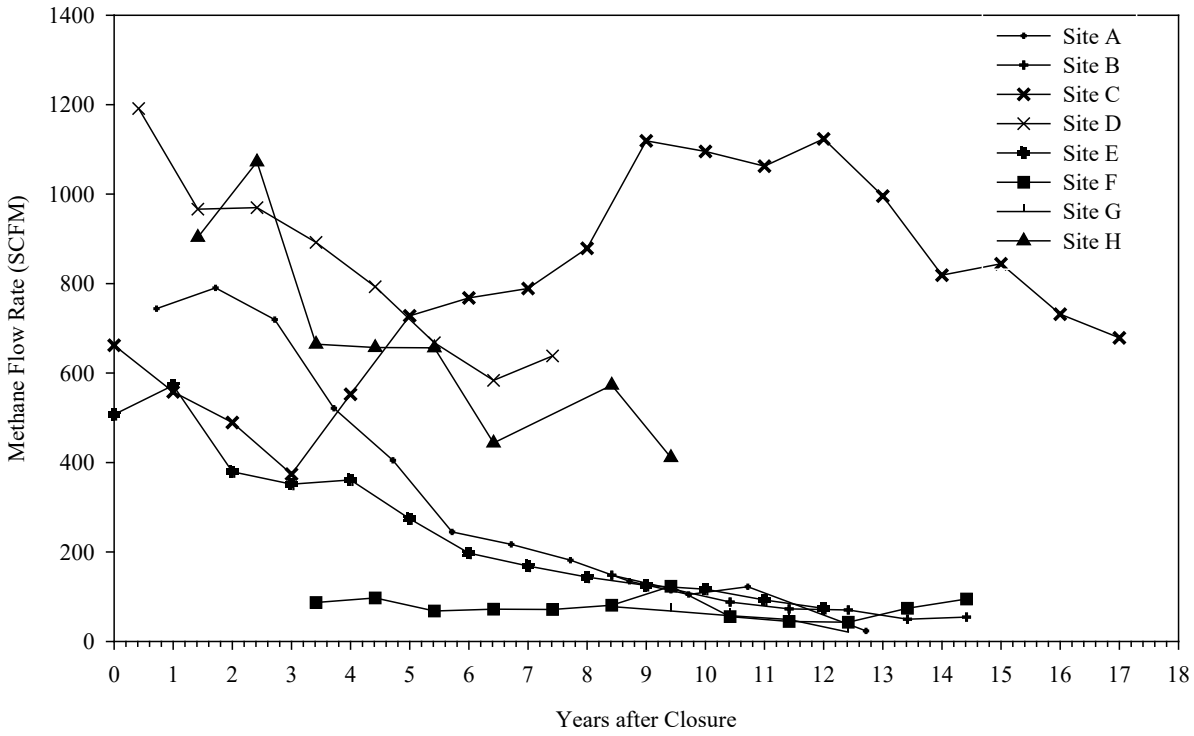


Figure 5-1. Temporal Variation of Methane Flow Data for Study Sites

5.3.3. Estimated Site-Specific First-Order Decay Rates and Methane Collection Potentials

The waste decay rate has a significant impact on future LFG/methane/NMOCs generation potential. Table 5-5 presents the average annual precipitation, the site-specific decay rate, and the methane collection potential estimates based on the regression analysis. The estimates based on the first-order decay model fit the measured data with an $r^2 > 0.7$ for six of the eight study sites modeled. An r^2 of 0.7 means that the modeling approach captures or explains 70% of the variations in the methane collection rate over time at a given site, and the rest 30% is due to factors (e.g., changes in GCCS operations) that are not accounted for by the model.

Based on the approach described in Section 5.2.4, the estimated k for the study Sites A-H ranged from 0.06 to 0.24 year $^{-1}$ with an average of 0.16 year $^{-1}$; LFG data for Site I were not available. The estimated decay rates for four out of eight sites were within a range of 0.18-0.21 year $^{-1}$. As expected, k estimated for Site A is highest among all of the sites, potentially due to the extensive leachate recirculation conducted at the site. Leachate was also recirculated at study cells C and E. However, the fraction of the collected leachate that was recirculated and the duration of recirculation for these cells was substantially lower than that of study cell A. Nonetheless, the estimated k for the site was approximately 20% lower than that recommended by AP-42 of 0.3 year $^{-1}$ for wet MSWLFs (US EPA 2008). The estimated k was lowest for Site F, which is located in the driest area among all the study sites. It should be noted that active LFG collection from the study cell at Site F did not start until 2010 (13th year after closure). The LFG collection rate from another cell (lined with a clay liner) was used for the regression modeling for this site. The

estimated k values are within the range of those reported for MSW (Pelt 1993, Faour et al. 2007, Tolaymat et al. 2010, Barlaz et al. 2010, Amini et al. 2012, Amini et al. 2013, Wang et al. 2013, Zhao et al. 2013, Vu et al. 2017), but substantially greater than the NSPS default of 0.05 year $^{-1}$. The estimated k values are also greater than the ones recommended by AP-42 (0.02 and 0.04 year $^{-1}$ for regions with less than 25 inches and more than 25 inches of precipitation, respectively) as well as values specified by GHG reporting regulations (0.02, 0.038, and 0.057 year $^{-1}$ for less than 20, 20-40, and more than 40 inches precipitation zones, respectively). The estimated k values for these sites were approximately three to four times the k specified by the GHG reporting regulations and three to five times the AP-42 recommended k (except for Site A). These high values suggest that the MSW at the study sites is decomposing at a substantially faster rate than the AP-42 recommended rate as well as the rate specified by the GHG reporting regulations.

The L_c estimates ranged from 36-152 m^3 CH_4 Mg^{-1} with an average of approximately 94 m^3 CH_4 Mg^{-1}. Krause et al. (2016) presented a comprehensive compilation and a critical review of MSW L_o and reported values to range from 20-223 m^3 CH_4 Mg^{-1} based on modeled values and those calculated based on waste composition and biodegradability of individual waste components. A range of 35-167 m^3 CH_4 Mg^{-1} was reported based on experimentally measured values of mixed MSW (Krause et al. 2016). NSPS prescribes a L_o of 170 m^3 CH_4 Mg^{-1} for determining whether or not a landfill will be subjected to its requirements. This value has been reported to be conservative and not reflective of the actual generation (Krause et al. 2016). Krause et al. (2016) reported an average L_o of 94 m^3 CH_4 per Mg for North America. The EPA Compilation of Air Pollutant Emissions Factors (AP-42) suggests L_o of 100 m^3 CH_4 per Mg for MSWLFs, which is very close to the average L_o reported by Krause et al. (2016) for North America.

The site-specific estimates for all the sites were in the range of experimentally measured of L_o values of mixed MSW compiled by Krause et al. (2016). The relatively low L_c values for the Sites E and G are potentially due to the disposal of considerable amounts of inerts in the study cells at these sites. As discussed in Attachments E and G, non-MSW materials (primarily inerts including contaminated soil, sludges, ashes, industrial waste) constituted over 30% (by total weight) of the in-place waste at these sites. In addition, limited gas collection infrastructure in one of the cells at Site G results in poor gas collection efficiency and lower L_c estimate; LFG was only collected from the leachate collection pipes cleanouts at one of the cells at Site G.

The use of site-specific L_c for estimating the future annual NMOCs collection rate and assessing the timeframe for the NMOCs rate to decline below the NSPS threshold is appropriate as the NSPS regulations allow using the measured LFG collection rate for estimating the NMOCs generation/emission rate for landfills with a GCCS. However, it should be noted that not all the LFG generated is collected even at sites with GCCS. The LFG and NMOCs generation rates are, therefore, expected to be greater than those estimated using L_c. The magnitude of the difference between the collection and generation rates would depend on the GCCS collection efficiency, which is typically not measured. It should be further noted, the percent remaining methane and NMOCs generation potentials and timeframes to achieve the LFG flow rate below a given percent of the peak flow rate estimates would not depend on the L_o (or L_c) value used. However, the use of L_c would result in an underestimation of the mass-based remaining methane and NMOCs generation potential.

Table 5-5. Estimated First-Order Decay Rate and Methane Collection Potential, and Average Annual Precipitation for the Study Sites

Site	Site-Specific Decay Rate, k (year $^{-1}$)	Site-Specific Methane Collection Potential, L_c (m^3 CH_4 per Mg Waste)	Coefficient of Correlation (r^2)	Annual Precipitation (inches/year)
A	0.24	152	0.93	51
B	0.21	87	0.89	56
C	0.20	127	0.86	47
D	0.09	69	0.86	46
E	0.18	58	0.86	30
F	0.06	132	0.21	22
G	0.21	36	0.71	46
H	0.11	94	0.55	47

5.3.4. Timeframes for Achieving Annual NMOCs Collection Rate below 50/34 Mg and LFG Flow Rate Below 5% and 10% of the Peak Rates

This section addresses the questions of whether the annual NMOCs collection rate from closed MSWLFs would be less than 50 (or 34) Mg within the minimum NSPS-required GCCS operating period of 15 years or within 30 years after closure. The analysis also addresses whether the LFG collection rate would decline below 5% or 10% of the peak LFG flow rate before achieving annual NMOCs collection rates of 50 or 34 Mg. As mentioned earlier, the NSPS-default NMOCs concentration in combination with the site-specific k and L_c estimates (Table 5-5) were used for the first-order decay modeling to estimate the future annual methane and NMOCs collection rate. The use of site-specific L_c estimates, which are representative of the LFG collection rate, is appropriate for this analysis since NSPS allows the use of the LFG collection rate for estimating the annual NMOCs generation rates for determining whether or not GCCS operation can be terminated.

Table 5-6 presents the estimated period after closure for an annual NMOCs collection rate to decline below the NSPS thresholds of 50 (or 34) Mg per year for the study sites. It also presents the waste amounts used for LandGEM modeling, the estimated years after closure for LFG flow rates to decline to 5% and 10% of the peak flow rates, as well as the estimated GCCS operating duration before the NMOCs collection rate decline below 50 Mg per year. The GCCS operating duration was calculated based on the actual GCCS start-up year and the year in which the NMOCs collection rate is expected to decline below 50 Mg per year; it does not represent the actual total GCCS operating duration. As expected, the duration for the annual NMOCs collection rate to decline below 50 Mg is lower than those associated with 34 Mg per year.

The GCCS at all of the sites except for the study Sites A, C, and G would have operated for more than 15 years before the annual NMOCs collection rate declines below 50 Mg year. The analysis suggests that the annual NMOCs collection rate threshold is the limiting NSPS constraint for the termination of GCCS operations. The expeditious decline of annual NMOCs collection rate below 50 Mg per year at Sites A, C, and G is potentially due to the relatively smaller amount of waste placed at these sites and extensive bioreactor operation of Site A and C. As mentioned earlier, entire waste mass deposited at the study site was used for the analysis presented in this section; only waste data from the study cells C1 and C2 were used for the analysis due to lack of waste disposal data for the other cells at Site C. The timeframes to achieve annual NMOCs collection rate reduction below 50 and 34 Mg per year are expected to be greater than those presented in

Table 5-6 for Site C if waste placement amounts for all the cells at this site were included. Site G was assumed to close in 2016.

Table 5-6. Estimated Timeframes for NMOCs Collection Rate and LFG Flow Rate Reduction

Site	Waste Amount (MT)	Landfill Gas Collection and Control System Operating Duration Until Annual NMOCs Collection Rate < 50 Mg per Year (Years)	Duration after Closure (Years)			
			Annual Nonmethane Organic Compounds (NMOCs) Collection Rate		Landfill Gas Flow Rate	
			< 50 Mg per Year	< 34 Mg per Year	<10% of Peak LFG Flow Rate	<5% of Peak LFG Flow Rate
A	2.05	10	11	13	11	13
B	3.27	21	8	10	10	13
C1 and C2	1.40	13	10	12	11	14
D	5.56	36	26	30	24	31
E	3.97	23	10	13	13	16
F	1.85	25	27	34	42	55
G	1.64	<0	<0	<0	7	10
H	3.46	32	21	24	20	26

The Sites A, F, and G contain less than 2.5 million metric tons of waste; MSWLFs containing less than 2.5 million metric tons of waste are exempt from the LFG control requirements of NSPS. The implementation of GCCS at Sites A, F and G were not dictated by NSPS. The termination of GCCS operation at these sites would, therefore, not be controlled by NSPS provisions. The GCCS at Site F was implemented to control subsurface migration of LFG, and energy generation from LFG appears to be the motive behind implementing a GCCS at Site G.

The GCCS operation at Site A was terminated after 13 years of operation due to flare operation issues once the LFG flow rate declined to 5% of the peak flow rate observed at the site. This suggests that GCCS operation may need to be terminated before the NSPS (Subpart WWW) 15-year operating duration requirement due to lack of adequate methane flow rate needed to sustain flare operation, especially for small MSWLFs such as Site A. At this point, the blower/flare system may need to be replaced with either a smaller capacity system or with alternative options such as biofilters to control any potential odor issues and/or to oxidize methane and NMOCs.

The analysis suggests that the annual NMOCs collection rate would be less than 34 Mg per year by the end of 30 years after closure for all of the study sites except Site F. The annual NMOCs collection rate for Site F is estimated to decline below 34 Mg per year 34 years after closure. However, the provisions of Subpart XXX are only applicable to the MSWLFs that commenced construction, reconstruction, or modifications after July 17, 2014. As mentioned earlier, Site F is not subjected to the requirements of NSPS (Subpart WWW or XXX). Given that all the study sites, except for Sites C and G, were closed before 2014, Subpart XXX provisions are not applicable to these study sites.

The timeframe to achieve LFG flow rates that are 5-10% of the peak flow rates are greater than those needed to achieve an annual NMOCs collection rate of 34-50 Mg per year for all the study sites. The analysis suggests that no major modification of the blower/flare system capacity would

be needed at any of the study sites to continue operating the GCCS until the annual NMOCs collection rate decline below the NSPS thresholds. The blower/flare systems at Sites D and H may also need to be replaced as the total estimated GCCS operating duration at these sites is over typical service life (15-25 years) of the blower/flare system.

5.3.5. Impact of In-Place Waste Amount and Decay Rate on Timeframes to Achieve Annual NMOCs Collection Rate Reduction below the NSPS Thresholds

The annual NMOCs collection rate is dependent on the LFG flow rate, which in turn is dependent on the in-place waste amounts, waste placement timing, and decay rate. The estimated timeframe for annual NMOCs collection rate reduction below 50 Mg per year for Site D is substantially greater than the other sites, which is potentially due to considerably higher in-place waste amounts compared to the other study sites (Table 5-6). The impact of the decay rate (k) on the duration for NMOCs collection rate reduction below the NSPS thresholds can be assessed by comparing these timeframes within study sites of similar size. For example, although Sites A and F are similar in size, the timeframes for annual NMOCs collection rate to decline below 50 and 34 Mg for Site A are lower than those for Site F. This distinction is potentially due to a considerably higher waste decay rate (k) at Site A than that of Site F. Similarly, the timeframe for the annual NMOCs collection rate to decline below the NSPS thresholds for Site B is substantially lower than that of the Site H, which is similar in size to Site B, potentially due to a substantially higher waste decay rate at Site B than that of Site H.

Figure 5-2 presents the distribution of the capacities of approximately 1,300 active and closed MSWLFs that reported annual GHG emissions to the EPA at least once during the 2010-2016 timeframe. Over 75% and 60% of the MSWLFs have a capacity of more than 4 and 5 million metric tons, respectively. The study sites sizes, therefore, are not representative of the size of approximately 75% of MSWLFs in the US, given that six of the study sites contain less than 4 million metric tons of waste. The total capacity of Site C, which was actively receiving waste at the time of this study, is unknown.

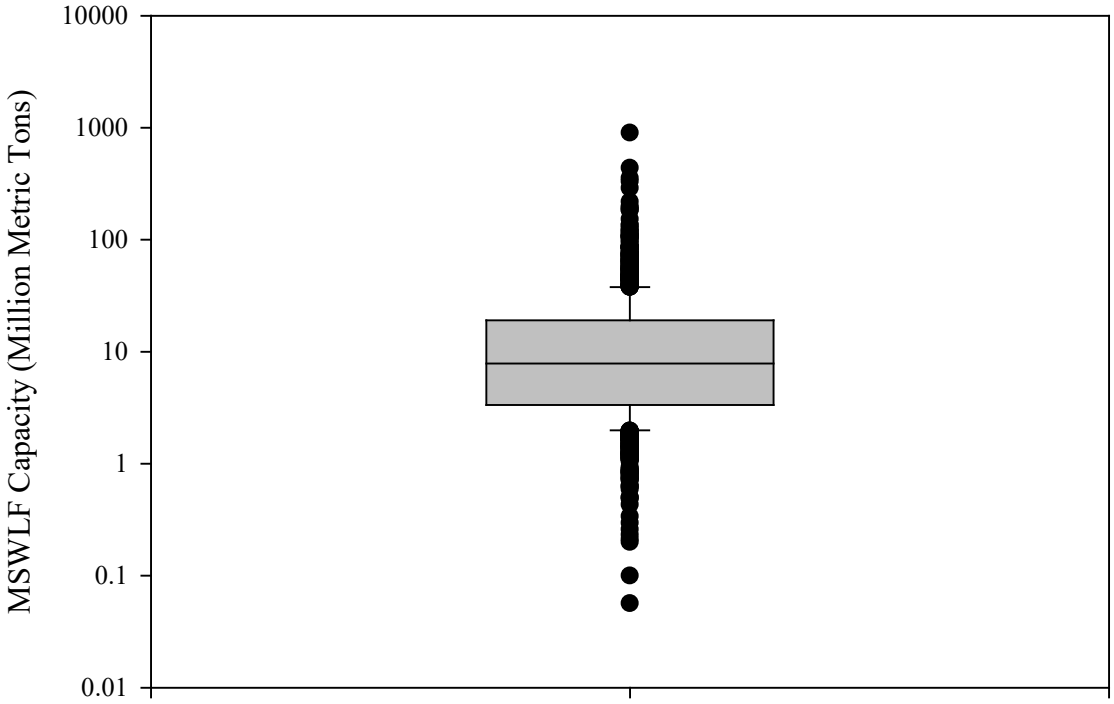


Figure 5-2. Distribution of Capacities of Active and Closed MSWLFs in the US

In order to assess the post-closure timeframes needed to achieve NMOCs reduction below the NSPS threshold for MSWLFs of varying sizes, the methane and NMOCs collection rates were estimated for three hypothetical MSWLFs that are representative of the 25th, 50th, and 75th percentile of the capacity of MSWLFs in the US. The 25th, 50th, and 75th percentile of the capacity of MSWLFs in the GHG database is approximately 3.35, 7.85, and 19.1 million metric tons, respectively. As mentioned earlier, the MSWLF containing in-place waste amounts corresponding to 25th, 50th, and 75th percentile MSWLF capacity in the US, is referred herein to as small, medium, and large MSWLF, respectively.

The annual disposal amounts for each of the three examples MSWLFs were calculated by dividing the total capacity by the respective median lifespan, then rounded to the nearest thousand. For the analysis presented in this section, the waste placement rate is assumed to be consistent over the life of the MSWLF. Table 5-7 summarizes the capacity, lifespan, and annual waste placement rate of the three example MSWLFs used for assessing the impact of the in-place waste volume and decay rate on the post-closure timeframe for achieving annual NMOCs reduction below 50 Mg per year. First-order decay modeling was conducted to estimate the annual methane and NMOCs collection rates for small, medium, and large MSWLFs using the lifespans and annual placement rate values for these MSWLFs as presented in Table 5-7.

Table 5-7. Capacity, Lifespan, and Annual Disposal Amounts of the Example MSWLFs

MSWLF Size Category	MSWLF Capacity (Million Metric Tons)	Capacity percentile Range for Lifespan Estimation	Median Lifespan (years)	Estimated Annual Placement Rate (Mg/year)
Small	3.35	15%–35%	43	78,000
Medium	7.85	40%–60%	58	135,000
Large	19.1	65%–85%	71	269,000

Table 5-8 presents the post-closure period to achieve annual NMOCs collection rate reduction below 50 Mg per year. The estimates suggest that the annual NMOCs collection rate of only small MSWLFs in the wet zone would decline below 50 Mg per year within 30 years after closure. The annual NMOC collection rates from the medium MSWLFs in moderate precipitation and arid zones are expected to decline below 50 Mg per year in 50 and 80 years after closure, respectively. As expected, the post-closure duration for annual NMOCs collection rate to decrease below 50 Mg per year is the longest for MSWLFs in the arid zone. It should be further noted site-specific conditions such as subsurface methane migration and odor issues may necessitate GCCS operation beyond the NSPS-required timeframe to ensure compliance with pertinent local, state, and federal regulations. The impact of the local and state regulations on GCCS operating duration was not assessed in this study.

The annual NMOC collection rates for all three MSWLFs sizes is expected to decline below 50 Mg per year within 20 years if these landfills are operated as a bioreactor. The bioreactor operation substantially reduces the post-closure duration for the annual NMOCs collection rate to decline below 50 Mg from 124 years to 13-17 years for large size MSWLFs in arid areas. As expected, the analysis suggests that the bioreactor operation can substantially reduce the GCCS operating duration for MSWLFs of all sizes, especially those located in arid and moderate climatic zones. Based on a review of data from ten Wisconsin MSWLFs that recirculated leachate and added external liquids waste, Bareither et a. (2018) reported that LFG generation rate is expected to decline below 5% of the peak rate and more than 75% of the LFG generation potential would be realized within 40 years of PCC at these sites.

The LFG flow rate declined to less than 10% of the peak LFG flow rate after the annual NMOCs collection rate reduced below 50 Mg per year for all simulation scenarios for small and medium MSWLFs. For large MSWLFs, the NMOCs collection rate declined to less than 50 Mg per year after the LFG flow rate declined to 10% of the peak LFG flow rate but before the LFG flow rate declined to 5% of the peak LFG flow rate. The analysis suggests that a blower/flare system that is designed based on the peak LFG flow rate can be used to collect and combust LFG until the annual NMOCs collection rate declines below 50 Mg. However, the service life (typically 15-25 years) of a blower/flare system is expected to be much shorter than the duration the GCCS would need to operate for NSPS compliance for the MSWLFs that are not operated as bioreactors.

The NSPS-default NMOCs concentration of 4,000 parts per million by volume as hexane was used for the analysis. NMOCs content of LFG has been reported to vary over a wide range. As a point of comparison, AP-42 suggests using NMOCs concentration of 2,420 parts per million volume as hexane for landfills known to have the co-disposal of MSW with non-residential waste and recommends 595 parts per million volume as hexane for landfill containing only MSW (EPA 1998). EPA (2007) reported NMOCs to range from 233 to 5,870 parts per million by volume as hexane based on measurements from five MSWLFs in the US. The annual NMOCs collection rate would decline below 50 Mg per year over a shorter duration than the estimates presented in the

chapter if the actual NMOCs concentration is lower than 4,000 part per million by volume as hexane. In addition, a constant NMOCs concentration was used for the entire duration as the data pertaining to the variability of NMOCs concentration in LFG over time are lacking.

Table 5-8. Impact of In-Place Waste Amount and Decay Rate on the Estimated Timeframes for Achieving Annual NMOCs Collection Rate Below 50 Mg per Year

Precipitation Zone/Operating Condition	Decay Rate, k (Year $^{-1}$)	Timeframe after Closure for Annual Nonmethane Organic Compounds (NMOCs) Collection Rate <50 Mg per Year (Years)		
		Small MSWLF	Medium MSWLF	Large MSWLF
Arid (< 20 Inches Annual Rainfall)	0.02	49	85	124
Moderate (20-40 Inches Annual Rainfall)	0.038	35	52	71
Wet (>40 Inches Annual Rainfall)	0.057	26	36	48
Bioreactor-Median of Reported k	0.17	9	13	17
Bioreactor-Average of Reported k	0.22	7	10	13

5.3.6. Potential of Elevated NMOCs Generation after GCCS Termination

As described earlier, NSPS allows termination of GCCS operation once the annual NMOCs generation rate has declined below 50 Mg (or 34 Mg) per year. The gas collection wells are, typically, retrofitted to passively vent the generated gas into the atmosphere once the GCCS operation is terminated. The annual NMOCs generation rate is expected to continue declining after GCCS termination until the methane generation potential of the waste is exhausted. Precipitation intrusion into the landfill due to a compromised final cover would increase the in-situ waste moisture content, and enhance the waste decomposition process and resultant methane and NMOCs generation/emission rates if the MSWLF at this point still contains undecomposed biodegradable organics with significant methane and NMOCs generation potential.

Figure 5-3 presents the estimated annual NMOCs generation rate for a small MSWLF in a moderate precipitation zone with no change and a 100% increase in decay rate ten years after the annual NMOCs generation rate declines below 50 Mg NMOCs per year. With an assumed 100% increase in decay rate to 0.076 year $^{-1}$, NMOCs generation rate increases to 70 Mg NMOCs per year and stays above the NSPS-threshold of 50 Mg NMOCs per year for four years. The assumed increased decay rate of 0.076 year^{-1} might not be unreasonable given that all the study sites except Site F, which is in an arid region, are estimated to have a decay rate of more than 0.076 year^{-1}.

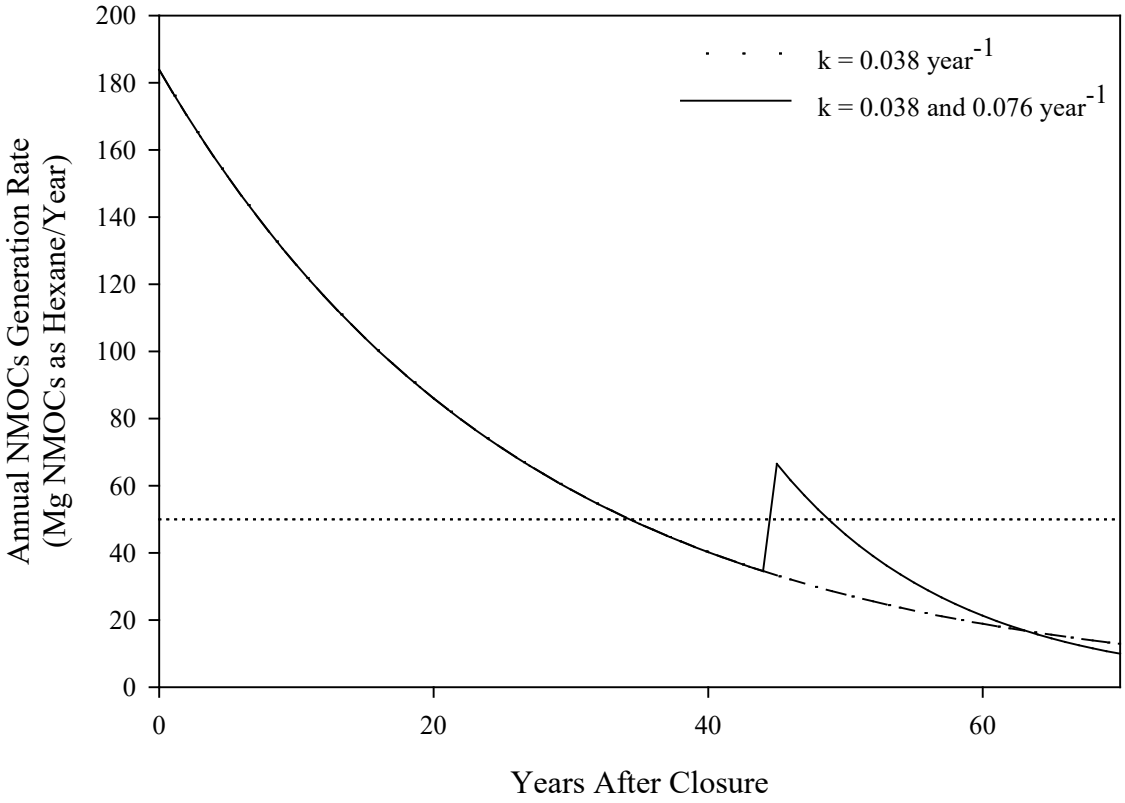


Figure 5-3. Estimated Annual NMOCs Generation Rate for Typical Small MSWLF in a Moderate Precipitation Zone with 100% Increase in Waste Decay Rate 10 Years after GCCS Termination

Figure 5-4 shows the estimated annual NMOCs generation rate after closure for a typical large MSWLF in an arid area. The annual NMOCs generation rate is estimated to decline below 50 Mg per around 124 years after closure. Per NSPS, GCCS operation can be terminated at this point. Assuming that PCC is also terminated and the final cover is no longer monitored and maintained at this point, any deterioration of the final cover system would result in an increase of moisture intrusion into the landfill. The NMOCs generation rate is estimated to increase to 81 Mg NMOCs per year and remain above 50 Mg NMOCs per year for 13 subsequent years if the waste decay rate doubles at the 134th year after closure due to moisture intrusion into the MSWLF. The oxidation of NMOCs and methane through the final cover soil is expected to be insignificant as the geomembrane underlying the final cover soil layer at most of the study cells would minimize gas migration through the final cover soil.

The decay rate was assumed to increase by 100% ten years after GCCS termination for this analysis. The likelihood of this increase within ten years of GCCS termination may be questionable due to relatively low permeability of MSW, and the presence of the final cover, even if compromised due to potential lack of maintenance, is expected to restrict the infiltration of moisture into the landfill. The intent of this analysis was to illustrate a scenario where the annual NMOCs emission rate can exceed the NSPS thresholds after GCCS termination with an increase in waste decay rate if the in-place waste has considerable remaining NMOCs and methane generation potential.

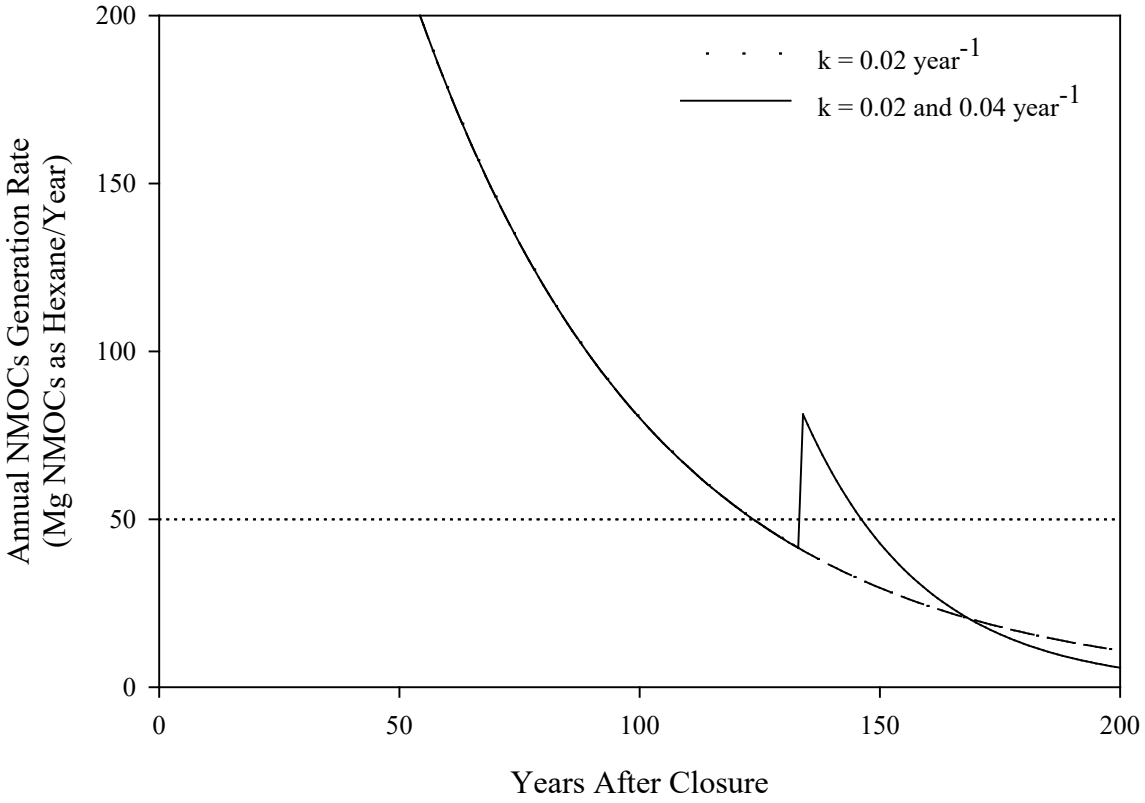


Figure 5-4. Estimated Annual NMOCs Generation Rate for Typical Large MSWLF in an Arid Area with 100% Increase in Waste Decay Rate 10 Years after GCCS Termination

The analysis presented above suggests that the annual NMOCs generation rate can spike back above the NSPS in years after GCCS operation is terminated. The final cover, therefore, should be rigorously maintained even after GCCS termination until the NMOCs generation potential and the leaching potential of the in-place waste has declined to levels that are unlikely to pose a risk to HHE. In addition, landfill owners and regulators should also continue surface and subsurface emissions and odor monitoring to proactively identify signs of an increase in LFG generation rate and have provisions in place to resume GCCS operation, if needed, to control these issues.

The reliance on the importance of the final cover to alleviate the long-term HHE impacts associated with GCCS operation termination can be mitigated by implementing operating strategies such as bioreactor that can stabilize waste while the final cover is actively monitored and maintained (Morris and Barlaz 2011, Bareither et al. 2018).

In addition to the NSPS criteria, the landfill owners and regulators should also consider evaluating the remaining methane and NMOCs generation potential of the in-place waste and assessing the likelihood of future emissions based on site-specific factors including but not limited to provisions for the final cover monitoring and maintenance, sub-surface and surface methane emission monitoring, and plan to resume GCCS operation any future odor/emission issues for decisions pertaining to GCCS termination.

5.3.7. Estimated Remaining Methane and NMOCs Generation Potential

The magnitude of the hypothetical spike and duration of elevated annual NMOCs generation rate would depend on the remaining methane and NMOCs generation potential of the in-place waste. An assessment of the remaining methane generation potential for a landfill at the end of 30 years after closure, therefore, is important. Table 5-9 presents the estimated percent of total methane generation potential remaining at closure and at the end of 30 years after closure. It also presents the methane generation potential remaining at closure and at the end of 30 years after closure. The remaining methane generation potential at closure ranged from 2.2 m^3 CH_4 per Mg of waste (for Site G) to 73.6 m^3 CH_4 per Mg of waste (56% of the total of 132 m^3 CH_4 per Mg of waste for Site F) for the study sites; Site G is an active site and was assumed to close in 2016 for the purpose of the analysis. The remaining methane potential at Site G declined below 10% before 2016 (assumed closure year) due to a significant decline in the MSW disposal amounts at the site since 2007.

The remaining methane generation potential was estimated to range from less than 0.1 m^3 CH_4 per Mg of waste to approximately 13.9 m^3 CH_4 per Mg of waste for the study sites 30 years after the closure. The remaining methane potential is estimated to be less than 2 m^3 CH_4 per Mg of waste for all of the sites except for Site F at the end of 30 years after closure. Site F, which is located in the arid area, is estimated to have the greatest remaining methane generation potential at closure and 30 years after closure due to the lowest estimated decay rate among the study sites.

Table 5-9. Estimated Percent Remaining Methane Generation Potential of the Study Sites at Closure and 30 Years after Closure

Site	Remaining Methane Generation Potential at Closure		Remaining Methane Generation Potential 30 Years after Closure	
	%	m^3 CH_4 per Mg	%	m^3 CH_4 per Mg
A	26%	39.2	0.02%	<0.1
B	13%	11.3	0.02%	<0.1
C	29%	36.8	0.07%	<0.1
D	48%	32.9	2.9%	2.0
E	23%	13.2	0.11%	<0.1
F	56%	73.6	11%	13.9
G	6%	2.2	0.01%	<0.1
H	43%	40.5	1.4%	1.3

The remaining methane generation potential of 25% and 10% are used by some states as a benchmark for GCCS operation modifications. All the study sites, except Site F, are expected to have less than 10% remaining methane generation potential within the timeframes estimated for NMOCs collection rates to decline below 50 Mg per year threshold. Site F is expected to achieve a remaining methane potential of 25% and 10% in 15 and 31 years after closure, respectively.

The percent remaining methane or NMOCs generation potential is not a reliable measure for assessing potential HHE impacts. Figure 5-5 presents the annual NMOCs generation rate as a function of the percent remaining methane generation potentials for a small MSWLF in a moderate precipitation zone and a large MSWLF in an arid zone. For the same percent remaining methane generation potential, the annual NMOCs generation/emission rate is greater for the large MSWLF compared to the small MSWLF. The percent remaining methane potential corresponding to the annual NMOCs generation rate of 50 Mg per year for the small MSWLF and the large MSWLF is approximately 13% and 4.6%, respectively.

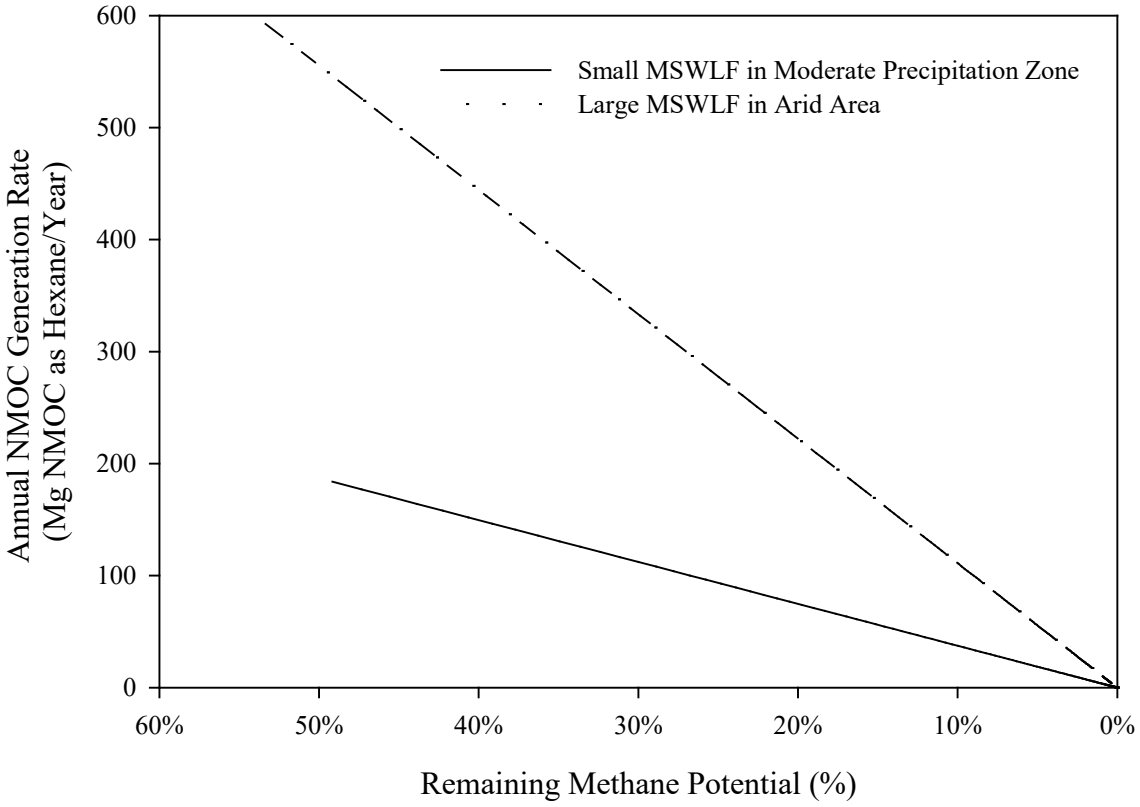


Figure 5-5. Annual NMOCs Generation Rate as a Function of the Percent Remaining Potential for Small MSWLF in Moderate Precipitation Zone and Large MSWLF in Arid Area

The mass-based remaining methane (Mg methane) and NMOCs generation potential (Mg NMOCs) are more appropriate indicators of the HHE impacts than the percent remaining methane and/or NMOCs generation potential. Table 5-10 presents the mass-based methane and NMOCs generation potential remaining at closure and 30 years after closure, and assuming an NMOCs concentration of 4,000 parts per million by volume as hexane. All of the sites except Sites D, F, and H are estimated to have less than 10 Mg of NMOCs generation potential at the end of the 30-year post-closure period. The analysis suggests that Sites A, B, C, E, and G would not have the potential to generate NMOCs above 50 Mg per year 30 years after closure. Sites D, F, and H are estimated to have approximately 321, 7380, and 134 Mg, respectively, of the total remaining NMOCs generation potential at the end of 30 years after closure.

The remaining NMOCs generation potential is dependent on the in-place waste amounts and the decay rate. Although the in-place waste volume at Site F (located in an arid area) is smaller than that of Site D, the remaining NMOCs generation potential for Site F is more than that of Site D. Based on the timeframes presented in Table 5-6, the GCCS at Site F can be terminated around the 27^{th} year after closure as annual NMOCs generation rate declines below 50 Mg per year at this point. However, without active LFG control, Site F has a potential for HHE impacts from methane NMOCs release if all the remaining NMOCs are released within a short period after 30 years of the post-closure period.

Table 5-10. Estimated Remaining NMOCs Generation Potential of the Study Sites at Closure and 30 Years after Closure

Site	Remaining Methane Generation Potential (MT methane)		Remaining NMOCs Generation Potential (MT NMOCs)	
	at Closure	30 Years after Closure	at Closure	30 Years after Closure
A	54,641	38	2,329	2
B	25,119	40	1,069	2
C	34,975	80	1,492	3
D	124,265	7,582	5,265	321
E	35,629	168	1,510	7
F	92,727	17,482	3,913	738
G	2,220	4	105	<1
H	94,998	3,158	4,034	134

It should be noted that the mass-based remaining methane and NMOCs generation potential are estimated based on site-specific L_c estimates, which corresponds to the LFG fraction that is captured by the GCCS. The use of L_c may result in an underestimation of the actual methane generation rate for the cases where part of the waste may not be decomposing due to conditions such as inadequate moisture content and for sites with poor gas collection efficiency. The LFG collection efficiency is not expected to be the case with NSPS compliant systems. For the cases with L_c estimate that is substantially lower than AP-42 recommended value of 100 m^3 CH_4 Mg^{-1} waste, the landfill owners and regulators should consider a comprehensive evaluation of site-specific data such as amounts of inerts in the landfill, level and frequency of surface and sub-surface methane emissions, landfill surface settlement rate and the volume loss rate to assess the appropriateness of L_c for estimating the remaining methane generation potential. For the sites that are evaluated to contain waste that is not decomposing and/or have substantial fugitive emissions, the use of AP-42 default L_o value (in instead of site-specific L_c estimate) should be considered for estimating the remaining methane and NMOCs generation potentials.

The remaining methane and NMOCs generation potential can also be estimated by collecting samples of deposited waste and measuring the biochemical methane and NMOCs generation potential in a laboratory. As the remaining methane generation potential depends on waste age and composition, samples across the landfill area and depth should be collected to account for the variability in the composition and the age of the deposited waste. Several studies (e.g., Townsend et al. 1996, Mehta et al. 2002, Tolaymat et al. 2010, Kim and Townsend 2012) have reported methane generation potential based on this approach. Potential damage to the final cover geomembrane and landfill infrastructure (e.g., horizontal LFG collector, buried LFG header and lateral), extensive sample collection and analysis, and cost are the major disadvantages of this approach.

5.4. Summary

LFG emissions are one of the primary pathways for the HHE impacts of MSWLFs. The RCRA Subtitle D regulations require the installation of subsurface monitoring probes around the periphery of the site and the quarterly monitoring of these probes and structures at the facility for methane. Subsurface methane monitoring probe data from seven study sites suggest relatively few exceedances detected in the subsurface and structural methane monitoring. Methane exceeded the

lower explosive limit of methane (i.e., 5% methane by volume in air) in 103 instances (i.e., approximately 1.4% of all 7,598 measurements included in the evaluation). Approximately 80% of these exceedances were observed at Site E and 10% of exceedances occurred at Site B. Both these sites contained either unlined cells or cells lined with a compacted clay liner, which are likely the cause of the observed methane exceedances at these sites. All the exceedances correspond to three sites during the first five years after closure for all the sites except for Site E. The GCCS at Site A was terminated in the 13th year of closure. The available perimeter probe monitoring data do not show any methane exceedance since the termination of GCCS at Site A. The structure methane concentration data (2,105 measurements) were also available for Sites B, F, and H. None of the methane measurements in structures at Sites B and H were above 25% of its LEL. Only 5 out to the total 1,155 methane measurements in Site F structures were above 25% of its LEL.

The federal NSPS requires active collection and control of LFG from MSWLFs that produce more than 50 Mg of NMOCs (as hexane) annually. The GCCS at the NSPS-regulated MSWLFs should be operated for at least 15 years, and until the annual NMOCs generation rate is more than 50 Mg per year. The future NMOCs generation rate can be estimated using the LandGEM model based on waste decay rate, in-place waste amount and placement timing, methane generation potential of waste, and NMOCs content of LFG. NSPS allows using the measured LFG collection rate and the site-specific NMOCs concentration, if available, for NMOCs generation rate estimation.

All of the study sites have/had a GCCS. The GCCS monitoring data were available for review only for eight of the case-study sites. A regression analysis was conducted to estimate a site-specific methane collection potential and decay rate that provided the best fit to the available monthly methane flow rate data when used with the first-order decay model. The methane flow rates were calculated using the LFG flow rate and methane content unless the methane flow rates were directly available. The estimated site-specific decay rates suggest that waste decomposition at all of the eight study sites (with GCCS data) is occurring more rapidly than the decay rates specified/recommended by NSPS, AP-42, and GHG Reporting regulations. The estimated decay rate for four out of eight sites was within a range of 0.18-0.21. As expected, k estimated for Site A is the highest among all of the sites, potentially due to the extensive leachate recirculation conducted at the site. The estimated k was lowest for Site F, which is located in the driest area among all the study sites. The estimated methane collection potential ranged from 36-152 m^3 CH_4 per Mg waste. Sites E and G had a relatively low methane collection potential due to the disposal of considerable inert materials at the study cells at these sites. Limited LFG collection infrastructure in one of the study cells also contributed to low methane collection potential at Site G.

The future annual methane and NMOCs collection rates were estimated for the study sites using the first-order decay model based on site-specific decay rate and methane collection potential, study site disposal amounts, and an NSPS-default NMOCs concentration of 4,000 parts per million by volume as hexane. The estimated NMOCs and methane collection rate data were then used to assess the post-closure period needed to achieve (1) the annual NMOCs collection rate below 50 (and 34) Mg per year, and to (2) LFG flow rates that are 5% and 10% of the peak LFG flow rate.

The annual NMOCs collection rates from all of the study sites are expected to decline below 50 Mg per year within 30 years after closure. The annual NMOCs collection rate estimated to decline below 34 Mg per year within 30 years after closure for all the study sites except Site F. The remaining methane generation potential at all of the study sites, except Site F, is expected to be

less than 10% of the total generation potential before the NMOCs collection rates decline below 50 Mg per year.

The study site sizes, however, are not representative of the size of approximately 75% of MSWLFs in the US as six of the study sites contain less than 4 million metric tons of waste. The NMOC generation/emission rate is dependent on the landfill size and, therefore, the required GCCS operating duration estimated for the study site may underestimate the duration for typical MSWLFs in the US. Over 75% of the MSWLFs included in the EPA GHG database have a capacity of more than 4 million metric tons. In order to assess the post-closure timeframes to achieve NMOCs reduction below the NSPS threshold for typical size MSWLFs in the US, NMOCs and methane flow rates were estimated for MSWLFs containing approximately 3.35 (small MSWLF), 7.85 (medium MSWLF), and 19.1 (large MSWLF) million metric tons MSW. These sizes correspond to the 25^{th}, 50^{th}, and 75^{th} percentile of the capacity of MSWLFs in the US, respectively.

In order to assess the impact of decay rate on the annual NMOCs collection rate modeling was conducted for each of the three example MSWLFs for the following decay rates: 0.02 (representative of arid areas with less than 20 inches annual precipitation), 0.038 (representative of moderate precipitation regions with 20-40 inches annual precipitation), 0.057 (representative of wet regions with more than 40 inches annual precipitation), 0.17 (reported median for bioreactor landfills), and 0.22 year $^{-1}$ (reported average for bioreactor landfills).

The analysis suggests that the annual NMOCs collection rate for MSWLFs containing more than 3.35 million MT waste and located in arid and moderate precipitation areas are not expected to decline below 50 Mg per year within 30 years after closure. The NMOCs collection rate for MSWLFs containing more than 7.85 million MT waste is not expected to decline below 50 Mg per year within 30 years after closure irrespective of its location. The annual NMOC collection rate of MSWLFs that are operated as a bioreactor (with a decay rate of 0.17 year $^{-1}$ or 0.22 year $^{-1}$) is expected to decline below 50 Mg per year within 20 years after closure. The bioreactor operation can substantially shorten the GCCS operating timeframe, especially for medium and large MSWLFs. The LFG flow rate is expected to decline below 5% of the peak flow rate after the annual NMOCs collection rate reduces below 50 Mg per year for all of the scenarios modeled.

The analysis suggests that the annual NMOCs generation rate can spike above the NSPS threshold of 50 Mg per year after GCCS operation termination with an adequate increase in the decay rate; the decay rate was assumed to increase by 100% for the analysis presented in this report. The deterioration of the final cover and ensuing moisture intrusion may result in an enhanced waste decay rate. The stakeholders should consider monitoring and maintaining the final cover even after GCCS termination until the remaining generation potential and leaching potential of the in-place have declined to a level that is not likely to pose a risk to HHE in the event of the final cover failure. In addition, landfill owners and regulators should also continue surface and subsurface emissions and odor monitoring after GCCS termination to proactively identify signs of an increase in LFG generation rate, surface and subsurface emissions, and odor issues, and have provisions in place to resume GCCS operation, if needed.

Several states use percent remaining methane generation potential as a criterion for assessing the impact of modifications to GCCS operation. The analysis suggests that the percent remaining methane potential, however, is not an appropriate metric to assess the HHE impacts. A smaller percent remaining methane generation potential at a large MSWLF may pose a greater risk than a

small MSWLF with relatively higher percent methane generation potential. A mass-based threshold (e.g, NMOCs threshold used by NSPS) would be a more appropriate metric than a percent-based criterion for GCCS modifications/termination.

In addition to the evaluation of the study sites, this chapter also presents approaches that the landfill owners/engineers can use to estimate the site-specific decay rate and remaining methane and NMOCs generation potential using the LFG flow rate and composition data that are typically measured at MSWLFs in the US.

5.5. Limitations

The analysis presented in this chapter has the following limitations:

1. The study sites/cells size (with respect to in-place waste amounts) are smaller compare to the size of currently operating MSWLFs in the US. Three of the sites contain less than 2.5 million metric tons of waste; MSWLFs containing less than 2.5 million metric tons of waste are exempt from the requirements of NSPS. The landfill gas assessments pertaining to these landfills may not be applicable to a majority of the MSWLFs operating in the US.
2. The available subsurface probes and facility structures data suggest infrequent detection and exceedances at subsurface probes and in facility structures. However, monitoring data were not available for all of the monitoring events conducted after closure. The facility structure monitoring data were available for only three sites.
3. Limited LFG methane content data were available for a few of the study sites. For example, methane content data were available for a limited duration for Site D. The methane content of the LFG can vary considerably (typically varies between 40% to 55% by volume) over time depending on modifications to the operating conditions of GCCS.
4. The site-specific estimates of k are based on the assumption that the LFG collection efficiency is constant over the timeframe that the LFG data are used for regression analysis. The actual measurement of LFG collection efficiency is typically not conducted for MSWLFs. Any variation in the collection efficiency over the duration of the data used for regression analysis would impact the accuracy of the decay rate estimates.
5. Site-specific methane collection potential (L_c) estimate was used for estimating the remaining methane and NMOCs potential for the study sites. It should be noted that L_c values are representative of the LFG collection rate and not the generation rate. The actual remaining methane and NMOCs generation potential are expected to be greater than the one estimated based on L_c values. AP-42 default L_o value of 100 m^3 per Mg waste was used to model LFG generation and corresponding NMOC generation at examples sites. Variations in the methane generation potential would lead to a concomitant change in total NMOC generation rate, which is modeled as a fraction of total LFG in this work. In addition, an LFG collection efficiency of 100% was assumed. A lower collection efficiency would result in proportionally smaller timeframes to achieve the NSPS thresholds for GCCS termination than estimated in this study.
6. The NSPS-default NMOCs concentration of 4,000 parts per million by volume as hexane was used for the analysis. NMOCs content of LFG has been reported to vary over a wide range. As a point of comparison, AP-42 suggests using NMOCs concentration of 2,420 parts per million volume as hexane for landfills known to have the co-disposal of MSW with non-residential waste and recommends 595 parts per million volume as hexane for landfill containing only MSW (EPA 1998). EPA (2007) reported NMOCs to range from 233 to 5,870 parts per million by volume as hexane based on measurements from five

MSWLFs in the US. The annual NMOCs collection rate would decline below 50 Mg per year over a shorter duration than the estimates presented in the chapter if the actual NMOCs concentration is lower than 4,000 part per million by volume as hexane. In addition, a constant NMOCs concentration was used for the entire duration as the data pertaining to the variability of NMOCs concentration in LFG over time are lacking. A change in NMOCs concentration over time would concomittantly change the NMOCs emission rate and GCCS operating duration.

7. The analysis only estimated the methane and NMOCs collection rates and the remaining generation potential for methane and NMOCs. These estimates can be used for an assessment of the HHE impacts. However, an HHE impact assessment was not conducted in this study.
8. NSPS annual NMOCs generation rate thresholds were used for the analysis. However, NMOCs emissions below the regulatory threshold may not necessarily suggest the absence of HHE impacts from the long-term emission of NMOCs. Modeling approaches such as life cycle assessment and contaminant fate and transport modeling coupled with risk assessment can be used to estimate the HHE impacts of methane and NMOC emissions associated with the termination of GCCS operation.
9. Only methane and NMOCs generation rates were modeled in this study. The LFG has been reported to contain other trace contaminants (e.g., mercury) that may have significant HHE impacts.
10. It should be noted that the literature-reported LFG data and/or values observed at other sites cannot be used to reliably assess site-specific impacts of terminating GCCS operation on HHE due to the large magnitude of variation reported in the literature. Therefore, the data from the study sites should not be used as a proxy for conducting a reliable site-specific post-closure period assessment. The approaches presented in this chapter can be used to estimate the site-specific remaining methane and NMOCs generation rate potential and rates that can be used as inputs for a more reliable assessment of the HHE impacts.
11. The site-specific estimates suggest that the decay rates recommended/specified by AP-42, and NSPS and GHG Reporting regulations underestimate the waste decomposition rates even at the study sites that were not operated as a bioreactor. Due to a small number of sites analyzed in this study, a statistical evaluation could not be performed to estimate the representative decay rate for different precipitation zones of the US. A majority of the MSWLFs are required to report the collected methane amounts along with other data such as annual disposal amounts to the EPA. Future research efforts should consider using the reported data from these MSWLFs to estimate a site-specific decay rate of a large number of MSWLFs in the US.
12. Site-specific conditions such as subsurface methane migration and odor issues may necessitate GCCS operation beyond the NSPS-required timeframe to ensure compliance with pertinent local, state, and federal regulations. The impact of the local and state regulations on GCCS operation duration was not assessed in this chapter. Furthermore, GCCS may be operated beyond the timeframe to comply with NSPS to support the ongoing LFG beneficial use project, if applicable. The GCCS operation in these cases would, probably, be not regulated by the NSPS and NESHAP regulations.

6. Landfill Leachate

6.1. Overview

Based on the PCC cost information gathered for the sites reviewed in this study, leachate management cost constituted a significant fraction of the total PCC cost for several of the study sites. The termination of leachate collection and management appears to represent a major cost saving for the closed sites. Subtitle D regulations require maintaining and operating the leachate collection system (referred herein to as LCS) in accordance with §258.40 until the "*owner or operator demonstrates that leachate no longer poses a threat to human health and the environment.*" Both the leachate generation rate and chemical characteristics dictate the potential loading rate of contaminants to groundwater or surface water systems associated with leakage through the bottom liner or seepage through landfill side slopes or toe after the LCS operation is terminated. This evaluation, thus, assessed the study sites with respect to both leachate generation rate (e.g, gallons per acre-day) and leachate chemical characteristics.

The federal Subtitle D landfill regulations do not specify numeric criteria for when a site's leachate generation rate is sufficiently low enough to terminate PCC. However, the range of state-specified criterion for LCS system operation termination/modification varies from no leachate generation to historically low or steady leachate generation rate (FDEP 2016; VDEQ 2007; UDSHW 2012; WADOC 2011; WDEQ 2000). The first objective of this chapter is to analyze the leachate collection rate data from the study sites to assess whether any of the study sites have achieved no or historically low and steady leachate generation rate after closure. As only collection rates and not generation rates are monitored at MSWLFs, The reported collection rates were used as a proxy for the leachate generation rate throughout the analyses presented in this chapter.

Objective 1. Analyze leachate collection rate data to determine whether any of the study sites have achieved a zero or historically low and stable leachate collection rate after closure.

One of the desired outcomes from landfill closure is a long-term reduction in leachate production from a site. As part of the process of developing PCC cost estimates, the long-term leachate generation rate for a site must be estimated. Estimates of future leachate generation rates can also be used to evaluate HHE impacts and necessary timeframes to achieve no or historically low leachate generation rate thresholds used by some states for LCS operation termination. The second objective of this chapter is to evaluate different approaches that can be used to estimate the long-term leachate generation rate from a closed MSWLF.

Objective 2. Assess approaches that can be used to estimate a site's long-term leachate generation rate. These approaches can also be used for estimating the timeframes for leachate generation to decline below a state-specific threshold.

As described above, HHE impacts associated with leachate emissions after LCS termination depend in part on the chemical characteristics of a landfill's leachate. Both the types of chemicals occurring in the leachate and their concentrations are important for HHE impact assessment. The Subtitle D regulations do not require routine characterization of leachate chemical composition, and only some state regulatory programs require the collection and reporting of such data. As groundwater contamination is the primary pathway for landfill leachate to impact HHE, some states recommend that MSWLF owners conduct a comprehensive characterization of leachate composition with respect to groundwater quality monitoring parameters (e.g., FDEP 2016,

UDSHW 2012). The third objective of this chapter is to evaluate the comprehensiveness of the available leachate quality data with respect to the parameters specified in the federal regulations for groundwater monitoring.

Objective 3. Evaluate the comprehensiveness of the available leachate quality data with respect to the parameters specified by the federal regulations for groundwater monitoring.

Leachate generated by a Subtitle D landfill, even at the end of the PCC period, is not expected to be of sufficient quality for human consumption. The typical practice for assessing HHE impacts from a closed landfill is to evaluate the quality of the water expected at the point of compliance (for example, a groundwater monitoring well at the perimeter of the landfill site). This process involves using the leachate generation rate and chemical characteristics as inputs into a chemical fate and transport model to estimate concentrations of chemicals of concern at the compliance point(s), and comparing these estimated concentrations to risk-based protection standards. Fate and transport models require much more site information than leachate generation rate and quality. A common screening step is to compare leachate concentrations directly with risk-based protection standards to assess the degree of dilution and attenuation necessary to meet HHE objectives at the point of compliance. This screening process also allows an assessment of which leachate constituents (referred herein to as the contaminants of potential concern [CPOCc]) are likely to be most limiting with respect to terminating LCS operation. The fourth objective of this chapter is to conduct a screening analysis to identify leachate contaminants at the study sites with concentrations greater than the respective risk-based threshold.

Objective 4. Conduct a screening analysis to identify the contaminants that have been frequently measured in leachate above the respective human health risk-based protection standards at the study sites after closure to identify the COPCs.

The concentrations of contaminants and the associated HHE impacts are expected to vary over time. The screening analysis described above for identifying the COPCs does not account for temporal variation of concentrations. The contaminants that were initially measured above the respective protection standard but have declined below the respective risk-based protection standards over time are not expected to be an HHE concern. Some state guidance documents recommend demonstration of declining or steady concentration for the COPCs as a criterion for evaluating termination or scaling back of LCS operation. The fifth objective of this chapter is to evaluate temporal trends of the COPCs concentrations to assess whether the concentration of these contaminants has declined below the respective risk-based threshold over time after closure.

Objective 5. Assess whether the concentration of the COPCs identified based on the screening analysis has declined below the respective protection standard over time after closure.

Robust analysis methods and approaches are critical for a reliable estimation of emissions. An additional objective of this chapter is to present and discuss the approaches that can be used for estimating the long-term leachate generation rate after closure. The analysis presented in the chapter should not be considered as a comprehensive evaluation due to various assumptions and limitations. These assumptions and limitations are presented along with the analysis. These limitations are also summarized in the last section of this chapter.

6.2. Data Sources

A review of peer-reviewed literature and government publications (Moody and Townsend 2017; Masoner et al. 2016; Townsend et al. 2015b; Masoner et al. 2014; Andrews et al. 2012; NCSU and ERG 2011; Barlaz et al. 2010; Sizirici et al. 2010; US EPA 2006; Benson et al. 2005; Statom et al. 2004; Ward et al. 2002; Ecobalance 1999) was performed to understand the extent of existing data on leachate quality from MSWLFs in the US. In summary, the majority of the sources identified did not present data from closed landfills. Moreover, the data for each study were limited to a small geographic region of the US or a relatively small timeframe.

An understanding of long-term leachate generation rate and quality are critical for assessing site-specific HHE impacts of terminating or scaling-back of LCS operation. The availability of leachate quantity and quality data was one of the criteria for selecting sites for this study. The study sites are located in different regions of the US, and each has leachate quantity and quality data available for several years since closure. While leachate quality data were available from all nine sites reviewed for this study, Site G did not separately track the quality of leachate from its closed cells.

As mentioned earlier, federal regulations do not require routine leachate generation rate and quality monitoring for MSWLFs. Some SEAs require routine leachate quantity and quality data tracking. In addition, leachate collection volumes and quality from MSWLFs are commonly monitored despite the absence of requirements to meet the contractual requirements of WWTPs in instances where leachate is managed through disposal at a WWTP. Although national-scale data are lacking, a handful of state-specific studies suggest that WWTP treatment of MSWLF leachate is the most prevalent management option for leachate management (Reinhart 2017; Townsend et al. 2015b; Wilson et al. 2011).

Table 6-1 shows a summary of the availability of leachate quality and quantity data from the nine study sites starting from the closure year (i.e., Year 0) to the latest available data (i.e., 2016) at the time of data collection for this study. Sites C and Site G had leachate data available from two separate cells. Of the nine sites, four are constructed with a double liner with a secondary leachate collection and removal system (herein referred to as *leak detection system* or *LDS*), and the rest were constructed with a single/composite liner system. Six of the sites had leachate quantity data available for more than ten years since closure. Two sites documented leachate recirculation after closure and the amount of leachate recirculated at these sites since the closure was available. Leachate quality data were available for eight of the sites, with most of the sites having at least ten years of post-closure leachate quality data. The leachate quality data for these eight sites except Site I were available for a majority of duration after closure. LDS leachate quality data were available for two sites.

Table 6-1. Summary of Availability of Leachate Quantity and Quality Data at the Nine Study Sites Since Closure

Site		A	B	C		D	E	F	G		H	I
				C1	C2				G3	G4		
Geographical Regions of the US [1]		SE	NW	NE	NE	SE	NE	SW	NE	NE	NE	NE
Closure Year		1998	2004	1998	1998	2009	2004	1997	1993	2000	2008	1997
Years since Closure (as of 2016)		18	12	18	18	7	12	19	23	16	8	19
Has LDS [2]		No	No	Yes	Yes	No	No	No	Yes	Yes	Yes	Yes
Area associated with Leachate Quantity and Quality Data (acres)		28	45	17.5	23.3	69	61	38	7.1	7.9	60	51
Leachate Quantity	**Years Data Available for LCS [3,4]**	0-16	4-12	0-18	0-18	0-7	0-12	1-6, 8-19	7-20, 22-23	0-13, 15-16	0-8	0-12
	Years Data Available for LDS [2,4]	N/A	N/A	No	No	N/A	N/A	N/A	3-23	0-16	0-8	0-12
	Was Leachate Recirculated After Closure?	Yes	No	No	No	No	Yes	No	No	No	No	No
	Years Leachate Recirculated after Closure[4]	5-16	N/A	N/A	N/A	N/A	7, 9-11	N/A	N/A	N/A	N/A	N/A
	Amount Leachate Recirculated (%) [5]	74%	N/A	N/A	N/A	N/A	8%	N/A	N/A	N/A	N/A	N/A
Leachate Quality	**Are Data Available for LCS/composite leachate for Closed Cell(s)? [3, 6]**	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	Yes	Yes
	Years Data Available for LCS/composite leachate [3,4,6]	0-16	0-11	0-18	0-18	0-7	0-12	0-19	N/A	N/A	0-8	0-12
	Data Available for LDS [2]	N/A	N/A	No	No	N/A	N/A	N/A	No	No	Yes	Yes
	Years Data Available for LDS [2,4]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3-8	0-12

Notes:

(1) SE: Southeast; NW: Northwest; NE: Northeast; SW: Southwest
(2) LDS: Leak detection system or secondary leachate collection and removal system
(3) LCS: Primary leachate collection and removal system
(4) Considering closure year as year zero (0)
(5) Percentage of leachate recirculated of the leachate collected since the closure
(6) Composite leachate represented mixed leachate collected from LCS and LDS

N/A Not applicable

6.3. Methodologies for Prediction of Long-Term Leachate Collection after Closure

6.3.1. Overview

A reliable estimate of the long-term leachate generation rate is needed to assess the impacts on HHE from termination of LCS operation; leachate would not be collected and treated after terminating the LCS operation. This section presents a description of the following three approaches that were used to estimate the future leachate generation rate from the study sites: first-order decay modeling, unsaturated flow modeling, and Hydrologic Evaluation of Landfill Performance (HELP) modeling.

6.3.2. First-Order Decay Modeling

The leachate generation rate is dependent on the moisture-holding capacity of the in-place MSW and in-place moisture content, which in turn depends on various factors, including the nature of the waste and the annual infiltration during operation. A bulk of the moisture-holding capacity of MSW is due to the presence of paper, cardboard, and other paper products. The moisture-holding capacity of MSW is expected to decline over time as these materials biodegrade over time at a rate that typically follows a first-order decomposition rate equation. The leachate generation rate is expected to mimic the moisture-holding capacity loss rate. Following the first-order decay model, the mass-loss rate and the associated rate of the moisture-holding capacity loss are expected to decline, which in turn would result in a concomitant decrease in the leachate generation rate over time. As presented later, leachate collection rate data for six of the study sites visually appears to follow a first-order decay pattern. The following first-order decay equation was used to model leachate generation for all the study sites except Site A, which recirculated 100% of the leachate for a majority of the period after closure. EPA (2017) used a similar exponential-decay modeling approach for estimating the future leachate collection rates based on the historical leachate collection rates for Subtitle C landfills.

$$Q_t = k_l \, V_o M_o e^{-k_l t} \qquad \text{(Eqn. 6-1)}$$

Where,

Q_t = leachate flow rate at a given time (m^3/year)

V_o = total leachate generation potential of MSW after closure (m^3/Mg). The total leachate generation potential represents the total leachate volume that would be generated over an infinite time horizon per unit waste mass. It is not expected to be significantly impacted by weather conditions such as rainfall and evapotranspiration after installation of the final cover as it is designed to minimize infiltration into the landfill.

M_o = total in-place mass of waste in place (Mg, wet weight basis)

k_l = first-order leachate collection decay rate (year $^{-1}$)

t = time since closure (years)

A regression analysis was conducted using the Excel Solver add-in to minimize the sum of the squared difference between the modeled and measured data to find V_o and k corresponding to the first-order decay curve that fit the reported leachate collection rate the best. Monthly leachate

collection rate data, if available, were used for the analysis. A V_o of 0.5 m^3/Mg and a k_l of 0.05 year $^{-1}$ were used as the initial conditions for Solver to initiate the calculations.

6.3.3. Unsaturated Flow Modeling

Once the landfill is closed, only the stored moisture and hydraulic properties of MSW are expected to impact the leachate percolation rate within the landfill. The weather conditions (specifically rainfall and evapotranspiration) are not expected to have a significant impact as the final cover minimizes precipitation infiltration and evapotranspiration out of the landfill. Assuming a unit gradient (i.e., flow is only driven by gravity and not leachate pressure in the landfill), the following equations can be used to model the annual/monthly leachate vertical percolation rate (leachate generation rate) through MSW. These equations are the basis of the approach used by the HELP model to simulate vertical percolation of moisture in the landfill (Schroeder et al., 1994). A similar approach for simulating vertical percolation in landfills has been used by other models (e.g., US EPA 1987).

$$Q_i = K_{ui} \times A \quad \text{(Eqn. 6-2)}$$

Where,

Q_i = leachate percolation rate (leachate generation rate) (m^3/s)

K_{ui} = unsaturated hydraulic conductivity (m/s) at moisture content θ_i, calculated using Eqn. 6-3 below)

A = landfill footprint (m^2)

$$K_{ui} = K_s \left[\frac{\theta_{id}}{\varphi_d}\right]^{3+\frac{2}{\lambda}} \quad \text{(Eqn. 6-3)}$$

$\theta_{id} = \theta_i - \theta_r$= drainage moisture content (v/v) (Eqn. 6-4)

$\varphi_d = \varphi - \theta_r$ = drainable porosity (v/v) (Eqn. 6-5)

Where, K_s = saturated hydraulic conductivity (m/s)

θ_i = total moisture content (v/v)

φ = total MSW porosity (v/v),

λ = MSW pore-size distribution index (-)

θ_r = MSW residual volumetric water content of the landfill (v/v)

A spreadsheet model was developed to conduct an iterative regression analysis to estimate the initial drainable moisture content (θ_{id}), pore-size distribution index (λ), drainable porosity (φ_d), and saturated hydraulic conductivity (K_s) values that provided the best-fit to the actual leachate collection rate measurements. The model uses the initial values specified in Table 6-2 for initial drainable moisture content (θ_{id}), saturated hydraulic conductivity (K_s), and residual moisture content (θ_r) to calculate unsaturated hydraulic conductivity and the leachate generation rate using Eqns. 6-3 and 6-2, respectively; Table 6-2 also includes HELP defaults for these parameters as a point of comparison. The waste moisture content at the start of the subsequent timesteps is calculated by subtracting the leachate generated per volume of waste from the volumetric moisture content of the preceding timestep; MSW in-place density of 900 kg/m^3 was used to calculate the in-place waste volume based on the in-place waste mass (Eqn. 6-6).

$$\theta_{i+1} = \theta_i - \frac{Q_i}{M} \times 900 \frac{kg}{m^3} \quad \text{(Eqn. 6-6)}$$

Where,

θ_{i+1} = total moisture content at the start of time step i+1 (v/v)

M= Mass of waste in the study cell (in kg)

Q_i = leachate generation rate for time step i (m^3/time)

The leachate generation rates were calculated for all timesteps (month or year) for which the actual measurements from the site were available. The square of differences between each point of modeled flow and actual site flow data are summed to calculate the SSE. The Excel function Solver was used to minimize the SSE by changing initial drainable moisture content, pore-size distribution index, drainable porosity, and saturated hydraulic conductivity values. The initial drainable moisture content, drainable porosity, pore-size distribution, and saturated hydraulic conductivity values that resulted in the lowest SSE provide the leachate generation rates that best-fit the measured data for the site. This analysis was conducted for all sites except Site A, which recirculated all of the collected leachate volume for a majority of the period after closure. Monthly leachate collection rates, if available, were used for calculating the SSE for each iteration.

Table 6-2. Initial Conditions and Constraints for Calculating K_{ui}

Parameter	HELP Defaults		Initial, Min and Max Values used for Modeling			Range Reference
	MSW	MSW With Channeling	Initial	Min	Max	
Drainable Porosity (φ- θ_r)	0.671	0.168	0.50	0.45	0.62	Townsend et al. (2015a)
λ	0.47	0.55	0.50	0.26	0.70	Calculated from default HELP θ_{WP} and θ_{FC}
Drainable Moisture Content ($\theta_{i=0}$ - θ_r)	Based on Climate		0.13	0.04	0.62	Townsend et al. (2015a)
Saturated Hydraulic Conductivity, K_s (cm/s)	0.001		10^{-5}	3×10^{-7}	0.25	Townsend et al. (2015a)

6.3.4. HELP Modeling

Neither the first-order decay nor the unsaturated flow modeling approach accounts for the impacts of moisture influx through the final cover defects or leachate recirculation and leakage through the liner on the leachate collection rate. The HELP model offers capabilities to assess the impacts of moisture fluxes associated with the final cover and bottom liner defects/damages, and leachate recirculation on leachate generation/collection rates (US EPA 1997). The HELP model is a commonly used model by landfill engineers and regulators to estimate the long-term leachate generation rate and head on the liner. Due to the prevalence of its use by landfill designers and its additional capabilities, the use of HELP for estimating the future leachate collection rate from closed landfills was evaluated.

For all the study sites, with the exception of Site A, simulations were performed with the HELP model to estimate leachate collection rates after closure. At Site A, leachate was extensively recirculated. Although HELP allows leachate recirculation modeling, it is difficult to specify time-varying leachate recirculation rates to simulate the site-specific operating conditions.

Perfect geomembrane-clay liner contact without any pinhole/defect was assumed for the final cover and bottom liners. The final cover and the bottom liner configuration for the study sites were used for HELP modeling. The actual study cell areas were used for the simulations. The waste height was estimated by dividing the in-place mass by an assumed density of 900 kg/m^3 (Townsend et al. 2015a) and the landfill area; the estimated waste height was used for the simulations.

Precipitation and temperature data were obtained from the National Oceanic and Atmospheric Administration (NOAA) weather station located closest to the site. The default values of other climate data (humidity, wind speed, and growing season duration) were used from the nearest HELP model default city to the site, and the site latitude was provided to estimate solar radiation. A fair grass condition (or bare grass condition for an exposed geomembrane cap), a site-specific evaporative zone depth, and maximum leaf area index was also used as a weather data input based on the HELP default values for geographic locations corresponding to the study sites. These inputs were used to simulate rainfall, temperature, and solar radiation for a 50-year period after closure using the HELP model. These weather and vegetative layers (overlying the final cover geomembrane) data are not expected to impact the modeled leachate generation rate as the geomembrane was assumed to be free of pinholes and defects. The impact of these inputs and compromises in the final cover can be modeled by specifying geomembrane defects in the HELP model.

HELP does not have a field to enter the pore size distribution index but rather calculates it based on the entered wilting point and field capacity of the waste (Eqn. 6-7). A HELP default field capacity for MSW with the channeling of 0.073 (v/v) was used to estimate the wilting point of MSW using Eqns. 6-7 and 6-8 below using the pore-size distribution index estimated based on the unsaturated flow model for each site.

The drainable porosity and initial drainable moisture content, estimated based on the unsaturated modeling, were used to estimate the total porosity and the initial moisture content using Eqn. 6-5 and 6-4, respectively, for each site; residual moisture content calculated using Eqns. 6-7 and 6-8, and the field capacity value described above were used for these estimations. Similarly, the best-fit saturated hydraulic conductivity, porosity, initial moisture content, and field capacity were estimated using the unsaturated flow model described above were used for the HELP model run for each site. The initial moisture content was varied, if needed, until the HELP model output most closely resembled the measured data. The initial moisture content of all the other layers apart from waste was specified to be the lesser of five times the field capacity or half of the porosity (presented in Table 6-4).

$$\theta_{FC} = \theta_r + (\theta_{WP} - \theta_r) \times \left(\frac{15}{0.33}\right)^{\lambda} \quad \text{(Eqn. 6-7)}$$

$$\theta_r = \begin{cases} 0.014 + 0.25\,\theta_{WP} & for\ \theta_{WP} \geq 0.04 \\ 0.6\,\theta_{WP} & for\ \theta_{WP} < 0.04 \end{cases} \quad \text{(Eqn. 6-8)}$$

Where,

θ_{FC} = field capacity of MSW (v/v)

θ_{WP} = wilting point of MSW (v/v)

θ_r = MSW residual volumetric water content of the landfill (v/v)

λ = MSW pore-size distribution index (dimensionless)

6.4. Results and Discussion–Leachate Collection Rate

6.4.1. Measured Leachate Collection after Closure

Annual leachate collection volumes were available for all of the sites; however, these data values were not necessarily available every year after closure. Also, the leachate collection rates for some of the sites were not available for the entire year for some years. The available data were added for each year for each site and divided by the number of days for which data were available to calculate an annual average leachate collection rate (gallons per day). Although the total collection volume from an MSWLF is expected to be dependent on the in-place volume, the leachate collection rate is driven by its footprint and the vertical hydraulic conductivity of the in-place waste. The annual average leachate collection rates were, therefore, normalized by the cell area to calculate the annual average leachate collection rate per unit area of the landfill (gallons per acre per day (GPAD)) for an equitable comparison of leachate collection rates among the study sites with varying footprints. Figure 6-1 and Table 6-3 below presents the annual leachate collection rate per unit area for the study cells (i.e., 11 total cells) from all nine sites. It should be noted that the leachate collection rates for two study cells are presented for Site C and Site G. Since the closure, the annual leachate collection rate across all 11 cells has varied from 3–2070 GPAD, with a median of 92 GPAD and an average of 190 GPAD; 90% of the annual collection rate estimates are less than 500 GPAD. Bonaparte et al. (2002) reported an average post-closure LCS leachate collection rate from 11 closed cells at 3 MSWLFs (located in the northeast region of the US) based on data collected over periods ranging from 8 to 64 months within ten years after closure. The average leachate collection rate from these cells was reported to range from 439 to 7,480 GPAD.

With the exception of Sites A and E, which recirculated leachate after closure, the most recent annual leachate collection rates for all the site cells were below 100 GPAD. The median annual leachate collection rate among the non-leachate recirculating sites varied between approximately 8 and 268 GPAD. The wide variation of the leachate collection rates among the sites is potentially due to differences in site-specific precipitations and operating conditions (e.g., leachate recirculation). Site A has consistently managed leachate through recirculation over a majority of the period since closure. As shown in Table 6-1, Site E recirculated leachate a substantially lower fraction of leachate and for a shorter period than Site A. Only 8% of the leachate collected since the closure has been recirculated at Site E.

The reasons for a spike in leachate collection rate from the study cell G3 in the 12^{th} and 13^{th} year after closure and from G4 in the 10^{th} and 11^{th} year after closure is unknown potentially due to complexity of monitoring the conditions that can spike leachate collection rates. These conditions include the final cover geomembrane damages/defects, and stormwater run-off channeling into the leachate collection system. The site records documented multiple instances of the flow meter drifting out of calibration at Site D. The leachate collection rates reported for Site D might, therefore, not be representative of the actual flow rates.

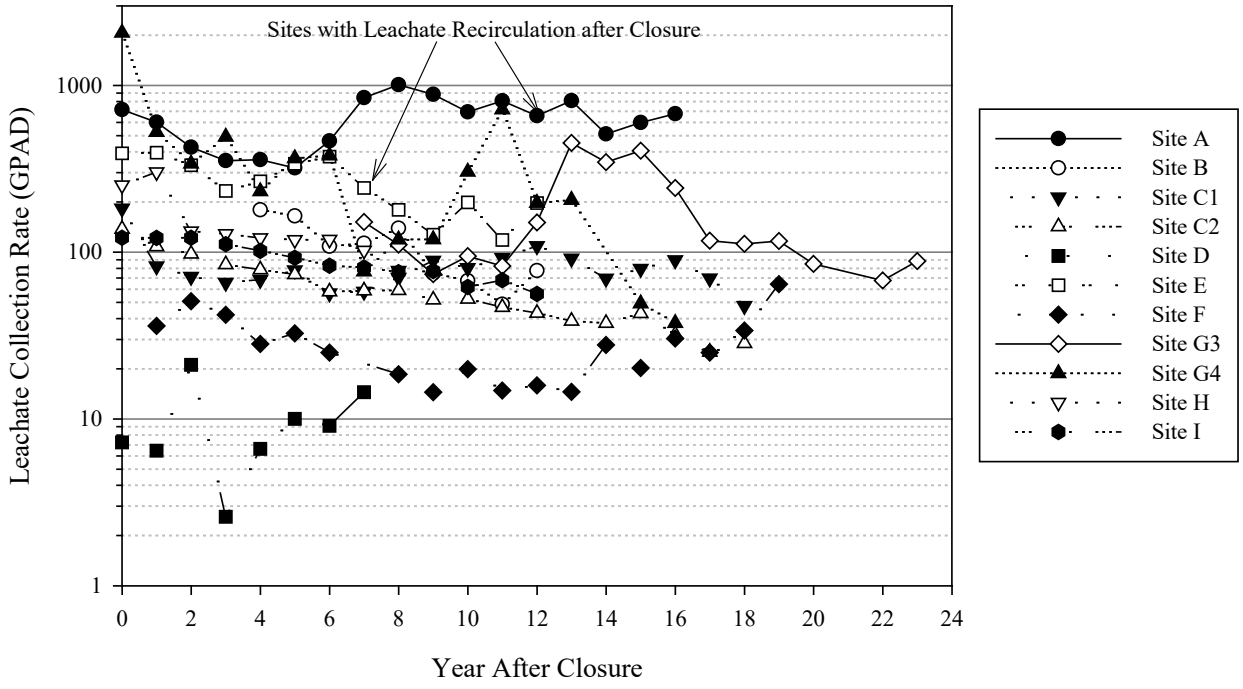


Figure 6-1. Temporal Variation of Annual Leachate Collection Rates at Study Cell(s) After Closure

Table 6-3 presents the average annual leachate collection rates for the study sites after closure. As expected, the leachate collection rate from Site A was greatest among all the study sites, potentially due to extensive leachate recirculation at the site. The leachate collection rates for each site except for Sites A, C1, and F show a generally declining trend. The leachate collection rate at Site A showed a declining trend for five years after closure and showed an increasing trend during the 68 years after closure following the start of leachate recirculation during the 6th year after closure. The leachate collection rate at Site A shows a general declining trend since the 8th year after closure; approximately 74% of the leachate collected since closure has been recirculated between year 616 since closure. The leachate collection rate from the study cell C1 declined from 183 to 82 GPAD within a year after closure and has fluctuated between 48 and 109 GPAD since the year after closure. None of the landfill cells examined exhibited trends indicative of reaching a steady state leachate collection/generation rate.

The leachate collection rate for Site F, in general, has declined during the first nine years after closure but has shown a gradually increasing trend during the subsequent 9-year period. The most recent annual leachate collection rate for Site F represents the maximum rate since closure. The monthly leachate collection rates were analyzed with respect to the monthly rainfall data to assess the impact of rainfall on the leachate collection rate at Site F. Figure 6-2 presents the monthly leachate collection rates and the monthly precipitation for the most recent two to three years for Site F. The monthly leachate collection rate during rainy seasons over this period was much greater (as high as ~120 GPAD) than those during dry periods (~ 20-30 GPAD). A spike in leachate collection rate with no or minimal time lag with rainy season is a strong indicator of stormwater channeling into the leachate collection infrastructure at the site. The leachate quality data collected at appropriate points of time (during the dry and rainy season), specifically the concentration of

leachate indicator parameters (e.g., chloride), can also be used to assess stormwater channeling into leachate collection infrastructure. The chloride concentration of leachate from one of the leachate collection sumps/wet wells in December 2014 (rainy season) was less than 100 mg/L compared to 1,000 mg/L during June 2015 (dry season). This suggests that leachate constituted less than 10% of the liquid extracted from this sump in December 2014, and stormwater constituted the rest; leachate quality at the site was characterized twice per year. The location or the cause of stormwater infiltration into the LCS could not be assessed based on the data available for the site for this study.

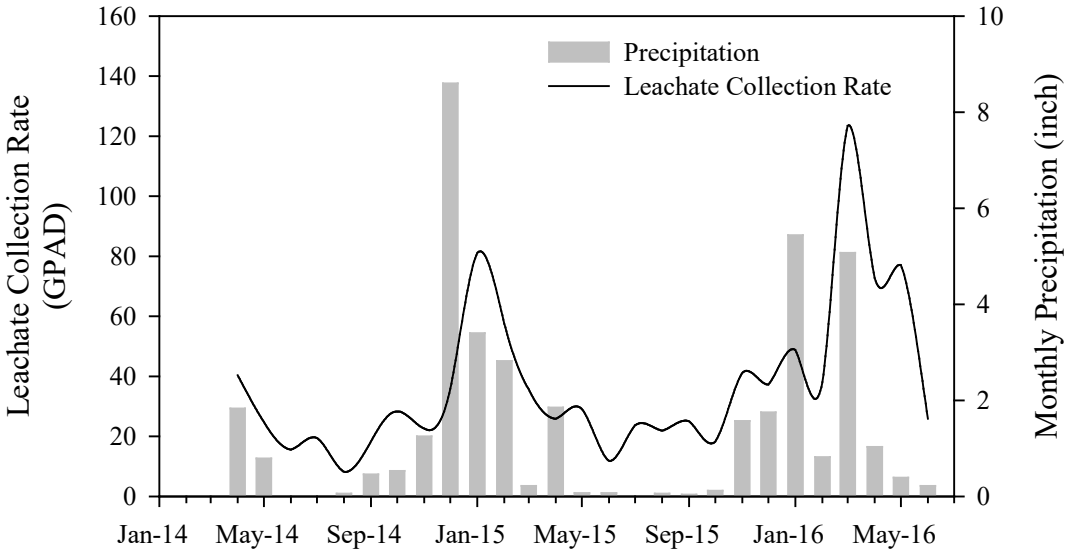


Figure 6-2. Temporal Variation of Monthly Leachate Collection Rate and Precipitation for Site F

As mentioned earlier, some states' guidance documents recommend LCS operation termination after cessation of leachate generation. Leachate generation continues at all the study sites. The most recent leachate collection rate for all of the sites except Sites A and E has been less than 100 GPAD, which is equivalent to 1.08×10^{-7} cm/s (1.34 inches/year). As a point of comparison, the leachate leakage rate through a compacted clay liner (without a geomembrane) with a saturated hydraulic conductivity of 10^{-7} cm/s is estimated (using Darcy's equation) to be 92.4 GPAD assuming a unit gradient and saturated flow conditions. The compacted clay liner underlying a geomembrane in a composite liner system is typically required to have a maximum hydraulic conductivity of 10^{-7} cm/s. All the leachate would leak through a compacted clay liner with a hydraulic conductivity of 10^{-7} cm/s and would not result in leachate accumulation over the liner once the leachate generation rate declines below 92.4 GPAD if the site is just lined with a compacted clay liner. The overlying geomembrane, due to its significantly lower equivalent hydraulic conductivity of ~10^{-13} cm/s (for HDPE liner) (equivalent to 9.2×10^{-5} GPAD), is expected to be the limiting factor for the leakage rate through a composite liner. Based on the equations used by the HELP model, leakage rate from a 1-cm^2 defect in geomembrane underlain by 10^{-7} cm/s hydraulic conductivity clay liner for good and poor liner contact case is approximately 0.1314 and 0.7234 gallons per day, respectively. The leachate is expected to accumulate above the liner and may cause elevated pore pressure if LCS operation is terminated before achieving a leachate collection rate of less than equal to the leakage rate through the primary liner. The leakage rate is

expected to increase over time with an increase in the head on the liner after LCS operation termination.

Excluding Site D data and the collection rate for Site G4 for Year 0 (2,070 GPAD), the annual leachate collection rate (GPAD) varies over almost two orders of magnitude, as shown in Figure 6-1 and Table 6-3. The leachate generation rate would be primarily dependent on precipitation intrusion into the landfill during its operation. The daily leachate collection rate per unit area (GPAD) was converted to an equivalent rate in inches per year (which is the unit used for annual precipitation) and divided by annual precipitation to assess the impact of precipitation on the variability of leachate collection rates among the study sites.

Table 6-3. Leachate Collection Rate Summary at the Nine Study Sites Since Closure

Site	A	B	C1	C2	D	E	F	G3	G4	H	I
Precipitation (in/year)	51	56	47	47	46	30	22	46	46	47	51
Years Since Closure as of 2016 (Considering closure year as year zero)	18	12	18	18	7	12	19	23	16	8	19
Years Since Closure	**Annual Average Leachate Collection Rate (gallons per acre per day)**										
0	715	-	183	137	7	427	-	-	2070	252	122
1	601	-	82	108	6	395	36	-	526	302	122
2	426	-	71	98	21	332	51	-	340	134	122
3	353	-	66	84	3	233	42	-	492	129	112
4	359	180	68	79	7	266	28	-	232	122	102
5	320	165	78	73	10	338	33	-	366	119	93
6	465	108	57	58	9	373	25	-	379	119	83
7	845	113	58	59	14	243	-	152	77	102	81
8	1009	139	70	59		179	19	111	119	78	76
9	883	75	89	52		127	14	74	119		77
10	693	67	81	52		199	20	95	303		62
11	808	49	92	47		118	15	83	718		68
12	658	78	109	43		197	16	151	197		56
13	810		92	39			15	452	206		-
14	511		70	38			28	347	-		-
15	599		80	43			20	406	49		-
16	676		90	32			30	242	38		-
17	-		70	25			25	117			-
18	-		48	28			34	112			-
19							64	117			-
20								85			
21								-			
22								68			
23								88			
Average (All Years)	631	108	82	61	10	264	29	169	389	151	90

"–" indicates that leachate collection data were not available for the site for that year

Shaded cells indicate future years (after 2016)

Figure 6-3 presents cell post-closure leachate collection rates as a percentage of the average annual precipitation of the sites. The annual precipitation for each site was derived from NOAA data (NOAA 2017) for the closest weather station to each site. The leachate collection rate for all the study sites except Sites A, E, and G has been less than 5% of the annual rainfall. Excluding Site D data and the collection rate for Site G4 at closure year, the precipitation-normalized leachate

collection rate varies over an order of magnitude (0.5 GPAD/inch precipitation to 19.8 GPAD/inches) among the study sites excluding Site D. The smaller magnitude of the variability in the precipitation-normalized leachate collection rate compared to the variability among the leachate collection rate presented in Figure 6-1 suggests precipitation has a considerable impact on leachate generation/collection as expected. Other factors, such as the effectiveness of the run-on/run-off control system, size of the working face area that is exposed to the precipitation during active operation, and waste composition, also impact the moisture influx into the landfill.

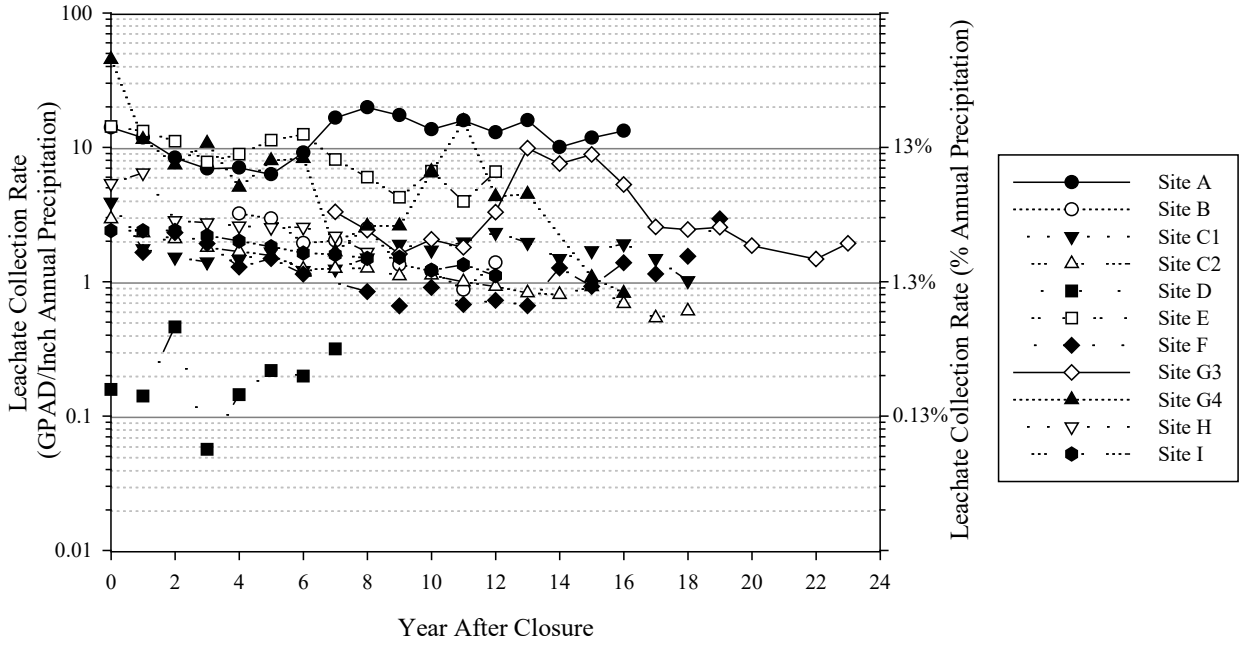


Figure 6-3. Annual Leachate Collection Rates as a Percentage of Average Annual Precipitation of Study Cells after Closure

6.4.2. Modeled Leachate Collection Rates

A comparison of the measured and modeled data using the three approaches described above are presented in Figures 6-4 through 6-13. In general, all three approaches provided comparable and reasonable fits to the measured study site data for all the study sites except Site C1, D, and G3. It should be noted that the y-axis scale and range used for Figures 6-4 through 6-13 are not the same but vary based on the range of the measured and model flow rates. As discussed earlier, the Site D measured leachate collection rate trend may not be representative of leachate generation due to leachate flow meter calibration issues on several occasions and the draining of the leachate storage pond before closure. It should also be noted that the Site F leachate collection rate diverges from a decay-trend and begins increasing due to intrusion of stormwater into the LCS (as discussed earlier). The leachate generation rate for Site B based on the HELP model shows a sharp decline to an insignificant rate around the 22nd year after closure. This is due to the decline in the waste moisture content below the field capacity used for simulation. As discussed earlier, a field capacity of 0.073 (v/v) for MSW was used for the HELP simulations. HELP models the vertical percolation of leachate only if the moisture content is above the field capacity. As discussed later, an initial moisture content of 0.13 (v/v), which was very close to the assumed field capacity, was used for MSW at Site B for the HELP model. As mentioned earlier, leachate generation modeling was not performed for Site A due to the complexity of modeling leachate recirculation using these

approaches. All of the leachate collected at Site A was recirculated for a majority of the period after closure.

The HELP model-based leachate generation rate for the first year was much higher than the second year for many of the sites. This is probably is due to drainable leachate associated with the assumed moisture content of the drainage layer. For some of the sites, the HELP model leachate generation rate sharply declined to an unreasonably low value and then increased gradually before starting a gradual declining trend; further analysis of the cause of this unexpected fluctuation in the modeled leachate generation rate was not conducted. These unreasonably low flow rates from the HELP model were disregarded and not included in the figures for Site C1, D, F, G3, and I.

Unlike the unsaturated flow and first-order decay modeling approached described above, HELP does not have a regression analysis feature that can automatically adjust various design inputs and media properties that would yield the leachate collection rates that match most closely with the measured data. The leachate collection/generation rates estimated using the HELP model are based on a mix of default parameters and the best-fit results from the unsaturated flow model. The estimates that best-fit the measured flow rates can be obtained using the HELP model by iteratively running the model by varying the waste properties (e.g., wilting point, field capacity, porosity, and hydraulic conductivity) values until the SSE between the modeled and measured data is minimized. The HELP model was not iteratively executed in this study due to time constraints to assess whether or not the HELP model estimates, which are presented in this section, represent the best-fit to the measured leachate collection rate data. The magnitude of the difference between the HELP model results and actual site data, therefore, should not be used to interpret that the first-order model and the unsaturated flow models are either more or less accurate than the HELP model for estimate leachate generation.

As mentioned earlier, among the approaches described above, only the HELP model provides the capability to assess the impact of precipitation intrusion into waste and leakage through the liner as the result of defects/damages to the final cover and bottom liner, respectively. The first-order decay and unsaturated flow modeling approaches do not account for the impacts of moisture influx through the final cover and leakage through the liner on the leachate collection rate.

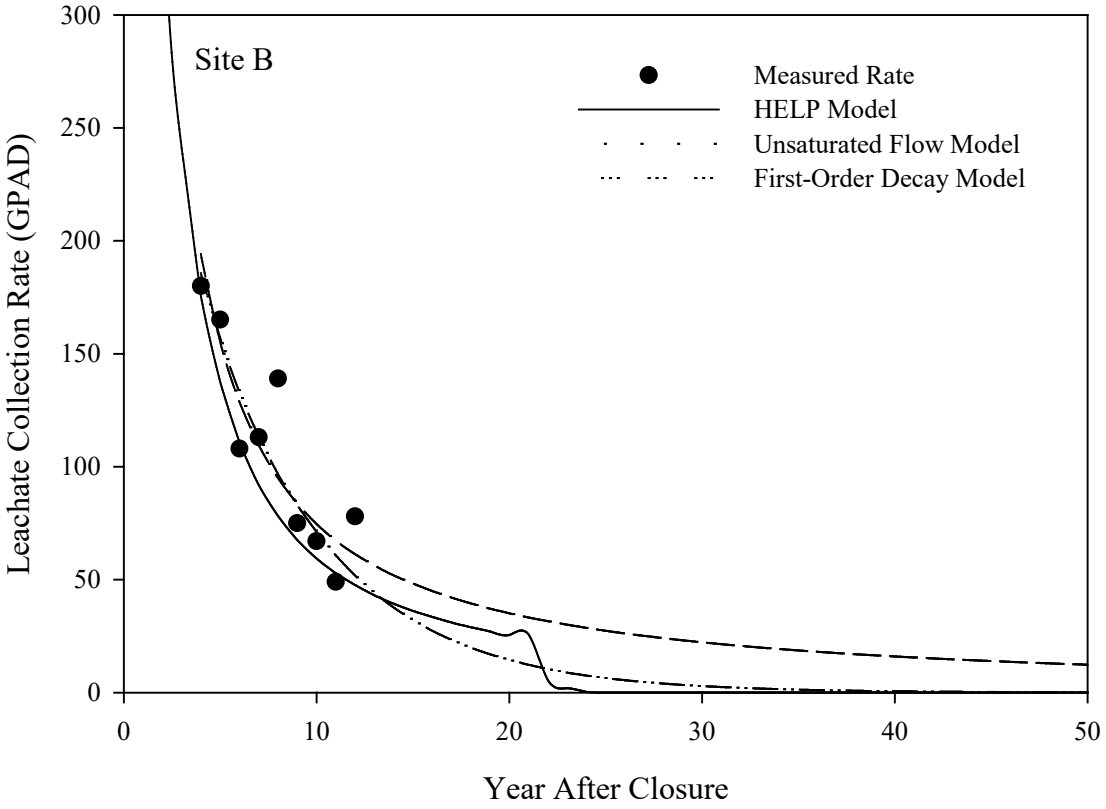


Figure 6-4. Measured and Modeled Leachate Collection Rates from Site B

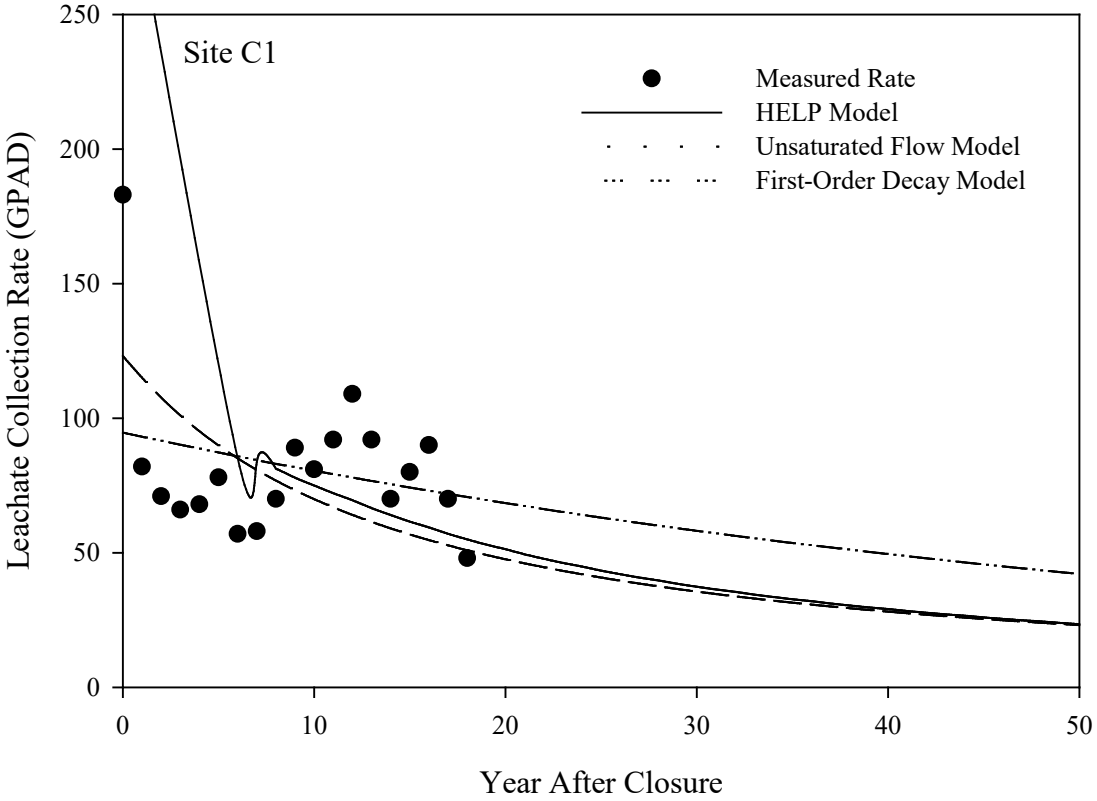


Figure 6-5. Measured and Modeled Annual Leachate Collection Rates from Site C1

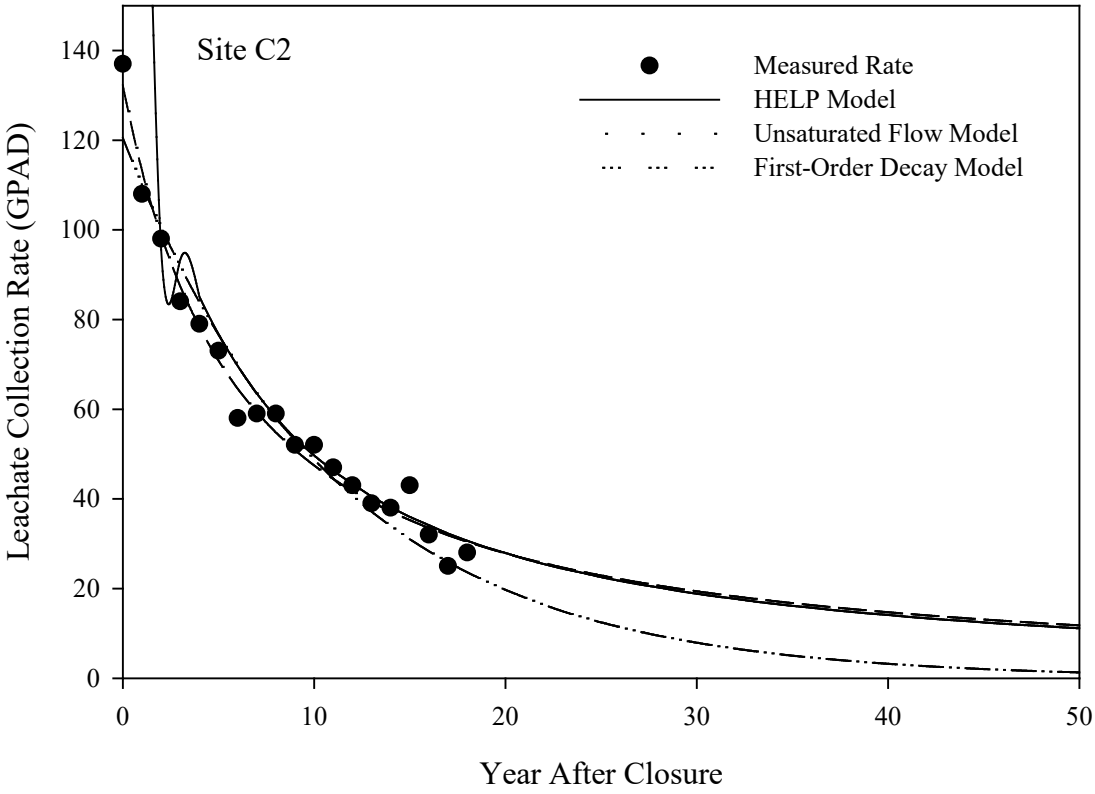


Figure 6-6. Measured and Modeled Annual Leachate Collection Rates from Site C2

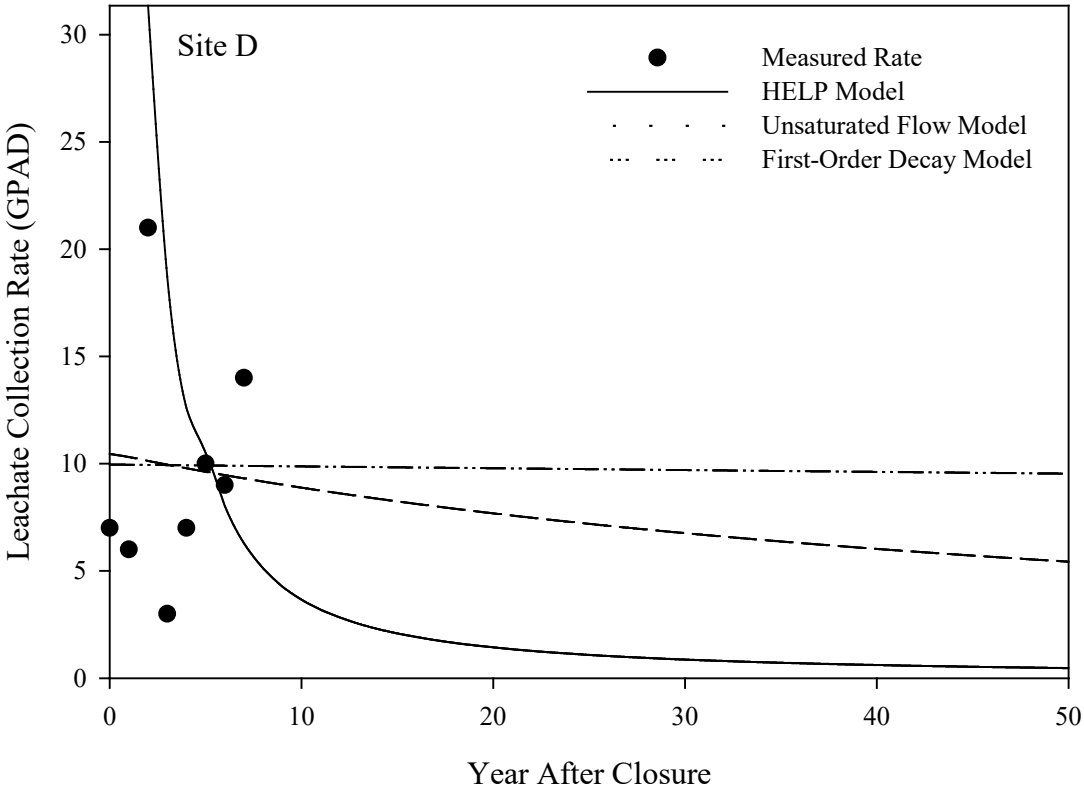


Figure 6-7. Measured and Modeled Annual Leachate Collection Rates from Site D

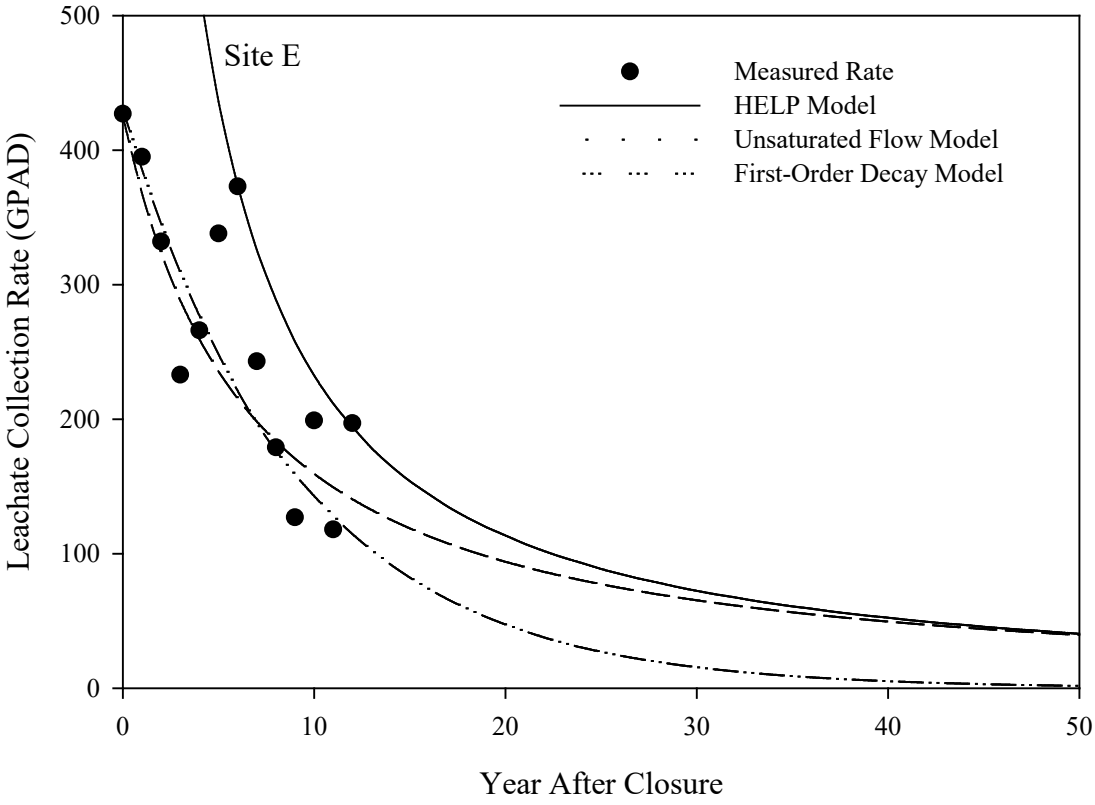


Figure 6-8. Measured and Modeled Annual Leachate Collection Rates from Site E

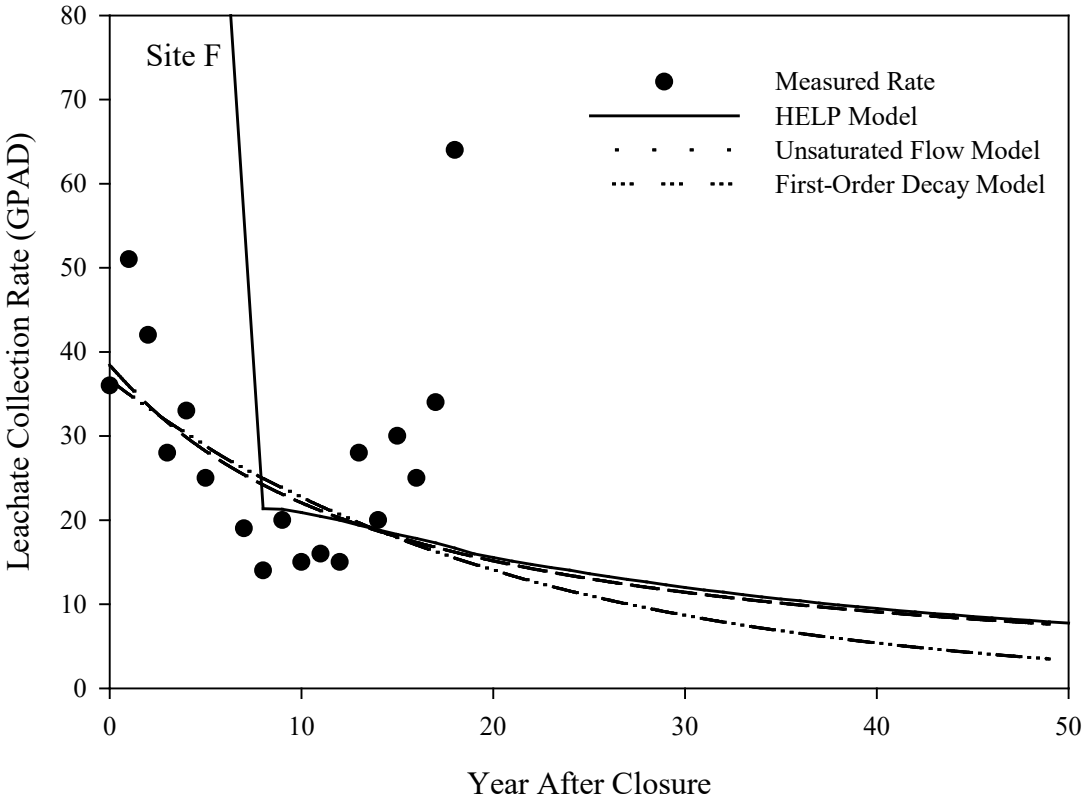


Figure 6-9. Measured and Modeled Annual Leachate Collection Rates from Site F

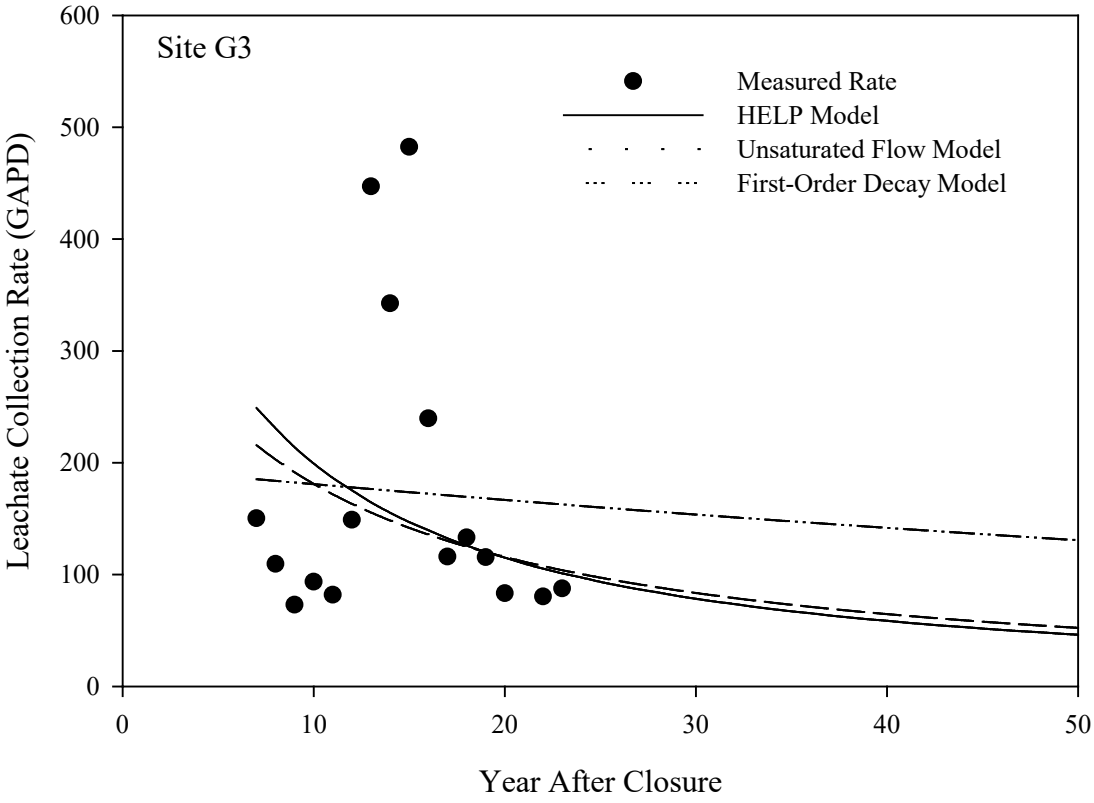


Figure 6-10. Measured and Modeled Annual Leachate Collection Rates from Study Cell G3

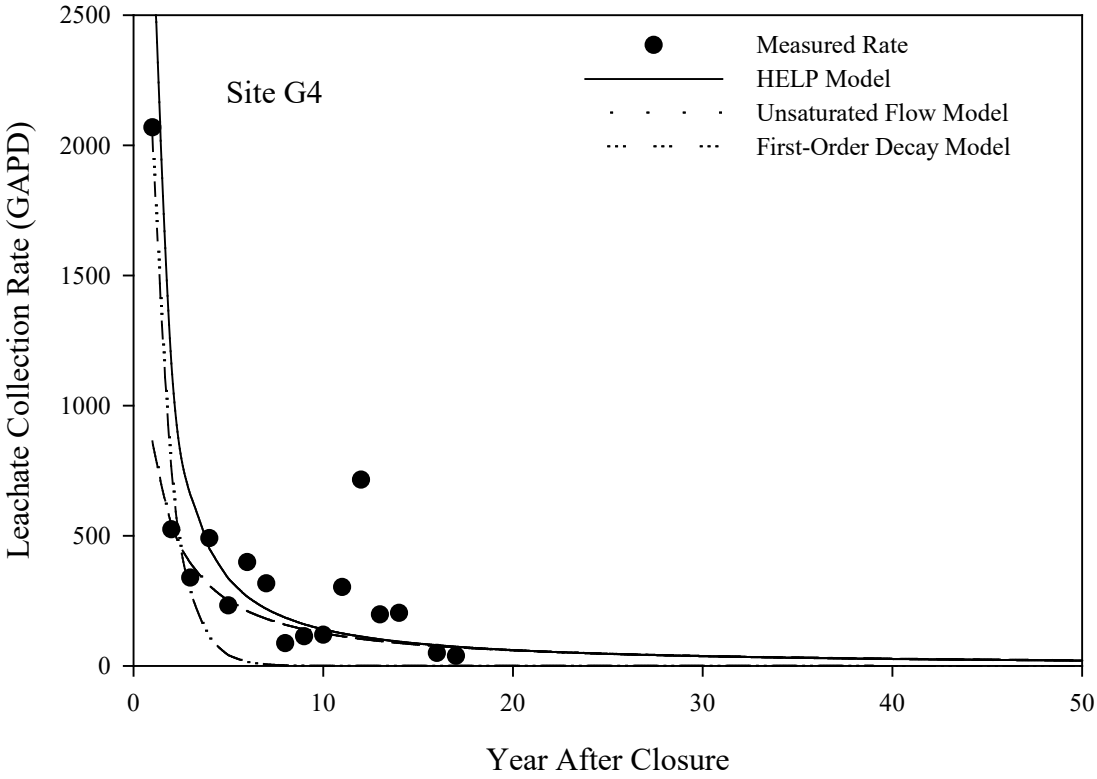


Figure 6-11. Measured and Modeled Annual Leachate Collection Rates from Study Cell G4

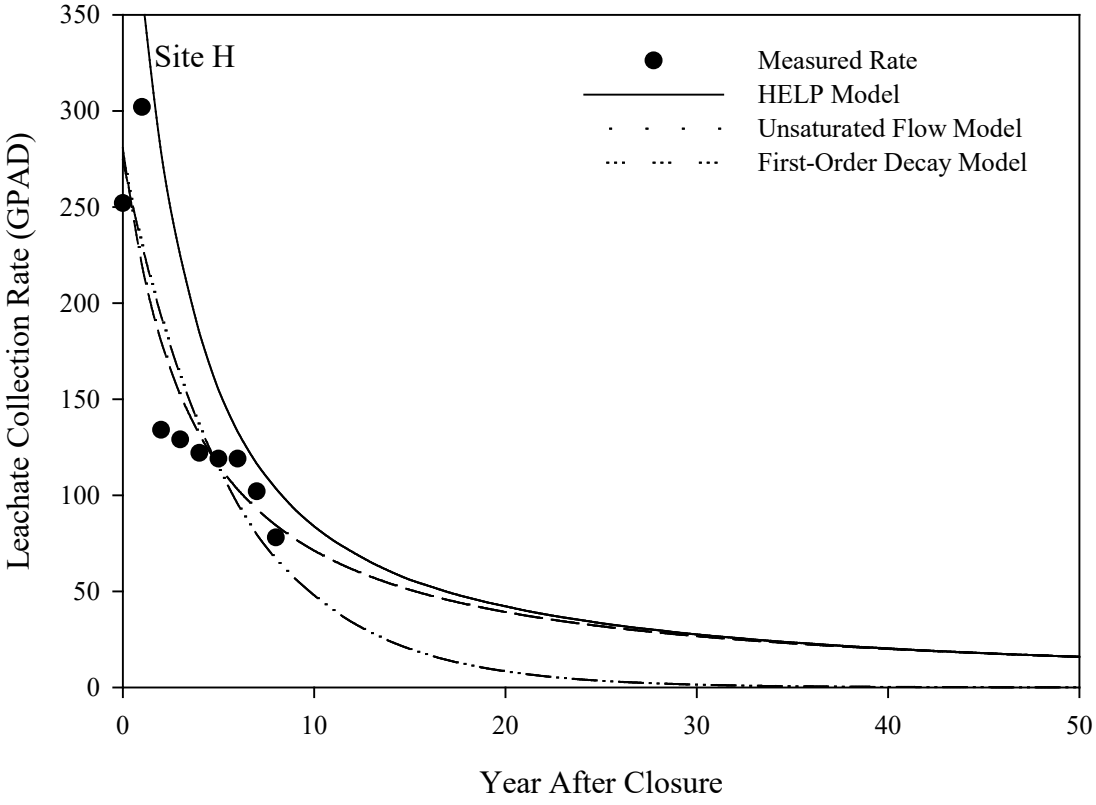


Figure 6-12. Measured and Modeled Annual Leachate Collection Rates from Site H

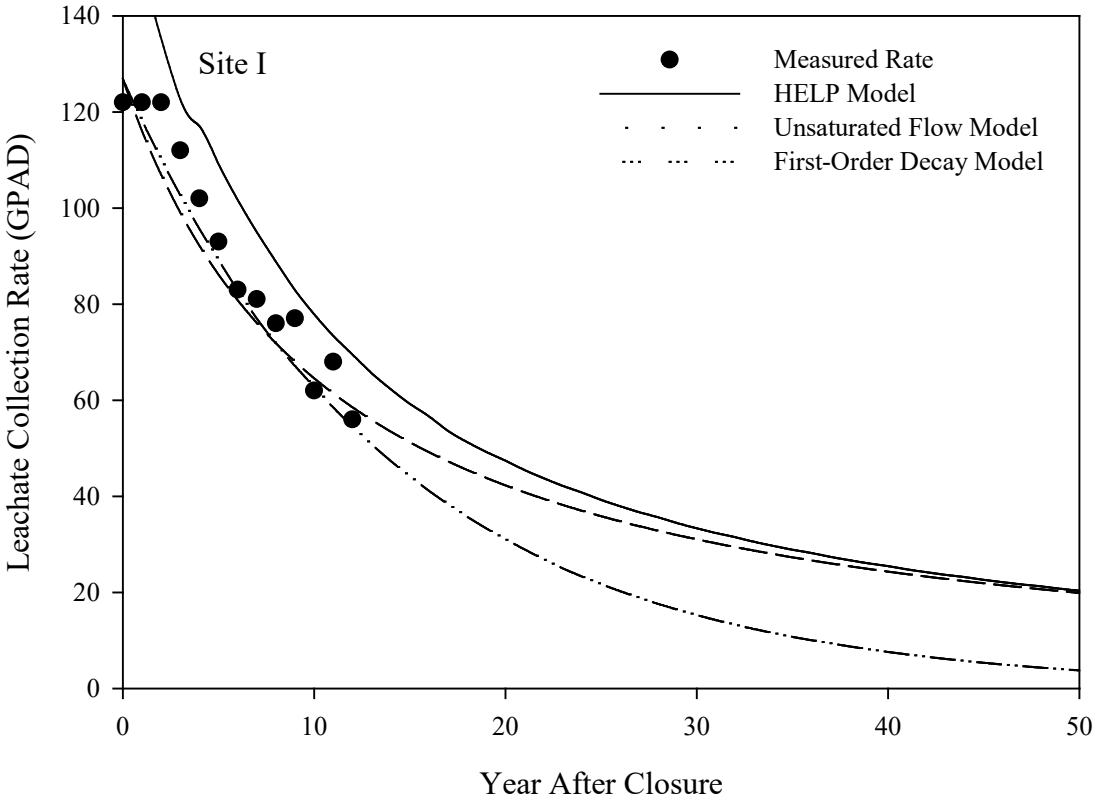


Figure 6-13. Measured and Modeled Annual Leachate Collection Rates from Site I

Table 6-4 presents the values of the input parameters corresponding to the best-fit based on the first-order decay, unsaturated flow, and HELP models. The estimates based on the unsaturated flow model fit the measured data with an $r^2 > 0.5$ for seven of the ten study cells modeled. An r^2 of 0.5 means that the modeling approach captures or explains 50% of the variations in the leachate flow rate over time at a given site, and the rest 50% is due to factors (e.g., stormwater intrusion into the landfill) that are not accounted for by the model. The first-order decay model also fits six of the ten study cells with an $r^2 > 0.5$. Sites C1, D, and G3 cells had the lowest r^2 value for both models; lower r^2 for Site G3 is probably due to a spike in leachate collection rate ten years after closure. Site H had five months of very high initial flow that rapidly declined, and Site E had periods of leachate flow that appeared to deviate from the overall general trend in the data during 2009 and 2010.

Note that the unsaturated flow model provides an estimate of drainable porosity and moisture content, which are different from the porosity and initial moisture content used by the HELP model. Table 6-4 also shows the half-life (i.e., timeframe for leachate collection to half of the rate at closure) corresponding to the estimated decay rate (k_l) for the first-order decay model. The half-life for all of the study cells (except C1, D, and G3) ranges from 2.3 to 14.5 years. Neither of the models provided a reasonable fit to the actual data for Sites C1, D, and G3 dues to the lack of a consistent trend in the leachate generation rate for these sites. The r^2 values corresponding to the HELP model flow rates were not calculated as the model was not iteratively run to minimize the SSE between the modeled and the measured data.

The in-place waste saturated hydraulic conductivity values estimated based on the unsaturated flow model for all sites with $r^2>0.5$ ranged from 9.0×10^{-6} to 3.7×10^{-3} cm/s, which is within the range reported in the literature (Townsend et al. 2015a). It should be noted that the default field capacity value was used for the HELP model run, and the wilting point was calculated using pore-size distribution (from the unsaturated flow model). The reasonableness of the estimates for the estimated wilting point could not be assessed due to a lack of data reported in the literature for this parameter.

Table 6-5 presents the leachate collection rate 30 years after closure estimated using different modeling approaches. The estimates based on the first-order decay model are, in general, substantially lower than those obtained using the HELP model and the unsaturated flow model. The HELP model estimates were very close to those from the unsaturated flow model. All the estimates except for the first-order decay model estimate for Sites C1 and G3 are lower than the most recent leachate generation rates.

Table 6-4. Values Used for Different Modeling Approaches

Parameter	Site									
	B	C1	C2	D	E	F	G3	G4	H	I
First Order Decay Model										
Total leachate generation potential of MSW, V_o (m^3/Mg)	18	91	12	66	23	14	179	88	11	10
Decay Constant, k_l (year $^{-1}$)	0.16	0.02	0.09	0.00	0.11	0.05	0.01	0.97	0.17	0.07
Half-Life (Years)	4.3	-	7.7	-	6.3	14.5	-	0.7	4.0	9.8
r^2	0.84	0.09	0.95	<0	0.65	0.48	<0	0.51	0.75	0.98
Unsaturated Flow Model										
Saturated hydraulic Conductivity, K_s ($\times 10^{-4}$ cm/s)	9.8	0.004	3.8	2500	0.09	0.15	0.007	0.07	37	1
Drainable Porosity, φ_d (v/v)	0.50	0.62	0.51	0.59	0.54	0.53	0.62	0.53	0.58	0.46
Initial Drainable Moisture Content, θ_{od} (v/v)	0.12	0.53	0.18	0.12	0.35	0.23	0.51	0.49	0.22	0.19
MSW Pore-size Distribution Index (λ)	0.68	0.70	0.44	0.26	0.57	0.48	0.70	0.52	0.31	0.46
r^2	0.80	<0	0.98	<0	0.62	0.55	<0	0.82	0.76	0.96
HELP Model										
Saturated Hydraulic Conductivity, K_s ($\times 10^{-4}$ cm/s)	9.8	0.004	3.8	2500	0.09	0.15	0.007	0.07	37	1
Porosity, φ (v/v)	0.508	0.627	0.522	0.615	0.554	0.541	0.62	0.53	0.607	0.477
Initial Moisture Content, $\theta_{o,}$ (v/v)	0.131	0.450	0.175	0.1	0.359	0.2	0.51	0.49	0.23	0.185
Field Capacity, $\theta_{FC,}$ (v/v)	0.073	0.073	0.073	0.073	0.073	0.073	0.073	0.073	0.073	0.073
Wilting Point, $\theta_{WP,}$ (v/v)	0.012	0.012	0.027	0.043	0.018	0.024	0.011	0.020	0.039	0.025

Table 6-5. Estimated Leachate Collection Rate at the End of 30 Years after Closure Based on Modeling Approaches Used

Site	Most Recent Leachate Collection Rate (gallons per acre per day)	Estimated Leachate Collection Rate 30 Years after Closure (gallons per acre per day)			Coefficient of Correlation (r^2) between the Measured and Modeled Flow Rates	
		HELP Model	Unsaturated Flow Model	First-Order Decay Model	Unsaturated Flow Model	First-Order Decay Model
B	78	~0	22	3	0.80	0.83
C1	48	39	36	58	<0	0.09
C2	28	20	19	8	0.98	0.95
D	14	1	7	10	<0	<0
E	197	75	65	16	0.62	0.65
F	64	12	12	9	0.55	0.48
G3	88	78	84	154	<0	<0
G4	38	37	38	<1	0.82	0.51
H	78	29	27	1	0.76	0.75
I	56	34	31	15	0.96	0.98

6.4.3. An Evaluation of the Performance of the Primary Liner System

The bottom liner is designed to intercept leachate and prevent or minimize migration to the underlying surficial aquifer. However, in reality, geomembranes are not entirely impervious as these allow vapor transmission and may have manufacturing and construction defects. Six of the study cells are lined with a double bottom liner system and have a secondary leachate collection system or leak detection system (referred herein to as LDS). Figure 6-14 presents a temporal variation of the leachate collection rate from LDS for the study cells with a double-liner system and available LDS collection rate data; LDS collection rates were not available for study cells C1 and C2.

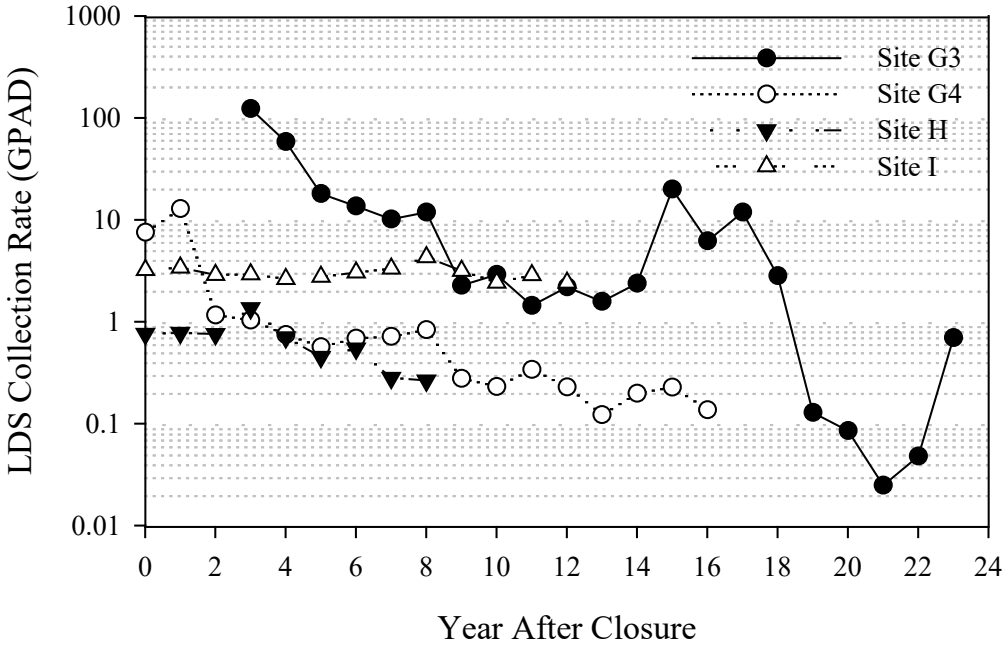


Figure 6-14. Temporal Variation of LDS Collection Rates at Study Cell(s) of Each Study Site After Closure

As shown in Figure 6-14, the LDS collection rates of these study sites ranged from approximately 0.1 GPAD (for study cell G4) to 123 GPAD (for study cell G3). Over 50% and 85% of the values across these sites are less than 1.5 GPAD and 8 GPAD, respectively. The LDS rates, in general, show an overall declining trend over time after closure. As shown in Figure 6-14, the LDS collection rate of G3, in general, was higher than the other sites. Bonaparte et al. (2002) reported an average post-closure LDS rate from 14 closed cells at 4 MSWLFs (located in the northeast region of the US) based on data collected over periods ranging from 4 to 64 months within ten years after closure. The average leachate collection rate from these cells was reported to range from 0 to 1,767 GPAD.

The aggregate primary liner efficiency was calculated by dividing the sum of annual LDS collection rates by the sum of the corresponding annual LCS and LDS rates for each study site for all years with available LCS and LDS collection rates. This analysis assumes that the leachate leakage through the primary liner is the source of liquids in the LDS. As will be discussed later in this section, in some cases, it appears that groundwater intrusion is a significant contributor to liquids in the LDS. The assumption that 100% of LDS flow is the result of leachate leakage through the primary liner provides the most conservative estimate of the primary liner performance. The

analysis further assumes that the sum of the LCS and LDS flow rates represent the total leachate generation rate. The analysis suggests that 2.8%, 0.4%, 0.4%, and 3.2% of leachate intercepted by the primary liner leaked through the primary liner after closure for study cells G3, G4, H, and I, respectively. In other words, the primary liner efficiency was estimated to be 97.2%, 99.6%, 99.6%, and 96.8% for study cells G3, G4, H, and I, respectively. The analysis assumes that the leachate leakage through the secondary liner is insignificant. The primary liner efficiency would be lower than the estimates presented above if the leakage through the secondary liner is significant.

There are notable differences in the configuration of the primary liner system among these sites, which may contribute to the differences in the LDS rates observed at these sites. One major difference in the primary liner system among the sites was geomembrane material. The study cell G3 was lined with a 36-mil Hypalon liner, while the primary geomembrane of G4, H, and I was 60-mil HDPE. The primary liner efficiency for G3 (97.2% as presented above) was in the range estimated for the other sites with an HDPE liner. It should also be noted that a part of cell G3 does not have a geomembrane underlaying its LDS, making it more susceptible to groundwater intrusion than other sites.

The second notable difference is the nature of the layer underlying the primary geomembrane. It should be noted that the part of the cell G3 bottom liner, which was on the slope of the adjacent, is a composite liner, and the rest of the area was constructed in two phases with a double liner with two varying configurations. The leachate collection rates were not available for these distinct areas of G3. The primary liner of G3 (a part), G4, and Site I was constructed with a geomembrane underlain by a low permeability GCL or compacted clay liner. The primary geomembrane of Site H and a part of G3 was underlain by a high permeability layer (geonet for Site H and sand/granular layer for part of G3); schematics depicting these liners configurations are presented in Attachments G, H, and I.

The low permeability layer underlying the primary geomembrane of the Sites G3 (a part), G4, and I is expected to impede leakage through any liner pinholes and defects, while a high permeability layer underlying the primary geomembrane of Site H is expected to enhance leakage through the geomembrane pinholes/defects. The leachate leakage rate through the primary geomembrane at Site H is, therefore, expected to be greater than those at Sites G3 (part), G4, and I. However, for the comparable primary leachate collection rates, the LDS collection rate at Site H was, in general, lower than the other sites. The analysis suggests that other factors such as the head on the liner, liner construction quality (e.g., frequency of construction defects), and groundwater intrusion rates, which are typically unknown, may have a greater influence on the LDS collection rate than the geomembrane material and/or the nature of layer underlying it.

The LDS collection rates at the study sites were substantially higher than those corresponding to the equivalent hydraulic conductivity of geomembrane (e.g., $\sim 10^{-13}$ cm/s for HDPE liner) suggesting that the liquids in the LDS are primarily contributed by leachate leakage through pinholes and construction defects in the primary liner and/or groundwater intrusion. The intrusion of groundwater into the LDS has been reported to impact the LDS collection rate for Subtitle C landfills (EPA 2017a). A review of groundwater well depth-to-liquid readings, site topography, and bottom liner design for both Sites H and I show that the groundwater table was near and (at least at certain times of the year) may have exceeded the bottom-most elevation of the liner. A potential, therefore, existed for groundwater intrusion into the LDS and, ensuing dilution of the LDS leachate. The available LDS leachate quality data were analyzed and compared with the LCS

leachate quality and groundwater quality to assess the relative contributions of these sources to the liquids collected from the LDS.

Leachate quality data were available for leachate collected from the LCS and LDS for Site I and from the LDS, and LDS-LCS composite for Site H. Because more than 99.5% of the leachate collected from Site H was primary leachate, it was assumed that composite leachate is representative of LCS leachate quality. Several commonly detected parameters that have been analyzed at both sites for primary leachate, secondary leachate, and groundwater were analyzed in this study to assess the potential of groundwater intrusion into the LDS. The leachate quality data for Site G were not analyzed as the available data represented the quality of leachate composited from the closed and active cell(s) at the site.

Figure 6-15 presents a comparison of the distribution of chloride, the concentration of leachate collected from LCS and LDS as well as groundwater at each of the two sites. It can be seen that chloride concentration in LDS leachate is approximately an order of magnitude lower than that of the respective LCS leachate for both the sites. The median chloride concentration in LDS leachate is approximately two orders of magnitudes greater than the groundwater chloride concentration. The leachate and groundwater quality data available after closure were used for the analysis; chloride concentrations from five and three upgradient groundwater monitoring wells were used for Site H and I, respectively.

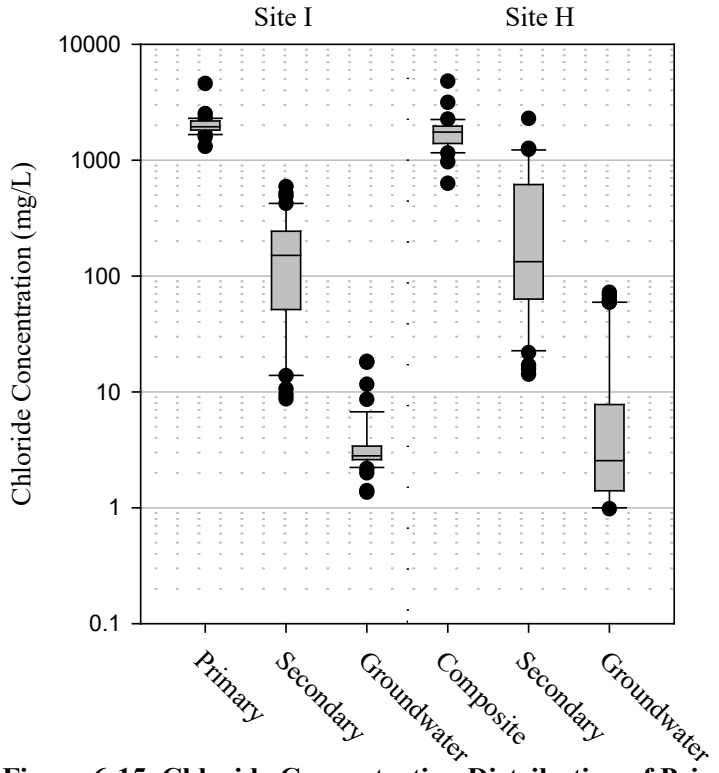


Figure 6-15. Chloride Concentration Distribution of Primary (or Composite) Leachate, Secondary Leachate, and Background Groundwater Quality for at Site I and Site H

Figure 6-16 presents a comparison of the concentration distribution of several organic compounds detected in more than 30% of the samples in both primary (or composite) and secondary leachate at each of the two sites. The detection limit was used as a surrogate value for non-detect measurements. Total organic carbon (TOC) was detected in all the leachate samples at Site H.

Tert-butyl alcohol was below the detection limit in ten out of 15 secondary and two out of 15 primary leachate samples at Site I. Tetrahydrofuran was not detected in four out of 15 secondary leachate samples for Site I. As shown in the figures below, the median LCS (or LCS-LDS composite) concentration of each compound is approximately an order of magnitude greater than the LDS concentration. This difference is similar to the difference in chloride concentrations shown in Figure 6-15.

Considerable differences in the concentrations of parameters that are indicators of MSW leachate (e.g., chloride, organic compounds) between leachate collected from the LCS versus the LDS suggest that leachate leakage from the primary liner may not be the primary source of liquids collected from the LDS. Assuming that the attenuation in chloride concentration is solely attributed to dilution by groundwater (and not due to precipitation as chloride salts), the magnitude of the difference in chloride concentration suggest that leachate leakage through the primary liner represents 10% of the liquids collected from the LDS for Sites H and I. Therefore, the primary liner leachate collection efficiency at these sites is expected to be higher than the estimates presented earlier assuming that the leachate leakage, if any, through the secondary liner is insignificant. A more detailed discussion of groundwater monitoring data for the study sites is presented in Chapter 7.

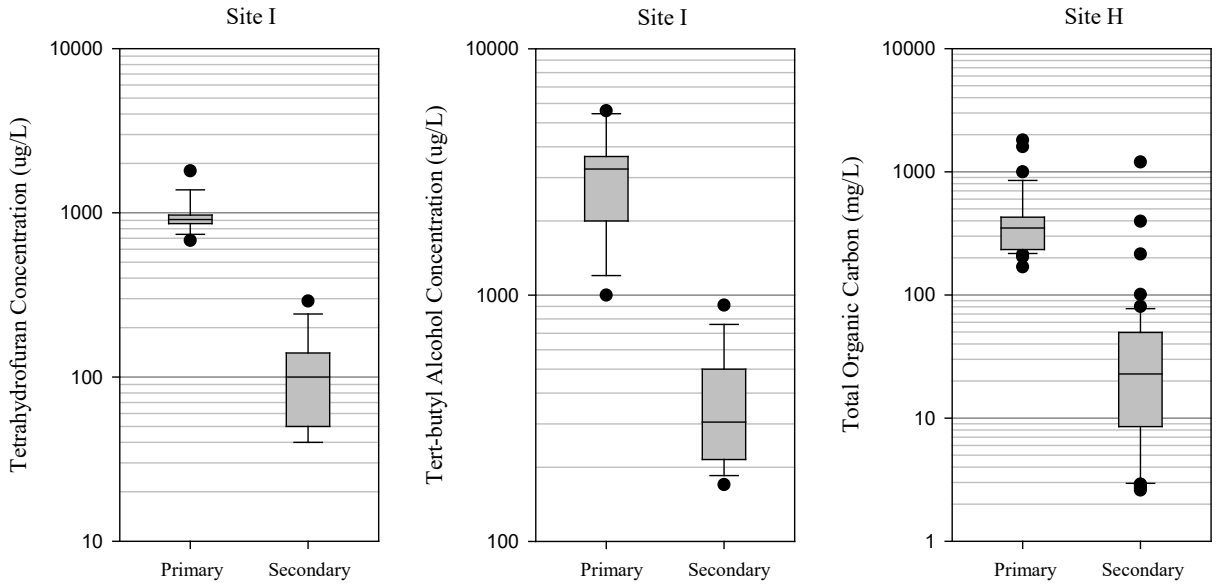


Figure 6-16. Organic Compound Concentrations in Primary (or Composite) and Secondary Leachate at Site H and I

6.5. Leachate Quality

6.5.1. Leachate Quality Data Available for the Study Sites

Leachate quality data were gathered from annual, quarterly, and/or monthly reports of the sites for the LCS and, if available/applicable, for the LDS. The data were compiled from the closure year through the most recently available data (as of 2016). Leachate quality data were available for all the study sites as availability of these data was one of the site selection criteria. Although leachate quality data were available for Site G, the data from this site represented the quality of leachate composited from closed and active cell(s) of the site. The data for Site G were not included in the analysis presented in this section as these were not truly representative of a closed MSWLF.

Leachate quality data were analyzed for eight of the nine study sites. Out of these eight sites, only LCS-LDS composite leachate quality data were available for one site (Site H). The chain-of-custody suggests that the composite samples were collected at the on-site plant intake at Site H, indicating that these samples contain the LCS and LDS liquids in the proportion of the LCS and LDS collection rates. The LDS flow collected over the quality data evaluation period represented approximately 0.4% of the total LCS and LDS liquids volume over this period; therefore, the composite leachate quality data are representative of the LCS leachate quality. Site B leachate quality data represented the quality of leachate collected from the site's closed lined cells, and leachate collected from the perimeter toe drain of the site's one unlined closed cell. Leachate quality data for study cells C1 and C2 were available separately. However, because C2 was piggybacked over C1 and both cells were closed together, leachate quality data of both the cells were analyzed as a single dataset to represent the leachate quality of study Site C.

The HHE impacts associated with leachate emissions after LCS termination would depend on the nature of the contaminants present in leachate and the associated concentrations. The impact to groundwater quality is expected to be the primary pathway of HHE impacts associated with the LCS operation termination/modification and the resultant leachate emissions. The RCRA Subtitle D regulations specify parameters for routine monitoring of groundwater quality at MSWLFs; App I and II of §258 list these parameters (referred herein to as App I and App II parameters). Some states recommend a comprehensive characterization of leachate with respect to groundwater quality parameters (e.g., FDEP 2016, UDSHW 2012). The leachate composition data for groundwater quality parameters would be an important input for contaminant fate and transport modeling for estimating concentrations of COPCs in groundwater at the point of compliance or the point of exposure resulting from potential leachate emissions from LCS termination or alternative leachate management options. This section presents an evaluation of the comprehensiveness of the available leachate quality data with respect to the App I and II groundwater monitoring parameters.

Figure 6-17 shows the count of all the parameters and that of App I and App II parameters measured for leachate quality at each study site at least once. App I and App II of §258 include 62 (15 metals and 47 organics) and 215 parameters (over 90% are organics), respectively. All of the App I parameters are also included in the App II list. A few of the sites measured the concentration of different congener compounds of App II parameters such as polychlorinated biphenyls (e.g., Aroclor-1016, Aroclor-1221) and chlordane (e.g., alpha-chlordane, beta-chlordane, and gamma-chlordane) in the leachate. While counting the number of App II parameters analyzed at each site, all congener compounds of a parameter measured at the site were grouped and counted as one App II parameter. However, all the congeners measured were counted individually and included in "other parameters."

As presented in Figure 6-17, the parameters monitored varied widely among the sites. Leachate was most comprehensively monitored at Site F among all the study sites. All but three App II parameters were analyzed for Site F leachate at least once after closure. In general, leachate at all of the sites except Sites A, B, and C was monitored for most of App I parameters at least once after closure. All the App I parameters were monitored for leachate at least once since closure at only three of the eight sites (Sites D, F, and H). Only 16, 24, 40, 49, and 54 out of 62 App I parameters were monitored for Sites A, B, C, E, and I, respectively. More than half of the parameters listed in App II were analyzed for Site E. Ten or less App II parameters (not included in App I) were monitored for Sites A, B, D, H, and I. At least one data point was available for 49 and 73 App II

parameters (not included in App I) for leachate at Sites C and E, respectively, after closure. Examples of the other parameters monitored at the study sites include field parameters (e.g., pH, dissolved oxygen), ions (e.g., chloride, ammonia, sodium), and water quality parameters (e.g., total dissolved solids (TDS), biological oxygen demand (BOD), chemical oxygen demand (COD), TOC). The primary reason for such a wide variability in the available leachate characterization data among the study sites is the lack of leachate monitoring requirements in the RCRA Subtitle D regulations. The leachate quality monitoring is primarily driven by the state regulations and the monitoring requirements of the WWTP(s) accepting the leachate for treatment and disposal.

The available leachate quality data suggest that sampling frequency varied widely among the study site from once per month at Site C to twice per year at Site D. The leachate samples were collected from a single location at all the study sites except Sites C and F. Leachate samples were collected from three and four locations at Sites C and F, respectively. The analysis frequency, in general, varied with parameters for all the study sites. Not all parameters were analyzed at every sampling event. For examples, major ions (e.g., chloride, sulfate) and field parameters were analyzed twice per year, metals were analyzed once in five years, and some organics (e.g., 1,4 Dioxane) were analyzed only once after closure at Site F. Only one measurement was available for each of the eight study sites for at least one organic compound after closure. The analysis frequency also appears to vary with time. For example, arsenic data for Site A are available once a month for a few years after closure, while only annual data are available for recent years.

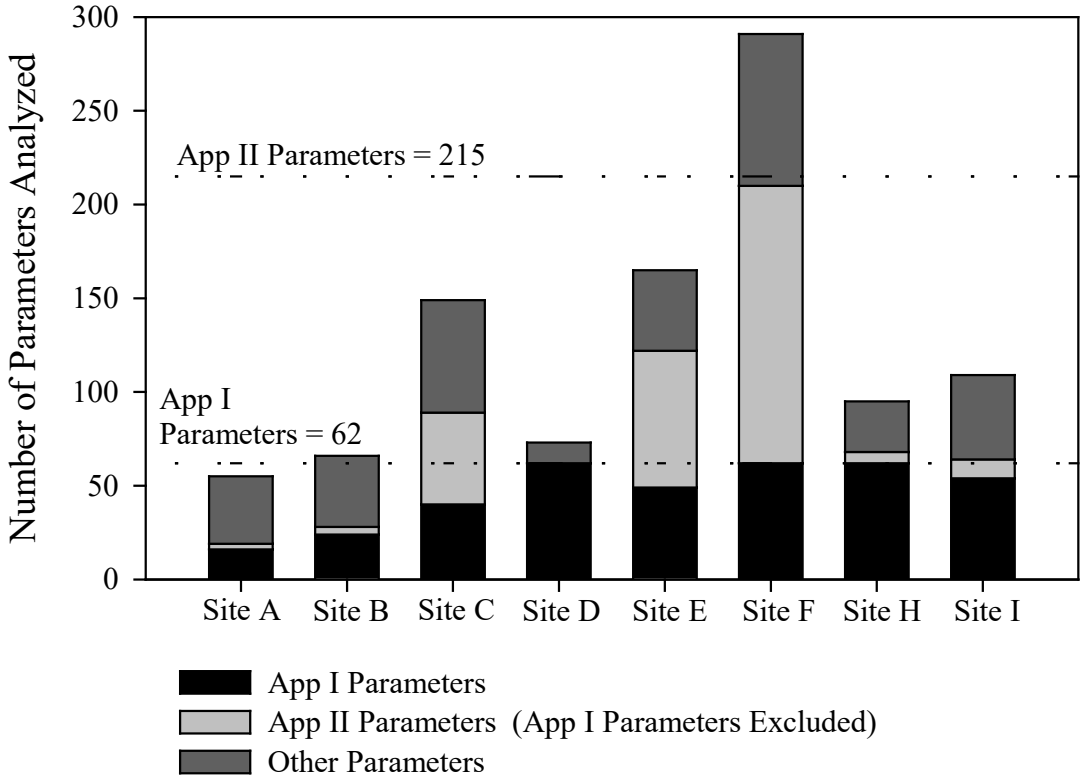


Figure 6-17. Number of §258 App I and App II Parameters Analyzed for Leachate at Eight of the Study Sites

In summary, leachate quality data were available for only three sites for all App I parameters and for only one site for all but three App II parameters. Two study sites had leachate characteristic

data available for less than half of the App I parameters. More than half of the study sites had leachate composition data available for ten or fewer App II parameters (excluding App I parameters). The characterization frequency varied among the sites from once per month to twice per year. It also varied with time and contaminants. Only a single measurement was available for a few of the organic compounds for each site after closure. Apart from the lack of the data for a large number of App I and II parameters, the number of measurements available for the constituents measured at the study sites may also limit a reliable HHE impact assessment.

6.5.2. Impact of Leachate Quality on Groundwater

The impact of leachate on HHE can be evaluated in three sequential steps, as suggested by the performance-based functional stability approach of HHE impacts evaluation (ITRC 2006). The first step is to identify the COPCs by comparing the contaminants concentrations in leachate to respective risk-based standards such as drinking water (40 CFR §141), surface water (40 CFR §445), and groundwater standards (40 CFR §258.54) as well as any state limits. The contaminants that have always been measured at concentrations below the respective risk-based thresholds are not expected to pose a risk to HHE even without any attenuation. In the second step, contaminant fate and transport modeling can be conducted to identify COPCs (identified in the first step) that would be above the respective risk-based standards at monitoring wells or at other points of compliance (e.g., surface water discharge outlet) in the event of leachate release associated with the termination or scaling-back of LCS operation. For the contaminants that are evaluated to have concentration above the respective risk-based standard at the point(s) of compliance, fate and transport modeling can be conducted to assess the concentration of the COPCs at the state-designated points of exposure in the event of leachate release from the MSWLF in the third step (ITRC 2006). As discussed in Section 6.4.1, leachate is expected to accumulate above the liner system after the LCS operation termination. The accumulated leachate would leak through the liner defects and percolate to groundwater over time. Any decline in liner performance over time or liner failure would increase the leakage rate.

The assessment presented in this chapter was limited to the first step as a detailed site-specific HHE risk assessment was beyond the scope of this study. Available leachate quality data for eight sites were analyzed to identify COPCs; the data from Site G were not analyzed as these represented the quality of leachate composited from the closed and active cell(s) at the site. This screening effort involves comparing contaminants concentration in leachate to relevant contaminant-specific and exposure-pathways-specific risk-based standards. The leachate discharge to surface water bodies (e.g., a wetland or creek near a landfill site) or to groundwater are two primary pathways for the HHE impacts associated with LCS operation termination. For the analysis presented in this report, groundwater was assumed to be the most likely water source affected by a leachate discharge after the termination of LCS operation. Thus, only the groundwater quality thresholds were used for comparison with the reported leachate constituents concentrations. This is one of the limitations of the analysis presented in this section. The HHE impacts associated with leachate releases to the surface water should also be evaluated for a comprehensive assessment.

Examples of risk-based thresholds relevant to groundwater include the Safe Drinking Water Act drinking water standards (MCLs promulgated under 40 CFR §141 or SMCL listed under 40 CFR §143), regional screening levels developed by EPA (EPA 2018), and the state-specified levels. The RCRA Subtitle D landfill regulations use MCLs (for contaminants with an MCL) as protection standards for groundwater. EPA (2017a) used MCL/SMCL to assess the leachate quality at the closed Subtitle C landfills. MCL and SMCL were used for the screening analysis presented in this

chapter. In most cases, these water quality thresholds are based on risk to human health upon consumption of the water. However, for some chemical constituents, water quality thresholds are based upon impacts to aesthetics (e.g., taste, odor) or to aquatic organisms. The study sites leachate quality data were evaluated for the following parameters:

a) Parameters with an MCL and secondary maximum contaminant level (SMCL)
b) §258 App I and II parameters as these are the parameters that are monitored for groundwater to assess impacts of MSWLFs to HHE
c) Parameters used by EPA (2017a) to assess the leachate quality of Subtitle C landfills

Overall, the leachate quality for a total of 272 parameters were evaluated for eight sites. A list of these parameters is presented in Table J-1 in Attachment J. Table J-2 in Attachment J shows a complete list of all the App I and App II parameters that were measured at least once in the leachate at each of the eight sites. The parameters are arranged in the table based on the monitoring frequency across the sites – the most frequently analyzed parameters are at the top of the table. Arsenic, cadmium, chromium, copper, nickel, selenium, and zinc were analyzed for leachate quality at all eight sites. Fourteen parameters (i.e., lead, 1,1,1-trichloroethane, p-dichlorobenzene, 1,1-dichloroethane, cis-1,2-dichloroethylene, trichloroethylene, vinyl chloride, acetone, barium, carbon disulfide, ethylbenzene, dichloromethane, silver, and toluene) were analyzed for leachate at least once at seven sites.

The concentration of all the leachate quality parameters measured at each of the eight sites (i.e., excluding Site G among the nine sites) were compared to their respective MCL. Of the 272 leachate quality parameters selected for evaluation in this study, only 100 parameters have MCL or SMCL, and among those 100 parameters, data were not available for 32 parameters (as listed in Table J-3 of Attachment J) for any of the study sites. Table 6-6 lists all the parameters measured above their respective MCL/SMCL at least once at the eight sites during the timeframe leachate quality was evaluated. Table 6-6 also shows the MCL/SMCL, the total number of data points, percent of samples that exceeded the MCL/SMCL, percent of samples that were measured above their method detection limit (MDL), percent of samples that were measured below their MDL but the MDL was greater than MCL/SMCL, and the number of sites where each parameter was evaluated. The parameters that were never measured above their MCL or SMCL are not reasonably expected to present a risk to HHE. The MDL was used as the concentration for the measurements that were below their MDL. For non-detect measurements where the MDL was not available (0.3% of all non-detects), the data point was evaluated as a non-detect without specifying any concentration (i.e., considered as a blank in the statistical analysis).

Among all the 100 parameters that have an MCL/SMCL, a total of 56 parameters exceeded their respective MCL/SMCL at least once; 43 and 30 of these parameters are App II and App I parameters, respectively. However, for 11 parameters (all organic compounds) that were reported as below detection for 100% of the samples, the reported detection limits were greater than the MCL/SMCL; these parameters cannot be conclusively determined to be below MCL/SMCL. This data quality issue represents a constraint for a reliable HHE impact analysis for a majority of the organic compounds and a few metals (e.g., antimony, beryllium, and thallium). Of the remaining 44 parameters, 29 parameters were measured above their MDL (i.e., detected) in less than 50% samples. Relatively low concentrations of a majority of contaminants, when compared to the respective MCL/SMCL, may not necessarily be indicative of stabilized conditions but may be due to the lack of exposure of landfilled waste to adequate moisture needed for hydrolysis/solubilization of contaminants from solid to liquid phase. The parameters that were

measured below their MDL in more than 50% of the samples were not further analyzed. For many parameters, MDLs varied considerably among the sites and even among sampling events at the same site.

The remaining 15 parameters (highlighted rows in Table 6-6) were measured above their MDL in more than 50% of the samples. Of these 14 parameters, six parameters (i.e., arsenic, TDS, iron, manganese, chloride, and color) were measured above their MCL/SMCL in more than 94% of the samples. Although color data were available for only one site and had only three data points, it is reasonable to assume that color would probably exceed its SMCL for MSWLF leachate. Turbidity was measured above its detection limit in all the data points. As presented in Table 6-6, over 85% of turbidity measurements were greater than 5 NTU (maximum allowable MCL for turbidity). Among the other eight parameters, aluminum was measured above its SMCL range (0.05 to 0.2 mg/L) in approximately 56% of samples, sulfate and fluoride were respectively measured above their SMCL and MCL in 25% and 13% of samples. Chromium, barium, copper, and toluene were observed above their MCLs in 0.3% to 6% of the samples. pH was measured outside of its SMCL range in 4% of all the data points. As presented in Table 6-6, there are a few parameters (e.g., mercury, 1,1,2-trichloroethane) for which all the exceedances were a result of the MDL being greater than the respective MCL. It should be noted that the number of measurements for each constituent are not evenly distributed among the study sites. For example, approximately 75% of 816 measurements for arsenic are for Site C (including C1 and C2).

Among the 15 parameters, five parameters (i.e., arsenic, chromium, barium, copper, and toluene) are also App I and App II parameters (Table J-2 in Attachment J) and were analyzed at seven of the eight study sites at least once. The 15 parameters that were measured above their MDL in more than 50% samples were further evaluated for the range of concentration, median, and arithmetic mean values among the study sites, as shown in Table 6-7. Additional leachate quality indicator parameters that do not have any MCL/SMCL (such as BOD, COD, ammonia-nitrogen, sodium, etc.) were also evaluated; the concentrations of these parameters also varied over a wide range among the sites. Table 6-7 also shows the total number of data points, percent of detected samples, and the number of study sites that analyzed for the respective leachate quality parameter. The MCL/SMCL for each parameter is also listed as a point of comparison with the results.

As shown in Table 6-7, the concentration of parameters generally varied over a wide range among the sites. The median concentration of seven parameters (i.e., arsenic, turbidity, TDS, chloride, iron, manganese, and color) was observed to be higher than their respective MCL/SMCL (highlighted rows in Table 6-7). The median concentrations of the other eight parameters with an MCL or SMCL shown in Table 6-7 were below their respective MCL or SMCL. It should be noted that the analysis presented in this report was conducted solely based on the federal MCL/SMCL. States may have state-specific risk-based standards, and landfill owners may be required to use state-specific standards for an assessment of impacts to HHE.

Furthermore, the leachate quality is typically reflective of the decomposition status of the bottom-most waste layer and does not necessarily represent the degree of stabilization of the entire landfill. A well-decomposed waste layer above the LCS may attenuate the concentration of parameters such as BOD and COD that are commonly used to assess leachate and waste stability (Kjeldsen et al. 2002). The biodegradable organics in leachate from fresher waste in the above layers would be consumed as it percolates through a well-decomposed carbon-limited waste layer above the LCS.

Table 6-6. Summary of Leachate Parameters that Exceeded the MCL or SMCL at Least Once at the Study Sites

Parameter	Units	MCL[1]/ SMCL[2]	Total Data Points	Samples (%)			Landfills Reporting
				Detected	Above MCL[1] or SMCL[2]	Not-detected and MDL[4, 5]> MCL/SMCL	
Contaminants with a Primary Drinking Water Standard							
Arsenic	mg/L	0.01	816	97%	94%	2%	8
Barium	mg/L	2	275	92%	5%	1%	7
Chromium	mg/L	0.1	665	97%	6%	1%	8
Copper	mg/L	1.3[3]	759	54%	1%	0%	8
Fluoride	mg/L	4	140	54%	13%	2%	4
Toluene	µg/L	1000	324	54%	0.3%	0%	7
Turbidity	NTU	5[6]	246	100%	86%	0%	4
Nitrogen, nitrate	mg/L	10	486	34%	6%	0%	7
Nitrogen, nitrite	mg/L	1	360	6%	56%	54%	3
Antimony	mg/L	0.006	256	38%	83%	57%	6
Beryllium	mg/L	0.004	254	6%	22%	21%	6
Cadmium	mg/L	0.005	567	17%	6%	4%	8
Lead	mg/L	0.015[3]	677	25%	6%	2%	7
Mercury (inorganic)	mg/L	0.002	424	3%	5%	5%	5
Selenium	mg/L	0.05	285	25%	16%	9%	8
Thallium	mg/L	0.002	196	6%	87%	85%	6
Cyanide (free cyanide)	mg/L	0.2	701	35%	0.1%	0%	5
1,1,2-trichloroethane	µg/L	5	187	0%	30%	30%	5
1,1-dichloroethylene	µg/L	7	187	0%	27%	27%	5
1,2,4-trichlorobenzene	µg/L	70	49	0%	22%	22%	3
1,2-Dibromo-3-chloropropane	µg/L	0.2	175	1%	99%	99%	5
1,2-dichloroethane	µg/L	5	230	17%	28%	28%	6
1,2-dichloropropane	µg/L	5	187	2%	29%	29%	5
Benzene	µg/L	5	324	38%	31%	27%	6
Benzo(a)pyrene	µg/L	0.2	26	0%	88%	88%	2
Carbon tetrachloride	µg/L	5	187	0%	28%	28%	5
Chlorobenzene	µg/L	100	263	34%	11%	11%	6
Cis-1,2-dichloroethylene	µg/L	70	284	24%	11%	11%	7
Di(2-ethylhexyl) phthalate	µg/L	6	80	10%	78%	69%	3
Dichloromethane	µg/L	5	312	11%	39%	34%	7
Endrin	µg/L	2	28	4%	25%	25%	2
Ethylene dibromide	µg/L	0.05	173	0%	99%	99%	5
Heptachlor	µg/L	0.4	53	6%	15%	15%	2
Heptachlor epoxide	µg/L	0.2	33	27%	24%	24%	2
Hexachlorobenzene	µg/L	1	26	0%	73%	73%	2
Hexachlorocyclopentadiene	µg/L	50	26	0%	8%	8%	2

Notes: (1) Maximum contaminant level (MCL) per 40 CFR §141; (2) Secondary maximum contaminant level (SMCL) per 40 CFR §143; (3) treatment-technique-specific action level; water systems are required to take additional steps if 10% of tap water samples exceed the action level; (4) Method Detection Limit; (5) Number of parameters that were not detected and had their MDL greater than MCL/SMCL; (6) varies with treatment technique; maximum allowable is 5 NTU. Highlighted rows are for the parameters that were detected in more than 50% of the samples.

Table 6-6 (contd.). Summary of Leachate Parameters that Exceeded the MCL or SMCL at Least Once at the Study Sites

Parameter	Units	MCL[1]/ SMCL[2]	Total Data Points	Samples (%)			Landfills Reporting
				Detected	Above MCL[1] or SMCL[2]	Not-detected and MDL[5, 6]> MCL/SMCL	
gamma-BHC	µg/L	0.2	26	4%	31%	31%	2
p-dichlorobenzene	µg/L	75	323	48%	10%	10%	7
Pentachlorophenol	µg/L	1	26	0%	92%	92%	2
Polychlorinated biphenyls	µg/L	0.5	18	17%	61%	50%	2
Styrene	µg/L	100	276	5%	9%	9%	6
Tetrachloroethylene	µg/L	5	263	1%	31%	31%	6
Toxaphene	µg/L	3	12	0%	67%	67%	1
Trans-1,2-Dichloroethylene	µg/L	100	225	0%	13%	13%	6
Trichloroethylene	µg/L	5	275	7%	31%	30%	7
Vinyl chloride	µg/L	2	286	28%	51%	47%	7
Contaminants with a Secondary Drinking Water Standard							
Total dissolved solids	mg/L	500[2]	723	100%	99%	0%	6
Sulfate	mg/L	250[2]	758	85%	25%	0%	8
Chloride	mg/L	250[2]	785	100%	97%	0%	7
Iron	mg/L	0.3[2]	626	99%	99%	0%	7
Manganese	mg/L	0.05[2]	599	99%	98%	0%	7
pH	S.U.	6.5-8.5[2]	1066	100%	4%	0%	8
Aluminum	mg/L	0.05-0.2[2]	114	73%	56%	0%	4
Color	C.U.	15[2]	3	100%	100%	0%	1
Silver	mg/L	0.1[2]	259	16%	3%	2%	7
Zinc	mg/L	5[2]	799	42%	1%	1%	8

Notes: (1) Maximum contaminant level (MCL) per 40 CFR §141; (2) Secondary maximum contaminant level (SMCL) per 40 CFR §143; (3) treatment-technique-specific action level; water systems are required to take additional steps if 10% of tap water samples exceed the action level; (4) Method Detection Limit; (5) Number of parameters that were not detected and had their MDL greater than MCL/SMCL Highlighted rows are for the parameters that were detected in more than 50% of the samples.

Table 6-7. Summary of Leachate Quality Parameters that were Detected in more than 50% of Samples and Exceeded the Respective Drinking Water Standard at least Once at the Study Sites

Parameter	Units	MCL[1]/ SMCL[2]	Mini-mum	Maxi-mum	Median	Mean	Total Data Points	Samples Detected (%)	Landfills Reporting
Contaminants with a Primary Drinking Water Standard									
Arsenic	mg/L	0.01	0.0047	0.997	0.05	0.06	816	97%	8
Fluoride	mg/L	4	0.03	19.5	0.6	1.8	140	54%	4
Barium	mg/L	2	0.003	100	0.45	1.2	275	92%	7
Chromium	mg/L	0.1	0.00041	10	0.03	0.08	665	97%	8
Copper	mg/L	1.3	0.0007	50	0.01	0.1	759	54%	8
Toluene	µg/L	1000	0.11	3,400	5	40	324	54%	7
Turbidity	NTU	5[3]	0.1	>1,000	27	95	246	100%	4
Contaminants with a Secondary Drinking Water Standard									
Total dissolved solids	mg/L	500[2]	200	17,000	4,140	4,297	723	100%	6
Chloride	mg/L	250[2]	1.8	4,580	1,260	1,284	785	100%	7
Iron	mg/L	0.3[2]	0.1	12,400	5.1	27	626	99%	7
Manganese	mg/L	0.05[2]	0.0004	34.1	0.29	0.8	599	99%	7
Color	C.U.	15[2]	180	800	520	500	3	100%	1
pH	S.U.	6.5-8.5[2]	4.94	11.54	7.5	7.5	1,066	100%	8
Sulfate	mg/L	250[2]	0.07	1,430	70	141	758	85%	8
Aluminum	mg/L	0.05 to 0.2[2]	0.023	27.3	0.20	0.6	114	73%	4
Other Contaminants									
Specific Conductance	µmho s/cm		3.08	21,750	8,395	8,983	1,066	100%	7
Alkalinity	mg/L		1.1	8,550	2630	2696	697	83%	6
Ammonia-N	mg/L		2.1	3,914	470	532	730	100%	5
BOD[4]	mg/L		3.7	4,620	76	153	787	98%	7
COD[4]	mg/L		51.6	7740	987	1,186	606	100%	7
TOC[4]	mg/L		17	2700	238	276	480	100%	5
Calcium	mg/L		20	470	71	89	587	100%	5
Magnesium	mg/L		10.7	360	85	100	588	100%	5
Potassium	mg/L		0.5	1,040	364	376	589	100%	6
Sodium	mg/L		58	25,000	960	1,095	648	100%	7
Nickel	µg/L		4.55	1,900	120	126	634	97%	8

Notes:

(1) Maximum contaminant level (MCL) per 40 CFR §141
(2) Secondary maximum contaminant level (SMCL) per 40 CFR §143
(3) Varies with treatment technique; maximum allowable is 5 NTU
(4) COD = chemical oxygen demand, BOD = biochemical oxygen demand, TOC = total organic carbon.

-Highlighted rows are for the parameters that had a median concentration greater than their respective MCL/SMCL

Significant figures used vary. The significant figures used for the minimum and maximum concentration are the same as those corresponding to the respective concentration values. The number of significant figures used for the mean and median for each parameter is the same as that of the concentration value with the least number of significant figures.

A common practice to assess the potential for waste-derived leachate to impact HHE is to compare the leachate concentration to a regulatory water quality threshold (which are usually based on safe water for human consumption). This analysis does not imply that waste-derived leachate would ever be consumed; instead, this analysis allows screening of the chemicals that might pose a concern if the waste-derived leachate mixed with a drinking water source and the degree of dilution and attenuation that would be required to alleviate such concerns. Parameters that are consistently measured above their respective MCL/SMCL may have the potential to impact groundwater quality, while those below should not have an impact. The concentrations of contaminants detected in more than one-third of the reported leachate samples, and at a concentration greater than MCL/SMCL, were divided by their respective MCL/SMCL. Leachate constituents with a larger ratio of concentration to MCL/SMCL would require more dilution and attenuation or treatment for mitigating the impact on groundwater. This ratio can be thought of as the dilution and attenuation factor (DAF) necessary to ensure that leachate mixed with groundwater fall below the MCL/SMCL for a COPC.

Figure 6-18 presents the distribution of the MCL normalized concentration of the frequently detected parameters with MCLs. The median MCL-normalized concentration is greatest for arsenic, followed by benzene. The detection limit was used as the concentration for the samples below detection. As presented in Table 6-6, benzene was detected in less than 40% of the samples and its MDL was greater than the MCL for 27% of the sample analyzed. Benzene was below detection in over 85% of the samples with MCL-normalized concentration values greater than one (1). Similarly, chlorobenzene and p-dichlorobenzene were below detection in all of the samples with MCL-normalized concentration values greater than one (1). Fluoride concentrations were below the respective MCL in more than 85% of the samples. Barium, chromium, and copper were below the respective MCL in more than 90% of the samples. Among all the constituents with MCL that are measured at the study sites (except for turbidity), arsenic exhibited the greatest DAF.

Figure 6-19 presents the distribution of the SMCL normalized concentration of the frequently detected parameters with SMCLs. The median SMCL-normalized concentration is greatest for color, followed by iron. As mentioned earlier, color data were available only for three sampling events at one site. Iron concentration is more than ten times its SMCL for over 75% of the measurements from the study site. A DAF of approximately 17 would be needed for the median iron concentration to decline below its SMCL. A DAF of less than ten would be needed for TDS, chloride, and manganese concentrations to decline below their respective SMCL.

It should be noted that the SMCLs of 0.3 mg/L for iron is based on aesthetics (e.g., color, taste) and technical considerations (e.g., impact to water treatment process) and are not based on human-health risk considerations. As a point of comparison, the human-health-risk-based regional screening level developed by EPA for tapwater ingestion for iron is 14 mg/L (for Hazard Index of 1.0) (more than 50 times the SMCL), which is greater than the median iron concentration of 5.1 mg/L measured at the study sites.

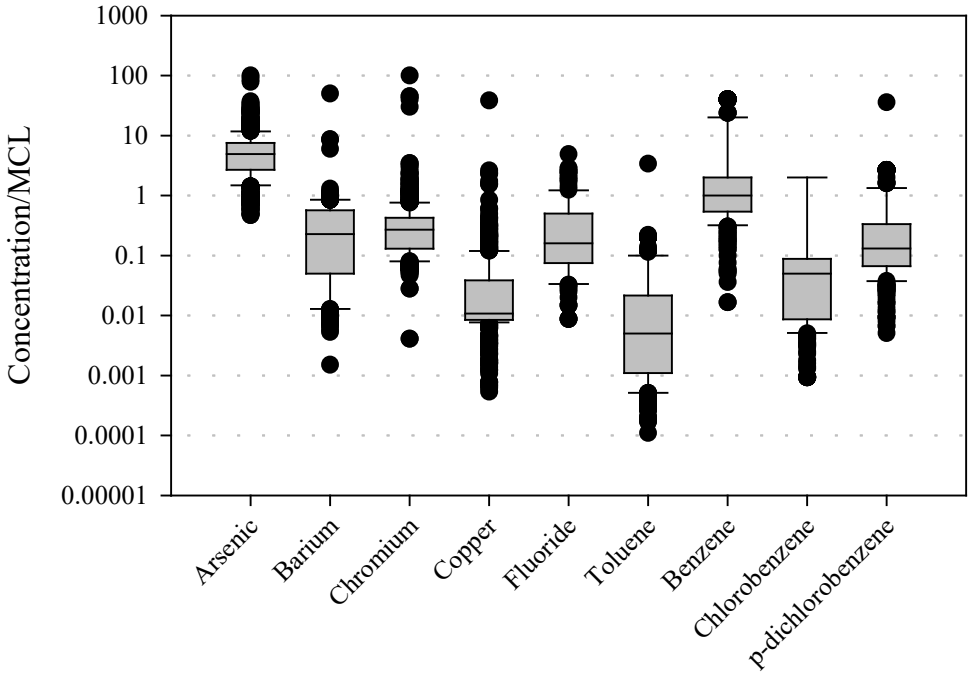


Figure 6-18. Distribution of MCL-normalized Concentrations of Frequently Detected Parameters

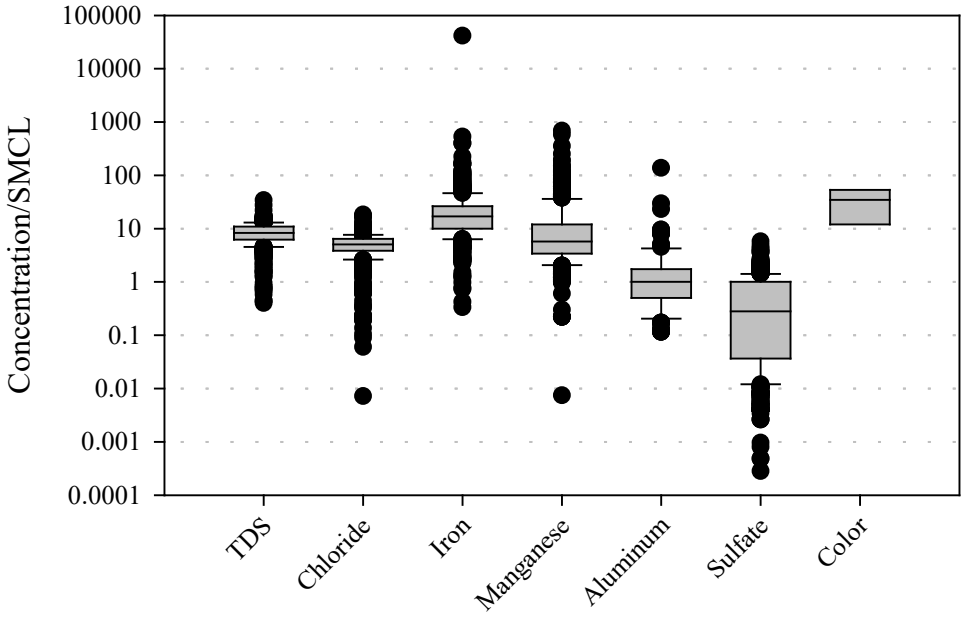


Figure 6-19. Distribution of SMCL-normalized Concentrations of Frequently Detected Parameters

It should be noted that leachate released from the base of the liner could potentially undergo natural attenuation in the surrounding soil and groundwater by processes such as oxidation, adsorption, hydrolysis, precipitation, biological degradation, and/or dilution by groundwater. Iron and manganese in leachate are typically in reduced forms and precipitate out as oxides with leachate oxidation. Iron precipitates are also expected to sorb arsenic and reduce the dissolved arsenic

concentration of leachate. Dilution with the groundwater is expected to the primary mechanism for the natural attenuation of TDS and chloride. The impact of these processes on concentrations at the point of compliance or receptor wells can be assessed using contaminant fate and transport models such as EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP). The fate and transport modeling should be performed to estimate the concentrations of the COPCs at the receptor wells (i.e., point of exposure) to assess the potential HHE impacts. As mentioned earlier, the analysis presented in this chapter only identifies the COPCs associated with leachate. A detailed fate and transport modeling was not performed to evaluate the concentrations of these CPOCs at the point of compliance or at the receptor wells for a comprehensive HHE impact assessment

6.5.3. Temporal Analysis of Leachate Quality

The observations made based on aggregate analysis conducted in the previous section may not conclusively be extended to individual sites due to issues such as wide variation in the number of data points from individual sites. For example, 75% of arsenic measurements were associated with Site C while less than 1% measurements were from Site I. The objective of the assessment presented in this section is to present an analysis of the data for CPOCs and other major leachate indicator parameters for individual sites. Site-specific distribution and temporal trends in leachate quality since closure until the most recent available data (as of 2016) were evaluated for five parameters (i.e., arsenic, iron, manganese, TDS, and chloride) that were recorded above their MDL in more than 50% of their measurements and which had a median concentration above their MCL/SMCL. Three major leachate quality indicator parameters (i.e., BOD, COD, and ammonia-N) that do not have any specific MCL/SMCL and pH were also evaluated. The leachate quality data for all sampling locations at a site were compiled chronologically since the closure. For example, Site C had three leachate sample collection points; all the leachate quality data for a parameter (e.g., pH) collected from the three sampling locations were organized in a chronological order starting the year of site closure. The first sampling event of the closure year is presented at Year 0 in the figures presented below. A similar data organization was performed for Sites E and F, which had three and four leachate sample collection points, respectively.

6.5.3.1. Arsenic

The distribution of arsenic (non-speciated) concentrations in leachate for the eight study sites (starting from the respective closure year) is shown in Figure 6-20. Among these sites, the arsenic concentration varied in the range of 0.0047 to 0.997 mg/L, with a median value of 0.050 mg/L. Approximately 94% of the total 816 measured concentrations among the sites were greater than the arsenic MCL of 0.01 mg/L. Townsend et al. (2015b) observed a median (of the mean for each site) leachate arsenic concentration of 0.044 mg/L from a review of data from 54 MSWLFs in Florida. Based on the median arsenic concentration, a DAF of approximately five would be needed to lower the leachate concentration of arsenic below its MCL. The arsenic concentrations varied among the sites over 1-2 orders of magnitude; for example, at Site C, arsenic concentrations were generally within the range of 0.01 to 0.1 mg/L, whereas, at Site H, arsenic concentrations were generally greater than 0.1 mg/L. The large variation in the observed arsenic concentrations among the sites may be the result of the variable nature of the waste deposited and/or maybe related to the type of cover soil used at the site.

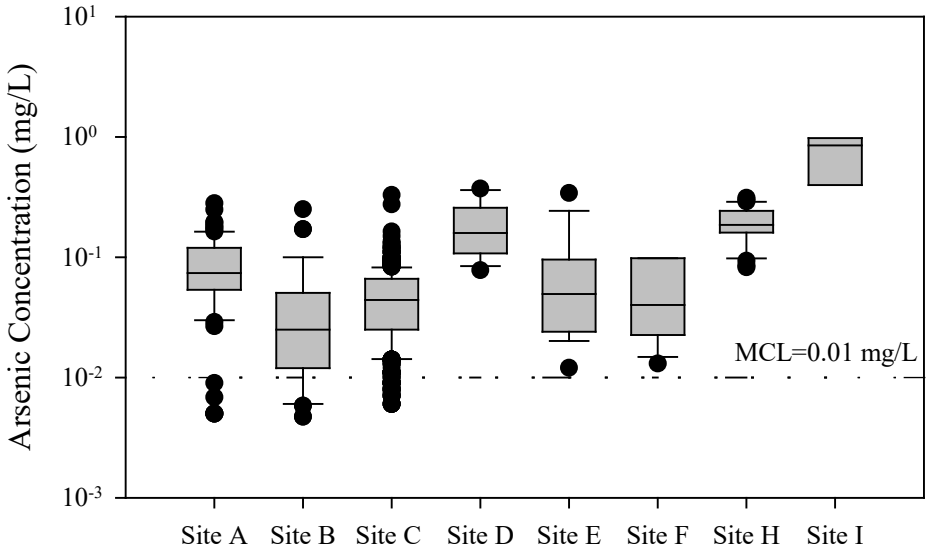


Figure 6-20. Distribution of Leachate Arsenic Concentration at Eight Study Sites Since Closure

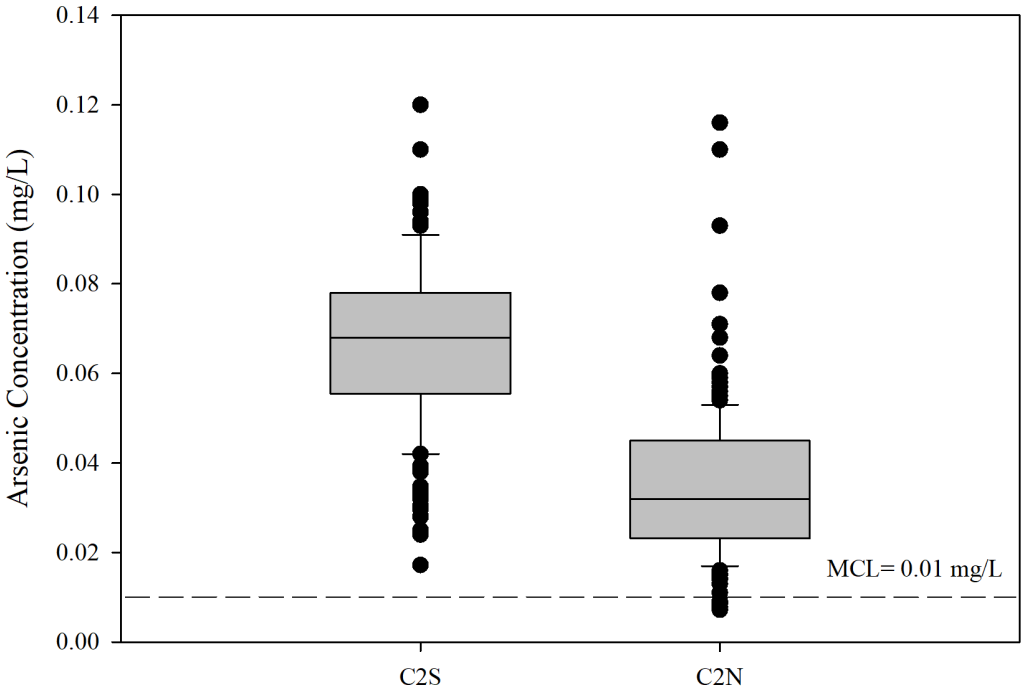


Figure 6-21. Distribution of Arsenic Concentration in Leachate from Two Sumps of the Study Cell C2

Although large differences appear to exist in arsenic concentration among several sites (e.g., Sites A, B, and H), these differences do not appear to be due to variation in precipitation or waste moisture content. For example, the arsenic concentration at Site A (with extensive leachate recirculation) is in the range of arsenic concentrations measured at Sites B and D, where leachate was not recirculated. Arsenic concentration at Sites C and D, which have similar annual rainfall, appear to be considerably different. The arsenic concentration of Site F, which is located in an

arid zone, does not appear to be significantly different than that of Site C, which is located in a higher precipitation zone. Large differences in concentration can exist among the leachate collected from different areas of the landfills. For example, arsenic concentrations in leachate collected from two of the sumps of cell C2 at Site C were substantially different, as shown in Figure 6-21. The median arsenic concentration for sump C2S leachate is more than double the median for sump C2N. The wide variation in arsenic concentrations of leachate collected from a single cell suggests that the nature and age of the deposited materials (MSW, daily cover soil) have a considerable influence on leachate quality.

A temporal trend of arsenic concentration in Sites B, D, and H is shown in Figure 6-22. As shown in Figure 6-22, the observed temporal trend of the arsenic concentration varied among the sites. In general, the arsenic concentration in the leachate at Site B appears to be decreasing over time after closure, whereas at Site D, the arsenic concentration showed an increasing trend. At Site H, arsenic concentration exhibited a variably increasing trend since closure. Measurements were available for only three and five sampling events for Sites F and I, respectively. Temporal trends were not evaluated for these two sites due to the small number of measurements available. Overall, arsenic showed a declining trend for Sites A, B, C2, and E. The four most recent arsenic measurements at Site B were below the MCL, as shown in Figure 6-22. The most recent set of measurements at Site A, C2, and E were above the MCL. Arsenic exhibited an increasing trend for Sites C1, D, and H.

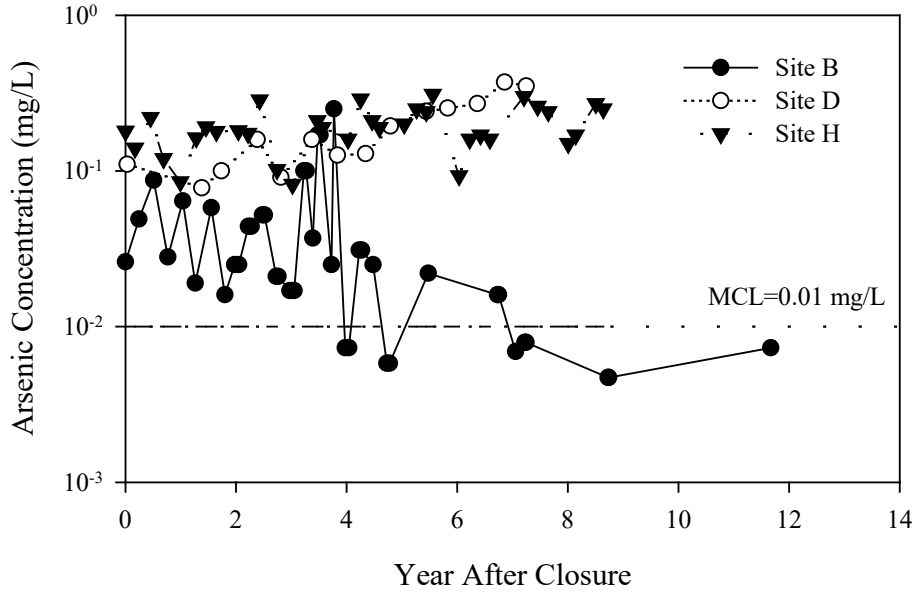


Figure 6-22. Temporal Variability in Leachate Arsenic Concentration of Sites B, D, and H Since Closure

The contaminant mass release rate, which would dictate the magnitude of impact to groundwater quality, may reduce over time due to declining leachate generation rate even in the case where the contaminant concentration is increasing. Figure 6-23 presents the arsenic release rate per unit area (lbs per acre per day) for Site H, which exhibited a variably increasing trend for arsenic concentration but a decreasing leachate generation rate trend. The arsenic release rates per unit area were estimated by multiplying the arsenic concentration to the corresponding daily leachate generation rate (estimated from the available month leachate generation data) and dividing by the landfill area. The arsenic release rate from Site H exhibited a slightly declining trend over time,

which is indicative of declining HHE impacts with respect to leachate arsenic emissions in the event leachate migrate into the environment after the termination of LCS operation.

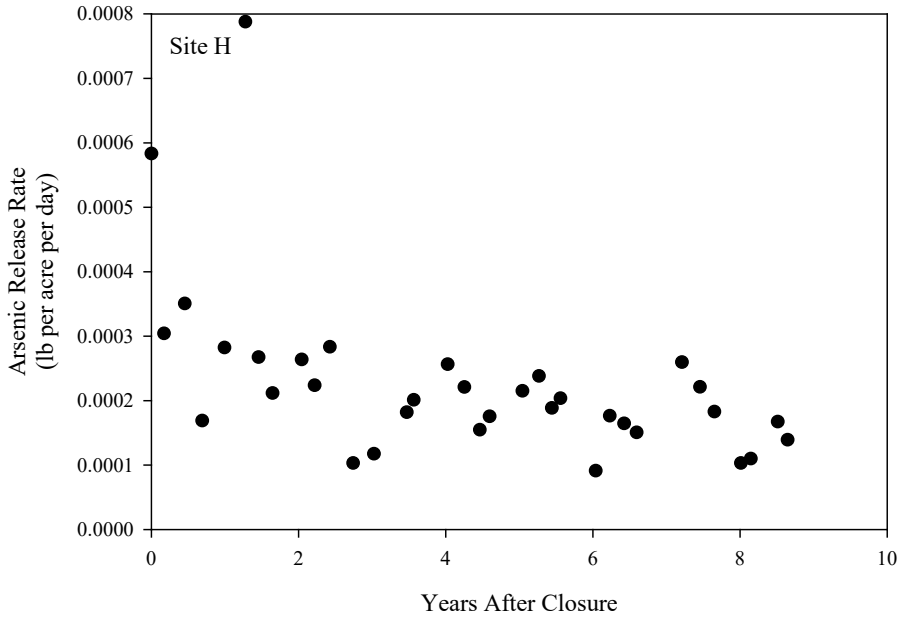


Figure 6-23. Temporal Variability in Arsenic Release Rate of Site H Since Closure

6.5.3.2. Iron

Figure 6-24 shows the distribution of leachate iron concentrations in seven study sites since the closure year. Among these sites, the iron concentration ranged from 0.1 to 12,400 mg/L, with a median value of 5.1 mg/L. Excluding an outlier (one data point with an iron concentration of 12,400 mg/L) at one site, iron concentrations generally varied between 0.1 and 121 mg/L. Over 99% of over 600 measured concentrations among the sites were greater than the iron SMCL of 0.3 mg/L. Townsend et al. (2015b) observed a median (of the mean for each site) iron concentration of 7.74 mg/L from a review of leachate iron concentration at 56 MSWLFs in Florida. Based on the observed median concentration at seven of the study sites, a DAF of approximately 17 would be needed to lower leachate iron concentrations to below the SMCL.

The temporal trend observed for leachate iron concentrations varied among the study sites. As an example, as shown in Figure 6-25, iron appears to be slightly decreasing over time in the leachate of Site I, while the iron concentration at Site H does not show a clear trend.

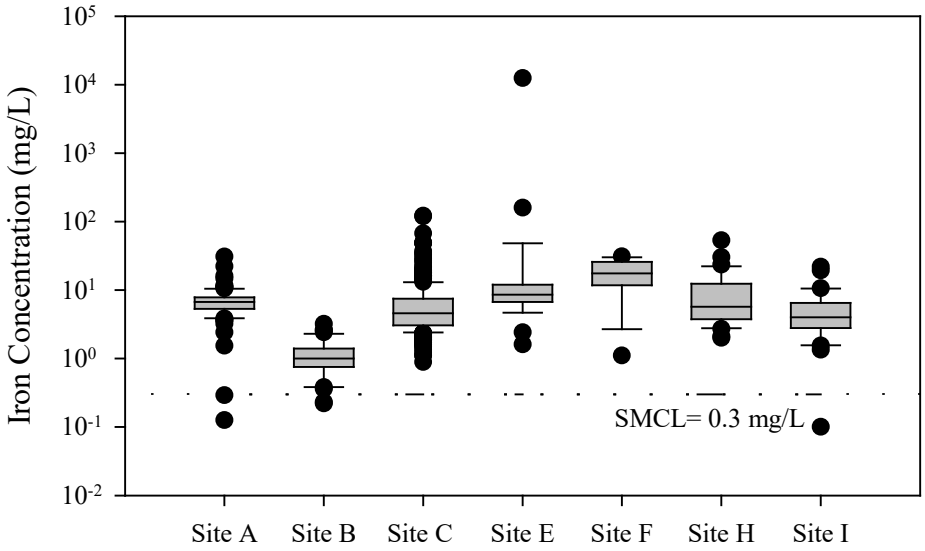


Figure 6-24. Distribution of Leachate Iron Concentration at Seven Study Sites Since Closure

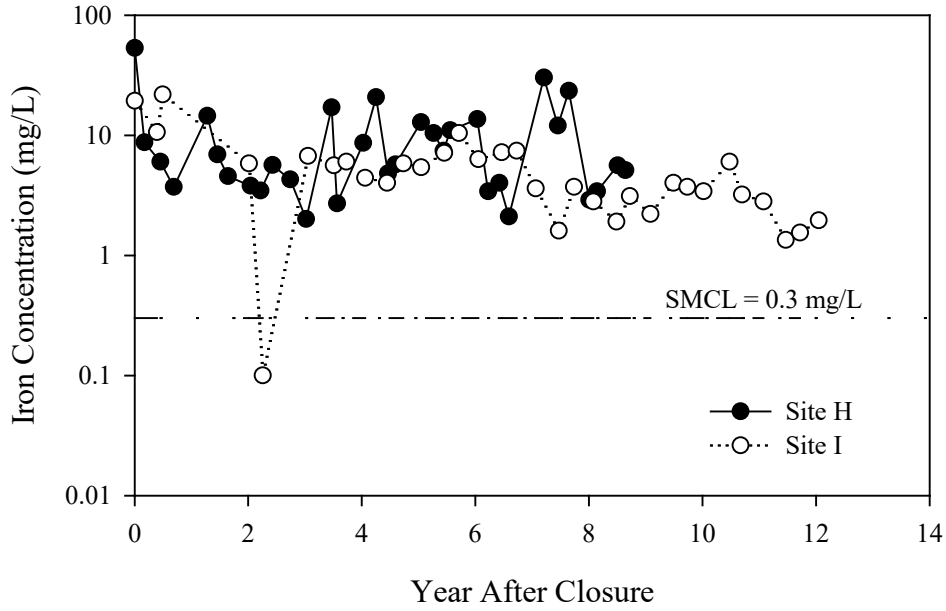


Figure 6-25. Temporal Variability in Leachate Iron Concentration of Sites H and I Since Closure

6.5.3.3. Manganese

Figure 6-26 shows the distribution of leachate manganese concentrations at seven of the study sites after closure. Among these sites, the manganese concentration varied in the range of 0.0004 to 34.1 mg/L, with a median value of 0.29 mg/L. Townsend et al. (2015b) observed a median (of the mean for each site) manganese concentration of 0.19 mg/L in leachate quality data from 11 MSWLFs in Florida. More than 98% of approximately 600 measured concentrations among the sites were greater than the manganese SMCL of 0.05 mg/L. Based on the observed median manganese concentration from the seven sites, a DAF of approximately 5.7 would be needed to lower median leachate manganese concentrations to below the SMCL.

Leachate manganese concentrations did not appear to stabilize with time at any site. As an example, Figure 6-27 shows the temporal variation in leachate manganese concentration at Site B and H. Site B appears to show manganese concentration varying in a smaller range just after closure followed by a decreasing trend, while manganese concentrations at Site H do not show an increasing or decreasing trend.

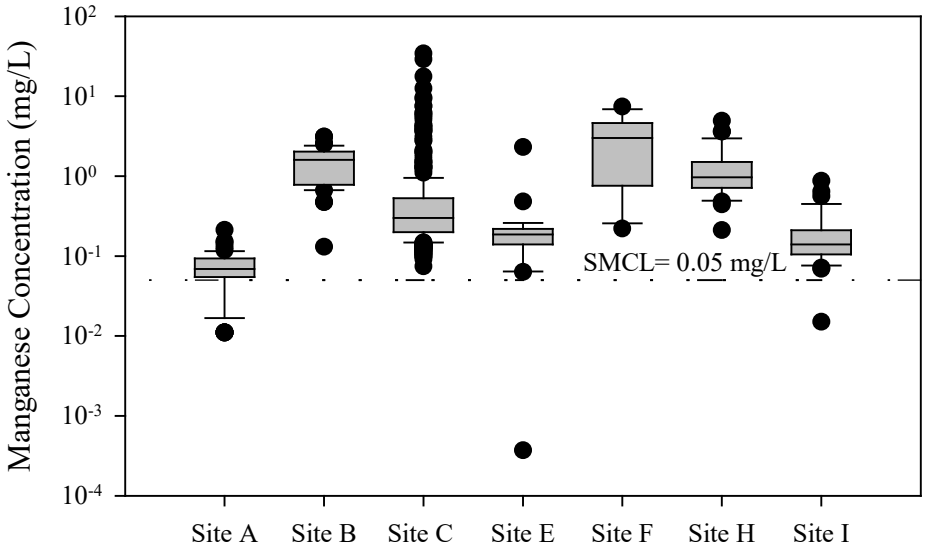


Figure 6-26. Distribution of Leachate Manganese Concentration at Seven Study Sites Since Closure

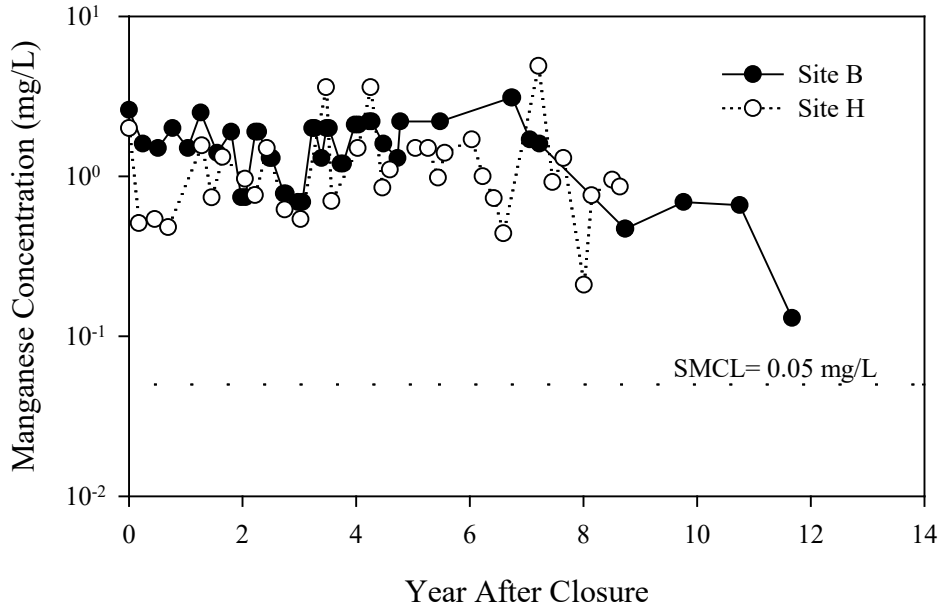


Figure 6-27. Temporal Variability in Leachate Manganese Concentration of Sites B and H Since Closure

6.5.3.4. Total Dissolved Solids (TDS)

Figure 6-28 shows the distribution of leachate TDS for six of the study sites starting from the closure year (Site I had only one TDS measurement in this duration). Among these sites, TDS varied over a wide range from 200 to 17,000 mg/L, with a median of 4,140 mg/L. As a point of

comparison, Townsend et al. (2015b) observed a median (of the mean for each site) TDS concentration of 3,723 mg/L in leachate quality data collected from 56 MSWLFs in Florida. More than 98% of over 700 measured TDS concentrations among these sites were greater than the SMCL of TDS (i.e., 500 mg/L). Based on the median leachate TDS concentration at six study sites, a DAF of 8.3 would be needed to lower leachate TDS concentrations to below the SMCL.

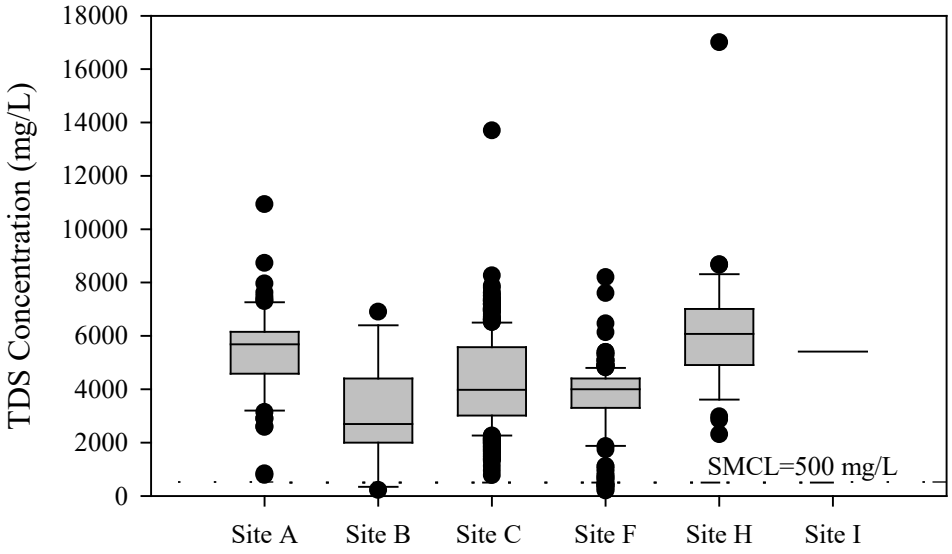


Figure 6-28. Distribution of Leachate TDS Concentration at Six Study Sites Since Closure

In general, TDS at the sites appears to be stable to slightly decreasing with time. As an example, Figure 6-29 shows the temporal trend of TDS concentrations in Site C leachate. The TDS concentration appears to be slightly decreasing without stabilizing in more than 18 years since closure.

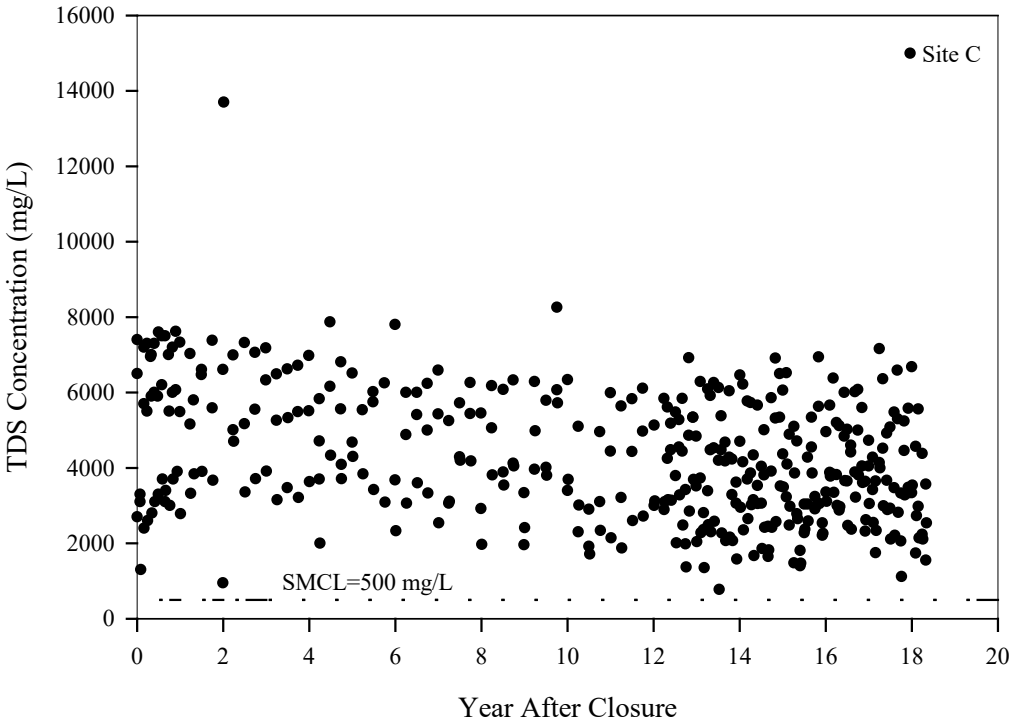


Figure 6-29. Temporal Variability in Leachate TDS Concentration of Site C Since Closure

6.5.3.5. Chloride

Figure 6-30 shows the distribution of leachate chloride concentration in seven of the study sites starting from the year of their closure. Among these seven sites, chloride concentration varied over a wide range of 1.8 to 4,580 mg/L with a median of 1,260 mg/L. Townsend et al. (2015b) reported a median (of the mean for each site) chloride concentration of approximately 863 mg/L based on the leachate quality data from 57 MSWLFs in Florida. More than 97% of over 750 measured chloride concentrations among these sites were greater than the chloride SMCL of 250 mg/L. Based on the median chloride concentration, a DAF of approximately five would be needed to lower leachate chloride concentrations to below the SMCL.

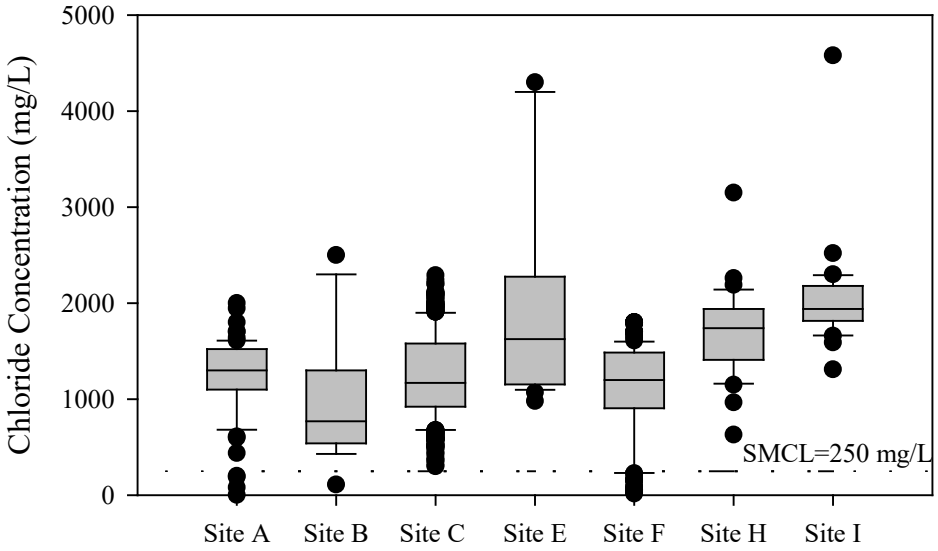


Figure 6-30. Distribution of Leachate Chloride Concentration at Seven Study Sites Since Closure

In general, the chloride concentration in leachate appears to be varying over a wide range after closure among the sites except for Site I. Figure 6-31 shows the temporal variation in leachate chloride concentration at Sites C and I as an example. Site I showed a slightly increasing trend of chloride concentration with time, whereas Site C had chloride concentration varying in a wide range without showing any specific trend. A similar trend in chloride concentrations was observed at the other sites as well (Site A, B, E, F, and H).

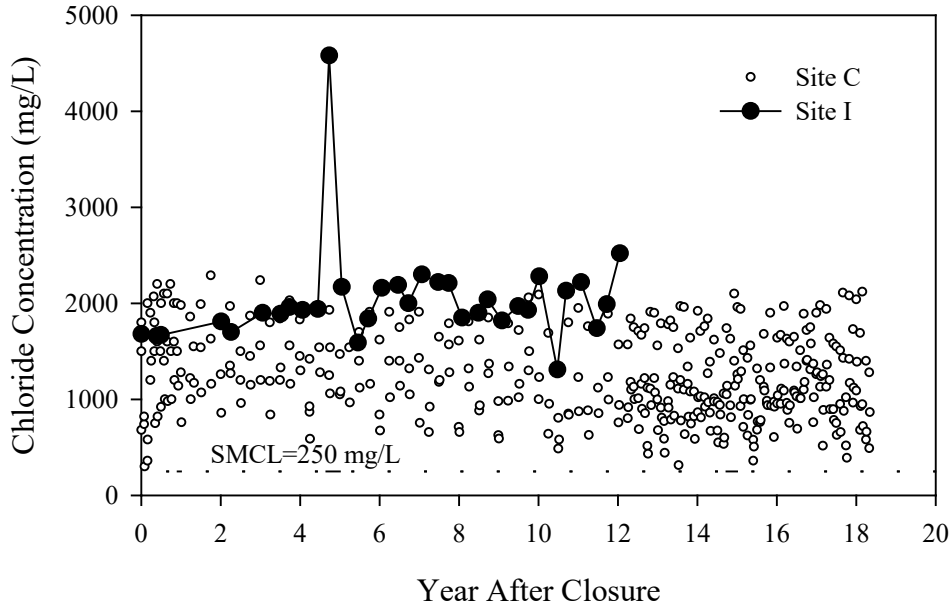


Figure 6-31. Temporal Variability in Leachate Chloride Concentration of Sites C and I Since Closure

6.5.3.6. pH

Figure 6-32 shows the distribution of leachate pH at eight study sites since closure. Leachate pH among the sites ranged from 4.9 to 11.5 s.u.; however, in general, the leachate pH remained within

the SMCL range of 6.5 to 8.5 s.u. Leachate at the study sites was generally alkaline, with a median value of 7.5. Leachate pH at all of the sites except Sites B and F ranged from 7.5 to 9 s.u., which is typical of the methanogenic phase, as reported by Kjeldsen et al. (2002). Townsend et al. (2015b) observed a median (of the mean for each site) pH of 7.28 s.u. based on leachate quality data collected from 57 MSWLFs in Florida.

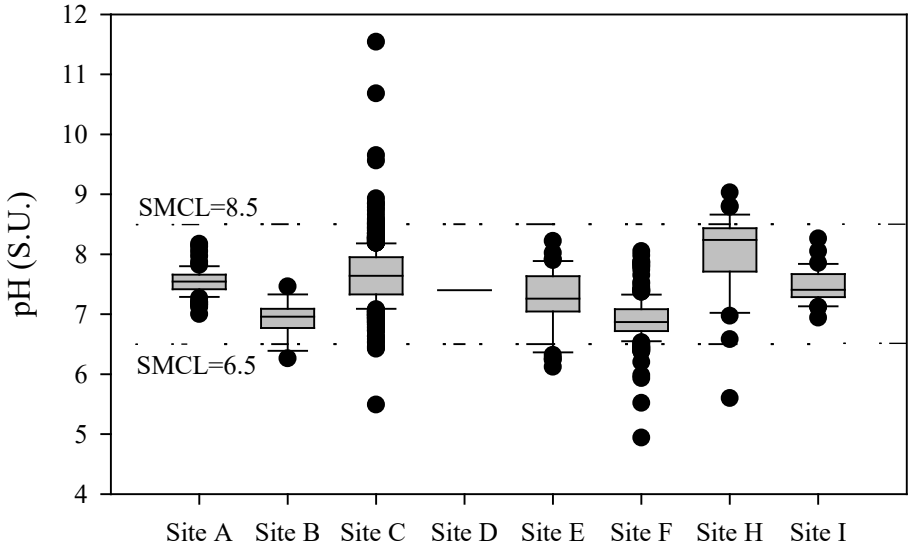


Figure 6-32. Distribution of Leachate pH at Eight Study Sites Since Closure

The leachate pH at each site temporally appears to be stable; an evaluation of the buffering system and its role on leachate pH was not conducted in this study. As an example, a temporal trend of leachate pH at Sites A and F are shown in Figure 6-33. The pH at both sites generally varied in a smaller range and was alkaline; pH at Site A was slightly higher than at Site F.

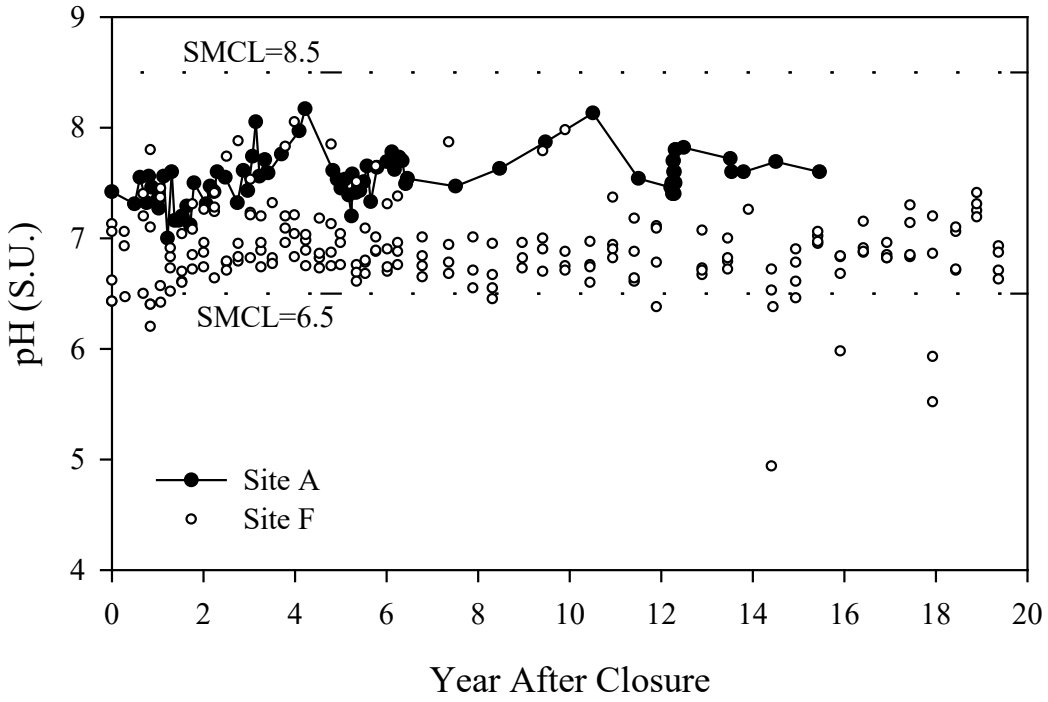


Figure 6-33. Temporal Variability in Leachate pH of Sites A and F Since Closure

6.5.3.7. Biochemical Oxygen Demand (BOD)

BOD is one of the key leachate quality indicator parameters that may be used to infer the extent of waste degradation. Morris and Barlaz (2011) reported that a statistical evaluation of BOD concentrations in leachate could be used as a primary measure of estimating leachate impact on HHE. Figure 6-34 shows the distribution of leachate BOD in seven of the study sites starting in the year of their closure. BOD data for Site F were not available, and very limited data were available for two other sites (i.e., Site H and I). BOD varied over a wide range from 3.7 to 4,620 mg/L, with a median of 76 mg/L. Townsend et al. (2015b) reported a median (of the mean for each site) BOD concentration of 84.5 mg/L based on data from 31 MSWLFs in Florida.

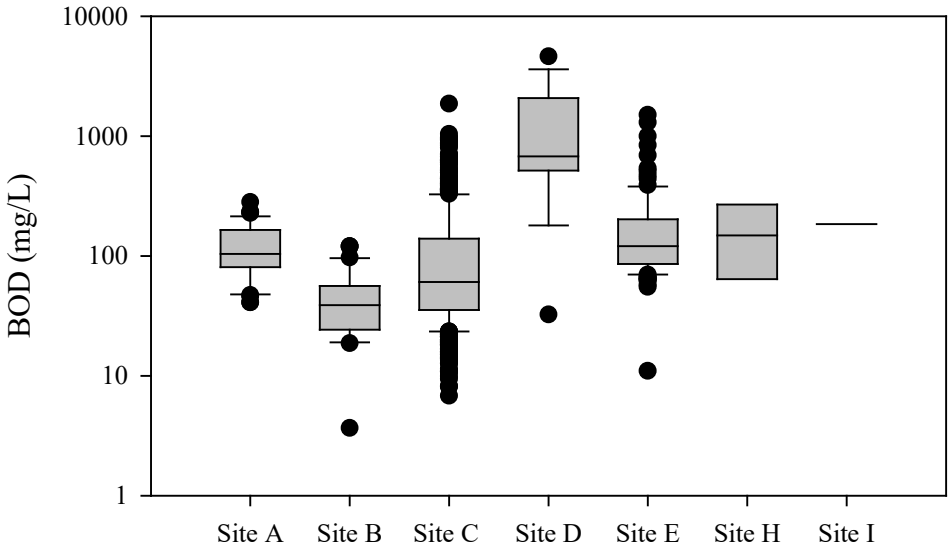


Figure 6-34. Distribution of Leachate BOD at Seven Study Sites Since Closure

In general, BOD appears to gradually decrease over time since closure for all the study sites where BOD data were available. Figure 6-35 shows the temporal variation in leachate BOD at Site B and Site C. A decrease in leachate BOD with landfill age is typical as the biodegradable carbon compounds contents such as cellulose and hemicellulose decrease in the waste. Statom et al. (2004) observed a similar decreasing trend of BOD in the leachate at a landfill located in Florida. It should be noted that biodegradable organics in leachate from fresher waste in the above layers would be consumed as it percolates through a well-decomposed carbon-limited waste layer above the LCS and thus attenuating leachate BOD (Kjeldsen et al. 2002). The leachate BOD, therefore, is typically reflective of the decomposition stage of the bottom-most waste layer and does not necessarily represent the degree of stabilization of the entire landfill.

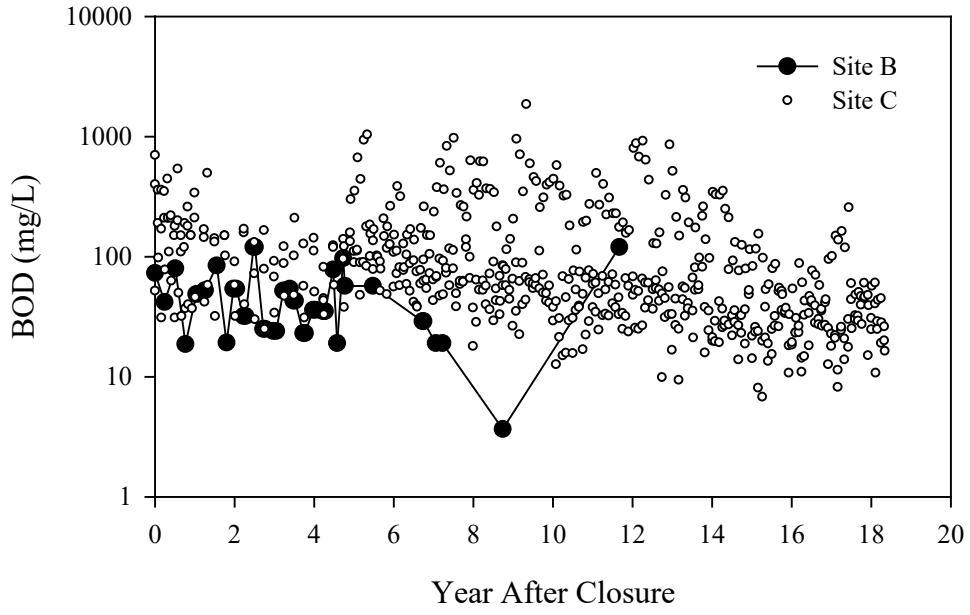


Figure 6-35. Temporal Variability in Leachate BOD of Sites B and C Since Closure

6.5.3.8. Chemical Oxygen Demand (COD)

Figure 6-36 shows the distribution of leachate COD at seven of the study sites since closure. COD data for one site were not available. COD ranged from 52 to 7,740 mg/L, with a median of 987 mg/L. Townsend et al. (2015b) reported a median (of the mean for each site) COD concentration of 907.7 mg/L based on leachate quality data collected from 26 MSWLFs in Florida. The ratio of the median values of BOD and COD after closure for all the study sites with the available BOD and COD data except for Sites D and H was less than 0.1, which is a threshold that is commonly used as an indicator of stabilized leachate. The ratio of median BOD to COD ratio for Sites D and H after closure were approximately 0.28 and 0.12, respectively.

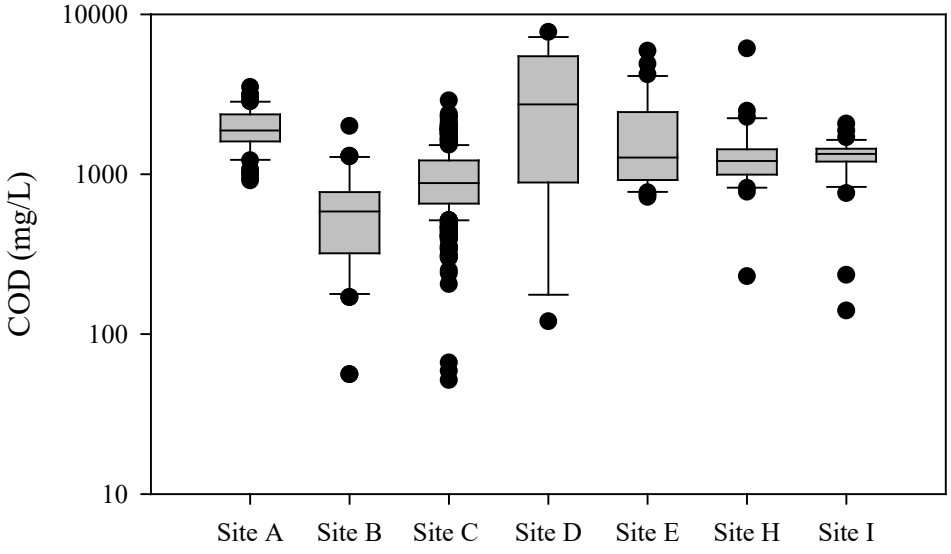


Figure 6-36. Distribution of Leachate COD at Seven Study Sites Since Closure

In general, leachate COD appears to be slightly decreasing after closure at all of the sites. Similar to BOD, the COD data beyond ten years after closure were only available for two sites (i.e., Sites B and C). Figure 6-37 shows the temporal variation in leachate COD at Site B and Site C. Site C leachate COD appears to be slowly but consistently decreasing and does not appear to have reached an asymptotic level in 18 years since closure. It should be noted that the leachate COD is typically reflective of the decomposition stage of the bottom-most waste layer and does not necessarily represent the degree of stabilization of the entire landfill. A well-decomposed waste layer above the LCS may attenuate the COD concentration as leachate percolates through this waste layer (Kjeldsen et al. 2002).

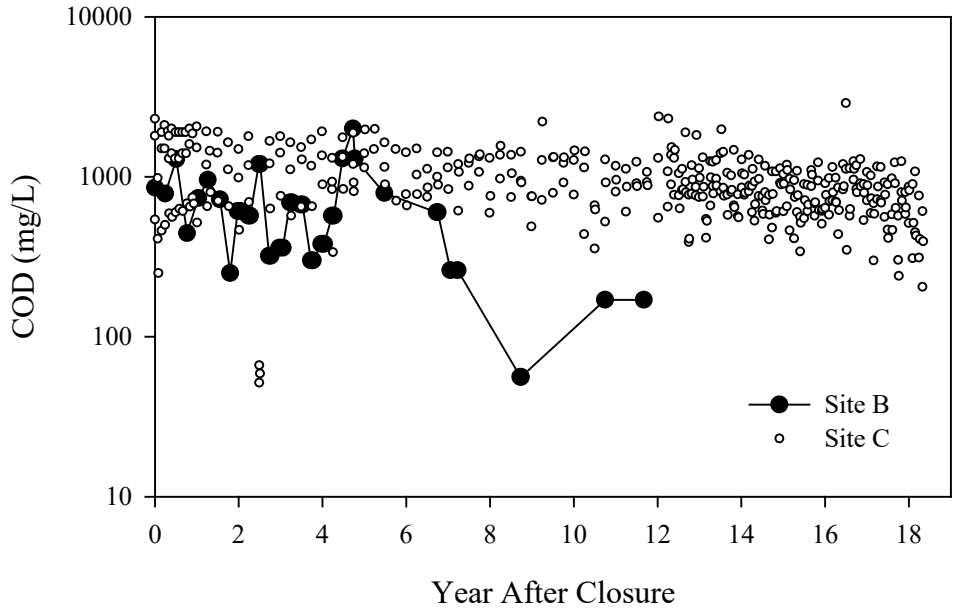


Figure 6-37. Temporal Variability in Leachate COD of Sites B and C Since Closure

6.5.3.9. Ammonia-Nitrogen

Although ammonia-nitrogen (hereafter referred to as 'ammonia-N') is not an App I or App II parameter and does not have any MCL or SMCL, it is one of the critical long-term pollutants from the HHE impact perspective(Kjeldsen et al. 2002). It has been reported as the primary cause of the acute toxicity of MSWLFs leachate (Kjeldsen et al. 2002). Figure 6-38 shows the distribution of leachate ammonia-N concentrations from five of the study sites after closure. Ammonia-N data were not available from the other study sites. Ammonia-N concentration ranged from 2.1 to 3,914 mg/L, with a median of 470 mg/L. As a point of comparison, Townsend et al. (2015b) observed a median (of the mean for each site) leachate ammonia-N concentration of 360 mg/L based on leachate quality data collected from 57 MSWLFs in Florida, which was over 120 times the risk-based standard for ammonia (2.8 mg/L) in Florida at the time of the study conducted by Townsend et al. (2015b). The DAF needed for the median ammonia-N concentration of 470 mg/L for the study sites to decline below this risk-based standard would be 168, which is the highest DAF among all the parameters evaluated in this study.

Site A ammonia concentration appears to be substantially greater than the other sites. Considerably higher ammonia concentration at Site A is potentially attributed to a higher degree of waste decomposition at Site A and progressive accumulation of ammonia in leachate due to its continual

recirculation into the landfill. Ammonia-N released from waste decomposition (such as proteins) does not degrade in the anaerobic environment of a landfill and is released with leachate.

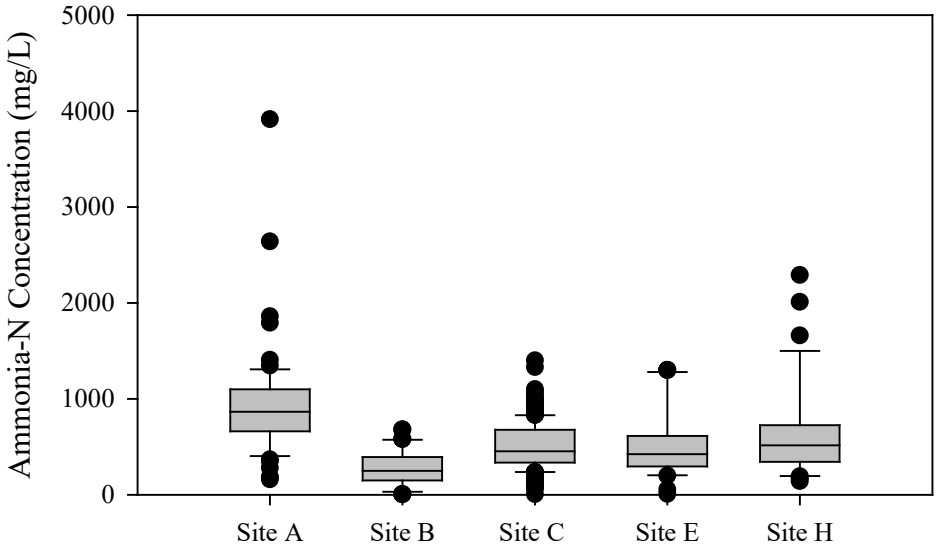


Figure 6-38. Distribution of Leachate Ammonia-N Concentration at Five Study Sites Since Closure

In general, the leachate ammonia-N concentration trends appear to be stable to slightly decreasing since closure. As an example, the temporal variation in leachate ammonia-N concentrations at Site C and Site H are shown in Figure 6-39. Statom et al. (2004) observed a similar stable to a slightly declining trend in leachate ammonia-N concentration in a landfill located in Florida. Based on an evaluation of data from 50 landfills in Germany, Krumpelbeck and Ehrig (1999) reported no significant decline in ammonia concentration even 30 years after closure (Kjeldsen et al. 2002).

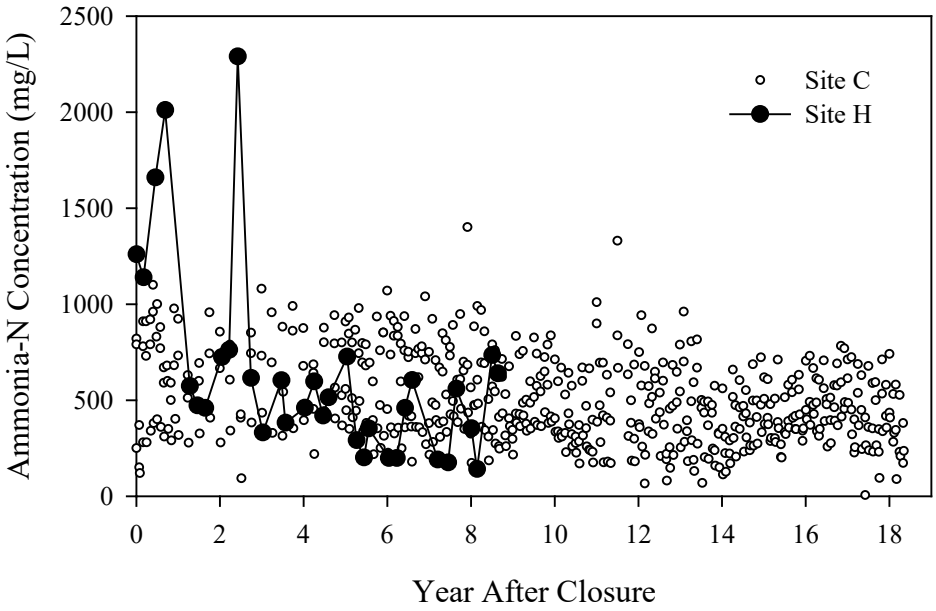


Figure 6-39. Temporal Variability of Leachate Ammonia-N Concentration of Sites C and H Since Closure

6.6. Impacts of Leachate Collection System Failure and Subsequent Leachate Leakage

Evaluation of primary liner leachate collection efficiency was performed and described in Section 6.4.3 using leachate quantity data from two of the nine sites. This evaluation suggests that only a small fraction of the leachate that collects over the primary liner system leaks through the liner. The sites/cells analyzed in this study have been closed for 8-19 years (as of 2016) and are expected to still be within the service life of the liner. However, the integrity of the liner and its effectiveness in intercepting leachate beyond its service life is a concern from a long-term HHE impact perspective. In addition, deterioration in the performance of the LCS due to factors such as drainage layer clogging would impact the ability to efficiently pump leachate out from the collection system. These issues would result in an increase in leakage from defects in the liner due to increased head on the liner. Additional liner deterioration and defects from liner aging may further impact groundwater quality and subsequently may pose a risk to HHE. Leachate leakage rate through the bottom liner defects can be estimated using the HELP model or published analytical and empirical mathematical equations (e.g., Giroud et al. 1997).

The seepage of leachate through the final cover defects on landfill side slopes in the event of leachate built-up in the landfill after LCS termination may subsequently impact surface water quality. The impacts of leachate releases through all potential pathways should be evaluated to assess the long-term impact of the termination of leachate collection and liner failure.

Leachate leakage is not expected to pose a risk to HHE if leachate analyte concentrations are below the respective risk-based health and ecological protection standards developed for the media of interest, such as groundwater or surface water (e.g., EREF 2006). However, it should be noted that leachate contains elevated concentrations of several contaminants (e.g., ammonia) that do not have a federal or state risk-based standard. Contaminant fate and transport modeling can be conducted to assess the impact of contaminant transformation and dilution on concentrations at the point of

compliance and receptor wells, which may be located well away from site groundwater monitoring wells. Additional information on leachate quality, as observed from leachate samples analyzed from the eight study sites evaluated in this study, is discussed in Section 6.4. Approaches for evaluating the risk associated with leachate leakage are discussed in more detail in Section 6.7.

6.7. Considerations for Assessing and Mitigating Leachate Impacts on HHE

As described by EREF (2006) and ITRC (2006), there are three general approaches that may be successively implemented to demonstrate that terminating LCS operations or reducing the frequency of leachate monitoring/management would not have HHE impacts. The first and most conservative approach is to show that the concentrations of all regulated leachate analytes are below regulatory standards (i.e., state or federal) for groundwater and surface water. The second is to demonstrate that groundwater or surface water concentrations cannot be reasonably expected to exceed the regulatory standards at permitted points of compliance (e.g., groundwater monitoring wells). The third is to demonstrate the concentrations would not exceed any regulatory or health-based standard at the nearest point(s) of exposure.

As described previously, out of the 68 parameters with an MCL/SMCL that were measured at the study sites, 15 parameters were detected in more than 50% of the samples. Of these 15 parameters, ten were found to exceed their regulatory limit in 10% or more of the samples, and only three of these ten parameters represent primary drinking water standards: arsenic, turbidity, and fluoride. While it should be noted that there were a number of regulated and non-regulated contaminants that were not analyzed, an analysis and comparison of historic leachate quality to regulatory standards provides a valuable first step for screening the contaminants that represent a risk to HHE.

Typical points of compliance for MSWLFs (that may be impacted by damage or imperfections in the bottom liner or final cover system) are groundwater monitoring wells downgradient of the landfill and stormwater outfalls (typically located at/near the site property boundary). For those leachate analytes with concentrations identified above the regulatory limit(s) during the initial step, site owners/operators may proceed to the second step and choose to demonstrate that these contaminants have never exceeded applicable standards at the site's permitted points of compliance. If exceedances have been observed, the temporal trends for these parameters should be analyzed to assess whether the concentration has consistently been decreasing to the point below and can be reasonably expected to remain below the regulatory protection standard. Examples of plots showing the temporal trend of arsenic and lead concentrations at groundwater monitoring wells are presented in Chapter 7.

However, if temporal trends in groundwater/stormwater contaminant concentrations cannot be used to demonstrate the absence of risk to HHE at the point of compliance, site owners/operators may proceed to the third step and conduct the contaminant fate and transport modeling to assess whether these would exceed the regulatory or health risk-based standards at the nearest point(s) of exposure. The EPA Industrial Waste Management Evaluation Model (IWEM) is one screening-level tool commonly used to assess contaminant fate and transport in groundwater. As described by EPA (2017b), for version 3.1 of the tool, the user inputs site-specific parameters, climate, and hydrogeological conditions to estimate the concentration of contaminants at specific downgradient groundwater monitoring wells. Specific states may have their own recommended/required modeling software programs. For example, the FDEP provides guidance and information on the

selection of risk-based corrective action fate and transport models in Florida Administrative Code (FAC) Chapter 62-780.100.

The analysis presented in this report included only the parameters that are required to be monitored for groundwater at MSWLFs and have a federal MCL/SMCL. However, leachate may contain constituents that are currently not required to be monitored for groundwater or for which regulatory standards do not currently exist. These parameters may have the HHE impacts. Some examples of these parameters include (but are not limited to) ammonia, pharmaceuticals, plasticizers, and certain types of pesticides and flame retardants (Moody and Townsend 2017; Masoner et al. 2016; Andrews et al. 2012; Musson and Townsend 2009; Barnes et al. 2004). A comprehensive risk assessment should include an evaluation of these parameters as well.

Finally, the low contaminant concentrations observed during operating and post-closure phases may not necessarily be indicative of stabilized conditions but may be due to partial stabilization/mineralization because of the lack of exposure of landfilled waste to adequate moisture. The considerations discussed above do not address the impacts of a potential increase in the emission of contaminants with leachate after the LCS operation termination. Moisture intrusion into the landfill due to compromises in the integrity of the final cover after PCC termination may trigger decomposition of biodegradable waste, if any, and result in a release of contaminants with leachate that poses a risk to HHE. The impacts of the potential future moisture exposure on leachate emissions can be evaluated and mitigated by actively recirculating leachate or adding other liquids sources (e.g., groundwater) during operating and post-closure phases when the site is actively monitored by owners/operators and regulators. Adequate leachate/moisture volume should be added to expose the waste mass deposited in the landfill to the elevated moisture conditions.

6.8. Summary

6.8.1. Data Availability

Leachate collection rate and chemical quality data were available for all the study sites as the availability of these data was a key site selection criterion. Leachate collection rates were available for two distinct cells at two of the sites (C and G), thus along with the nine other single-celled study sites, leachate collection rates were analyzed for 11 total landfill cells. Of the 11 study cells, six are equipped with a double liner and LDS. Leachate quantity and quality data were available for most of the period after closure for all of the sites except for Site I; these data were available only for the first 12 of 19 years since closure for Site I. Two sites documented leachate recirculation after closure and the volumes of leachate recirculated at these sites after closure were available. LDS leachate quality data were available for two of the sites. The leachate quality data for Site G were not included in the analysis as these were not truly representative of the closed MSWLF.

6.8.2. Measured Leachate Collection Rate

The available data were added for each year for each site and divided by the number of days data were available and the area of the cell to calculate an annual average leachate collection rate per unit area of the cell in GPAD. Since the closure, the annual leachate collection rate across all 11 cells varied from 3–2,070 GPAD, with a median of 92 GPAD and an average of 190 GPAD; 90% of the annual collection rate measurements were less than 500 GPAD. Leachate generation continues at all of the sites. Except for Sites A and E, which recirculated leachate since the closure, the most-recent annual leachate collection rate for the landfills examined was below 100 GPAD,

with the median annual leachate collection rate varying between approximately 6.6 and 159.2 GPAD among the non-leachate recirculating sites. The leachate collection rates for all sites except for Sites A, C1, and F exhibited a general declining leachate production trend. Based on reported chloride concentrations and observed spikes in leachate collection during the rainy season, the recent spike in leachate collection rate at Site F appears to be a result of stormwater intrusion into the LCS infrastructure. None of the landfill cells examined exhibited trends indicative of reaching a steady state leachate collection/generation rate.

6.8.3. Leachate Collection Rate Modeling Approach Evaluation

Three approaches were used to estimate the future leachate collection rate from the study sites: first-order decay modeling, unsaturated flow modeling, and HELP modeling. In general, all three approaches provided comparable and reasonable fits to the measured study site data for all the study sites except Sites C1, D, and G3. The estimates based on the unsaturated flow model fit the measured data with an $r^2 > 0.5$ for seven of the ten study cells modeled. The first-order decay model also fit six of the ten study cells with an $r^2 > 0.5$. Sites C1, D, and G3 cells exhibited the lowest r^2 value for both models. Unlike the first-order decay and unsaturated flow model, the HELP model was not iteratively executed to obtain the best-fit to the measured leachate collection rate data but was based on a mix of default parameters and the best-fit results from the unsaturated flow model.

6.8.4. Hydraulic Efficiency of Primary Liner

Six of the study cells are lined with a double bottom liner system. The LDS collection rates at four of these sites ranged from 0.1 GPAD (for study cell G4) to approximately 123 GPAD (for study cell G3); LDS data were not available for the study cells C1 and C2. Over 50% and 85% of the values (across these sites) were less than 1.5 GPAD and 8 GPAD, respectively. The LDS rates, in general, show an overall declining trend over time after closure. The aggregate hydraulic efficiency of the primary liner was calculated by dividing the sum of annual LDS collection rates by the sum of the corresponding annual LCS and LDS rates for all years with available LCS and LDS collection rates. The primary liner efficiency was calculated to be 97.2%, 99.6%, 99.6%, and 96.8% for study Sites G3, G4, H, and I, respectively. A comparison of LCS and LDS leachate quality data available for two of the sites suggests groundwater intrusion into the LDS might be a significant source of liquids collected from LDS. Therefore, the primary liner efficiency at these sites is expected to be higher than the estimates presented in Section 6.4.3.

6.8.5. Available Leachate Quality Data

The comprehensiveness of the available leachate quality data for eight study sites was evaluated with respect to the parameters specified in the federal regulations for groundwater monitoring at Subtitle D landfills (App I and App II of §258). A total of 272 leachate constituents were monitored at least once among the study sites (excluding G). The number of chemical constituents that must be monitored for groundwater as part of App I and II of §258 are 62 and 215, respectively; all of the App I parameters are also included in the App II list. The parameters monitored varied widely among the sites. Only three sites reported leachate constituent data for every App I parameter, and one of these sites reported at least one measurement for all but three of the App II parameters. Two study sites had leachate characteristic data available for less than half of App I parameters. More than half of the study sites reported leachate constituent concentration data for ten or less App II parameters (excluding App I parameters). The characterization frequency varied among the sites from once per month to twice per year, and further varied with time and contaminant. Only a single

measurement was available for a few of the organic compounds for each site after closure. Apart from the lack of data for a large number of App I and II parameters, the small number of measurements available for some of the constituents measured at the study sites may limit a reliable HHE impact assessment.

6.8.6. Contaminants of Potential Concern

A screening analysis was conducted to identify the contaminants that have been frequently measured in leachate above the respective risk-based protection standards at the study sites after closure to identify the COPCs. Parameters that were never measured above their MCL or SMCL are not expected to present a risk to HHE. Drinking water standards were used as the risk thresholds for this evaluation.

Of the 272 leachate quality parameters selected for evaluation in this study, only 100 parameters have MCL or SMCL, and among those 100 parameters, data were not available for 32 parameters (as listed in Table J-3 of Attachment J) for any of the study sites. Among all 100 of the parameters with an MCL/SMCL, a total of 56 parameters exceeded their respective MCL/SMCL at least once. However, 11 parameters (all organic compounds) reported as below detection, but the reported detection limits were greater than the MCL/SMCL; these parameters cannot be conclusively determined to be below MCL/SMCL. MDL was greater than the respective MCL for a majority of organic compounds for which data were available for the study sites. This data quality issue limits the HHE impacts assessment for these parameters.

Of the remaining 44 parameters, 29 parameters were measured above their MDL (i.e., detected) in less than 50% samples. The remaining 15 parameters were measured at concentrations above their MDL in more than 50% of the samples. Of these 15 parameters, six parameters (i.e., arsenic, TDS, iron, manganese, chloride, and color) were measured above their MCL/SMCL in more than 94% of the samples. Among all the constituents with MCLs that were measured at the study sites, arsenic and turbidity were the only primary MCL parameters that consistently exceeded the MCL. Among the secondary parameters, greatest dilution and attenuation would be needed for iron for its concentration to decline below its SMCL of 0.3 mg/L. A majority of the iron measurements were below the regional screening level developed by EPA for tapwater for the ingestion pathway for iron (14 mg/L).

Relatively low exceedance frequency of a large number of contaminants above the respective MCL/SMCL may be due to incomplete flushing out of contaminants associated with limited waste stabilization because of the lack of exposure of landfilled waste to adequate moisture. Moisture intrusion into the landfill due to compromises in the integrity of the final cover after PCC termination may trigger further decomposition of biodegradable waste, if any, and result in a release of contaminants with leachate at levels that pose a risk to HHE. The impacts of the potential future moisture intrusion on leachate emissions can be evaluated and mitigated to some extent by actively recirculating leachate or adding other liquids sources (e.g., groundwater) during operation and after closure while the site is actively monitored by owners/operators and regulators.

6.8.7. Temporal Trends of the Contaminants of Potential Concern

The temporal trend for most of the COPCs varied among the study sites. Arsenic showed a declining trend for Sites A, B, C2, and E. Four of the most recent arsenic measurements at Site B were below the MCL. The most recent set of measurements at Sites A, C2, and E were above the MCL. Arsenic concentrations exhibited an increasing trend for Sites C1, D, and H. The arsenic

mass release rate from Site H exhibited a slightly declining trend over time. TDS and ammonia at the sites, in general, appeared to be stable to slightly decreasing with time. In general, BOD appears to gradually decrease since closure for all the study sites for which BOD data were available. Leachate COD appears to be slightly decreasing after closure at all of the sites.

6.9. Limitations

The analysis presented in this chapter has the following limitations:

1. The primary liner performance evaluation analysis assumes that the leachate leakage through the secondary liner is insignificant. The primary liner efficiency would be lower than the estimates presented if the leakage through the secondary liner is significant.
2. Only limited leachate composition data were available for a few of the study sites. For example, only five arsenic measurements collected during the 3-4 years after closure were available for Site I. The contaminant concentration can vary substantially over time.
3. The MDL for a large number of organic compounds monitored for the study sites was greater than the respective protection standard. These parameters cannot be conclusively determined to be below MCL/SMCL.
4. The screening analysis conducted to identify the COPCs was based on the assumption that groundwater was the water source affected by a leachate discharge after the termination of LCS operation. Impacts on surface water may be a more probable pathway for some sites. The use of surface water-specific risk-based thresholds would have resulted in a different set of the COPCs than presented in this chapter.
5. An analysis of the historical leachate collection/emission rates and the leachate characterization data are presented in this chapter. These estimates can be used for an assessment of the HHE impacts. However, an HHE impact assessment was not an objective of the study. Modeling approaches such as life cycle assessment and contaminant fate and transport modeling coupled with risk assessment can be used to estimate the HHE impacts of leachate emissions associated with the termination of LCS operation at the point of compliance or at the point of exposure.
6. The screening analysis conducted to identify the COPCs was based on MCLs/SMCLs. More than 60% of the parameters that were measured at least once do not have an MCL/SMCL. Some of these parameters may be of HHE concern. As mentioned above, the analysis only included groundwater contamination pathway. The surface water contamination pathway was not evaluated.
7. It should be noted that the literature-reported leachate collection rate and composition data and/or values observed at other sites cannot be used to reliably assess site-specific impacts of terminating LCS operation on HHE due to the large variation reported in the literature. Therefore, the data from the nine study sites should not be used as a proxy for conducting a reliable site-specific impact assessment. The data analysis approaches presented in this chapter can be used to estimate the site-specific leachate collection rates, which can be used as inputs for a more reliable assessment of the HHE impacts.
8. Due to the small number of sites analyzed in this study, statistical evaluation was not performed to estimate the representative leachate collection rates for different precipitation zones of the US. Several states require mandatory routine reporting of leachate collection rates and composition data. Future research should consider using these data for statistical evaluation to assess the variation of leachate collection rates and composition with weather conditions (e.g., precipitation).

7. Groundwater Monitoring Data

7.1. Overview

Groundwater monitoring data serves as one of the key criteria for assessing the MSWLF containment system performance and the impacts on HHE. RCRA Subtitle D regulations require the monitoring of groundwater quality per §258(e) for MSWLFs, which describes the requirements for groundwater monitoring and corrective action, as one of the PCC requirements for MSWLF sites. Groundwater monitoring systems for MSWLFs are typically designed to monitor groundwater of the uppermost aquifer. The monitoring system consists of a network of background (or upgradient) and downgradient wells. The background well(s) are located to monitor groundwater quality that has not yet been, nor expected to be impacted by the landfill, while downgradient wells are installed at locations to detect groundwater impacts from the landfill. Groundwater monitoring events are required to be conducted on a semi-annual basis throughout the operating and PCC period unless an alternative frequency is approved by the appropriate regulatory agency.

Groundwater monitoring is conducted in two phases: detection monitoring and assessment monitoring. As discussed in Chapter 2, detection monitoring includes the routine sampling of 62 parameters (App I) (though 15 inorganic parameters in this list may be modified by States with an approved program). As required by §258.54(c), assessment monitoring is initiated if a statistically significant increase over background concentrations is identified for one or more of the detection monitoring parameters in any downgradient monitoring well. To assess whether a statistically significant increase has occurred, §258.53(g) requires the selection of a statistical method to analyze the concentrations of the various groundwater monitoring parameters. Additional guidance on the use of these statistical methods is provided in EPA (2009).

Assessment monitoring includes the monitoring of 215 parameters (included as App II of 40 CFR §258); over 90% of these parameters are organic compounds. All of the App I parameters are included in the App II list. The owner may be required to implement corrective measures (e.g., groundwater remediation) if one or more of App II parameters are detected at statistically significant levels above the respective groundwater protection standard. The MCL is the groundwater protection standard for the parameters for which an MCL has been promulgated. For constituents for which the background concentration is higher than the MCL, the background concentration is used as the protection standard §258.55(h). The background concentration or a state-specified human health risk-based standard should be used as the groundwater protection standard for the parameters without an MCL. The first objective of the analysis presented in this chapter is to identify parameters that were detected above the respective MCL.

Objective 1. Analyze groundwater quality data available for the study cells to identify the parameters that were detected above the respective MCL.

Contaminant concentrations and the associated HHE impacts are expected to vary over time. The screening analysis described above for identifying the contaminants that were detected above the respective MCL at the study sites does not account for temporal variation of concentrations. The contaminants that have been frequently measured above the respective MCL initially but have declined below the respective MCL over time are not expected to be an HHE concern. Some state guidance documents recommend a demonstration that the contaminants have not been detected above the respective MCL or state standard for the past several years in order to reduce the

groundwater monitoring requirements. The second objective of this chapter is to evaluate temporal trends of contaminant concentrations that have been frequently detected above the respective MCL to assess whether the concentration of any of these contaminants has declined below the respective MCL over time after closure.

Objective 2. Assess whether the concentrations of the contaminants that were frequently detected above the respective MCL have declined below the MCL over time after closure.

It should be noted that only 51 out of 215 App II parameters (including lead and copper) have an MCL. The groundwater protection standard for the parameters without an MCL is the background concentration established based on background well(s). The groundwater protection standards for these parameters were not available for the study sites. Some of these parameters may have a state-specific risk-based standard, which may be used to assess the HHE impacts of these parameters. The third objective of the analysis presented in this chapter is to present examples of parameters that were detected above the respective state-specific risk-based standard.

Objective 3. Analyze groundwater quality data available for the study cells to identify examples of parameters that do not have an MCL and that were detected above the respective state-specific risk-based standard/criteria.

This chapter also presents a discussion about the impact of biogeochemical changes in the aquifer system, LCS failure, and subsequent leachate release on groundwater quality.

7.2. Data Sources

Groundwater quality data are valuable in assessing both current and long-term groundwater impacts at the site. The groundwater monitoring data from the nine study sites were analyzed to assess the nature and frequency of groundwater issues observed after closure; only the data collected after closure were analyzed in this report. As discussed in the individual site descriptions (Attachments A-I), groundwater monitoring reports (including statistical analysis of groundwater monitoring data) were not available from all nine sites–data from several sites were only available in an unprocessed form (e.g., were downloaded from a tabulated state database, were received as a spreadsheet from the site owner/operator).

Of the nine sites reviewed, groundwater/environmental monitoring reports were available for six of the sites. Of these six sites, the most recent groundwater monitoring report suggests that three of the sites appear to be in detection monitoring, and two are in assessment monitoring (or the state-equivalent monitoring phase), and one is under the corrective action phase. The presence of an unlined or a lined cell not constructed under §258 regulations near the study cell(s) at two of the three sites under assessment monitoring and corrective action phase complicates a reliable assessment of the groundwater impacts of the study cells. The following data and information were compiled and analyzed for each of the study sites:

- Number of upgradient and downgradient groundwater monitoring wells
- Groundwater flow direction
- The date range of available groundwater monitoring data
- The available groundwater quality data. Only concentration data for parameters that were historically detected in groundwater samples were available for some sites.
- Parameters observed to have exceedances of the respective MCLs after the closure

- Examples of parameters observed to have exceedances of the respective and state-specific risk-based standards after the closure
- Year(s) since closure when the exceedance(s) was observed

Due to data inconsistencies or unavailability at most of the sites, the following groundwater information was not reviewed and summarized as part of the analysis:

- Groundwater data analysis with respect to the respective state groundwater protection standards
- Groundwater flow gradients (change in total head over the associated horizontal distance)
- Groundwater well maintenance logs/procedures
- Adjacent or on-site land use activities that may be impacting groundwater quality or flow direction around the study site

Table 7-1 provides a summary of the key features for the study cells of all nine sites pertaining to the groundwater quality data analyzed in this chapter. The groundwater quality data were available for the entire duration since closure for all the study sites. The groundwater sampling frequency at the study sites is either semiannually or quarterly. The available data suggest that the laboratory analysis frequency appears to be contaminant specific for a majority of these sites. For example, some parameters appear to be analyzed on a quarterly basis, while a majority of the organics are analyzed once every five years at Site F.

Table 7-1. Summary of Groundwater-Related Features for Study Cells of Each Studied Site

Feature	A	B	C	D	E	F	G	H	I
Years since Closure (as of 2016)	18	12	18	7	12	19	23 (G3), 16 (G4)	8	19
Years of Available Groundwater Data	14	13	19	8	13	20	18	9	19
Piggybacked Over Closed, Non-Subtitle D Cell	✓	✓			✓				
Adjacent to Active Cell			✓				✓		
Consistent Groundwater Flow Direction		✓		✓	✓	✓	✓	✓	✓
# Upgradient Wells Analyzed	1[1]	4	4	2	11	4	9	10	3
# Downgradient Wells Analyzed	23	17	6[2]	16	49[2]	4	17	20	14
Typical Groundwater Monitoring Frequency (#/year)	2	4	4	2	2	4	4	4	4

[1] the Site had a variable groundwater flow direction.
[2] includes all the wells without a known location relative to the site.

It should be noted that a typical groundwater data evaluation process, as required by §258.53(g), involves the selection and implementation of a detailed statistical method to evaluate groundwater quality impacts. The analysis of groundwater monitoring results presented in this section and described below is not intended to meet or serve as a substitute to the requirements of §258.53(g) but was used for screening groundwater quality to identify parameters with an elevated potential to cause groundwater quality impacts.

7.3. Challenges to Isolating and Understanding Study Cell(s) Groundwater Impacts

Many of the study sites received waste before the promulgation of §258 regulations and contain cells that are not required to meet requirements of §258 (e.g., unlined or lined cells that were closed before the promulgation of these regulations). At these sites, the cells under the §258 regulations are commonly constructed immediately adjacent to or piggybacked over an old cell (i.e., a cell excluded from §258 regulations) for more efficient use of available space. In these cases, it becomes challenging to conclusively assess whether groundwater impacts (if existent) are associated with the cell(s) under §258 or cells that are not under §258 regulations.

Groundwater monitoring data were reviewed for all the monitoring wells around the study cells for those sites where the studied cell(s) was physically separated from any unlined cell (e.g., the studied cell was not piggybacked over an unlined cell) and/or could be analyzed independent of all other non-Subtitle D cells and active cells. Only five studied sites (Site C, D, F, H and I) had closed lined cell(s) that were completely isolated from other cells (i.e., active cells under §258 regulations as well as closed cells not under §258 regulations), appeared to have a consistent groundwater flow direction, and had groundwater wells installed at both upgradient and downgradient locations from the closed study cell(s) of interest. The data from all the monitoring wells were reviewed for the other four sites (Sites A, B, E, and G). The closed lined cell at Site A is located immediately adjacent to an unlined cell. Issues such as this complicate the evaluation of groundwater data for an individual cell that is contiguous to other closed MSWLF cells not constructed under §258 regulations. States (e.g., Florida) may allow the initiation of PCC for a closed cell or group of cells only if these cells are maintained and/or monitored separately from the rest of the landfill.

Change in groundwater flow direction is another factor that may complicate a reliable groundwater impact assessment. For example, while the groundwater flow direction for Site A was fairly consistent during the first portion of the PCC period, however, the groundwater reports suggested a highly variable groundwater flow direction in the recent years. Changes in land use/groundwater use around the site and long-term weather patterns (e.g., sequencing of dry and wet years) are some of the factors that may result in localized changes in surficial aquifer groundwater flow direction. MSWLF owners/operators attempting to make a demonstration in support of reducing the frequency of or terminating PCC should consider an evaluation of these changes around the site and the long-term impact of these changes on groundwater flow direction and quality.

7.4. Groundwater Impacts at the Studied Sites

In this study, existing impacts on groundwater were identified at the sites for three sets of parameters: parameters with MCLs, parameters with SMCLs, and example of parameters without an MCL or SMCL. First, the data were analyzed only for the App I and II parameters with an MCL. An MCL is a federal human-health risk-based standard. All the parameters that are measured at concentrations above the respective MCLs at least once in a groundwater monitoring well after closure were identified.

The second set of parameters that were evaluated are silver and zinc exceedances (i.e., with respect to the SMCL); silver and zinc are the only parameters included in the App I and App II lists that have an SMCL. SMCLs are based on aesthetics (e.g., color, taste), cosmetics (e.g., skin

discoloration), and technical considerations (e.g., impact to water treatment process) and are not based on human-health risk considerations.

A large number of App I and II parameters do not have an MCL/SMCL; only 32 and 51 of App I and App II parameters, respectively, have an MCL, and only two parameters (i.e., silver and zinc) have an SMCL. The groundwater impact with respect to these parameters without any MCL/SMCL can be assessed by comparing the concentrations measured at downgradient wells to those of upgradient well(s) or to a state standard. An analysis of a few parameters at a couple of sites is presented to highlight the importance of analyzing the parameters without MCL/SMCL for assessing the groundwater impacts.

Table 7-2 summarizes the MCL exceedances identified at each of the sites. No exceedances were identified at Site F for the range of data available for review (i.e., post-closure Year 8 through Year 19). The exceedance counts presented in the table do not include groundwater monitoring data for which the MDL was above the respective MCL. Several instances were found where the parameter analysis detection limit was greater than the respective MCL. For example, antimony, beryllium, thallium, and arsenic detection limits were higher than the respective MCL on several sampling occasions at Site C. Several organics measured at Site F had detection limits greater than the respective MCL.

As shown in the table, only occasional MCL exceedances were identified following cell closure among the studied sites. A majority of the organic compounds measured at the study sites were not detected in groundwater samples. Over 65% of the observed vinyl chloride exceedances at Site E occurred at a well close to the unlined cell at the Site. The arsenic exceedances observed at Site B are potentially associated with the unlined cells as these occurred at the downgradient wells that are closer to the unlined cell than the study cell. Only Sites G, H, and I appear to have had reoccurring groundwater exceedances of one or two constituents. More than 50% and 10% of measurements at Site H had beryllium and thallium concentration above its respective MCL. However, approximately 40% and 30% of the observed beryllium and thallium exceedances occurred at nine and five upgradient wells, respectively. Due to the exceedances observed above the respective MCL at several upgradient wells, the exceedances at the downgradient wells cannot be conclusively attributed to lined cells at Site H.

Although 107 arsenic exceedances at Site G is the third-largest exceedance count among all the study sites, these exceedances represent less than 15% of the arsenic measurements at the site; arsenic was not detected in more than 80% of the groundwater samples collected at the site. Approximately 11% and 52% of arsenic exceedances observed at Site G occurred at four upgradient and five wells downgradient to the unlined cell, respectively, at the site. The number of arsenic exceedances at the other wells at Site G ranged from two to eight, after the closure of the study cell. It should be noted that the site had an active cell at the time of the study.

All the parameters, except for mercury, listed in Table 7-2 for Site G, also exceeded the respective MCL at the upgradient well(s). Approximately 13% and 38% of the lead exceedances occurred at upgradient wells and wells downgradient/close to an unlined cell, respectively, at the site. Less than 20% of the lead exceedances at Site G have occurred in the last ten years. Due to the exceedances observed above the respective MCL at several upgradient wells, the exceedances at the downgradient wells cannot be conclusively attributed to unlined and/or lined cells at Site G. Only one arsenic measurement for a single well was available for Site E after closure.

Table 7-2. Site Summary of App I and II Parameters with Exceedances Above the Respective Maximum Contaminant Level After Closure

Site	Parameter	Total number of wells data were available	Total Number of Measurements	Number of Detected Measurements	Number of Exceedances	Number of Upgradient Wells with Exceedances	Number of Downgradient/ Side-gradient Wells with Exceedance	Post-Closure Duration Data Available (Years)	Year (after Closure) of Exceedance(s)
A	Arsenic	23	214	61	16	0	9	13	0,1,3,13
	Cadmium	23	177	9	1	0	1	9	1
	Chromium	23	229	37	1	0	1	14	11
	Lead	23	178	29	3	0	2	10	0-2
	Thallium	23	226	78	2	0	2	10	12,14
	Bis(2-Ethylhexyl) Phthalate	1	1	1	1	0	1	1	0
	Vinyl Chloride	23	207	10	5	1	1	14	0,1,3,8
B	Arsenic	21	911	597	18	0	2	7	0-1, 7-11
	Vinyl chloride	21	1042	183	1	0	1	8	1
C	Antimony	9	149	1	1	0	1	19	18
	Arsenic	9	149	5	1	1	0	19	3
	Beryllium	9	149	15	3	0	1	19	14, 15, 18
	Cadmium	9	149	7	1	0	1	19	11
	Thallium	9	149	1	1	0	1	19	15
D	Beryllium	18	288	118	2	2	0	8	0
	Chromium	18	288	284	2	0	2	8	1, 3
	Lead	18	288	183	7	1	3	8	0-2, 4, 5, 7
	Thallium	18	288	115	5	0	5	8	0
E	Arsenic	1	1	1	1			1	10
	Tetrachloroethylene	45	482	8	1	0	1	12	0
	Vinyl chloride	45	482	39	15	0	3	12	1-11
F		8						12	
G	Arsenic	24	799	128	107	3	18	17	5-13, 16-22
	Cadmium	24	990	22	6	1	3	18	6, 7, 11, 12, 19
	Lead	24	975	320	53	3	12	18	5-9, 12-14, 16, 17, 19-22
	Selenium	22	234	32	7	1	6	16	6, 7
	Antimony	22	239	13	13	3	10	16	7, 11, 19

Table 7-2 (contd.). Site Summary of App I and II Parameters with Exceedances Above the Respective Maximum Contaminant Level After Closure

Site	Parameter	Total number of wells data were available	Total Number of Measurements	Number of Detected Measurements	Number of Exceedances	Number of Upgradient Wells with Exceedances	Number of Downgradient/ Side-gradient Wells with Exceedance	Post-Closure Duration Data Available (Years)	Year (after Closure) of Exceedance(s)
G	Thallium	22	235	21	19	3	7	16	7, 11-13
	Mercury	22	241	4	2	0	2	16	9, 16
H	Arsenic	30	255	141	5	0	1	9	3-5, 7-8
	Beryllium	30	255	232	149	9	14	9	0-8
	Cadmium	30	255	129	1	0	1	9	2
	Chromium	30	255	103	1	1	0	9	8
	Copper	30	256	97	1	0	1	9	7
	Lead	30	256	81	12	0	3	9	0-7
	Thallium	30	255	27	27	5	13	9	1,2,4,5,8
I	Arsenic	16	297	114	114	0	7	19	0-18
	Vinyl Chloride	14	14[1]	1	1	0	1	1	0
	Methylene Chloride	15	134	14	14	0	1	8	0-7

Constituents in the shaded rows were above MCL in upgradient well(s)

[1] Data points only for the sampling events with at least one detected measurement.

As shown in Table 7-2, elevated arsenic levels have been observed at seven different downgradient groundwater monitoring wells located at Site I. Arsenic exceeded the MCL only once for two of these wells after closure. The arsenic concentrations at the rest of the five wells were regularly observed above the MCL (i.e., 0.010 mg/L), and one of these five wells has not been monitored for the most recent five years. A temporal plot of arsenic concentrations at downgradient wells for the remaining four wells is presented in Figure 7-1; the laboratory reporting limit of 0.010 mg/L was used for the values below the reporting limit. Two data points that appeared to be outliers (one point from Well A and one from Well C) were not included in the plot for clarity; these concentrations were approximately an order of magnitude above the concentrations measured during the sampling events immediately prior to and after the outlier measurement.

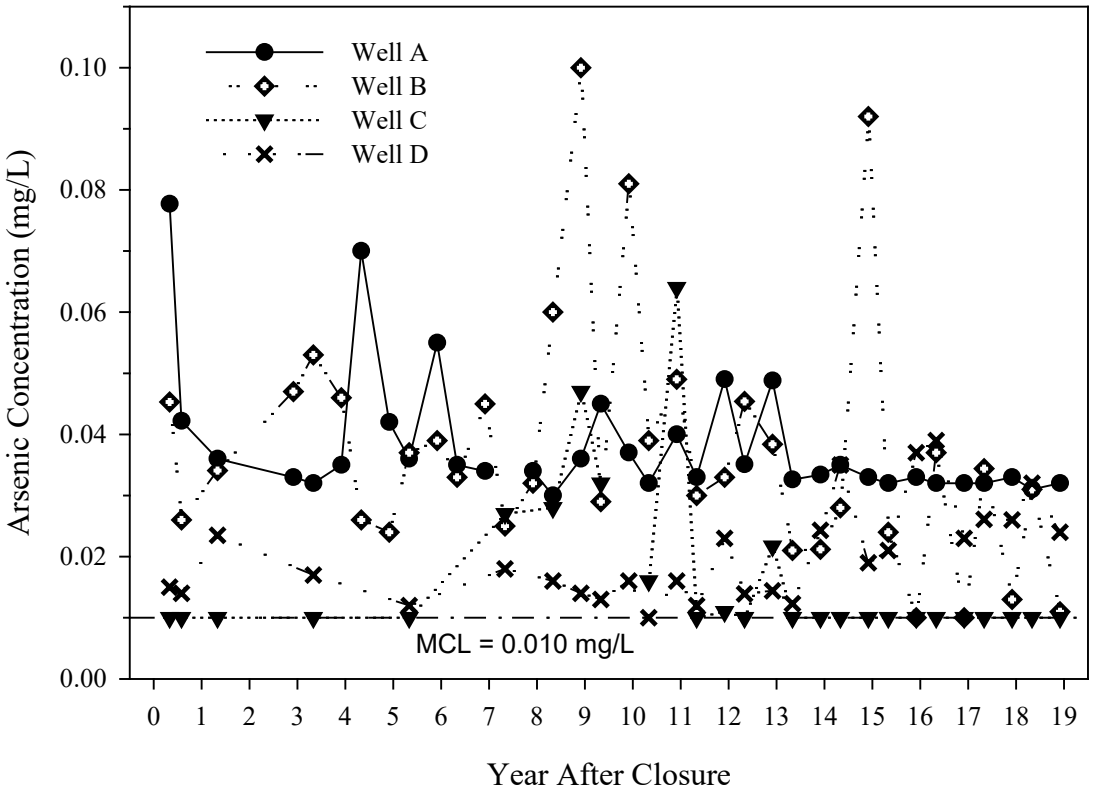


Figure 7-1. Temporal Variation of Arsenic Concentrations Observed Above the MCL at Four Downgradient Wells at Site I After Closure

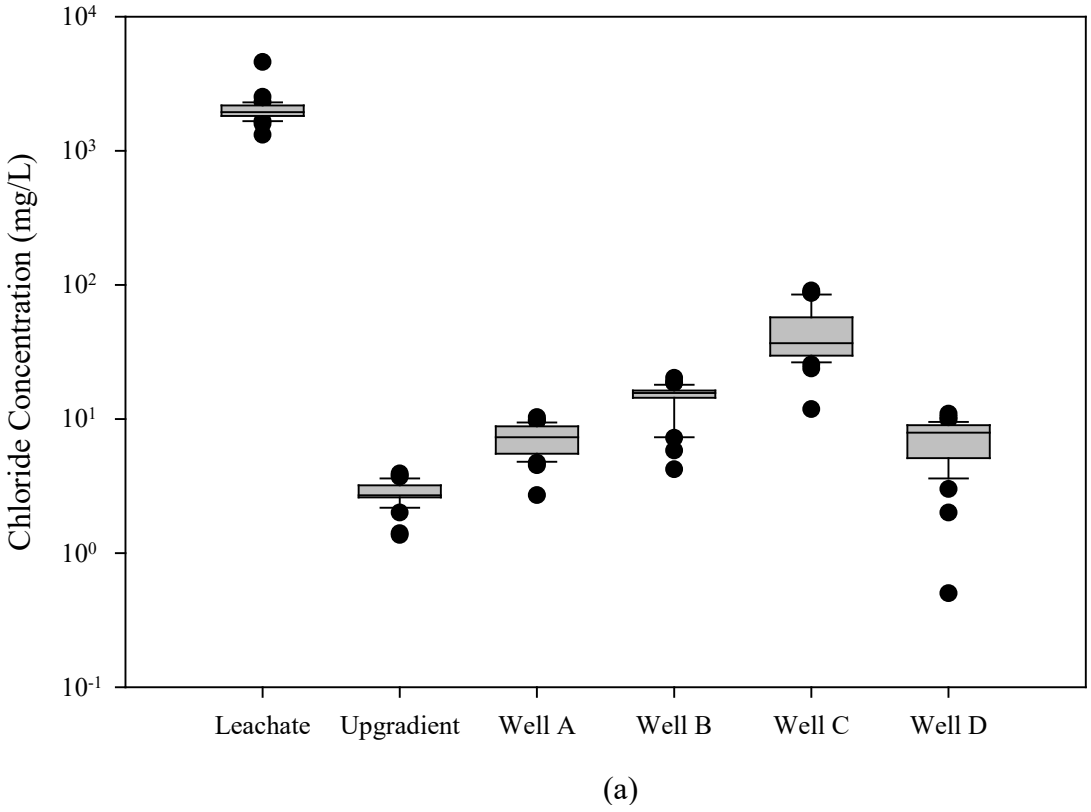
As shown in Figure 7-1, the arsenic concentrations observed at Well C have been consistently below the reporting limit/MCL since year 13 after closure. Arsenic concentrations have been consistently above the MCL at Wells A, B, and D. Arsenic concentrations exhibit an increasing trend at Well D. Arsenic concentrations observed at Well B vary over a relatively wide range compared to the other wells while the arsenic concentrations observed in Well A have been relatively consistent since year 13. In summary, the variable arsenic concentration trends at these wells (i.e., variable at one location, increasing and consistent at the other locations) suggest a need for continued groundwater monitoring at the site to assess long-term impacts on groundwater quality.

A comparison of the leachate indicator parameter (e.g., total dissolved solids, chloride, and ammonia) concentration in downgradient wells with the respective background concentration can be used for assessing the magnitude of the groundwater impacts associated with leachate release. Figure 7-2 presents chloride and ammonia distribution in leachate, an upgradient well, and four downgradients wells that exhibited arsenic concentrations above MCL after closure and were actively monitored at the time of the study. As expected, the groundwater chloride and ammonia concentrations are 2-3 orders of magnitude lower than the respective concentration in leachate. The chloride concentration in downgradient wells was much greater than that for the upgradient well suggesting potential impacts from the study cell. The chloride and ammonia concentrations in Well C were greatest among all the downgradient wells. However, as shown in Figure 7-1, arsenic concentration observed at Well C was lowest among the downgradient wells suggesting leachate release, if any, does not appear to be the primary cause of arsenic exceedances at this well. The site monitoring reports documented that these elevated levels may be a result of subsurface geochemical changes resulting from the development of the site, which included the elimination of groundwater recharge (i.e., not due to leachate release into groundwater).

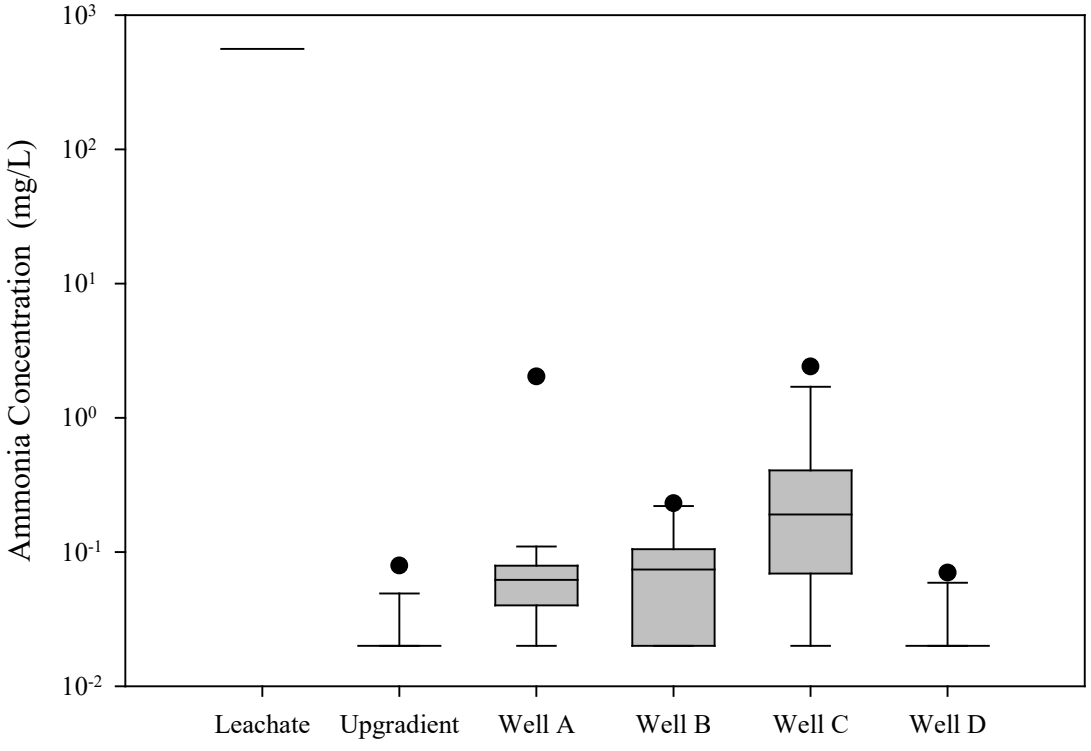
The availability of chloride and/or ammonia data for groundwater is requisite for this analysis. It should be noted that neither chloride nor ammonia is an App II parameter. Chloride data were available for all of the study sites. Chloride was measured only for upgradient well at Site D. Ammonia data were available for all the study sites except for Sites D and F. These data were not analyzed for the study sites as the analysis was primarily focused on App I and II parameters with MCL/SMCL and examples of App I and II parameters with state standard but no MCL/SMCL. Both chloride and ammonia should be collected and analyzed to assess potential leachate release from MSWLFs and ensuing HHE impacts.

The analysis presented above was conducted only for parameters that have a federal MCL. In addition to reviewing the groundwater parameters with an MCL, groundwater quality data were also reviewed for App I and II list parameters with an SMCL–silver and zinc. These data were not available for all of the sites. For example, zinc and silver measurements were not available for Sites I and E, respectively. Only one site had an exceedance of one of these parameters; Site G had a silver exceedance three years after closure at a single well. As described previously, only 32 out of 62 App I and 51 out of 215 App II parameters have an MCL and two parameters in these lists have an SMCL.

The parameters that do not have a federal MCL/SMCL would also need to be analyzed for a comprehensive HHE impact assessment. The state may have a human-health risk-based standard for parameters with no federal MCL. An analysis was conducted for App I and II parameters without an MCL/SMCL for a study site to emphasize the importance of analyzing these parameters for assessing the groundwater impacts. As of the time of this evaluation, all three of the sites under assessment monitoring/corrective action had instances where parameters without an MCL contributed to groundwater impacts. Examples of these parameters include vanadium (Site A), 1,1-dichloroethane (Site F), and 1,4-dioxane (Site I). Establishment of a site-specific groundwater protection standard for the App II (which includes all App I parameters) parameters that do not have an MCL is required by §258. The site-specific groundwater protection standard may be based on the background concentration (established based on background/upgradient well groundwater quality) or the state-specified human-health risk-based standard.



(a)



(b)

Figure 7-2. Distribution of (a) Chloride, and (b) Ammonia for Leachate and Four Downgradient Wells and an Upgradient Well at Site I

In order to provide an example evaluation of a parameter with a state-mandated risk-based standard (but no MCL), parameters at Site I without an MCL were reviewed to identify parameters with a state-designated groundwater protection standard that appeared to be significantly elevated over background quality. Parameter 1,4-dioxane was one that has never been detected in background wells but was detected in three downgradient wells. Although there is no federal MCL/SMCL, a state-specified risk-based groundwater protection standard of 3 µg/L for 1,4-dioxane applies to Site I. The concentrations of 1,4-dioxane in these three downgradient wells are presented in Figure 7-3.

It is interesting to note that two of the three downgradient wells with detected concentrations of 1,4-dioxane at Site I are the same wells (i.e., Wells B and C) that exhibited arsenic exceedances, as presented in Figure 7-1. Well C, which exhibited the greatest 1,4-dioxane concentration, also had the greatest median chloride and ammonia concentration among all downgradient wells, as shown in Figure 7-2, suggesting potential impacts of the landfill on groundwater quality. Elevated concentrations of 1,4-dioxane in the downgradient groundwater monitoring wells (particularly in Well B and Well C), when compared to the concentrations observed at the background wells, suggest impacts from the closed study cell at Site I. The exhibited exceedances and temporal trend of concentrations suggests a need for continued groundwater monitoring of 1,4-dioxane for a reliable long-term HHE impact assessment.

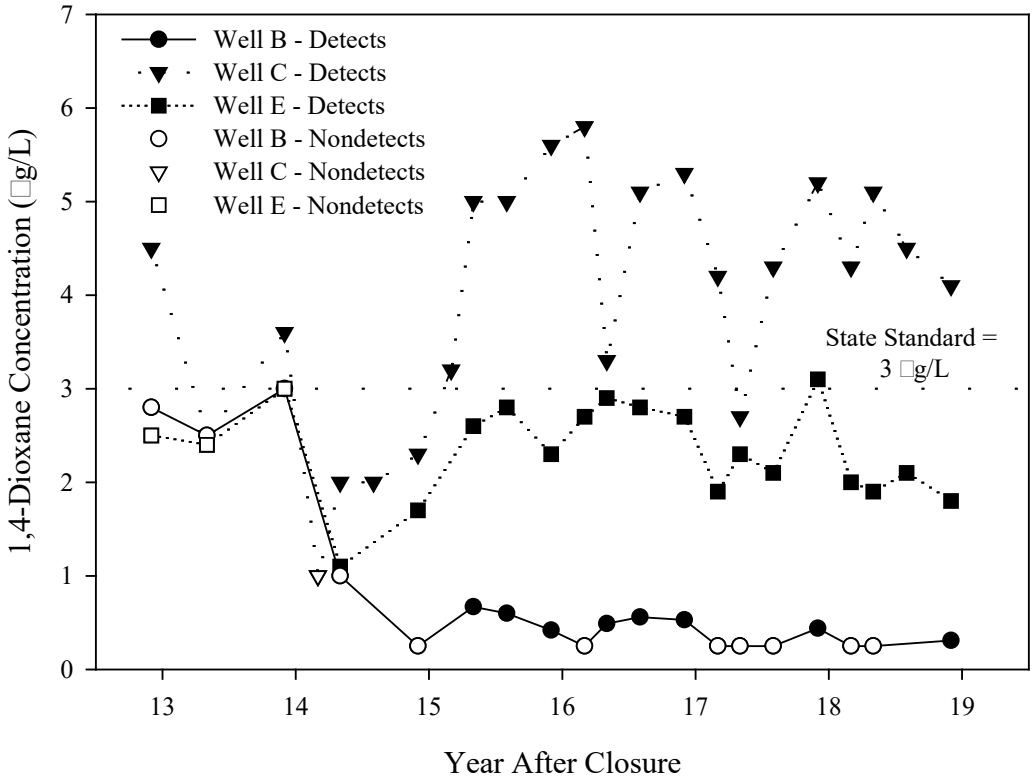


Figure 7-3. Temporal Variation of Detected 1,4-Dioxane Concentrations Observed at three Downgradient Wells of Site I After Closure

The groundwater 1,4-dioxane concentrations can be compared to that in leachate. If the concentration in leachate is statistically lower than those in groundwater, it is likely that the sources other than leachate are contributing to 1,4-dioxane exceedance at the site. A statistical comparison

of concentrations in groundwater and leachate could not be performed due to insufficient leachate 1,4-dioxane data. As presented in Attachment I, 1,4-dioxane concentration was available for only one leachate sampling event after closure.

Groundwater monitoring reports documented groundwater impacts associated with elevated vanadium concentrations at downgradient wells at Site A. Vanadium is included in both App I and II and does not have an MCL. Figure 7-4 presents a comparison of the distribution of vanadium concentrations at upgradient wells and downgradient wells after closure. Please note that only downgradient wells that have shown vanadium concentrations over the reporting limit on at least two sampling events are presented in the figure. As shown in Figure 7-4, the vanadium concentration at three downgradient wells appears to be noticeably greater than at the upgradient wells. In addition, there have been two occasions where a downgradient well exceeded the State's risk-based standard – this well has generally shown vanadium concentrations approximately an order of magnitude above background concentrations. Vanadium concentrations at this well have exhibited a declining trend since these exceedances.

For all non-detect measurements, the detection limit was used as a surrogate for the concentration value. Over 80% of the measurements at the upgradient wells were below the method detection limit. Over 75% of the measurements at two downgradient wells (D-2 and D-3) were reported below the detection limit.

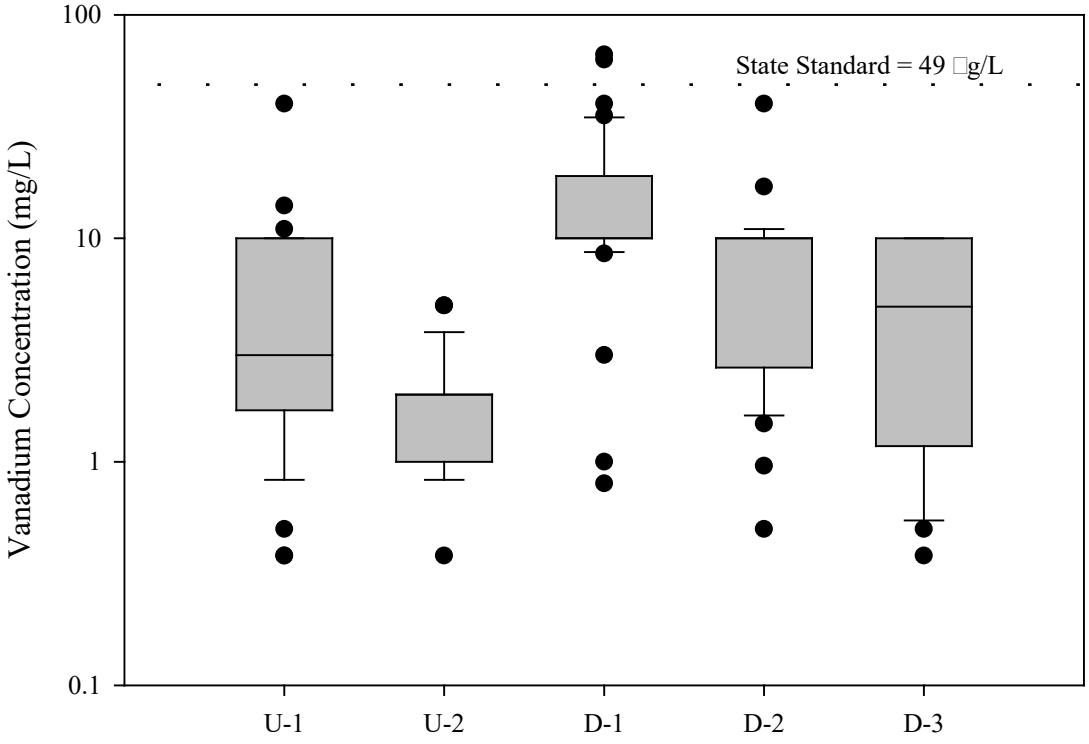


Figure 7-4. Distribution of Vanadium Concentration at Upgradient (U) and Downgradient (D) Groundwater Monitoring Wells of Site A After Closure

The groundwater impact assessment presented in the section is by no means a comprehensive assessment and has several limitations. First, only App I and II parameters with MCL/SMCLs were analyzed. Although an example of other parameters (e.g., 1,4-dioxane at Site I and vanadium

at Site A) are presented to emphasize a need to examine these parameters for assessing groundwater impacts, not all the groundwater monitoring data available for the site were analyzed. Site owners should consider collecting and analyzing data for all of the App I and II parameters, and leachate quality indicators (e.g., ammonia, chloride) as well as emerging contaminants such as per- and polyfluoroalkyl substances and pharmaceuticals, which are not be currently included in App I or II.

Second, the analysis was conducted based on federal MCL and SMCLs for App II parameters with these limits. The HHE impact analysis would need to be conducted based on the state-specified risk-based standards, which, if available, are expected to be lower than the respective federal MCL. The outcome of the analysis based on state-specified standards may be different from that based on federal limits. For example, Site B does not appear to pose a threat to HHE for arsenic and vinyl chloride with respect to MCLs. However, the site is under corrective action for frequent exceedances of several parameters, including arsenic and vinyl chloride. The state-specific vinyl chloride and arsenic standard used for this site are an order of magnitude smaller than the respective federal MCL. The state may require monitoring of additional parameters of HHE concern. For example, exceedances of manganese, which is not an App II parameter, were documented for Sites B and I.

Third, the available groundwater data had several inadequacies. Only limited data were available for a few of the study sites. For example, only one mercury measurement was available for each monitoring well after the closure of the study cell at Site C and only a single arsenic measurement was available for Site E. Adequate numbers of data points are essential for reliable statistical analysis and analyzing trends over time. The unavailability of data for review for this study does not necessarily mean that these data were not collected. MSWLFs owners should consider collecting and cataloging the groundwater quality data for easy retrieval for statistical and trends analysis.

Fourth, the data for several parameters had data quality issues specifically pertaining to the method detection limit used for the laboratory analysis. Several instances were found where the parameter analysis detection limit was greater than the respective MCL. These data were excluded from the analysis presented in this section. Finally, only data collected after closure were examined. All the available data, including data collected before waste placement activities, should be analyzed for a more comprehensive impact assessment. This analysis approach can be complemented with more rigorous fate and contaminant transport modeling to assess the impacts of contaminant transformation and dilution on groundwater quality at the point of compliance and the point of exposure over time.

7.5. Impact of Biogeochemical Changes on Groundwater Quality

Biogeochemical changes in the aquifer system due to liner construction can also impact groundwater quality. Several landfills throughout Florida have reported iron exceedances in groundwater monitoring wells. The process commonly believed to be responsible for iron releases at landfill sites is known as “reductive dissolution.” In this process, reducing conditions develop in the surficial aquifer at a landfill site and transform the oxidized solid-phase of naturally-occurring iron to reduced, dissolved iron. This phenomenon has been observed and described at landfill sites around the country. In many cases, the reductive dissolution of iron minerals triggers exceedances with respect to other parameters (e.g., arsenic); as iron reduces to its soluble ferrous form, arsenic sorbed to the iron is also released.

Many of the sites where iron dissolution has occurred are older, unlined landfills. However, growing evidence shows that similar problems can occur at lined landfill sites as well. The presence of a liner system may result in reducing conditions by limiting oxygen transport from the atmosphere to groundwater. Townsend et al. (2015c) observed an increase in groundwater iron concentration following the construction of a liner system of a test cell at a landfill in Florida. The test cell was constructed in an area away from other landfill cells at the site, and no waste was placed in the cell.

Figure 7-5 presents the iron concentrations distribution in landfill leachate and at several groundwater monitoring wells at a lined MSWLF–this site does not have an unlined cell (IWCS 2010). As can be seen from the figure, the iron concentrations recorded at groundwater monitoring wells are much greater than that of leachate, suggesting that leachate release alone could not result in the groundwater iron concentrations observed at this site.

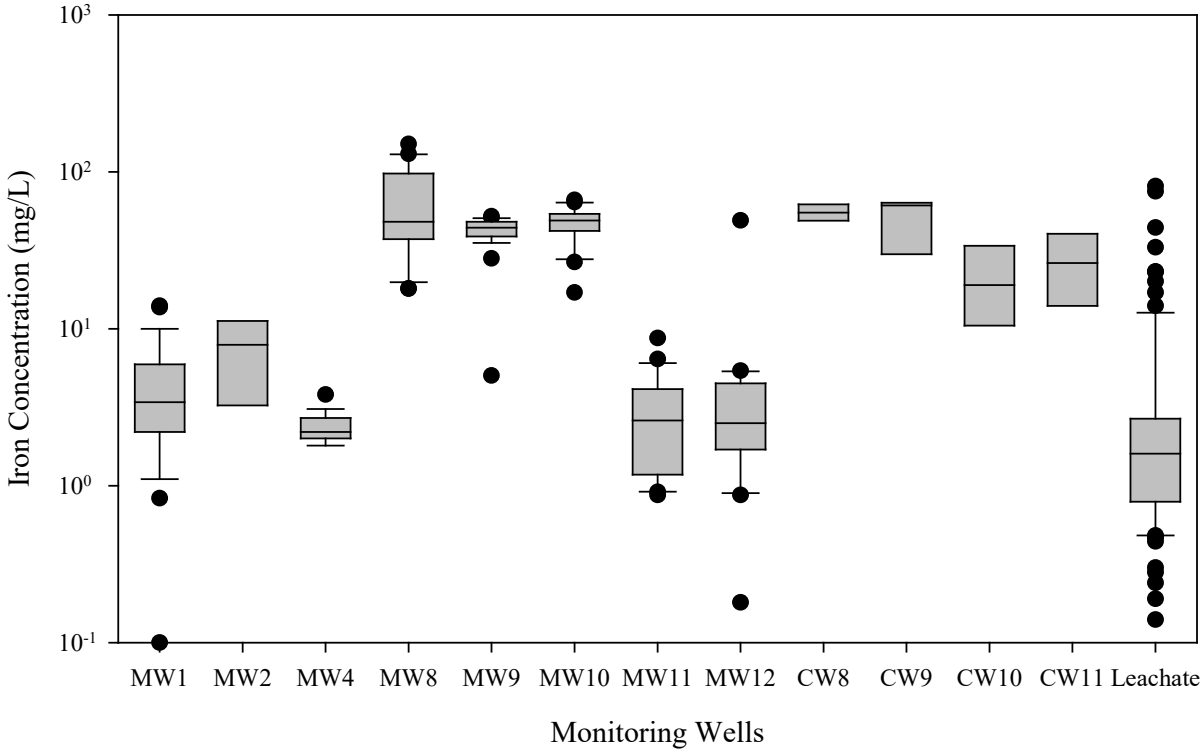


Figure 7-5. Distribution of Iron Measured at Various Groundwater Monitoring Wells (identification labels with MW and CW prefixes are groundwater monitoring wells), and Leachate at an Active Lined MSWLF in Florida (IWCS 2010)

Figure 7-6 shows the distribution of iron concentration at the four downgradient groundwater monitoring wells at Site I that showed elevated arsenic concentrations after closure. Two of the groundwater wells have median iron concentrations above that of leachate from the closed study cell, suggesting leachate may not be the primary contributor to the elevated iron concentrations observed at Site I groundwater.

Several indicator parameters can be used to assess whether leachate release and migration is the cause of groundwater contamination at MSWLFs. Strong indicators of leachate contamination include ions of soluble salts (such as chloride), ammonia, and organic chemicals. As discussed

above, a comparison of chloride and ammonia concentrations in downgradient wells with the respective background concentration (in upgradient wells) can be useful for assessing leachate impacted on the groundwater quality. The contaminant and chloride concentration ratio in leachate can be compared with that for groundwater to assess whether leachate release has resulted in the groundwater impacts with respect to the contaminant. In the event a leachate release is the major contributor to groundwater contamination, the chemical concentration ratio(s) for parameters such as iron or arsenic to chloride in groundwater should be similar to or (due to attenuation) lower than the ratio(s) for leachate.

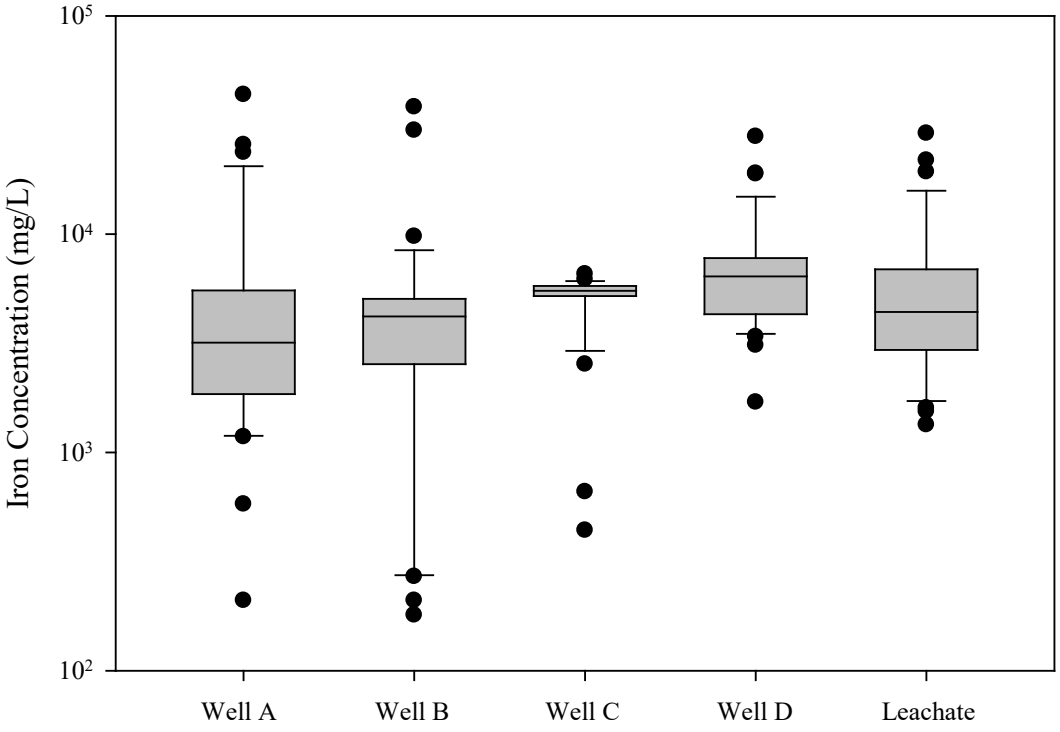
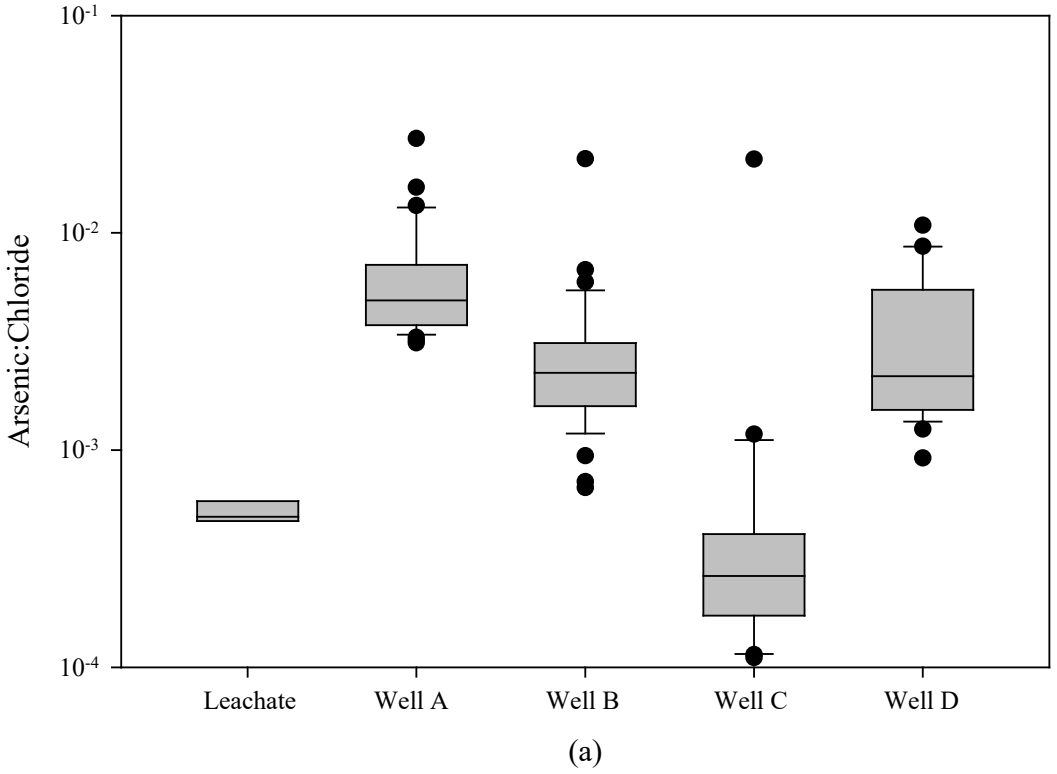


Figure 7-6. Distribution of Iron Concentrations Measured in Leachate and in Downgradient Groundwater Monitoring Wells with Elevated Arsenic Concentrations at Site I after Closure

Figure 7-7 presents the distribution of arsenic:chloride and iron:chloride ratios for four monitoring wells with consistent arsenic exceedances and for leachate from the Site I study cell. The arsenic:chloride ratios for three of these monitoring wells are substantially more than that of leachate, which also suggests that a leachate release, if any, is not the primary contributor to arsenic exceedances. Similarly, the iron:chloride ratio for all monitoring wells is approximately 2-3 orders of magnitude greater than for leachate, suggesting that leachate release, if any, is not the primary contributor of elevated iron concentrations in groundwater. This analysis approach cannot be used to conclude that leachate release has not contributed to the impact altogether as the magnitude of the arsenic mobilized from reducing condition is not known. Leachate release, if any, can also cause reducing conditions or further amplify prevailing reducing conditions and arsenic mobilization.



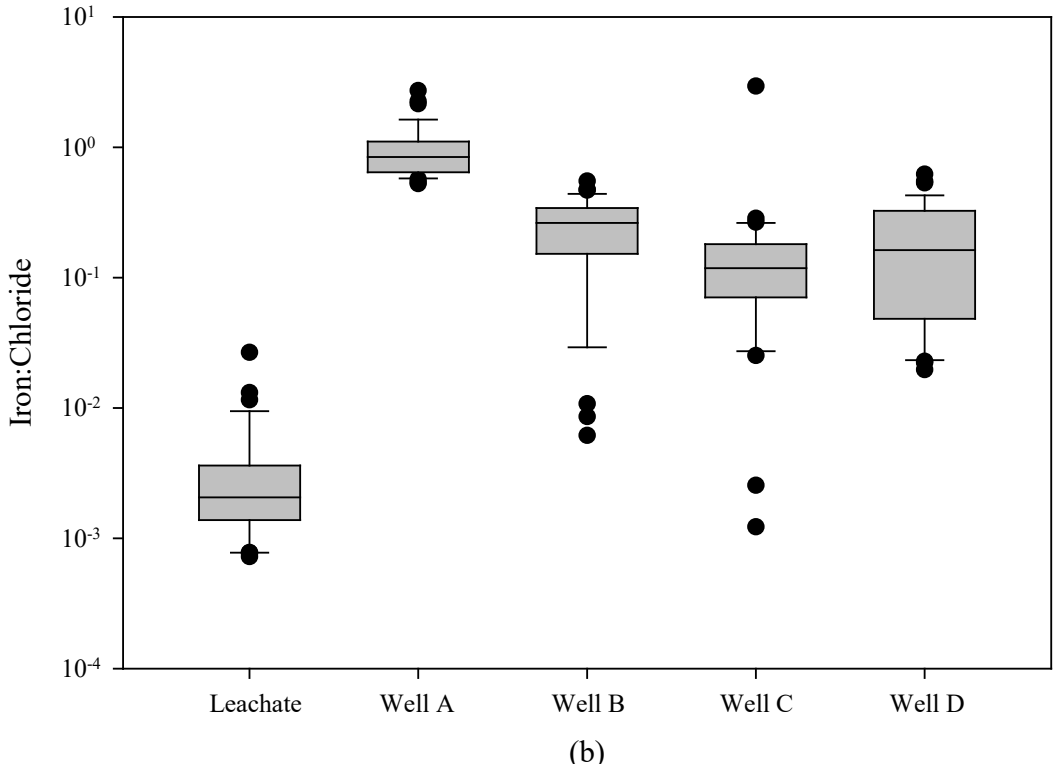


Figure 7-7. Distribution of (a) Arsenic-to-Chloride Ratio, and (b) Iron-to-Chloride Ratio for Leachate and Four Downgradient Wells at Site I after Closure

7.6. Variation in Groundwater Flow

As discussed above, MSWLF groundwater impacts are analyzed by statistically comparing the groundwater quality observed at downgradient wells with that of upgradient wells, which represent the background quality and are not expected to be impacted by landfill cell(s). The direction of groundwater flow determines the classification of wells as upgradient or downgradient. Groundwater flow direction and velocity are routinely estimated by plotting potentiometric maps using water level measurements at the wells during groundwater sampling events. Temporal changes in groundwater flow directions can complicate comparisons of upgradient and downgradient water quality and may make identifying the true impacts of MSWLFs challenging to assess. Of the nine study sites, periodic groundwater monitoring reports (which document groundwater flow direction) were available from five sites (i.e., Site A, C, D, F, and H). Of these sites, records for three sites (i.e., Site D, F, and H) suggest that there has been little/no variability in groundwater flow direction after closure. Of the other two sites, reports suggest that groundwater flow direction appears to be varying seasonally for one site (i.e., Site C) and may be experiencing long-term changes in groundwater movement at the other site (i.e., Site A). Changes/consistency in site groundwater flow directions at the study sites were not analyzed for each sampling event.

Factors such as weather patterns (e.g., sequencing of dry and wet years), changes in land use/zoning and associated changes in consumptive use of groundwater surrounding the sites (e.g., pumping from the municipal water supply or other well fields located in the vicinity of the landfill), or shifting gradients resulting from seasonal variations or tidal influences can result in localized changes in groundwater flow. The impact of factors like these should be considered while assessing the long-term impact of PCC duration reduction/termination on HHE.

7.7. Integrity and Performance of Monitoring Wells

The ability to collect representative groundwater samples is paramount to the reliability of groundwater data and the contingent analysis/decision making. Factors such as the clogging of groundwater monitoring well screens with sediments and precipitates (USGS 2016, OH EPA 2009, and EPA 1988) and accidental well damages (e.g., from equipment or vehicular traffic, surface drainage) casing can impact the quality of the data collected. As groundwater well maintenance records were not requested from the owners/operators of the study sites included in this study, an evaluation of the nature and prevalence of issues that may impact the performance of groundwater monitoring wells was not performed. The site owner should consider implementing a routine groundwater monitoring well inspection and maintenance program to ensure the collection of representative samples for an accurate and reliable assessment of groundwater water quality. The site operator should consider maintaining and including these records as part of the demonstration to request PCC termination or an alternative PCC duration.

7.8. Summary

The groundwater monitoring data for all the study sites were analyzed for the App II parameters with a federal MCL or SMCL. The groundwater data were analyzed for all the wells at the study site regardless if the study cell was contiguous to the other active or unlined cells (Sites A, B, E, G). Only data from the wells around the study cells were analyzed for the site if the study cell was not contiguous to active or unlined cells (Site C, F, H, and I). A majority of the organics measured at the sites were not detected in groundwater. No exceedances were identified at Site F for the

range of data available for review (i.e., post-closure Year 8 through Year 19). Only occasional MCL exceedances were identified following cell closure among the studied sites. Only Sites G, H, and I appear to have had reoccurring groundwater exceedances. More than 55% and 10% of 255 samples at Site H had beryllium and thallium concentration greater than the associated MCL, respectively. Approximately 40% and 30% of beryllium and thallium exceedances observed at Site H occurred at the upgradient well(s), respectively. Site G has 107 and 53 samples with arsenic and lead concentrations greater than the associated MCL, respectively. More than 10% of arsenic and lead exceedances observed at Site G occurred at the upgradient well(s). More than 50% of the arsenic exceedances observed at Site G correspond to the wells that were downgradient or close to the unlined cell at the site. The beryllium and thallium exceedances at Site H and arsenic and lead exceedances at Site G cannot be conclusively attributed to lined cells at these sites due to the exceedances observed above the respective MCL at several upgradient wells for these parameters at these sites.

The arsenic exceedances at Site I have been consistently observed at several downgradient wells after closure. An analysis of the arsenic:chloride ratio in groundwater and leachate at Site I suggest that arsenic exceedances at the site are not likely associated with leachate release, if at all. The change in the biogeochemical environment of the surficial aquifer resulting from landfill construction can also result in mobilization of naturally-occurring constituents and impact to the groundwater quality.

The available silver and zinc were analyzed to identify exceedance with respect to the SMCL. These data were not available for all of the sites. For example, zinc and silver measurements were not available for Sites I and E, respectively. Only one site with these data had an exceedance of one of these parameters; Site G had a silver exceedance three years after closure at a single well.

Only 51 out of 215 App II parameters have an MCL and two parameters in these lists have an SMCL. The parameters that do not have a federal MCL/SMCL may also need to be analyzed for a comprehensive HHE impact assessment. The state may have a human-health risk-based standard, which may be used for assessing the risk to HHE for these parameters. As of the time of this evaluation, all three of the sites under assessment monitoring/corrective action had instances where parameters without an MCL contributed to groundwater impacts. Examples of these parameters include vanadium (Site A), 1,1-dichloroethane (Site F), and 1,4-dioxane (Site I). In addition, leachate indicator parameters such as chloride and ammonia should be routinely monitored for groundwater. These parameters are helpful for assessing whether or not the observed groundwater impacts are associated with leachate emission from MSWLFs.

7.9. Limitations

The analysis presented in this chapter has the following limitations:

1. Only limited groundwater data were available for a few of the study sites. For example, only one mercury measurement was available for each monitoring well after the closure of the study cell at Site C and only a single arsenic measurement was available for Site E. The contaminant concentration can vary considerably over time.
2. Only the groundwater quality data available after closure of the study cell(s) was analyzed. The analysis presented in this chapter does not include the data collected before the cell closure.
3. The MDL for a large number of organic compounds and some metals monitored for the study sites was greater than the respective protection standard. These parameters cannot be

conclusively determined to be below MCL/SMCL These measurements were not included in the analysis presented in this chapter.

4. The analysis was conducted for App II parameters with MCL/SMCL. More than 60% of the parameters that were measured at least once do not have an MCL/SMCL. These parameters, which may be of HHE concerns, were not comprehensively evaluated in this study.
5. The analysis was conducted based on federal MCL and SMCLs for App II parameters with these limits. The HHE impact analysis would need to be conducted based on the state-specified risk-based standards, which, if available, are expected to be lower than the respective federal MCL. The outcome of the analysis based on state-specified standards may be different from that based on federal limits. For example, Site B does not appear to pose a threat to HHE for arsenic and vinyl chloride with respect to MCLs. However, the site is under corrective action for frequent exceedances of several parameters, including arsenic and vinyl chloride. The state-specific vinyl chloride and arsenic standard used for this site are an order of magnitude smaller than the respective federal MCL.
6. The analysis was conducted only for App II parameters (all App I parameters are included in App II). The state may require monitoring of additional parameters of HHE concern. For example, exceedances of manganese, which is not an App II parameter, were documented for Sites B and I.

8. Final Cover Performance

8.1. Surface Emissions Monitoring

SEM data were available for review from six of the nine sites. However, a complete dataset covering the entire period from the closure date to the time of this study was only available for two of the sites (Sites D and H). Table 8-1 presents the number of locations with SEM exceedances (a surface emissions concentration elevated over the 500-ppm standard) as a function of years since closure. A "-" indicates that SEM data were not available for the site for that year. Shaded cells indicate future years. Please note that the values presented in the table only represent exceedances identified over the cover system of the site's study cell(s).

Table 8-1. Number of Annual Surface Emissions Monitoring Exceedances Over Study Cells as a Function of Time after Closure

Site	C[1]	D	E	F	H	I	Years Since Closure
Years of Available Surface Emission Monitoring Data	2014 - 2016	2009-2016	2008-2015	2012-2015	2008-2016	2004-2016	
Area (Acres)	40.8	69	30 (Cell 2- 24 and Cell 3-6)	38	60	51	
Closure Year	1998	2009	2004	1997	2008	1997	
Exceedances per Year	-	3	-	-	6	-	0
	-	0	-	-	2	-	1
	-	0	-	-	1	-	2
	-	0	-	-	2	-	3
	-	0	5	-	4	-	4
	-	0	2	-	2	-	5
	-	0	2	-	0	-	6
	-	0	1	-	2	61	7
	-		0	-	0	21	8
	-		0	-		0	9
	-		3	-		0	10
	-		0	-		0	11
	-			-		0	12
	-			-		0	13
	-			-		0	14
	-			0		0	15
	13			0		0	16
	10			0		0	17
	10			0		0	18
				-		0	19

[1] Data were available for three quarters of year 16, 17, and 18 after the closure

Study cells at Sites C, E, and H had the most frequent exceedances based on the available data. The highest number of exceedances was observed over two monitoring events at Site I. Well penetrations through the final cover were documented as the primary factor contributing to these exceedances. More than 80% of these exceedances were corrected within 30 days of the initial SEM exceedance observation; details about measures taken to correct the exceedances were not

available. As shown in the table above, the final SEM exceedance observed at five out of six sites were within ten years since closure. SEM exceedances beyond the initial ten years of PCC were only observed at one site (Site C), and exceedances were noted for all three years that data were available (years 16-18). It should be noted the study cells at Site C are covered with an exposed geomembrane cap, which lacks a soil cover layer where methane oxidation could occur. At Site E, recent exceedances occurred near wellheads, which were repaired either by excavating the cover soil and backfilling with bentonite or by installing an additional small well tied into the geomembrane just outside the well casing. At Site H, two exceedances were observed near two different gas wells during one of the monitoring events in 2015 (seven years after closure). The PVC pipe collar between the gas well casing and the geosynthetic closure system was adjusted at one exceedance location and additional soil filling was provided at the other location as a corrective measure to address these exceedances. Exceedances were not observed in the following 10-day monitoring event at both Site H locations.

8.2. Settlement

With the exception of Site I (where annual settlement data were available), limited settlement data were available from occasional surveys of the landfill surface for Sites B, C, E, and F. For Site C, topographic surfaces generated from surveys conducted in the 0, 1st, 4th, 6th, and 18th year after closure were available; however, settlement or elevation data at specific survey points were not available. Similarly, for Site E, the topography of the final cover at the 0, 7th, and 8th year after closure were available for review; however, the available data were insufficient to analyze temporal trends in the rate of settlement. Site F topographic maps were available for PCC years 5, 9, and 15 as images; however, the topographic survey data for the closure year were not available; therefore, an analysis of trends in the settlement rate for Site F was not conducted. Landfill settlement measurements at ten settlement plates installed on the final cover at Site B were available for 9th and 11th year after closure. The location details of these plates were not available.

Site I topographic survey data was available for PCC years 1 to 13, 15, and 17 for 20 discrete survey points located on the study cell. Eleven (11) of these points were on the top deck. Distribution of the annual settlement rate at the survey points located on the top deck of the cell is shown in Figure 8-1. The overall settlement rate exhibits an overall declining trend after closure. The median annual settlement decreased from 2.4 ft in the first year to 0.13 ft in the 17th year after closure. The range of the settlements observed at the survey points appears to decrease over time from 2.5 ft to approximately 0.5 ft during the first year and 17th year after closure, respectively. The range of the measured settlement rate at a given point of time is indicative of differential settlement. As discussed previously, the differential settlement is one of the primary contributing factors that impact the integrity of the final cover. As shown in Figure 8-1, a few points exhibited a negative annual settlement rate, which might be associated with regrading activities that are typically conducted on an as-needed basis to fill depressions and maintain positive surface drainage. Several approaches and models have been used for modeling waste settlement (Townsend et al., 2015a). Some of these models account for mass/volume loss over time due to the decomposition process (e.g., Hettiarachchi et al. 2007).

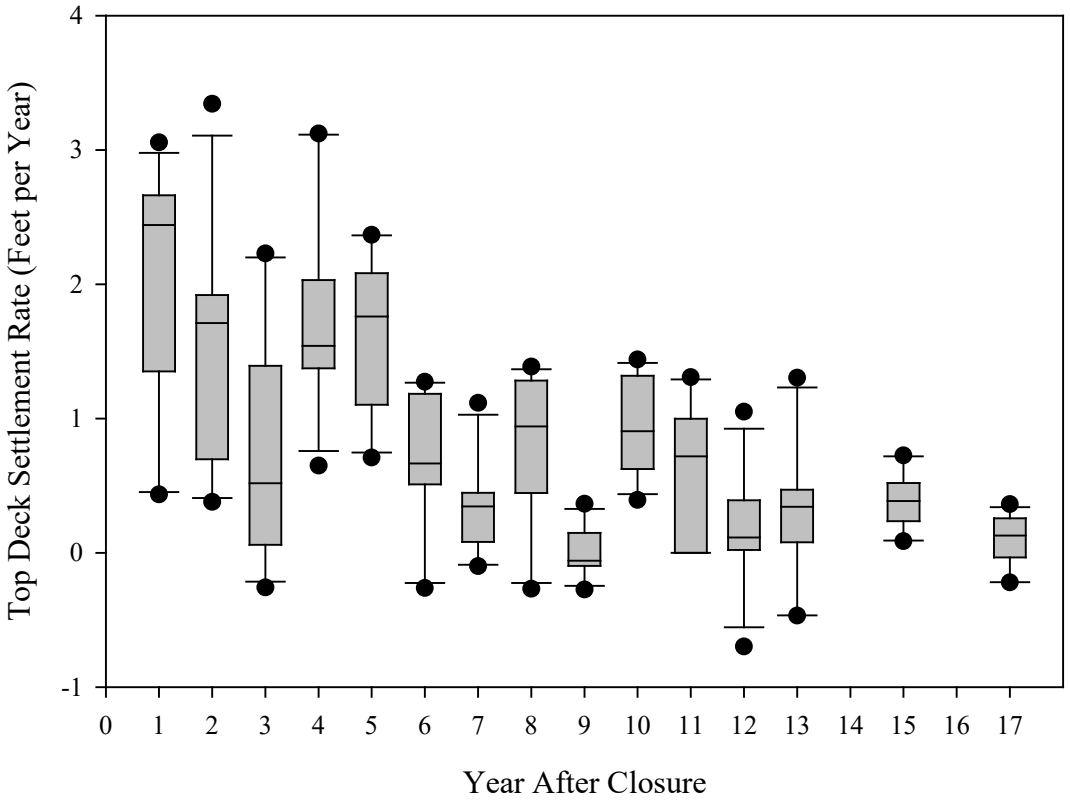


Figure 8-1. Distribution of Yearly Point-to-Point Settlement Rate at Site I

Bottom grade construction drawings and topographic information were used to develop a topographic surface representative of the base grades of the cell at Site I (i.e., using AutoCAD Civil 3D 2013). The topographic surfaces were compared with bottom grade surfaces to estimate in-place waste volume for each year, which were used to evaluate the annual volume loss rate. The waste volume loss rate is expected to be driven by the waste decomposition rate, which typically is modeled using a first-order decomposition rate equation (Hettiarachchi et al. 2007). Several researchers have modeled MSW mass and volume loss rates (e.g., Sheridan 2003, Kim 2005). Many of these either are empirical or do not resemble the first-order decay model used for landfill gas modeling. In order to estimate a volume loss decay rate that is analogous to the waste decay rate used for LFG modeling, Eqn. 8-1 was used to model the volume loss rate; this equation resembles the equation (Eqn. 5-3) used for methane generation modeling for MSWLFs. This approach is similar to the one used by Hettiarachchi et al. (2007) to model the volume loss rate using the first-order decomposition rate equation.

$$V_t = k_v \cdot V_s (e^{-k_v \cdot t}) \qquad \text{Eqn. 8-1}$$

where,

V_t = waste volume loss rate at time t (yd^3 per year)
V_s = waste volume loss over an infinite time horizon (yd^3)
k_v = first-order decay rate for volume loss ($year^{-1}$)
t = time (years)

An iterative regression analysis was conducted using the Excel Solver function in order to calculate the decay rate constant and waste volume loss over an infinite time horizon that would best-fit the model to the estimated volume loss based on the settlement data. Figure 8-2 shows the estimated and the modeled volume loss rate data for Site I. Once calculated, the model parameter values (as shown in the figure) can be used to estimate the annual volume loss rate at a site for future years. The results of the regression analysis suggest a total volume loss of 670,000 yd^3 over an infinite time horizon and a first-order decay rate of 0.149 $year^{-1}$. Approximately 526,000 yd^3 of volume loss (representing approximately 79% of the total estimated volume loss) had occurred within 17 years of closure. The model indicates that 99% of the volume loss is estimated to occur within 30 years of closure. Due to the lack of landfill gas data for Site I, the correlation between the estimated volume loss rates and the methane collection rates, and that between the methane generation and volume loss decay rates could not be assessed.

The settlement and volume loss estimates presented above assumed a constant topographic condition for the landfill bottom and did not account for the foundation settlement at the site. The settlement and volume loss estimates are regarded as overestimations due to unaccounted settlement of subsurface soils below the landfill as a result of overburden pressure.

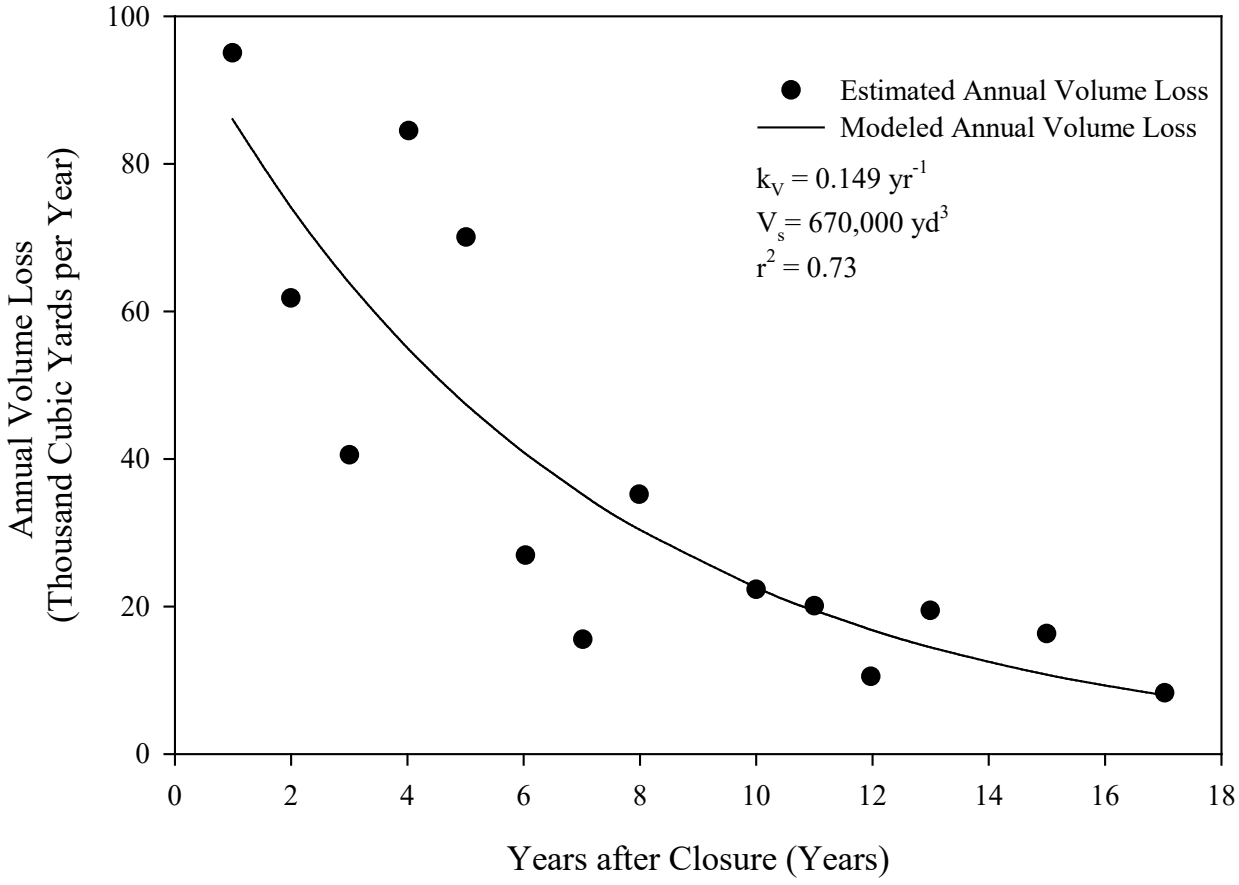


Figure 8-2. Site I Estimated Annual Volume Loss Based on Topographical Data and Modeled Volume Loss

9. Summary and Considerations

9.1. Summary of Findings

9.1.1. Overview

The MSWLFs owners are required to perform PCC for a period of 30 years after closure (§258.61(a)), unless the time period is decreased or increased by the Director of an approved State as necessary to protect HHE. Subpart F of §258 lists PCC-specific requirements for the MSWLFs. Although the RCRA Subtitle D regulations allow modification of PCC duration, there is no federal guidance or specific direction on approaches that can be used for making a demonstration pertaining to HHE impacts for supporting the extension or termination of PCC activities. The EPA collected, reviewed, and analyzed environmental monitoring data from the closed cell(s) of nine MSWLFs located across the US to use as examples to:

(1) Assess the nature of the data available for MSWLFs that can be used for HHE impact evaluation
(2) Present approaches that site owners/operators and engineers can use to evaluate monitoring data for identifying the COPCs and estimate emission rates of these contaminants
(3) Identify data gaps, and
(4) Present operating and monitoring considerations for MSWLFs owners to evaluate and mitigate the long-term impacts of MSWLFs.

Five of the sites are located in the northeast, two are located in the southeast, one in the northwest, and one in the southwest region of the US. Five of the study sites are publicly owned and operated, and the rest four are privately owned and operated. The selected study sites each possessed at least one cell that has been closed for five years or more and has maintained environmental monitoring records (e.g., groundwater monitoring, landfill gas quantity and quality, leachate quantity and quality) available for review. The area and capacity of the studied cell(s) at these study cells/sites ranged from 6 to 69 acres and from 0.5 to 4.9 million tons of MSW, respectively.

9.1.2. Post-Closure Care Cost

The available PCC cost data from the nine study sites were analyzed to evaluate the financial impact of different MSWLF PCC activities. The available data were organized into the following six major cost categories: (1) leachate management, (2) GCCS management, (3) final cover maintenance, (4) groundwater and subsurface gas monitoring and maintenance, (5) engineering support and administration, and (6) other miscellaneous expenditures.

Limited PCC cost data were available for the study sites. Actual PCC cost data were available only for three sites (Sites B, D, and E); only estimated cost data were available for the other locations. Additionally, only three sites (Sites G, H, and I) provided cost data exclusive to the study cells. For the other sites, the available cost data included the cost of maintaining other site cells as well. All the available PCC cost data were adjusted to 2017 dollars based on consumer price index values.

For the sites with available study cell-specific data, the annual PCC cost varied from approximately $1,200 to $11,000 per acre of waste footprint, with an average of $5,300 per acre. For the sites where PCC cost data represented the entire site, annual PCC cost ranged between $1,550 to $37,000 per acre with an average of $6,450 per acre. Inconsistencies in cost categories used for tracking PCC cost, and the necessity for occasional system upgrades (e.g., GCCS expansion at Site E, and construction of a sewer connection for pumping leachate to the local WWTP) appears to be one of the primary reasons for such a wide variation in annual PCC cost at the study sites.

Annual leachate management (including LCS operation and maintenance, hauling, onsite/off-site treatment, and leachate sampling and analysis) cost represented the greatest category cost at six of the nine sites with a range of approximately 3 % to 68% of PCC cost. The leachate management ranged from $5.5 to $219 per 1,000 gallons of leachate collected with an average of approximately $79 per 1,000 gallons of leachate collected. GCCS management costs ranged from approximately 11% to 44% of the annual average PCC cost. The annual GCCS management cost ranged from approximately $2,700 to $593,000 per year, for seven of the nine sites based on available data. The average annual groundwater and subsurface gas monitoring cost varied between approximately 5% to 26% of the total average annual PCC cost. The annual monitoring cost among the sites ranged from approximately $2,400 to $169,600. Annual final cover maintenance cost among the sites (except Site H) ranged between approximately 0.2% to 14% of the total average annual cost. The annual cover maintenance cost of two sites (Sites G and I), which had cell-specific cover maintenance cost data available, ranged from approximately $56 to $600 per acre of the waste footprint. The final cover maintenance cost at Site I was observed to decline with the amount of differential settlement, which was observed to reduce at Site I over time. For remaining sites, the annual cover maintenance cost ranged from approximately $24 per acre to $3,400 per acre.

The available data suggest that the cost varied over a wide range due to several factors, including inconsistencies in cost categorization, mixed availability of actual cost and cost estimates, and the necessity for occasional capital-intensive system upgrades. Because of these limitations, landfill owners and engineers should consider tracking and using the site-specific cost data for evaluating the financial impacts of PCC activities instead of using the data presented in the report as proxies.

9.1.3. Landfill Gas

LFG emissions constitute one of the primary pathways for potential HHE impacts resulting from MSWLFs. The available LFG collection and flow rates and subsurface probe monitoring data for the study sites were analyzed to:

(1) Assess the frequency of subsurface methane migration
(2) Assess the timeframe needed for the annual NMOCs collection rate from study sites to drop below the NSPS threshold of 50 (or 34) Mg per year and the timeframe for the LFG flow rate to decline below 5% and 10% of the peak flow rates
(3) Assess whether MSWLFs would have sufficient NMOCs generation potential to generate 50 (or 34) Mg of NMOCs annually after the termination of GCCS operation, and
(4) Assess the remaining methane generation potential at 30 years after closure and assess the timeframes needed for the remaining methane potential to drop below 25% and 10% of the

total generation potential. Some states use the percent remaining methane potential as criteria for assessing the PCC termination.

The subsurface methane monitoring probe data from seven of the study sites suggest exceedances in the subsurface and structural methane monitoring is a relatively rare occurrence. Approximately 1.4% of all 7,598 methane measurements at the subsurface perimeter monitoring probes exceeded the lower explosivity limit of methane. Approximately 90% of these exceedances were observed at Sites B and E. The observed methane exceedances at these sites likely caused unlined cells or cells lined with a compacted clay liner at these sites. All the exceedances occurred during the first five years after closure for all the sites except for Site E. The presence of a geomembrane liner and an active GCCS limit subsurface migration of landfill gas and are the likely primary reasons for the relatively infrequent methane detection/exceedances. The GCCS at Site A was terminated in the 13th year of closure and no exceedances were reported since. The structure methane concentration data (2,105 measurements) were also available for Sites B, F, and H. None of the methane measurements in structures at Sites B and H were above 25% of its LEL. Only 5 out to the total 1,155 methane measurements in Site F structures were above 25% of its LEL.

A regression analysis was first conducted using data from all of the study sites (except Site I) to estimate site-specific decay rates and methane collection potential of waste that provided the best-fit of the first-order decay model to the available monthly methane flow rates data. LFG data exclusive to the closed cell at Site I were not available. The estimated site-specific decay rates suggest that the waste decomposition at all of the eight study sites (with GCCS data) occurred more rapidly than the decay rates specified/recommended by NSPS, AP-42, and GHG Reporting regulations. This suggests that the post-closure methane generation rate from MSWLFs are expected to be lower than those estimated based on regulatory default or AP-42 recommended decay rates.

The future annual methane and NMOCs generation rates were then estimated for the study sites using the first-order waste decomposition rate equation based on site-specific decay rates and methane collection potentials, disposal amounts, and NSPS-default NMOCs concentration of 4,000 parts per million by volume as hexane. The annual NMOCs collection rate at each study site was estimated to decline to below 50 Mg per year within 30 years after closure. The annual NMOCs collection rate was estimated to decline below 34 Mg per year within 30 years after closure for all the study sites except one (Site F). The study sites are not representative of the size of approximately 75% of MSWLFs in the US, as six of the study sites contain less than 4 million metric tons of waste. In order to assess the post-closure timeframes for typical size MSWLFs to achieve NMOCs reduction below the NSPS threshold, NMOCs and methane flow rates were estimated for MSWLFs containing approximately 3.35 (small MSWLF), 7.85 (medium MSWLF), and 19.1 (large MSWLF) million metric tons of MSW, which correspond to the 25th, 50th, and 75th percentile of the capacity of MSWLFs in the US, respectively. In order to assess the impact of decay rate on the annual NMOCs collection rate trend, first-order decay modeling was conducted for each of the three hypothetical MSWLFs for the five decay rates ranging from 0.02 to 0.22 year $^{-1}$ to represent waste decay conditions ranging from arid climate to bioreactor operation.

The analysis suggests that the annual NMOC collection rate for small MSWLFs located in arid to moderate precipitation areas are not expected to decline below 50 Mg per year within 30 years after closure. The NMOCs collection rate for medium and large MSWLFs is not expected to decline below 50 Mg per year within 30 years after closure irrespective of location. The annual NMOCs collection rate of MSWLFs operated as a bioreactor is expected to decline below 50 Mg per year within 20 years after closure. The LFG flow rate is expected to decline below 5% of the peak flow rate after the annual NMOCs collection rate reduces below 50 Mg per year for all of the typical-size MSWLFs scenarios modeled.

The analysis suggests that the annual NMOC generation rate can surge above the NSPS threshold of 50 Mg per year after GCCS operation termination with an adequate increase in the decay rate. The deterioration of the final cover, if any, after GCCS operation termination could allow moisture into the landfill, which is expected to enhance waste decay rate, uncontrolled methane, and NMOCs emissions. The final cover should be monitored and maintained until the remaining generation potential and the leaching potential of the in-place waste have declined to the levels which are unlikely to pose a risk to HHE. In addition, landfill owners and regulators should also continue surface and subsurface emissions and odor monitoring to proactively identify signs of an increase in LFG generation rate and have provisions in place to resume GCCS operation, if needed, to control these issues.

Estimates of the remaining methane generation potential were predicted to decline to below 10% of the total potential within 30 years after closure for all the sites except for Site F. The percent remaining methane potential, however, is not an appropriate metric to assess the HHE impacts. A smaller percent remaining methane generation potential at a large MSWLF may pose a greater risk than a small MSWLF with relatively higher percent methane generation potential. A mass-based threshold (e.g., NSPS threshold of 50 Mg/year for annual NMOCs generation rate) is a more appropriate metric for HHE impact assessment than a percent-based criterion.

9.1.4. Landfill Leachate

Leachate collection rate and quality data available for the study sites were analyzed to:

(1) Determine whether any of the sites have stopped generating leachate or achieved a historically low and stable leachate collection rate after the closure
(2) Assess approaches that can be used to estimate a site's long-term leachate collection rate
(3) Evaluate the comprehensiveness of the available leachate quality data with respect to the parameters specified by the federal regulations for groundwater monitoring
(4) Conduct a screening analysis to identify the contaminants that have been frequently measured in leachate above the respective human health risk-based protection standards at the study sites after closure to identify the COPCs, and
(5) Assess whether the concentration of the COPCs identified based on the screening analysis has declined below the respective protection standard with time after closure.

Leachate collection rates, quality, and quantity data were available for all study sites. Leachate collection rates were also available for two separate and distinct cells at two of the sites (C and G). The leachate collection rates were analyzed for 11 closed cells.

The annual leachate collection rate across all 11 cells varied from 3 – 2,070 GPAD after closure, with a median of 92 GPAD and an average of 190 GPAD; 90% of the annual collection rate measurements were less than 500 GPAD. Leachate generation continued at all of the sites. The leachate collection rates for all sites except for Sites A, C1, and F exhibited a generally declining trend. Based on reported chloride concentrations and observed spikes in leachate collection during the rainy season, the recent spike in leachate collection rate at Site F appears to be a result of stormwater intrusion into the LCS infrastructure. None of the landfill cells examined exhibited trends indicative of attaining a steady-state leachate collection/generation rate.

An estimate of the long-term leachate generation rate would be needed to reliably assess the HHE impacts. Three approaches were used to estimate the future leachate generation rate from the study sites: first-order decay modeling, unsaturated flow modeling, and HELP modeling. In general, all three approaches provided comparable and reasonable fits to the measured study site data for all the study sites except Sites C1, D, and G3. Sites C1, D, and G3 cells exhibited the lowest r^2 value for both the first-order decay and the unsaturated flow models. Unlike the first-order decay and unsaturated flow model, the HELP model was not iteratively executed to obtain the best-fit to the measured leachate collection rate data but was based on a mix of default parameters and the best-fit parameters from the unsaturated flow model.

The comprehensiveness of the available leachate quality data at the study sites was analyzed with respect to the parameters specified in the federal regulations for groundwater monitoring at Subtitle D landfills (App I and App II of §258). The leachate quality data for Site G were not analyzed since the data from this site represented the quality of leachate composited from closed and active cell(s) of the site. The data for a total of 272 leachate constituents were evaluated to assess the leachate quality of the study sites. The number of chemical constituents that must be monitored for groundwater as part of App I and App II 40 CFR §258 are 62 and 215, respectively; all the App I parameters are also included in the App II list. The parameters monitored varied widely among the sites. Only three sites reported leachate constituent data for every App I parameter, and one of these sites reported at least one measurement for all but three of the App II parameters. Two study sites were encountered with leachate characteristic data available for less than half of App I parameters. More than half of the study sites reported leachate constituent concentration data for ten or fewer App II parameters (excluding App I parameters). The monitoring frequency varied among the sites from once per month to twice per year, and further varied with time and contaminant. Only a single measurement was available for a few of the organic compounds measured at each site after closure. Apart from the lack of data for a large number of App I and II parameters, the small number of measurements available for some constituents reported may limit a reliable HHE impact assessment.

A screening analysis identified chemical constituents that have been frequently measured in leachate above the respective risk-based protection standards at the study sites after closure to identify the COPCs. Parameters that were never measured above the respective risk-based standards are not expected to present a risk to HHE. The federal primary and secondary drinking water standards were used as the thresholds for this evaluation. Fifteen parameters were measured at concentrations above the respective MDL in more than 50% of the samples. Of these 15 parameters, six parameters (i.e., arsenic, TDS, iron, manganese, chloride, and color) were

measured above their MCL/SMCL in more than 94% of the samples. Among all the constituents with an MCL, arsenic and turbidity were the only primary MCL parameters that consistently exceeded the MCL. Among the secondary parameters, greatest dilution and attenuation would be needed for iron for its concentration to decline below its SMCL of 0.3 mg/L. A majority of the iron measurements were below the regional screening level developed by the EPA for tapwater for the ingestion pathway for iron (14 mg/L).

The temporal trend for most of the COPCs varied among the study sites. Arsenic showed a declining trend for Sites A, B, C2, and E. Four of the most recent arsenic measurements at Site B were below the MCL. The most recent set of measurements at Sites A, C2, and E were above the MCL. Arsenic exhibited an increasing trend for Sites C1, D, and H. The arsenic release rate (lbs per acre per day) from Site H exhibited a slightly declining trend over time. TDS and ammonia at the sites, in general, appeared to be stable to slightly decreasing with time. In general, BOD appeared to gradually decrease since closure for all the study sites for which BOD data were available. Leachate COD appears to be slightly decreasing after closure at all of the sites. It should be noted that the leachate quality is typically reflective of the decomposition status of the bottom-most waste layer and does not necessarily represent the degree of stabilization of the entire landfill. A well-decomposed waste layer above the LCS may attenuate the concentration of parameters such as BOD and COD that are commonly used to assess leachate and waste stability.

The hydraulic efficiency of the primary liner was also evaluated for four of the study cells that are lined with a double bottom liner system. The LDS rates, in general, show an overall declining trend over time after closure. The aggregate hydraulic efficiency of the primary liner was calculated by dividing the sum of annual LDS collection rates by the sum of the corresponding annual LCS and LDS rates for all years with available LCS and LDS collection rates. The primary liner efficiency was calculated to be 97.2%, 99.6%, 99.6%, and 96.8% for study cells G3, G4, H, and I, respectively. However, a comparison of several leachate indicator parameters (chloride, and trace organics or TOC) in primary and secondary leachate and groundwater suggests groundwater intrusion into the LDS might be a significant source of liquids collected from LDS. The primary liner efficiency at these sites is thus expected to be higher than the estimates presented above.

9.1.5. Groundwater Monitoring Data

The groundwater monitoring data were analyzed for each of the study sites for App II parameters to:

(1) Identify the parameters that were detected above the respective MCL or SMCL
(2) Assess whether the concentration of the contaminants that were frequently detected above the respective MCL has declined below the MCL over time after closure, and
(3) Analyze groundwater quality data available for the study cells to identify examples of the parameters that do not have an MCL and that were detected above the respective state-specific risk-based standards.

It should be noted that the groundwater data were reviewed for all of the monitoring wells at the study site for five sites (Sites A, B, E, and G). The data only from the wells around the study cells at Sites C, D, F, H, and I were analyzed as these study cells were standalone cells and did not adjoin an unlined or active cell.

The impacts to groundwater were identified at the sites for three sets of parameters: parameters with MCL, parameters with SMCL, and examples of parameters without MCL or SMCL. A large number of App I and II parameters do not have an MCL/SMCL; only 32 and 51 of App I and App II parameters, respectively, have an MCL, and only two parameters (i.e., silver and zinc) have an SMCL. The groundwater impact with respect to these parameters without any MCL/SMCL can be assessed by comparing the concentrations measured at downgradient wells to those of upgradient well(s) or to a federal or state recommended risk-based standards.

Only occasional MCL exceedances were identified following cell closure among the studied sites. No exceedances were identified at Site F for the range of data available for review (i.e., post-closure Year 8 through Year 19). Only Sites G, H, and I were observed to exhibit reoccurring groundwater exceedances. More than 55% and 10% of samples at Site H had beryllium and thallium concentration greater than the associated MCL, respectively. Approximately 40% and 30% of beryllium and thallium exceedances observed at Site H occurred at the upgradient well(s), respectively. Arsenic and lead concentrations in 107 and 53 samples were greater than the associated MCL at Site G, respectively. More than 10% of arsenic and lead exceedances observed at Site G occurred at the upgradient well(s). More than 50% of the arsenic exceedances observed at Site G correspond to the wells that were downgradient or close to the unlined cell at the site. The beryllium and thallium exceedances at Site H and arsenic and lead exceedances at Site G cannot be conclusively attributed to lined cells at these sites due to the exceedances observed above the respective MCL at several upgradient wells for these parameters at these sites. The laboratory detection limit used was greater than the respective MCLs for several reported measurements (e.g., several organics at Site F, several arsenic and thallium measurements at Site C).

The arsenic exceedances at Site I have been consistently observed at several downgradient wells after closure. The site monitoring reports suggest that these elevated arsenic levels at Site I may be a result of subsurface geochemical changes. An analysis of the arsenic:chloride ratio in groundwater and leachate at Site I also suggest that leachate release, if any, is not the primary contributor of arsenic exceedances at the site. The change in the biogeochemical environment of the surficial aquifer resulting from landfill construction can also result in mobilization of naturally-occurring constituents and impacts to groundwater quality.

The available silver and zinc data were analyzed to identify exceedance with respect to the SMCL. These data were not available for all of the sites. For example, zinc and silver measurements were not available for Sites I and E, respectively. Only one site with these data had an exceedance of one of these parameters; Site G had a silver exceedance three years after closure at a single well.

It is recommended that the parameters that do not have a federal MCL/SMCL should also be monitored for a comprehensive HHE impact assessment. The site-specific background concentration and/or state-specified human-health risk-based standard can be used for assessing the risk to HHE for these parameters. As of the time of this evaluation, all three of the sites under assessment monitoring/corrective action had instances where parameters without an MCL contributed to groundwater impacts. Examples of these parameters include vanadium (Site A), 1,1-dichloroethane (Site F), and 1,4-dioxane (Site I).

9.1.6. Final Cover Performance

The final cover performance at six of the nine study sites was evaluated by analyzing available SEM and settlement data. However, a complete dataset covering the entire period from the closure date to the time of this study was only available for two of the sites (Site D and Site H).

The available data suggest that the surface emissions exceedances frequently occurred at study cells at Sites C, E, and H. The highest number of exceedances was observed over two monitoring events at Site I; wells penetrations through the final cover were documented as the primary factor contributing to these exceedances. Overall, site SEM data suggests a decreasing trend of SEM exceedances over time. The final SEM exceedance observed at five out of six sites were within ten years of closure. SEM exceedances beyond the initial ten years of PCC were only observed at one site, (Site C), and exceedances at Site C were noted throughout the three years of available data (years 16-18) potentially due to the lack of a soil layer where methane oxidation could occur over the exposed geomembrane cap.

Limited settlement data were available for only a few study sites (Sites C, E, and F), with the exception of Site I, which had annual settlement data available. The topographic data for the closure year of Site F were not available; therefore, an analysis of trends in the settlement rate for Site F was not conducted. The median annual settlement for Site I decreased from 2.4 ft in the first year after closure to 0.13 ft in the 17th year after closure. The range of the settlements observed at the survey points appears to decrease over time from 2.5 ft during the first year to approximately 0.5 ft in the 17th year after closure for Site I. The topographic surfaces at Site I were compared with bottom grade surfaces to estimate in-place waste volume for each year, which were used to estimate the annual volume loss rate. The volume loss rate was modeled using a first-order decay equation. The results of the best fit analysis suggest a total volume loss of 670,000 yd^3 over an infinite time horizon at Site I. Approximately 99% of the total settlement is estimated to occur within 30 years after closure. The settlement and volume loss estimates are regarded as overestimations due to unaccounted settlement of subsurface soils below the landfill as a result of overburden pressure.

9.2. Considerations for Assessing and Mitigating Long-term Impacts of MSWLFs

9.2.1. Operating Considerations

The risk to the human health and the environment is contingent on the contaminant(s) mass loading rate, which is a combination of the flow rate (e.g., leachate generation rate or landfill gas flow rate) and contaminant(s) concentration, into the environment (Morris and Barlaz 2011). Compromises in the integrity of the final cover would potentially result in moisture intrusion into the landfill and subsequent increase leachate collection/generation rates. The concentration of leachate constituents and landfill gas flow rate and composition are expected to be contingent on the biodegradability of the deposited waste and/or leaching potential when exposed to moisture. The possibility of future final cover compromises and ensuing emissions is one of the key concerns of PCC termination. These concerns can be alleviated by implementing operating strategies (e.g., bioreactor operation) that can stabilize waste and flush out or stabilize leachable contaminants

before terminating PCC and consequently mitigate the importance of the containment system performance in protecting the HHE after PCC termination (Morris and Barlaz 2011).

As bioreactor landfill operations enhance the waste decomposition rate and substantially reduce the time period over which waste decomposition occurs, leachate contaminants (except for contaminants such as ammonia and major ions such as chlorides) concentrations and landfill gas generation rate are relatively high during the early phases of bioreactor operation and comparatively low once the decomposition has occurred. The amount of in-place waste that can potentially degrade in the future is expected to be lower with bioreactor operation than that associated with the conventional dry tomb operation. In addition, the cap of a bioreactor landfill is expected to be less susceptible to damage from the differential settlement than a dry tomb landfill as a majority of differential settlement associated with waste decomposition has occurred before closure or during the PCC period when the site is actively monitored. Bioreactor operation is also expected to result in lower leachate management costs as some of the leachate recirculated into the landfill would be absorbed by the waste. The bioreactor operation, however, also has design, operating, and monitoring challenges including but not limited to unavailability of moisture source especially in arid areas, the complexity of adding liquids to achieve uniform moisture distribution in the landfill, flooding of gas collection devices, and a need to collect and manage excess leachate at the end of bioreactor operation (Townsend et al. 2015).

9.2.2. Monitoring Considerations

The ability for the owner and/or operator of an MSWLF unit to evaluate the HHE impacts following completion of PCC is dependent on having quality environmental monitoring data specific to the MSWLF unit of interest. For several of the study sites evaluated in this report, various categories of monitoring data were not available for analysis for this study. The following monitoring data would be useful for assessing whether or not an MSWLF unit would be protective of HHE in the event of termination or reduction of PCC:

Waste Tonnage and Composition – while waste-specific (e.g., MSW, CDD) tonnages are typically well-documented and were available for a majority of the study sites, detailed composition studies or records documenting specific incoming waste types were scarce. The waste composition study data were not available for any of the study sites except Site D. In addition, the amount and chemical characterization data of different non-MSW materials such as industrial waste, combustion residuals, and contaminated soils deposited in the landfill are valuable for HHE impact assessment.

GCCS – Typically, GCCS at MSWLFs (e.g., Site C) with active cells are progressively expanded with time to collect LFG from newly filled areas, and the LFG from several closed and active cells are routed to a blower/flare system. Although LFG flow rate and composition data are collected from each well, these are not typically tracked for individual cells. Study cell-specific LFG flow rate and composition data were not available at several sites (e.g., Sites B, C, E, and G). From a PCC monitoring standpoint, MSWLF owners/operators would benefit by installing devices that allow frequent monitoring of the flow rate and quality of LFG collected from standalone closed MSWLF units as these data would allow estimation of decay rate and the remaining methane potential of these cells.

LCS – The leachate quality (e.g., Site G) and quantity (e.g., Site B, Site E) at some of the study sites were not independently tracked for the study cell. Depending on the timing of the closure of various cells and the PCC goals, the owner/operator may consider PCC termination of standalone cells or cluster of cells while operating the other cells at the site. However, the lack of monitoring data from standalone individual or clusters of cells may preclude early PCC termination of these cells. Like LFG, leachate is typically routed to a single location (e.g., for leachate storage/pre-treatment) before transporting/pumping off-site for treatment and disposal. The study site data suggest that leachate generation/collection rate and quality may vary substantially among various collection points. MSWLFs owners/operators should consider monitoring leachate collection rates and quality from various collection points. These data would allow identification of individual cells (e.g., with elevated leachate collection rates and/or contaminant levels) that may need to be monitored and operated (e.g., cell-specific leachate recirculation) more rigorously than the other cells/areas. These data over time would also be helpful in identifying localized stormwater intrusion, if any, into LCS/leachate sump.

As leachate quality tracking at individual leachate collection points can be cost-intensive, contaminant-specific monitoring frequency should be considered. For example, field parameters (e.g., pH and specific conductivity) can be monitored more frequently than the laboratory parameters. The laboratory parameters that are frequently measured above the respective groundwater protection standards can be monitored more frequently than the parameters that are consistently undetected, thus reducing the cost burden while collecting adequate leachate quality data for a robust HHE impact assessment.

MSWLFs owners and operators should consider routine monitoring of leachate quality even though it not required by RCRA Subtitle D regulations. For a more comprehensive HHE impact assessment, MSWLF owners/operators should consider harmonizing the list of monitoring parameters for groundwater and leachate and ensuring that the laboratory reporting limits of the monitored parameters are lower than the respective groundwater protection standard. Leachate quality should be monitored for the parameters that are required for groundwater monitoring as well as for constituents that occur at elevated levels in leachate (e.g., chloride, ammonia) and those that can be used for assessing the waste stability (e.g., BOD and COD).

Groundwater Monitoring System – A specific challenge was identified when attempting to analyze groundwater impacts associated with the study cells at the sites with the presence of adjoining unlined or active cells. While isolating the groundwater impacts of adjoining cells may not be practically feasible, independent groundwater monitoring of standalone closed cells would allow identification of the sources of groundwater impacts, if any. A periodic review of changes such as surrounding land use/zoning changes that can impact the groundwater flux and flow direction should be considered while assessing the long-term impacts of modifying or terminating PCC. Monitoring of groundwater quality with respect to leachate indicator parameters such as chloride, ammonia, BOD, and COD would help to assess the contribution of leachate release, if any, on groundwater quality impacts.

Settlement – Differential settlement of the landfill surface represents one of the most probable risks to the integrity of the final cover. Routine settlement monitoring data can be used to estimate

the future settlement rate. In addition, settlement data, when used in conjunction with a temporal analysis of LFG collection and leachate quality, can provide an indication of waste stabilization. Settlement data were not available for several sites, including Sites A, D, and G. Routine topographic surveys of the final cover during PCC, including a survey of the final cover immediately following the closure, provide an opportunity to evaluate the rate of settlement and waste stabilization.

Monitoring Records – Some monitoring data (e.g., subsurface probe monitoring and SEM), which are typically required to be routinely monitored for MSWLFs, were not available for several monitoring events for review/analysis for this study. For example, SEM data were not available for review for several years for Sites C, E, and I. The state regulators and MSWLFs owners/operators should consider implementing systems for cataloging monitoring data for prompt retrieval and analysis.

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