REPORT

2019 ANNUAL REPORT

Wildwood Landfill, Powell River, BC

Submitted to:

Catalyst Paper

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19128752-003-R-Rev0

27 April 2020

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Executive Summary

Groundwater and air quality monitoring at the Wildwood Landfill in Powell River, British Columbia, continued in 2019 with monitoring programs that were carried out in accordance with Landfill permit PR-4565 (amended on 19 March 2015). Catalyst Paper Corporation operates the Landfill, which consists of the closed Phase I Landfill and an operational Landfill.

Background

The Phase 1 Landfill was operated from the 1960s until its closure in 1995 by landfilling with waste-material from the mill site. Leachate from the Phase 1 Landfill is extracted from groundwater wells and treated at the mill site. Discharge of waste-material from the mill site was conducted at the Mini-Landfill, which is contained by an engineered leachate collection system, commencing in 1996. Construction of a Phase 2 Landfill (Landfill expansion) above the Mini-Landfill and the Phase 1 Landfill was carried out in 2013 and discharge of waste to the Phase 2 Landfill commenced on 15 October 2013. During 2019, all waste was discharged to the Mini-Landfill and the Phase 2 Landfill.

Historically, groundwater downgradient of the Phase 1 Landfill has included several chemical constituents at elevated concentrations, including pH as high as 12.1, total organic carbon as high as 68,325 mg/L, chlorinated phenols as high as 0.122 mg/L, dioxins and furans as high as 26,706 pg/L, and several metals in exceedance of the standards set by the BC Contaminated Sites Regulations (CSR). Since closure of the Phase 1 Landfill and commencement of operation of the groundwater extraction system in 1995, water quality at the site has improved considerably.

Monitoring of the Mini-Landfill and Phase 2 Landfill

The total Landfill airspace consumed in 2019 was 19,580 m^3. Similar to previous monitoring years, grab samples of leachate collected by the Mini-Landfill collection system were characterized by elevated specific conductivity and pH. Leachate from the Phase 2 Landfill had a higher pH than the Mini-Landfill during the March and November sampling rounds. The specific conductivity of the leachate collected from the Phase 2 landfill was also higher than that of the Mini-landfill. The biological oxygen demand (BOD_5) continues to be significantly reduced in the Phase 2 Landfill as compared to the Mini-Landfill.

No detectable dioxin and furan total toxicity equivalent was measured in either the Mini-Landfill or the Phase 2 Landfill leachate in this monitoring year.

Analysis of leachate from the Mini-Landfill and Phase 2 Landfill for a more comprehensive suite of parameters (not required by the Landfill permit) in 2015, 2016 and 2019 indicates the Phase 2 Landfill leachate may be characterized by higher concentrations of chloride, potassium and some trace metals (aluminum, arsenic, chromium, copper, molybdenum and vanadium) than the Mini-Landfill leachate. Although, there is some uncertainty in the chloride results for the Phase 2 Landfill because of the large disparity in the 2019 results relative to the 2015 results; therefore additional monitoring of the leachate will be required to resolve the chloride fingerprint of the Phase 2 Landfill. Most of these constituents have been found previously at similar or higher concentrations in the Phase 1 leachate at the south edge of the Landfill. Exceptions are potassium, molybdenum, and in particular chloride, which is significantly higher in the Phase 2 Landfill (chloride concentration of 2,920 mg/L to 11,900 mg/L) than the Phase 1 Landfill (89 mg/L) and Mini-Landfill (1560 mg/L to 4700 mg/L). As a result, chloride is considered a representative indicator parameter for the Phase 2 Landfill.

Leachate was detected beneath the Mini-Landfill liner over the period of 4 January through 6 May and a sample was collected on 27 March 2019 for analysis in accordance with the Landfill permit.

Groundwater Monitoring

A total volume of 1,321 m^3 of groundwater was removed from the Wildwood Landfill's groundwater extraction system in 2019. The total volume of water extracted is approximately 50% of the water extracted in 2018 (2,541 m^3) due to mechanical/electrical issues with PW99-5 which resulted in the pump being offline for approximately 7 months of 2019. However, of the total extraction rate, approximately 99% was extracted from pumping well PW99-5, completed in the 29 m flow zone. This well, together with PW99-2 in the 19 m flow zone, appears to have aided in the partial dewatering of the upper two flow zones. The well rehabilitation program conducted in March 2014 appears to have resulted in a sustained increase in pumping rate at PW99-5. Groundwater recovered by the extraction system was below the Contaminated Sites Regulation (CSR) for all constituents.

Monitoring of groundwater levels at the entire network of 85 wells at the Wildwood Landfill site indicates that groundwater at the site generally flows toward the southeast, toward Powell River. Flow patterns through several perched groundwater flow zones, the regional flow zone, and the bedrock flow zone in 2019 were generally similar to previous years.

Overall, the chemistry results from groundwater and surface water monitoring in 2019 were similar to previous years. The in-operation of PW99-5 over 7 months of 2019 had not caused any apparent changes in the chemical concentrations in the downgradient groundwater in 2019.

Phenols remained at non-detectable or low-levels, below applicable standards, at both monitoring locations (AH3 and AH-6L). Dioxins and furans were found at concentrations considered to be distinguishable from the laboratory method blanks at 89-5 (south edge of the Landfill) and AH-6L (immediately downgradient of the Landfill) in November 2019, with the exception of 89-5 and 94-19L where only a single parameter (Total-PeCDD) is distinguishable from the laboratory method blank. Some metals concentrations continue to exceed CSR standards immediately downgradient of the Landfill and at the leading edge of the Landfill, with monitoring wells AH-6L and 89-5 representing the most impacted locations.

In groundwater at the south edge of the Landfill at 89-5, an increase in alkalinity, chloride, sodium and associated specific conductivity has been apparent in recent years, with a marked rise in chloride observed in 2018 followed by a relatively minor decline in 2019. Because leachate from the Phase 2 Landfill may be characterized by elevated chloride relative to the Phase 1 Landfill, chloride concentrations in downgradient groundwater will be monitored closely in future monitoring events. There is no evidence to date to suggest that the Phase 2 Landfill has impacted groundwater quality further downgradient.

Water samples from Springs S1 and S2, which represent water downgradient of the Landfill near the closest point of discharge, met the CSR standards.

The results of the 2019 groundwater sampling program confirm that the original capping and leachate collection controls, together with the enhanced leachate collection measures implemented during the 2000 monitoring year, have resulted in the reduction of contaminants downgradient of the Landfill.

Dustfall and Air Quality Monitoring

In the summer and fall of 2019 (July through September 2019), Catalyst Paper Ltd. (Catalyst) conducted a dustfall monitoring program in compliance with the amended permit requirements for the Wildwood Landfill. A laboratory error in preparing the algaecide for the dustfall sampling program resulted in the distorted dustfall results during the 2019 program and considerably reduced confidence in the measurements. Because the sampling program did not yield trustworthy data, the 2019 dustfall monitoring data has not been presented in this report.

In 2019, sampling of particulate matter 2.5 microns and smaller ($PM_{2.5}$) and particulate matter 10 microns and smaller (PM_{10}) was conducted at the Wildwood School from 1 January 2019 through 31 December 2019. The TEOMs at the Wildwood School air quality monitoring station were replaced in January 2019 with an API T640 that measures both $PM_{2.5}$ and PM_{10}. Concentrations measured at the Wildwood School remained well below all applicable air quality objectives over the 2019 monitoring period. The forest fire season was less active in 2019 and thus elevated concentrations that would typically be observed were not seen in summer and early fall of 2019.

Discrete sampling of PM_{10} was also conducted at the Landfill in 2019 using a high-volume (Hi-Vol) sampler. The average PM_{10} results from Hi-Vol sampling program at the Landfill remained well below the BCO at approximately 9% of the BCO.

Recommendations

It is recommended that the groundwater and air quality monitoring program for the Wildwood Landfill continue for 2020. Amendments in the addendum letter to the Landfill permit allow for the cessation of PM_{10} monitoring at Wildwood School following favourable results in 2019. Additional recommendations for the 2020 monitoring year are presented in Section 7.0.

Study Limitations

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To prepare this report, we have relied in good faith on information provided by others as noted. We assume that the information provided is factual and accurate. We accept no responsibility for any deficiency, misstatement or inaccuracy contained in this report as a result of omissions, misinterpretations or fraudulent acts of persons interviewed or contacted.

The services performed as described in this report were conducted in a manner consistent with the level of care and skill normally exercised by other members of the engineering and science professions currently practising under similar conditions, subject to the time limits and financial and physical constraints applicable to the services. The content of this report is based on information collected during our monitoring program, our present understanding of site conditions, the assumptions stated in this report, and our professional judgement in light of such information at the time of this report. This report provides a professional opinion and, therefore, no warranty is expressed, implied, or made as to the conclusions, advice and recommendations offered in this report. This report does not provide a legal opinion regarding compliance with applicable laws. With respect to regulatory compliance issues, it should be noted that regulatory statutes and the interpretation of regulatory statutes are subject to change. The findings and conclusions of this report are valid only as of the date of the report. If new information is discovered in future work, or if the assumptions stated in this report are not met, Golder Associates Ltd. should be requested to re-evaluate the conclusions of this report, and to provide amendments as required.

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1.0 INTRODUCTION

This report presents the results of groundwater and air quality monitoring conducted at the Catalyst Paper Wildwood Landfill (Landfill) over the 2019 monitoring year. The Wildwood Landfill receives waste from the Catalyst Paper mill site located in Powell River, BC.

The 2019 monitoring program was carried out in accordance with the requirements outlined in the Landfill permit (PR-4565), which was amended by the BC Ministry of Environment & Climate Change Strategy (ENV) on 19 March 2015. A copy of the amended permit is provided in Appendix A. A summary of the contents of this report relative to the reporting requirements specified by the Landfill permit is presented below in Table 1.

Table 1: Summary of Reporting Compliance with Landfill Permit

Section of Permit	Permit Requirement	Report Reference
Section 3.7	Remaining design capacity of the Landfill (uncompacted cubic metres) and an estimation of closure date at the current rate of disposal	Section 4.2
Section 2.8	Efforts in waste reduction and alternative disposal	Section 5.1.1
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A summary of the scope of work carried out in 2019 in excess of the permit requirements, together with the associated sections of this report where these results are presented, is summarized in Table 2.

Table 2: Scope of Work and Reporting in Excess of Permit Requirements

Scope of Work	Report Reference
Sampling and analysis for halogens (Cl, F and Br) at all available monitoring locations for assessment purposes	Section 6.4.2.9
Comprehensive trend analysis (since the start of monitoring) of groundwater and surface water results	Section 6.4

2.0 BACKGROUND

A variety of waste-material from the Powell River mill site, including flyash from a power boiler, was placed in a sand and gravel quarry, beginning in the 1960s, to create the Wildwood Landfill. The location of the Landfill is shown in Figure 1 and site plans are presented in Figures 2 and 3. MacMillan Bloedel Ltd. owned and operated the Landfill from the 1960s until its closure in 1995. The closed portion of the Landfill is referred to as the Phase 1 Landfill. In January 1996, a new, smaller Landfill (referred to as the “Mini-Landfill”) was established within the footprint (on the northeast corner) of the older Landfill (Figure 3). In June 1998, MB Paper Ltd. sold the pulp and paper mill located in Powell River, BC to Pacifica Papers Inc. Pacifica Papers Inc. merged with Norske Skog Canada on 27 August 2001, forming a new company, NorskeCanada. In October of 2005, NorskeCanada changed its name to Catalyst Paper Corporation (Catalyst). The Wildwood Landfill has been included in all of these corporate transactions.

Closure of the Phase 1 Landfill consisted of covering the Landfill with a low-permeability asphalt cap, to minimize recharge from precipitation, and installation of a leachate collection system. The asphalt cap covers an area of approximately 4 hectares immediately south of the Mini-Landfill. The leachate collection system initially consisted of three pumping wells located within the uppermost perched groundwater flow system (PW95-1, PW95-8 and PW95-9) along the south edge of the Landfill (Figure 3). Three additional pumping wells (PW99-2, PW99-4 and PW99-5) were installed to greater depths along the east edge of the Landfill in 1999 (and became operational in March 2000) to intercept leachate that may have been by-passing the original collection system (Figure 3). The groundwater recovered by these wells is conveyed by pipeline to the Catalyst Paper mill site wastewater treatment plant, where it is treated along with the mill’s wastewater prior to discharge.

Waste from the mill site was placed in the Mini-Landfill commencing on 27 August 1996. The Mini-Landfill is lined with a HDPE geomembrane and soil bentonite liner and has provisions for leachate detection and collection.

Construction of a Phase 2 Landfill (Landfill expansion) above the Mini-Landfill and the Phase I Landfill was carried out in 2013. The Landfill expansion consists of a 1.5 mm (60 mil) HDPE geomembrane liner above the asphalt cap of the Phase I Landfill, an engineered leachate collection system, a below-liner drainage system, and a lined pond (Active Pond) for receipt of leachate runoff (Figure 3). Waste types permitted under the Environmental Management Act (Permit PR-4565) include flyash and a minor proportion of spent bed sand, miscellaneous mill waste and waste asbestos. Discharge of waste to the Phase 2 (expanded) Landfill commenced on 15 October 2013. During the 2019 monitoring year, all waste was discharged to the Mini-Landfill and Phase 2 Landfill.

3.0 GEOLOGY AND HYDROGEOLOGY

3.1 Geological Setting

The Wildwood Landfill is located at the southeastern edge of a coastal terrace, near the top of a minor watershed, approximately 700 m by 700 m, which drains toward Powell River. This minor watershed consists of a bedrock valley filled with deposits of gravel, sand, and silt. The ground surface slopes downward (at about 4 degrees) in a southeast direction from the southeast edge of the Landfill to Powell River. The portion of the hillside immediately above Powell River is a steep bank with a slope on the order of 30 degrees. The northwest edge of the Landfill appears to represent a saddle point in the surface topography, with the ground surface sloping downwards towards the west and southwest at some distance northwest of the Landfill (Figure 2).

Prior to operation, the Landfill site was a sand and gravel quarry. Inspection of the 6-metre high western quarry wall in 1977 revealed large-scale cross-stratification and at least one fine-grained marine sediment layer (Pit No. 36, McCammon, 1977). The following interpretations are inferred from borehole logs recorded during the drilling investigations at the site. Sediment thickness, from the present surface to the top of the underlying bedrock, ranges from approximately 20 m to 60 m. Several stratigraphic sequences are bedded approximately parallel to the present topography. These sequences appear to extend over large portions of the site, but some units may pinch-out locally.

The sediments are likely to have been deposited in the last c. 125,000 years during one or more series of glacial activity, glacial melt, and sea inundation associated with icecap isostasy. While specific stratigraphic units have not been identified at the site, deposits of the Fraser Glaciation (c. 10,000 to 28,000 ya), possibly including the Coquitlam Sequence and Vashon Sequence, are likely to compose the sediment fill atop the bedrock (Ryder et al., 1991). The steep slope southeast of the Landfill and adjacent to Powell Lake may have formed as the river cut through much of the glacial sediments during isostatic rebound following melting of the icecap. Some recent colluvium and soils overlie the glacial sediments.

The area occupied by the Landfill is bounded to the west and east by ridges of granitic bedrock, with exposed bedrock along the southwest edge of the Landfill. Beneath the site, the bedrock surface slopes in a southeast direction from an elevation of approximately 100 m above sea level at the south edge of the Landfill to an elevation of about 60 m above sea level just before the break in slope above the shoreline.

3.2 Hydrogeology

A conceptual hydrogeological model is provided in Figure 4 as a schematic cross-section through the site. A medium sand to sand and gravel layer is present above the bedrock surface. Regional groundwater flows in a southeast direction at the Landfill site within the sand unit and the upper, fractured zone of the bedrock. The thickness of the saturated zone above the bedrock is inferred to range between 5 m and 10 m in the central portion of the Landfill to between 2 m and 20 m downgradient of the Landfill. Groundwater within the regional zone is estimated to flow under an average gradient of 0.011, resulting in a velocity on the order of 1 m/day. Hydraulic conductivity of the regional flow zone has been estimated to be approximately 4×10^{-4} m/s from a single-well response test (Golder Associates Ltd., 2007).

Groundwater is inferred to discharge from the upper portion of the regional aquifer to springs located on the slope above Powell River and likely also directly to the river by seepage beneath the water level in the river (i.e., underflow). Groundwater within bedrock is inferred to discharge as underflow to Powell River.

The regional aquifer is overlain by a sequence of sands and gravels that contain layers of clay, silt and fine sand ranging from 0.1 m to 1.5 m in thickness. The presence of these stratigraphic units has created perched groundwater conditions. Perched aquifers have been identified beneath the southern crest of the Landfill above fine-grained sediment layers at depths of approximately 11 m, 19 m, 29 m and 38 m below ground surface. Groundwater is inferred to move laterally in the aquifers and vertically at any boundaries of the perched aquifers, as illustrated in the schematic in Figure 4. Hydraulic conductivity in the perched flow zones is estimated to range from approximately 3×10^{-7} m/s to 2×10^{-4} m/s, with a mean of 2×10^{-5} m/s, based on single well response tests and pumping tests at eight wells (Golder Associates Ltd., 2007).

Based on this mean value and the hydraulic gradient at the site, groundwater flow through the perched aquifers in the region of the Landfill is estimated to be between approximately 5 m^3/day and 20 m^3/d. Some groundwater from the shallower perched zones may discharge to the steep slope above the spring zone.

4.0 DESIGN AND OPERATING PLAN

4.1 Changes to the Operating Plan in 2019

The latest Design and Operating (D&O) Plan for the Wildwood Landfill is dated 26 November 2012 and the amended Landfill Permit 4565 is dated 19 March 2015.

According to the survey by Polaris Land Surveying Inc. on 29 January 2020, waste placement in 2019 appeared to have occurred across the entire Landfill.

A site visit was carried out in the afternoon of 29 October 2019 by Colin Wong, P.Eng. of Golder to observe Landfill conditions in relation to the Design and Operating Plan. Holes in the exposed 2.5 mm thick high-density polyethylene (HDPE) geomembrane liner for the pond were evident. It appears that one or more mammals, likely a bear, bit into the liner. Recommendations to repair the liner and to prevent entry of animals to the area were provided in Golder's letter dated 1 November 2019.

Based on observations during the site visit, it is Golder's opinion that landfilling was observed to be generally consistent with good landfill operational practice. No visual evidence of Landfill instability was observed.

There is a considerable amount of sand being discharged to the Landfill. Apparently, vegetation is growing in the sand that has been placed on the north slope. Due to the large quantity of sand, the operating plan for the Landfill is being updated and will be issued as a standalone report.

4.2 Landfill Capacity and Closure Plan

The capacity of the Landfill authorized by the 19 March 2015 amended permit is 620,000 m^3, with a maximum rate of discharge of 25,000 m^3/year. In 2019, this capacity was accommodated by discharging waste to the Phase 2 Landfill and Mini-Landfill.

Catalyst has estimated that up to 22,000 m^3 of flyash and 500 m^3 of miscellaneous mill waste could be disposed of annually in the Landfill and expects that this rate of disposal may remain constant over the next few years. The total volume (waste and associated cover material) placed over the period of 7 January 2019 through 29 January 2020, based on Polaris Land Surveying's annual quantity survey, was 19,580 m^3 (Polaris Land Surveying site plan, provided in Appendix B). The remaining design capacity of the Landfill, as of the end of 2019, is estimated to be approximately 285,000 or 320,000 m^3 (including the volumes of waste and a 0.75 m thick final cover), depending on the location selected for the truck wash. The projected closure date of the Landfill is estimated to be approximately 2032 or 2034, assuming that the annual tonnage of waste entering the Landfill in and after 2019 will be the same as the average for 2014 to 2019 of 19,028 m^3 per year.

The estimated cost of future closure, post-closure, and other regulatory requirements was reviewed by Catalyst and the accrued obligation on Catalyst's books on 31 December 2019 was approximately $715,517.

The closure plan for the Landfill is being updated and will be included in a standalone report.

5.0 SCOPE OF WORK AND FIELD METHODS

5.1 Monitoring of Mill Refuse in 2019

5.1.1 Alternate Disposal: Assessment of Flyash Utilization Options

In the 2018 monitoring year, laboratory analysis was conducted on the spent bed sand that is currently being disposed of at the Landfill to determine, based on its chemistry, whether the sand could be diverted from the Landfill. Further testing of the spent bed sand is scheduled for 2020.

Efforts undertaken to assess options for alternative flyash disposal prior to 2009 were presented in the July 2007 Environmental Assessment report and the 2008 Annual report. Options that were the subject of past investigations for flyash utilization are listed below:

- Research on the ability of flyash to oxidize hydrogen sulphide from sour gas wells is being undertaken by the Northern Alberta Institute of Technology.
- Flyash (and spent sand) was sent to FPInnovations, a research project funded by the Federal Government to look at alternative uses for flyash waste.
- Discussions with Natural Resources Canada regarding the potential for application of flyash on forested soils.
- Learning about alternative use of ash at Catalyst's sister mill in Rumford, Maine.
- Commercial use as a concrete additive.
- Participation in the BC bio economy network on the commercialization of forest-based materials.
- Use as a soil amendment or fuel additive for markets available to Sliammon.
- Use as a supplement in fertilizer.
- Use at Site C by BC Hydro.
- Use as a catalyst in the production of biofuel.
- Use in the ocean for growing kelp.
- Use in the manufacture of lock blocks.
- Uses with Target.
- Use as a soil conditioner.
- Use for construction purposes.
- Liming agent and fertilizer in agriculture.
- Backfill material for oil wells.
- Additive to a patented composite binder.
- Additive to building bricks.
- Construction of an artificial reef.
- Solidification of mine tailings.

5.1.2 Efforts to Minimize Leachate Formation

Efforts to minimize leachate formation during the 2019 monitoring year were consistent with previous years. Specifically, care was taken not to use excessive amounts of water when preparing the water/flyash slurry and when washing the mixer trucks on-site. In addition, Catalyst reports that the minimum one metre freeboard in the active pond was met during the current reporting year, as well as all prior years going back to the expansion.

5.1.3 Discharge Monitoring: Mini-Landfill

5.1.3.1 Discharge Types and Quantities

Catalyst recorded each truck load of mill refuse that was discharged to the Mini-Landfill and Phase 2 Landfill in 2019 and recorded the types and quantities of material. Authorized types of refuse include flyash, spent bed sand, waste asbestos, and a maximum of 3,000 m^3 per year of miscellaneous waste. Under the amended (19 March 2015) Landfill permit, the maximum rate of discharge to the Landfill authorized is 25,000 m^3 per year and the total volume permitted is 620,000 m^3. Results of the discharge monitoring are presented in Section 6.2.

The remaining capacity of the Landfill was monitored by conducting a land survey.

5.1.3.2 Leachate Monitoring

Leachate draining from the Mini-Landfill is collected by an engineered collection system located on top of an HDPE liner and is conveyed to the mill via pipeline for treatment prior to discharge. Leachate draining from the Phase 2 Landfill is also collected by an engineered collection system located on top of a HDPE liner. The leachate collected from this system is conveyed to a pipeline that runs down the slope south of the Phase 2 Landfill and discharges into the Active Pond. The liquid in the pond then flows into the leachate conveyance pipeline which transports the liquid to the mill for treatment prior to discharge.

In accordance with the Landfill permit, samples of the leachate from both the Mini-Landfill and the Phase 2 Landfill were collected by Catalyst during the months of March (Phase 2 and Mini-landfill respectively) and November 2019 and were analyzed for pH, specific conductance, and biological oxygen demand (BOD_5) by Catalyst Paper's laboratory. The March samples were also analyzed for dioxins and furans by Pacific Rim Laboratories Ltd. The collection system includes a leak detection monitoring port, which is monitored by Catalyst. A sample of leachate from the leakage detection system of the Mini-Landfill was collected by Catalyst in March 2019 and analysed for pH, specific conductance, and biological oxygen demand (BOD_5) as well as Dioxin Furans by Catalyst Paper's laboratory. Results of the leachate monitoring are presented in Section 6.2.

5.2 Groundwater Monitoring Program

The 19 March 2015 amended permit for active waste disposal at the Mini-Landfill and Phase 2 Landfill requires the following groundwater monitoring activities be carried out at the Wildwood Landfill site:

- Collection of groundwater samples from at least six groundwater monitoring wells and at least six groundwater pumping wells (when sufficient water is available).
- Monitoring for leachate indicator parameters ("snap-shot" sampling) at 14 monitoring locations on an annual basis.

In addition to the monitoring outlined above, Catalyst Paper monitors 1) the volumes of groundwater removed from the pumping wells on a weekly basis, and 2) water samples from the active pumping wells for field indicator parameters on a monthly basis.

A summary of the 2019 approved groundwater program, based on the 19 March 2015 amended permit, is provided in Table 3.

Table 3: Summary of Groundwater Monitoring Program

Sample Location	Dry Weather Monitoring Event (September 2019)	Wet Weather Monitoring Event (November 2019)
MW 93-2B (upgradient)	GP	GP, D. MET, D&F
MW 89-5	GP	GP, D. MET, D&F, S
MW AH-3	GP	GP, CP, D. MET, D&F
MW AH-6L	GP	GP, CP, D. MET, D&F
MW 94-1/3	GP	GP, D. MET, D&F
MW 94-16B	GP	GP, D. MET, D&F
MW94-19L	GP	GP, D. MET, D&F
PW-Composite (includes PW95-1, PW95-8, PW95-9, PW99-2, PW99-4 and PW99-5)	GP	GP, D. MET, D&F, S
Springs S1	GP, D&F	GP, T. MET, D&F
Spring S2	GP, D&F	GP, T.MET, D&F

Notes:

Field parameters monitored at all Sample Locations during each Monitoring Event: Specific Conductance (Electrical Conductivity at 25ºC), pH, Oxidation-Reduction Potential (ORP), and dissolved oxygen.

GP (General Parameters) = Total Suspended Solids (TSS), Total Dissolved Solids (TDS), total alkalinity, hardness, bromide, chloride, fluoride, ammonia, nitrate, nitrite, sulfate, Total Organic Carbon (TOC). Sampling for bromide, chloride, fluoride, ammonia, nitrate and nitrite are not required by the Landfill permit.

S = Sulfide

T.Met = Total Metals

D.Met = Dissolved Metals

CP = Chlorinated/non-chlorinated Phenolics

D&F = Dioxin & Furans (includes selected congeners)

Snapshot Wells: 93-2B; 94-16U, 94-16M, 94-16L, 94-16B, 94-17L, 94-17B, 94-18U, 94-18M, 94-18L, 94-19U, 94-19M, 94-19L, 94-19B; Snap-shot analysis: pH, specific conductivity, alkalinity, temperature, redox potential, dissolved oxygen and sulfate.

* carried out for quality assurance/quality control purposes and not required by Landfill permit.

The scope of the 2019 program also included monitoring of halogens (Cl, F, and Br) and nitrogen species (ammonia, nitrate and nitrite) at all sampling locations for assessment purposes to allow a comparison with the BC Water Quality Guidelines.

Groundwater sampling is conducted on two occasions during the year; a dry weather sampling event between July and September and a wet weather sampling event in November or December. During each sampling event, the water level is measured in each of the 85 monitoring points of the monitoring network to determine the direction of groundwater flow at the site. These 85 monitoring points are located at 55 unique well locations that include multilevel and nested wells. Locations listed in Table 3 consist of two groundwater seeps (Spring S1 and S2) and a set of monitoring wells selected from the network of wells. The selected monitoring wells aid in analysis of (1) the groundwater quality between the Landfill and the Wildwood residential community to the north and the freshwater aquatic environment of Powell River to the South; (2) the performance of the groundwater extraction system; and (3) the geochemical evolution of groundwater flowing towards Powell River.

Geochemical analyses include monitoring of metals, phenolics, and dioxins and furans. General geochemical parameters such as total dissolved solids and alkalinity, as well as major anions and common halogens (including chloride), are monitored to allow an assessment of the evolution of the geochemistry at the site. A full suite of analyses is done annually on samples from one well located north of the Landfill, four monitoring wells and the active extraction wells located immediately south of the Landfill, two monitoring wells near the shoreline of Powell River, and two groundwater seeps near the shoreline of Powell River.

An annual “snap-shot” sampling event is conducted to monitor the indicator parameters in one well located north of the Landfill and an additional 14 wells[1] located at various depths near the shoreline.

Groundwater at the site consists of moisture in unsaturated soil, perched water tables located at several depths, a regional aquifer, and a bedrock aquifer. Groundwater flow zones have been defined, based on sediment logs prepared from notes taken during drilling for monitoring well installations and from the historical water level measurements obtained at wells throughout the site. Sample locations have been selected based on the conceptual model for groundwater flow discussed in Section 3.2. A summary of the sample locations and their respective groundwater flow zones is presented in Table 4, and the locations are shown in Figure 3.

Table 4: Sample Locations and Associated Groundwater Flow Zones

Groundwater Flow Zone	Sample Location	Pumping Wells
11 m Perched Zone	AH-3, 89-5, 94-16U	
19 m Perched Zone		
29 m Perched Zone	AH6L	PW99-2, PW99-5
38 m Zone	94-1/3, 94-16M, 94-18M, 94-18U, 94-19M, 94-19U, Spring S1, Spring S2	
Regional Aquifer	93-2B (upgradient), 94-16L, 94-17L, 94-18L, 94-19L	
Bedrock Aquifer	94-16B, 94-19B, 94-17B	

[1] Of the 14 “snap-shot” locations, complete snap-shot data were obtained from 12 locations because 94-16U was dry, 94-16M could not be sampled due to an obstruction in the well.

During each monitoring event, groundwater levels are measured at all monitoring and pumping wells (the entire 85-well network), using an electronic water-level meter. Totalizer flow meters monitor the discharge rate of the water pumped from the extraction wells. The totalized discharge is recorded weekly, and the electrical conductivity and pH of the discharge is recorded monthly, where available.

Prior to sampling, monitoring wells were purged of three standing well volumes with dedicated lengths of Waterra^{TM} tubing. Field indicator parameters (pH, specific conductivity, temperature, dissolved oxygen and redox potential) were measured in the field at the time of sampling and documented on standard field sampling forms for each location (copies of the forms are presented in Appendix D). Grab samples were obtained from the pumping wells, and surface water samples were collected from discharging seepage faces at their inflows to Powell River(Springs S1 and S2).

Laboratory analyses were carried out by ALS Environmental. Quality control measures were taken during the sampling events and laboratory analyses conducted in 2019 to assure that the methods provided results that may be incorporated into analytical interpretations. A duplicate sample was taken in September from 94-19L for analysis of general parameters and a duplicate sample was taken in November from AH-6L for analysis of general parameters, and dioxins and furans.

5.3 Dustfall and Particulate Monitoring

Catalyst conducted an ambient dustfall monitoring program to determine dust deposition rates adjacent to the Wildwood Landfill. The methods and results of the dustfall monitoring are discussed in Appendix G. The dustfall sampling devices were placed in the field for three sampling periods, with each period lasting approximately 30 days (i.e., July, August and September). Dustfall samplers were installed at four separate locations on the northwestern and northeastern boundaries of the Landfill. A laboratory error in preparing the algaecide for the dustfall sampling program resulted in the distorted dustfall results during the 2019 program and considerably reduced confidence in the measurements. Because the dustfall data are not considered accurate or helpful from the 2019 sampling effort they are not presented in the annual report.

Sampling of particulate matter with a mean aerodynamic diameter less than 2.5 microns ($PM_{2.5}$) and particulate matter with a mean aerodynamic diameter less than 10 microns (PM_{10}) was conducted at the Wildwood School from 1 January 2019 through 31 December 2019. Continuous particulate instruments were updated in January 2019 from Tapered Element Oscillating Microbalance (TEOM) units to a singular Teledyne API T640, which monitors both $PM_{2.5}$ and PM_{10}. The $PM_{2.5}$ TEOM continues to remain operational at Wildwood School for co-location monitoring at the Ministry's request.

Sampling of PM_{10} was also conducted at the Mini-Landfill using a high-volume (Hi-Vol) sampler. Samples were collected over a 24-hour period once every 6 days throughout 2019.

5.4 Landfill Settlement

In 2013, three slope inclinometers were installed across the base of the Landfill expansion, three surface monuments were installed on the asphalt cover along the toe of the Landfill expansion, and five settlement gauges were installed on the slope south of the Landfill expansion to monitor settlement related to operations of the Phase 2 Landfill. These monitors were first surveyed on 27 January 2014 to provide baseline measurements, and have been re-surveyed annually since then to allow an assessment of Landfill settlement and slope stability. Table 5 summarizes the elevations of these settlement monitors measured by Emery and Rae Land Surveying Ltd. in 2014 and 2015; and by Polaris Land Surveying Inc. in 2019 and 2020. A copy of the most recent survey plan (by Polaris Land Surveying Inc.) is provided in Appendix B.

Table 5: Settlement Monitoring Results

Location	Settlement Monitor ID	Description	Elevation (m)				Settlement 2014-2020 (m)
			27 Jan 2014	6 Jan 2015	7 Jan 2019	29 Jan 2020	
W Toe	10106	Surface monument	135.157	135.158	135.158	135.155	0.002
W Toe	10105	Surface monument	134.423	134.425	134.419	134.420	0.003
East	10007	Surface monument	133.254	133.260	133.246	133.247	0.007
LF	10107	Slope inclinometer	140.757	140.752	No data	No data	0.007*
LF	10108	Slope inclinometer	138.687	138.679	No data	No data	0.011*
LF	10109	Slope inclinometer	137.424	137.420	No data	No data	0.011*
SE Slope	10008	Settlement gauge	131.108	131.107	131.100	131.093	0.015
SE Slope	10009	Settlement gauge	127.761	127.761	127.752	127.745	0.016
SE Slope	10010	Settlement gauge	125.648	125.646	125.638	125.631	0.017
SE Slope	10011	Settlement gauge	123.692	123.690	123.681	123.674	0.018
SE Slope	10012	Settlement gauge	121.209	121.206	121.200	121.194	0.015

Note: * Settlements at slope inclinometer locations 10107, 10108, and 10109 were between January 2014 and January 2016. No survey data at these locations have been available since 2017.

The settlement monitoring over the past five years can be summarized as follows:

- The Landfill experienced 7 to 11 mm of settlement at the inclinometer locations from January 2014 to January 2016. Survey data at these inclinometer locations have not been available since 2017.
- Settlements of 10 mm or less have occurred at the monitoring stations along the Landfill perimeter.
- Settlements of 18 mm or less have occurred at the settlement monitoring points on the southeast slope of the old Landfill. It is noted that these settlement monitoring points on the southeast slope of the old Landfill had apparently uniform settlements of 6 to 7 mm between January 2019 and January 2020, which suggests that the apparent settlements in this one year interval may be due to survey tolerances.

Based on the settlement monitoring results to date and the results of our 29 October 2019 site visit, it is Golder's opinion that the Landfill is geotechnically stable with respect to potential failure surfaces through the foundation of the Landfill.

6.0 RESULTS OF 2019 MONITORING PROGRAM

6.1 Precipitation

Total precipitation at the Wildwood Landfill in 2019 was estimated to be 802 mm, based on measurements at the Powell River weather station, located 8 km to the southeast of the Landfill (Environment Canada, 2019). Precipitation in 2019 was 23% below the mean annual value (1045 mm), as calculated from measurements at the station from 1924 through 2019 (Figure C-1). Precipitation during the majority of individual months were well below the mean value for those months, with the exception of January, April, July, and September (Figure C-2).

6.2 Monitoring of the Mini-Landfill and Phase 2 Landfill

6.2.1 Discharge Quantities

Catalyst Paper provides the types and volumes (based on truck counts) of uncompacted refuse discharged to the Mini-Landfill and Phase 2 Landfill, and these are summarized in Table B-1, through to the end of 2019. A total volume of approximately 19,620 m^3 of refuse material was placed in the Landfill during the 2019 monitoring year, the majority of which was flyash (15,570 m^3). Miscellaneous waste accounted for 4,018 m^3 of the material deposited which included sand while asbestos accounted for the remaining 32 m^3 of the material deposited.

Based on surveys carried out by Polaris Land Surveying Inc., the total Landfill airspace consumed between 7 January 2019 and 29 January 2020 was 19,580 m^3 and the total volume of waste and associated cover material that has been deposited in the Mini-Landfill and Phase 2 Landfill since the start of landfilling on 27 August 1996, is 316,500 m^3 (Appendix B). For only 2019, the airspace consumed is estimated to have been 18,074 m^3. Refuse was discharged in 2019 as part of a vertical expansion of the Phase 2 Landfill. The total volume of Landfill material tracked by continuous recordings of each truckload is estimated to be 287,349 m^3 (Table B-1) to 29 January 2020. The difference between the volume reported by Catalyst paper and the volume based on the survey is likely due to consolidation of the waste.

6.2.2 Leachate from the Mini-Landfill and Phase 2 Landfill

In accordance with the monitoring requirements, leachate from the collection systems of the Mini-Landfill and Phase 2 Landfill cells was sampled by Catalyst in March and November 2019. The results from the 2019 analyses of pH, specific conductance, BOD_5, and dioxins and furans are summarized in Table 6A for the Mini-Landfill and Table 6B for the Phase 2 Landfill. The historical and 2019 results for both cells are presented graphically in Figure 5.

In addition, leachate from the Mini-Landfill leak detection system was collected in March 2019 and analysed for the same constituents (pH, specific conductance, BOD_5, and dioxins and furans). The results are discussed below in Section 6.2.3.

Table 6A: Analysis of Leachate from the Mini-Landfill in 2019

Date of Sample Collection	pH	Specific Conductance (µS/cm)	BOD_5 (mg/L)	Dioxin/Furan	
				NATO TEQ (DL=0) (pg/L)	WHO TEQ (DL=0) (pg/L)
27 March 2019	11.63	7150	56.52	0	0
27 November 2019	10.35	6645	65.90	-	-

Table 6B: Analysis of Leachate from the Phase 2 Landfill in 2019

Date of Sample Collection	pH	Specific Conductance (µS/cm)	BOD_5 (mg/L)	Dioxin/Furan	
				NATO TEQ (DL=0) (pg/L)	WHO TEQ (DL=0) (pg/L)
27 March 2019	12.26	17150	1.17	0	0
27 November 2019	12.11	15410	1.55	-	-

The pH of the leachate from the Mini-Landfill measured in March 2019 was higher than the 2018 levels, however measurements in November 2019 was consistent with the 2018 levels. Results from 2017 to 2019 were higher than previous years. The pH of the leachate from the Phase 2 Landfill measured in March and November 2019 were both lower than the previous November 2018 measurement (Figure 5).

The specific conductance of the leachate from the Mini-Landfill was consistent with historical measurements in both March and November 2019. The specific conductance of the leachate collected from the Phase 2 Landfill in both March and November 2019 was within previous historical ranges (Figure 5). These results suggest that the February 2018 spike in specific conductivity was anomalous.

BOD_5, the biological oxygen demand, is the amount of dissolved oxygen consumed in five days by the microbiological decomposition of organic matter. The BOD_5 results for the Mini-Landfill leachate (56.5 mg/L and 65.9 mg/L in 2019) indicate that the leachate contains dissolved organic matter that could be decomposed by microbes. Historical trends in BOD_5 (Figure 5) show that it increased in the early 2000s, reaching a peak of 704.8 mg/L in 2004, and subsequently declined, possibly due to a decrease in disposal of biodegradable material in the Mini-Landfill and because there was relatively very little deposition of flyash compared to miscellaneous waste in the early 2000s (Table B-1). Flyash consists of approximately 1% organic carbon, and leachate extracted from a flyash sample in the laboratory had total organic carbon concentration of 0.50 mg/L (Golder Associates, 2007). BOD_5 results for the Phase 2 Landfill, since testing began in 2014, are one to two orders of magnitude lower than the results for the Mini-Landfill. These results suggest that there is very little biodegradable material in the Phase 2 Landfill in comparison to the Mini-Landfill.

No detectable dioxins and furans were measured in the leachates from the Mini-Landfill or the Phase 2 (expanded) Landfill in 2019.

In addition to the leachate sampling and analysis undertaken as part of the Landfill permitting requirements (described above), additional characterization of the leachate has been undertaken as a means of "fingerprinting" the leachate derived from the three parts of the Landfill (i.e., the closed Phase 1 Landfill, the Mini-Landfill, and the Phase 2 Landfill). Golder collected leachate samples from the Phase 2 Landfill and Mini-Landfill on 26 November 2019 for the analysis of dissolved chloride, dissolved metals, alkalinity, sulphate, nitrate, and total dissolved solids (TDS). Results of these samples were compared to historical data (Table C-2, Appendix C).

A comparison of the Phase 2 leachate with the Mini-Landfill leachate shows that specific conductance, pH and sodium values are similar in the two leachates. However, chloride concentrations in the Phase 2 leachate were substantially higher than the Mini-Landfill leachate based on 2015 results, but within the range observed at the Mini-Landfill based on 2019 results. The large disparity between the 2015 and 2019 chloride results for the Phase 2 Landfill have resulted in some uncertainty in the characteristic of this leachate with respect to chloride. Additional monitoring of the leachate will be required to resolve the chloride fingerprint of the Phase 2 Landfill. Concentrations of dissolved aluminum, arsenic, chromium, molybdenum, potassium and vanadium are higher in the Phase 2 leachate than the Mini-Landfill leachate.

Based on this review, leachate indicator parameters that may provide an indication of potential downgradient effects from the Phase 2 Landfill are chloride and potassium. Dissolved metals including aluminum, arsenic, chromium, molybdenum and vanadium may also provide some insight into potential leachate impacts; however, their occurrence should be interpreted with caution since these metals are subject to geochemical changes along the groundwater flow path.

6.2.3 Monitoring of Leakage Detection System

The leakage detection pump located beneath the liner of the Mini-Landfill is activated weekly by Catalyst to determine whether leachate has leaked through the collection system. Liquid was detected beneath the Mini-Landfill over the period of 4 January to 6 May 2019 and was associated with the high water-level in the wash pond over this period. A sample of the leachate was collected on 27 March 2019 and submitted for laboratory analysis for pH, specific conductance, BOD and dioxins and furans in accordance with the Landfill permit requirements. No liquid was detected in the collection system for the remainder of the 2019 monitoring year.

The results of analysis of the permitted parameters are presented below in Table 7. The results show that the leachate appears to be characterized by lower pH and higher specific conductance than typical leachate from the Mini-Landfill. No detectable dioxins and furans were measured. The cause of the elevated specific conductance is unknown.

Table 7: Analysis of Leachate from Mini-Landfill Leak Detection System in 2019

Date of Sample Collection	pH	Specific Conductance (µS/cm)	BOD_5 (mg/L)	Dioxin/Furan	
				NATO TEQ (DL=0) (pg/L)	WHO TEQ (DL=0) (pg/L)
27 March 2019	7.12	1,108,000	0.25	0	0

6.3 Monitoring of Groundwater Discharged from the Extraction Wells

The current leachate recovery system consists of one pumping well (PW99-2) that extracts groundwater from the 19 m flow zone and one pumping well (PW99-5) that extracts groundwater from the 29 m flow zone. PW99-5 represents the only recovery well where fully saturated conditions are present and the static water-level is above the well screen. Pumping well PW99-5 was offline from 9 April 2019 to 3 September 2019 and again from 3 December 2019 until the end of the year due to maintenance issues described below in Section 6.3.1.

Pumping wells PW95-8 and PW95-9, which are completed in the 11 m flow zone, have remained dry or nearly-dry since June 2000 and May 2002, respectively, and pumping well PW95-1 (11 m flow zone), which is usually nearly dry, had approximately 0.21 m and 0.13m of water above the base of its screen in August and November 2019, respectively. Pumping well PW99-4, which extracts groundwater from the 19 m flow zone, remained dry over the past year as it has in subsequent years.

Catalyst Paper recorded totalizer flow meter pumping volumes for the pumping wells on a weekly basis during 2019. The results were sent to Golder for input into the historical dewatering database and routine assessment of the performance of the dewatering wells. In addition, field measurements of pH and specific conductivity of the discharge from each pumping well was measured by Catalyst Paper on a monthly basis. The results of these measurements for the 2019 monitoring year are plotted in Figures 6, 7, 8 and 9.

6.3.1 Discharge Volumes and Discharge Rates from the Extraction Wells

Discharge rates in this section assume a pumping period from 8 January through 26 December 2019 (353 days). A total volume of approximately 1,321.4 m^3 of groundwater was removed by the extraction wells in 2019. The 2019 total volume of water extracted is approximately half of the water extracted in 2018, where 2,541 m^3 of groundwater was removed. This is primarily because PW99-5, which typically accounts for more than 99% of groundwater extraction, was not pumping for approximately 7 months (9 April 2019 to 3 Sept 2019 and 3 Dec 2019 to 31 Dec 2019). Additionally, there was less precipitation in 2019 than in 2018 (Figure C-1, Appendix C). Maintenance was required for PW99-5 because the pump stopped working in early April 2019. On 19 August 2019 Canwest Well Drilling Ltd. Installed a new pump and Catalyst repaired the electrical control system.
The pump was restarted between 3 September 2019 and 4 October 2019; however, on 26 November 2019, while Golder and Catalyst were inspecting the pump, it was noticed that the pump was cycling on and off every 10 to 15 seconds despite there being water in the pumping well. The pump was shut off for the remainder of 2019 to prevent the pump from further damage while awaiting repair.

Over the life of the extraction system (1995 through 2019), a total volume of approximately 58,747 m^3 has been extracted. Approximately 9,646 m^3 was extracted from the 11 m flow zone by PW95-1, PW95-7, PW95-8, PW95-9, PW95-10, and PW95-12 to effectively dewater the zone by 1997. In 2019, 13.3 m^3 of water was pumped from PW99-2 in the 19 m flow zone. Pumping well PW99-5, completed in the 29 m flow zone, extracted 1,243.1 m^3 of groundwater, representing the remaining 99% of the groundwater removed from the Landfill site in 2019. The discharge volume from the extraction well system has increased significantly since the well rehabilitation program conducted in March 2014 (Figures 6 and 7A).

Figure 6 presents the total pumping rate since installation of the extraction system in 1995, as a sum of the rates from each well. The maximum daily pumping rate in 2019 was 9.1 m^3/d measured on 20 February 2019.

Figures 7A, 7B, and 7C present the pumping rates for each of the pumping wells. The pumping rate at PW99-5 decreased from a maximum of approximately 27 m^3/d in 2000 to a low of approximately 2.4 m^3/d in 2013 and has since increased following the well rehabilitation program in 2014 (Figure 7B). Average discharge at PW99-5 was 3.6 m^3/d in 2019, which was lower than previous years due to problems with the pump encountered in early April that resulted in the pumping well being off for approximately 7 months. Discharge from PW99-2 is influenced by seasonal variations in precipitation and was an average of 0.043 m^3/d in 2019. Extraction of groundwater from the 1995 and 1999 pumping wells, together with reduction in recharge by the asphalt cover, resulted in the partial dewatering of the 11 m and 19 m flow zones along the southeast edge of the Landfill.

6.3.2 Quality of Pumped Groundwater

Measurements of specific conductance and pH, made on a monthly basis when available, are provided in Figures 8 and 9. A more detailed analysis of the samples collected from the pumping wells during the wet season is presented below in Section 6.4.2.

Specific conductivity at PW99-5 gradually declined over the period of record from 1999 through 2013; however, it has remained relatively constant following well rehabilitation in March 2014, with concentrations ranging from 0.244 mS/cm to 0.261 mS/cm in 2019. Similar to previous years, specific conductivity in groundwater discharged from PW99-2 varies on a seasonal basis, with higher conductivity in the late winter and early spring. Specific conductivity at PW99-2 varied between 0.233 mS/cm and 0.321 mS/cm in 2019.

In 2019, monthly values of pH at PW99-5 ranged from 5.5 to 6.9. PW99-2 was characterized by slightly higher pH, with values ranging from 6.8 to 7.4 (Figure 9). For reference purposes only, the guideline for the acceptable pH of drinking water established by the BCWQG is between 6.5 and 8.5.

6.4 Groundwater Monitoring Results

Groundwater monitoring was carried out at the Wildwood Landfill during the months of September and November 2019. During each monitoring round, groundwater levels were measured manually at all monitoring locations, and groundwater samples were collected from select monitoring wells in accordance with the monitoring program outlined in Section 5.2. As discussed in Section 5.2, a round of “snap-shot” sampling for leachate indicator parameters was also conducted in September 2019. The results of these tasks are presented below. Similar to previous years, during the snap-shot sampling, monitoring well 94-16U was nearly dry and had insufficient water to sample, and monitoring well 94-16M could not be sampled due to an obstruction in the well.

6.4.1 Water-level Monitoring

6.4.1.1 Groundwater Flow Patterns

The results from monitoring groundwater levels in 2019 are provided in Table 14. Groundwater level contours inferred from measurements made in November 2019 in the 11 m, 19 m, 38 m, regional, and bedrock flow zones are presented in Figures 10 through 14, respectively.

Figure 10 presents groundwater contours in the 11 m flow zone. Twelve of the monitoring wells in this shallow zone were dry in November 2019. Groundwater in the 11 m flow zone is inferred to flow toward the south to southeast.

Groundwater flow in the 19 m zone (Figure 11) occurs predominantly along the east edge of the site, where the 19 m silt layer is present. The figure shows that several of the wells installed within the 19 m flow zone are dry, which may be a result of the partial dewatering of this flow zone due to the operation of pumping well PW99-2. Groundwater in the 19 m zone flows toward the southeast.

A figure of the 29 m flow zone is not presented because only three wells (AH4, AH6L and PW99-5) are completed in that zone. The water levels in AH6L and PW99-5 are shown in Figure 11, as their water levels are similar to levels in nearby wells in the 19 m flow zone.

Figure 12 presents groundwater contours in the 38 m flow zone. Groundwater flow in this zone is inferred to flow toward the southeast.

Regional groundwater flow (Figure 13) appears to be influenced by the topographic protrusion immediately west of the Landfill. Water level measurements indicate that unsaturated conditions may exist at depths as great as approximately 40 m. To the south of the Landfill, groundwater in the regional aquifer, located 2 to 20 m above bedrock, flows east to southeast toward Powell River.

Groundwater in the uppermost portion of the bedrock (Figure 14) is inferred to flow east to southeast, away from the topographic protrusion west of the Landfill and toward Powell River.

6.4.1.2 Variations in Water-level Elevations

A summary of water-level measurements obtained at select monitoring locations is illustrated in Figures 15A through 15F for each groundwater flow zone. Gaps in the records indicate that the water level was too low to be measured.

In the upper (11 m) perched zone, water levels along the south edge of the Landfill decreased at most wells by approximately 3 m from 1995 to 2000, and the levels have remained relatively stable since then (Figure 15A). The 11 m flow zone has largely been dewatered, and water level measurements are indicative of standing water in the sumps of the wells located in the silt layer below the flow zone. An exception to this was apparent in December 2016, when the pumping wells were shut down for maintenance of the leachate discharge line and an associated rise in groundwater levels in the 11 m flow zone along the south edge of the Landfill was observed. Groundwater levels at these locations returned to pre-December 2016 levels during the monitoring events in 2017, 2018 and 2019.

Unlike most monitoring wells completed in the 11 m flow zone along the south edge of the Landfill, groundwater levels at 89-5 and 95-2 declined only a modest amount when the extraction system was first activated. Based on this response and the low hydraulic conductivity at 89-5 (3×10^{-7} m/s), these wells are inferred to represent low permeability sediments at the 11 m depth. Groundwater appears to enter the 11 m flow zone in the wet season and flow toward low points of the silt layer which is 11 m below the original land surface, to locations such as PW95-1.

Water levels in wells in the 11 m perched zone at other locations at the site remained relatively stable (Figure 15A), with some seasonal fluctuations evident.

Slight variations in water-levels are evident in the 19 m (Figure 15B), 38 m (Figure 15C), regional (Figure 15D) and bedrock (Figure 15E) flow zones that are attributed to annual variations in precipitation (Appendix C, Figure 1).

Similar to the 11 m flow zone, groundwater in the 19 m perched zone at the south edge of the Landfill appears to flow to low points in the silt layer. A perched flow zone at the 29 m depth has been identified only in the region near PW99-5, based on well logs. This flow zone is inferred to extend across a larger region or be hydraulically connected to other flow zones, as would be required to enable the sustained discharge from this zone observed at PW99-5. Water levels in the region near PW99-5 (Figure 15C) indicate that water flows from AH6L to PW99-5, but water at AH4 (approximately 30 m downgradient from PW99-5) is not captured by PW99-5. An estimated capture zone for PW99-5 (Golder Associates, 2000) extends laterally a width of approximately 70 m to the north and approximately 20 m to the south (longitudinally). However, the region in the immediate vicinity of AH6L shows a limited hydrogeologic response to pumping at PW99-5, located about 3 m away. An increased pumping rate at PW99-5 following the well rehabilitation program does not appear to have induced any significant lowering of the groundwater level in AH6L. This suggests there is a relatively limited hydraulic connection between PW99-5 and AH6L.

Water levels in the 38 m flow zone (Figure 15C) have exhibited similar trends as other zones that are not impacted by pumping.

Water levels in the regional flow zone (Figure 15D) have historically demonstrated the strongest response to annual variations in precipitation (Figure C-1).

Water levels in the bedrock flow zone are illustrated in Figure 15E and an assessment of the vertical flow direction between the regional flow zone and the underlying bedrock flow zone is provided in Figure 15F. This history of the vertical flow direction indicates that little gradient is evident between the bedrock and regional flow zone in the immediate vicinity of the Landfill, while downgradient of the Landfill some amount of groundwater in the bedrock typically discharges upward into the regional flow zone.

6.4.2 Chemistry Results

The results of the 2019 groundwater sampling are tabulated in Tables 8 through 13. Figures 16 A-Q present time-series plots of selected geochemical constituents and general parameters. Historical results of the dry season “snap-shot” sampling round of field parameters, alkalinity, and sulfate are plotted for each groundwater flow zone in Figures 17 A-G. Laboratory reports are provided in Appendix E.

The pumping wells completed in the 11 m zone (PW95-1, PW95-8 and PW95-9), together with those in the 19 m zone (PW99-2 and PW99-4) were dry or nearly dry at the time of sampling in the September and November 2019 sampling rounds. As a result, the composite samples (PW-Composite) obtained from the recovery system in September and November were representative of groundwater captured by PW99-5 (29 m zone) only.

Indicator parameters of the leachate from the Phase I Landfill have been identified over the course of groundwater monitoring since 1990 and are summarized in Table C-2. The following parameters are presented with their maximum concentration/level observed historically in groundwater from the recovery wells and 89-5 (located in the 11 m flow zone immediately downgradient of the closed Landfill) because concentrations representative of pure leachate are not available: specific conductance (16,700 µS/cm), pH (12.1), sulfate (2,890 mg/L), alkalinity (>15,400 mg/L), total organic carbon (TOC) (68,325 mg/L), metals, chlorinated phenols (0.122 mg/L), and dioxins and furans (26,706 pg/L[2]). Leachate from the Phase I Landfill is alkaline and carbon-rich, with a high ionic load relative to freshwater.

Some indicator parameters for leachate from the Mini-Landfill and Phase 2 Landfill are available from the leachate testing conducted in 2015 through 2019. The pH from the Phase 2 Landfill is comparable to the maximum historical pH recorded in the Phase 1 Landfill (12.1); however, in recent years a seasonal variation in pH has been observed, where the pH from the Phase 2 Landfill measured in February/March (9.4 in February 2017, 8.8 in February 2018, and 12.3 in March 2019) has been lower than the pH measured in November (11.6 in 2017,

[2] The TEQ reported was calculated using the North Atlantic Treaty Organization (NATO) method where values less than the detection limit were set equal to zero.

13.2 in 2018 and 15.4 in 2019). The pH from the Mini-Landfill is generally lower than the Phase 2 Landfill but appears to have been increasing in recent years (to 11.6 in March 2019), as shown in Figure 5. The specific conductance in the Mini-Landfill (6,645 µS/cm in March 2019 to 7,150 µS/cm in November 2019) leachate is lower than the maximum observed historically in the Phase I Landfill leachate (16,700 µS/cm). Instead, leachate from the Phase 2 Landfill up until 2019 has been slightly lower or within range of the maximum historically in the Phase I Landfill leachate, where a specific conductance of 17,150 µS/cm was measured in March and 15,410 µS/cm in November, 2019. Most of the constituents that were identified in the Phase 2 leachate, which are generally higher than the Mini-Landfill leachate (Section 6.2.2), have been found previously at similar or higher concentrations in the Phase 1 leachate at the south edge of the Landfill. Exceptions are potassium, molybdenum, and in particular, chloride, which is higher in the Phase 2 Landfill (11,900 mg/L in 2015 and 2,920 mg/L in 2019) than the Phase 1 Landfill (89 mg/L) and Mini-Landfill (2820 mg/L in 2015, 4700 mg/L in 2016, and 1,560 mg/L in 2019). As described previously in Section 6.2.2, there is some uncertainty in the chloride results for the Phase 2 Landfill because of the large disparity in the 2019 results relative to the 2015 results; therefore additional monitoring of the leachate will be required to resolve the chloride fingerprint of the Phase 2 Landfill. As such, chloride has the potential to be a representative indicator parameter for the Phase 2 Landfill, but further confirmation is required.

In general, for each geochemical constituent representative of the Phase I Landfill, concentrations along the flow path are highest at the Landfill and decrease downgradient of the extraction system. No leachate indicators have been identified downgradient of the Landfill that are considered representative of the Phase 2 Landfill. Temporal trends in groundwater chemistry based on historical results and the 2019 groundwater sampling and snap-shot monitoring results, are discussed below for individual constituents relative to baseline conditions.

6.4.2.1 Specific Conductance

Specific conductance (SC) remained at levels similar to recent years at all sample locations (Figure 16A and Figure 17B), with the exception of 89-5, where an increasing trend in SC has been observed starting in about 2016 (Figure 16A).

6.4.2.2 pH

The constituent pH has trended downward or remained between 6.2 and 8.5 in nearly all sampling locations since groundwater extraction began in 1999 (Figure 16B and Figure 17C). One exception is the pH at AH-6L which has remained largely above 8.5; however, observations since 2014 have shown a slight decrease in pH (Figure 16B). The elevated pH at AH-6L may be related to the leachate from the Phase 1 Landfill directly (associated with either flyash or lime dregs), or due to biodegradation of carbon, similar to a reaction zone in leachate from a typical carbon-rich Landfill.

6.4.2.3 Sulfate

Sulfate concentrations at the south edge of the Landfill and downgradient of the Landfill are generally higher than levels at locations upgradient of the Landfill (Figure 16C and Figure 17D). Historically, sulfate has typically ranged between 10 and 500 mg/L (but has been detected as high as 2890 mg/L) at 89-5, AH-3 and AH-6L. In 2019, sulfate ranged from 54 mg/L to 196 mg/L in samples from these wells (Figure 16C). Sulfate concentrations at

AH6L, which demonstrated a gradual upward trend commencing in 2007, was relatively stable from December 2013 to September 2018, and appear to have declined slightly from November 2018 to November 2019. Sulfate concentrations indicate that sulfate minerals in the Phase 1 Landfill continue to dissolve. Sulfate is then captured by the extraction wells and attenuates downgradient of the Landfill, possibly due more to mineral precipitation than sulfate reduction.

6.4.2.4 Alkalinity and Calcium

Alkalinity (reported as $CaCO_3$ for all samples) in the 11 m flow zone at the south edge of the Landfill has decreased since operation of the extraction system began in 1995, from a concentration at 89-5 as high as 15,400 mg/L to a concentration of approximately 254 mg/L in 2010 (Figure 16D). The increasing trend in alkalinity at 89-5 since 2010 continued in 2019 where an alkalinity of 1200 mg/L and 1230 mg/L was recorded in the September and November 2019 sampling rounds, respectively. Alkalinity in the discharge water from the pumping wells (PW-Composite) has remained relatively constant at a relatively low level in the last several years and was 64.4 mg/L in November 2019. Alkalinity upgradient of the Landfill at 93-2B was 65.8 mg/L and 68.3 mg/L in the September and November 2019 sampling rounds, respectively. Alkalinity in the 11 m and 19 m flow zones downgradient of the Landfill has decreased from between approximately 1200 and 1720 mg/L prior to 1995 to between 224 mg/L and 485 mg/L in 2019 (AH-3 and AH-6L, respectively). The decreasing trend in alkalinity in the 38 m flow zone from a high of 287 mg/L in 1994 was also observed in samples from 94-1/3 and Spring S1 up until 2004; alkalinity at those locations, together with Spring S2, has been relatively stable for the past decade, with concentrations between 70.3 mg/L and 84.8 mg/L in 2019 (Figure 16D).

Alkalinity in the 40 to 115 mg/L range upgradient of the Landfill (and up to 68.3 mg/L at 93-2B in 2019) is likely due to the mineralogy of the glacial sediments. Alkalinity elevated above these levels, where they occur downgradient of the Landfill, is likely due to dissolution of calcite and other carbonate compounds in Phase 1 Landfill material.

While calcium is a metal and is presented with metals in Tables 9 and 10, its geochemical relevance as an alkaline earth metal is more similar to alkalinity than to the other metals discussed below. Plots of historical calcium concentration are presented in Figure 16E. Calcium is the most prevalent alkaline earth metal in water samples collected at the site (Tables 9 and 10). Calcium concentrations have been fairly constant at each sample location since monitoring began, with the exception of 89-5 and AH6L. At 89-5, calcium concentrations exhibited an increasing trend until 2010 (74.1 mg/L), a decreasing trend through 2016 (36.4 mg/L) and have been relatively stable over the past two years (32.7 mg/L in 2019). At AH6L, a rise in calcium had been evident since 2014 with a maximum concentration of 80.2 mg/L recorded in December 2016 followed by a decline to 39.6 mg/L in November 2018 and then an increase to 64.4 mg/L in November 2019.

Calcium concentrations across the site ranged from 17 mg/L to 64.4 mg/L in groundwater and surface water samples in 2019. The highest concentrations of calcium along the flow path are usually present at the south edge of the Landfill at 89-5 (32.7 mg/L in 2019) and downgradient of the Landfill at AH6L (64.4 mg/L in 2019).

Overall, calcium concentrations at the site are significantly lower than alkalinity levels and may be limited by the solubility of calcite. With high alkalinity, calcium concentrations in the groundwater might be expected to remain at approximately the present levels for a considerable period of time.

6.4.2.5 Metals

Sodium, arsenic and vanadium have been identified as metals and cationic elements of interest in the groundwater at the Wildwood Landfill, based on their presence at concentrations above either the CSR or BCWQG at one or more locations in the past few years. While a comparison to the CSR and BCWQG is provided below for reference purposes only, these standards are only considered to be applicable at the point of discharge (Spring S1 and Spring S2), as described further in Section 6.4.2.10. Results from sampling in 2019 are listed in Tables 9 and 10, and the historical trends are discussed below.

Alkaline Metals

Sodium concentrations (Figure 16F) are measured to be between approximately 5 and 40 mg/L under background groundwater conditions at the site and approximately 1 mg/L in Powell River. Sodium concentrations have trended downward or remained stable since 1999 at all sample locations except AH-6L and 89-5. Sodium concentrations have been somewhat variable at AH-6L. At 89-5, located at the south edge of the Landfill, concentrations declined from 1770 mg/L in 1998 to 161 mg/L in 2011, but have demonstrated a rising trend in recent years including in 2019 where the concentration increased to 664 mg/L. Sodium concentrations have declined at the extraction wells (PW-Composite from 1,140 mg/L in 1998 to 13.7 mg/L in 2018) and at AH-3 (from 420 mg/L in 1998 to 84.0 mg/L in 2019). Concentrations at Spring S1 have been relatively stable since 2004 after declining from 114 mg/L in 1998 to 30 mg/L in 2004 (28.4 mg/L in 2019). Similarly, concentrations in Spring S2 have declined from 199 mg/L in 1994 to 22.9 mg/L in 2019.

In samples collected in 2019, sodium concentrations remained elevated above background at the edge of the Landfill and immediately downgradient of the Landfill. Sodium concentrations at background levels near the shoreline suggest that sodium is captured by the extraction wells and attenuated between the Landfill and Powell River, possibly by precipitation as sodium-carbonate, -oxyhydroxide, or -sulfate minerals.

Potassium concentrations (Figure 16G) are typically below 4 mg/L across the Site, with the exception of monitoring well 89-5, which has always been characterized by higher concentrations of potassium than elsewhere at the Site. No significant increases in potassium concentrations at 89-5 have been observed over the past 6 years. Potassium concentrations at AH-6L that were traditionally below laboratory detection limits were measurable in 2017 through 2019, reaching 8.99 mg/L in 2018 and then decreasing to 1.83 mg/L in 2019, suggesting that the 2018 concentration may have been anomalous.

Redox-Sensitive Metals

Manganese concentrations (Figure 16K) at 89-5 trended upward until 2010, downward from 2010 through 2016, and were elevated again in 2017 (1.75 mg/L) which was above the CSR drinking water (DW) standard of 1.5 mg/L. In 2018 the concentrations declined to 1.4 mg/L to below the CSR DW standard and continued to decline in 2019 to 0.774 mg/L. Manganese concentrations at all other locations have remained stable for several years, with the exception of 94-16B, where concentrations have been variable, with a downward trend since 2009 (0.042 mg/L in 2019). Elevated and fluctuating manganese levels at 94-16B prior to 2009 may be related to changes in vertical flow direction between the bedrock and overlying sediments in that area (see Figure 15F). Elevated manganese concentrations are typical of groundwater flow through a carbon-rich Landfill, due to dissolution of manganese from sediments under reducing conditions.

Trace Metals

Aluminium concentrations (Figure 16H) have been found below the CSR DW standard (10 mg/L) at all sampling locations since 1999, and below BCWQG standards (0.05 mg/L) at Spring 1 and 2 since 1999. In the past three years concentrations have remained stable relative to historical concentrations with the exception of AH-6L, where concentrations increased to 0.773 mg/L in 2017 and declined to 0.0576 mg/L in 2019.

Arsenic concentrations (Figure 16I) have been found below the CSR DW standard (0.01 mg/L) since 2004 at all sample locations except AH-6L (0.0348 mg/L in 2019). Changes in arsenic concentration at AH-6L reflect changes in redox at that location. Arsenic concentrations at the Spring S1 sample location have decreased from a maximum of 0.02 mg/L in 1992 to well below the CSR standards and BCWQG since 2004. Similar low concentrations are present at Spring S2.

Chromium concentrations (Figure 16J) have been found blow the CSR FW standards for Chromium III (0.09 mg/L mg/L) and Chromium VI (0.01 mg/l) since 2010. Concentrations have remained stable over the past nine years.

Molybdenum concentrations (Figure 16L) are typically below 0.0015 mg/L upgradient of the Landfill, with the exception of 06-2L which had concentrations of 0.0126 mg/L in 2008. Molybdenum concentrations across the Site are generally consistent with historical background groundwater concentrations with the exception of 89-5 at the south edge of the landfill and AH-3 and AH-6L downgradient of the landfill. Concentrations increased to 0.0863 mg/L at 89-5 in 2008, then decreased to 0.0005 mg/L in 2009 and remained consistent or slightly higher than background until 2019. Concentrations at AH-6L increased to 0.005 mg/L in 2009, then gradually decreased to 0.0005 mg/L in 2014 and remained consistent with background until 2019. Concentrations at AH-3 decreased from 0.0145 mg/L in 2008 to 0.0013 in 2012 and increased to 0.0082 mg/L in 2019.

Vanadium concentrations were found above the recently established CSR DW standard (0.02 mg/L) at 89-5 (0.173 mg/L) and AH-6L (0.127 mg/L) in 2018. Vanadium concentrations (Figure 16M) have trended downward since 2001 at 89-5 and AH-6L (from maximum concentrations of 5.1 and 1.06 mg/L, respectively). Research into the factors controlling the regional distribution of vanadium in groundwater in California attributed the occurrence of vanadium to areas where both the source rock and favourable geochemical conditions occurred, rather than to anthropogenic activity (Wright and Belitz, 2010). These conditions included areas characterized by granitic bedrock and alkaline groundwater conditions, which is consistent with site conditions.

6.4.2.6 Total Organic Carbon

Total Organic Carbon (TOC; Figure 16N) has typically been between 0 and 7 mg/L upgradient of the Landfill and has decreased significantly at the south edge of the Landfill from typical concentrations greater than 1,000 mg/L and as high as 68,325 mg/L to typical recent values of less than 50 mg/L. In the past three years there has been a slight increase in these concentrations at 89-5 to 112 mg/L in 2018 and 62.7 mg/L in 2019. Concentrations have recently ranged between 1 and 60 mg/L downgradient of the Landfill, and from 1 to 25 mg/L near the shoreline.

6.4.2.7 Chlorinated and Non-chlorinated Phenols

The 19 March 2015 amended Landfill permit requires that only AH-6L and AH-3 be sampled for chlorinated and non-chlorinated phenols. Results of analysis of phenols from sampling in 2019 are listed in Table 11 and the historical trends of total chlorinated phenols are presented in Figure 16O. In 2019, there were no detectable concentrations of chlorinated and non-chlorinated phenols at AH-3 or AH-6L.

6.4.2.8 Dioxins and Furans

Dioxins and furans are compounds that are similar in chemical structure to 2,3,7,8- tetrachlorodibenzo para dioxin (TCDD) that are present throughout the environment. The World Health Organization (WHO) and the North Atlantic Treaty Organization (NATO) have developed a method of quantifying the toxic equivalency (TEQ) of 17 dioxin and furan congeners by multiplying the concentration of each congener by a corresponding toxic equivalency factor (TEF). No guidelines for dioxins and furans in water have been established in Canada. A maximum contaminant level of 30 pg/L for 2,3,7,8-TCDD has been established by the United States Environmental Protection Agency for drinking water, and is noted here for reference purposes only. Analysis of water samples at such low concentrations requires highly-specialized laboratory equipment. The dioxin and furan analyses were conducted by ALS Environmental since 2017, while in previous years the analyses were conducted by Pacific Rim Laboratories.

Table 12 lists results of dioxin and furan analyses conducted on samples collected in 2019. Congeners were detected at all monitoring locations; however, of these, the concentrations measured at AH-6L and 89-5 are the only concentrations considered to be distinguishable from the laboratory method blank, with the exception of 94-1/3 and 94-19L where a single parameter (Total- PeCDD) is distinguishable from the laboratory method blank.

Figure 16P presents the historical trends in total TEQ for dioxins and furans. In the figure, the total TEQ is presented using the NATO TEF's and an assumption that non-detection represents a concentration of 0 pg/L. This approach tends to highlight instances in which the TEQ for dioxins and furans is above detection limits. Dioxins and furans concentrations were generally similar in 2019 to recent years, with the exception of upgradient well 93-2B where the TEQ concentration increased to 0.348 pg/L. The TEQ concentration at AH-6L has been relatively stable (with a TEQ of 5.48 pg/L in 2019) since 2006 after exhibiting a prior decreasing trend. Dioxins and furans TEQ at 89-5 were consistent with those measured in recent years (16.7 pg/L in 2019) and well below pre-2004 levels.

6.4.2.9 Halogens

Monitoring of the halogens chloride, bromide, and fluoride at sample locations began in 2008. Halogens are often non-reactive solutes in groundwater and may therefore be useful as tracers when present in a source of contamination. Sources of chloride could include salts used for road de-icing, Phase 1 Landfill material, degradation of chlorinated chemicals such as dioxins, chlorinated phenols, and PCBs, flyash, and potential continued dissolution of marine anhydrite precipitates formed in glaciofluvial sediments during isostatic rebound 5,000 to 10,000 years ago. More importantly, analysis of the Phase 2 leachate indicates that Phase 2 Landfill material is characterized by particularly high chloride concentrations, although further monitoring is required to confirm this.

Results from samples collected in 2019 are listed in Table 8, and Figure 16Q presents chloride results since 2008. In 2019, chloride concentrations ranged from 164 mg/L at the South Edge of the Landfill to 10.2 mg/L downgradient of the Landfill. The background concentrations upgradient of the Landfill have typically been less than 4 mg/L, however background concentrations increased slightly to 10.3 mg/L in 2019. The concentrations of chloride at Springs S1 and S2 were consistent with historical trends. Spring 1 and Spring 2 had concentrations of 19.9 mg/L and 22.9 mg/L, respectively, in September 2019 and 23.1 mg/L and 22.9 mg/L, respectively, in November 2019.

A review of chloride concentrations over time (Figure 16Q) shows that a rise in chloride has occurred in 89-5, increasing from 32.6 mg/L in August 2013 to 200 mg/L in December 2018, and decreasing to 164 mg/L in November 2019. Because leachate from the Phase 2 Landfill may be characterized by elevated chloride, this trend of rising chloride at 89-5 may be attributable to the Phase 2 Landfill. The chloride concentrations appear to vary seasonally, with higher concentrations measured during the wet season than the dry season.

Bromide concentrations in 2019 ranged from less than laboratory detection (<0.05 mg/L) to 0.31 mg/L at AH-6L. Fluoride concentrations ranged from levels below the laboratory detection limit (<0.020 mg/L) to a maximum of 0.28 mg/L in a sample taken from the 29 m flow zone at AH-6L.

6.4.2.10 Comparison to Applicable Criteria

The groundwater chemical analysis results were assessed relative to the groundwater standards specified in the B.C. Contaminated Sites Regulation (B.C. Reg. 375/96) last amended on 24 January 2019 by B.C. Reg. 13/2019. for reference purposes only.

The British Columbia Approved and Working Water Quality Guidelines (Criteria) (BCWQG), updated March 2018, represent criteria that surface water should meet. Since the BCWQG were derived for discharge directly to surface water, they are not applicable to groundwater at distances greater than or equal to the 10 m setback from the high water mark of an aquatic receiving environment (ENV Technical Guidance on Contaminated Sites, Document No. 15, Version 2.0, 1 November 2017). For this site, the only sampling locations within 10 m of Powell River are Springs S1 and S2. The results from these locations were compared with the drinking water criteria (DW) contained within the Summary of Water Quality Guideline for Drinking Water Sources, and with the freshwater aquatic life criteria (FAW) contained within the Summary of Water Quality Guidelines for Aquatic Life, Wildlife & Agriculture.

Table 13 summarizes the exceedances of the CSR standards and the BC Water Quality Guidelines. In 2019, exceedances of the CSR were at the south edge of the Landfill at 89-5 and immediately downgradient of the Landfill at AH-6L.

Groundwater Exceedances

While there were no exceedances of the CSR standards at the approximate point of discharge (Spring S1 and Spring S2), other locations along the groundwater flow path where exceedances of the CSR were observed are described below for reference purposes.

Metals concentrations in groundwater exceeded the CSR standards at two locations in 2019: 89-5 (11 m flow zone) and AH-6L (29 m flow zone). The metals listed in Table 13 refer to the dissolved form for groundwater samples. BCWQG for metals refer to the total form.

Arsenic exceeded the CSR DW standard (0.01 mg/L) in the 29 m flow zone at AH-6L (0.0348 mg/L). Arsenic has exceeded the DW standard at AH-6L since 1998 with two exceptions. The concentrations peaked in 2001, followed by a decline to 2005, and have remained relatively stable from 2006 through 2019. The increase and subsequent decline may be due to changes to redox conditions caused by Landfill leachate.

Sodium exceeded the CSR DW standard (200 mg/L) at AH-6L (319 mg/L) and at 89-5 (664 mg/L). Sodium is a leachate indicator and is elevated at these wells located in one of the most impacted regions of the site.

Vanadium exceeded the CSR DW standard of 0.02 mg/L at AH-6L (0.127 mg/L) and 89-5 (0.173 mg/L). Vanadium may have been released from the native overburden and bedrock materials under the geochemical conditions that are characteristic of the Landfill.

Surface Water Exceedances

There were no exceedances of the CSR standards nor the BCWQG's in the surface water samples taken from Spring S1 and Spring S2.

6.4.2.11 Summary of the Water Quality Monitoring in 2019

The 2019 water quality results are generally consistent with previous years.

Phenols remained at non-detectable or low levels. Dioxins and furans were found at concentrations that were indistinguishable from laboratory method blanks at all locations except 89-5, located at the south edge of the Landfill, and AH-6L, located immediately downgradient of the Landfill. Some metals concentrations continue to exceed CSR standards immediately downgradient of the Landfill and at the leading edge of the Landfill, with monitoring wells AH-6L and 89-5 representing the most impacted locations.

Increases in chloride concentrations at monitoring well 89-5 suggest that the Phase 2 Landfill may be affecting the groundwater chemistry at the leading edge of the Landfill. However, no other changes in groundwater chemistry have been observed downgradient of the Landfill or in the receiving environment that can be attributed to the Phase 2 Landfill.

The apparent decline or stabilization in specific conductivity, pH, sulphate, alkalinity, TOC, dissolved metals, total chlorinated phenols and dioxins and furans at most monitoring locations downgradient of the Landfill and in Spring S1 is an indication that the leachate control measures are resulting in some renovation of the shallow groundwater quality.

6.4.3 Quality Assurance/Quality Control

One set of blind field duplicate samples was collected in each of the September and November 2019 monitoring rounds and submitted for analysis. For duplicate analyses, a control limit of 20% for the Relative Percent Difference (RPD)[3] is applied to the original and duplicate samples that are greater than or equal to five times the method detection limit. Where one or more concentrations were less than five times the detection limit (MDL), values that differed by less than a Difference Factor (DF) of 2 were considered acceptable (where DF is the absolute difference between the two values divided by the method detection limit).

A summary of the relative percent difference (RPD) for the duplicate samples collected in 2019 is provided in Appendix F, Tables F1 and F2. Due to the very low concentrations of dioxins and furans, control limits are not applied to duplicate analyses for each congener. A review of the QA/QC data for all constituents except the dioxins and furans at 94-19L and nitrate at AH-6L indicates that the results were considered acceptable according to the criteria outlined above.

[3] Where RPD = |A-B|/((A+B)/2)*100

6.5 Results of Dustfall and Air Quality Monitoring

The results of the dustfall and air quality monitoring carried out in 2019 are presented in Appendix G. A brief summary of the results is presented below.

6.5.1 Dustfall Monitoring at Wildwood Landfill

A laboratory error in preparing the algaecide for the dustfall sampling program resulted in the distorted dustfall results during the 2019 program and reduced confidence in the measurements. Because we cannot have confidence in the dustfall data from 2019, the results of the program are not presented.

6.5.2 Air Quality Monitoring in the Community of Wildwood

The TEOMs at the Wildwood School air quality monitoring station were replaced in January 2019 with an API T640 that measures both $PM_{2.5}$ and PM_{10}.

The $PM_{2.5}$ and PM_{10} concentrations measured at the Wildwood School remained well below all applicable air quality objectives over the 2019 monitoring period. The forest fire season that typically produces higher than average ambient concentrations of PM was less active in 2019 due to precipitation in June and July, and thus the consequent elevated concentrations that would typically be observed were not observed in summer and early fall.

Average PM_{10} results from Hi-Vol sampling program during the summer remained below the BCO and were approximately 9% of the BCO.

Amendments in the addendum letter to the Landfill permit allow for the cessation of PM_{10} monitoring at Wildwood School following favourable results in 2019.

The contribution of the Landfill to ambient $PM_{2.5}$ and PM_{10} remains minimal.

7.0 CONCLUSIONS AND RECOMMENDATIONS

The results of the 2019 groundwater sampling program continue to confirm that the closure and groundwater recovery controls related to the Phase 1 Landfill have resulted in the reduction of contaminants downgradient of the Landfill. Groundwater monitoring results from 2019 indicate that groundwater discharging to Powell River and groundwater recovered by the extraction system (PW-Composite) continue to have water of sufficient quality to meet the standards of the BC Contaminated Sites Regulation.

Leachate from the Phase 2 Landfill is characterized by higher concentrations of potassium, molybdenum, and in particular, chloride, than the Phase 1 Landfill, although there is some uncertainty in the chloride fingerprint. Accordingly, chloride may be a good indicator of Phase 2 leachate, but further confirmation is required. In groundwater at the south edge of the Landfill at 89-5, an increase in alkalinity, chloride, sodium and associated specific conductivity has been apparent in recent years, with a marked increase in chloride in 2018 followed by a relatively minor decline in 2019. There is no evidence to suggest that the Phase 2 Landfill has impacted groundwater quality further downgradient.

Liquid has been detected in the leak detection system of the Mini-Landfill since 2017. The source of the liquid is currently unconfirmed. The leak detection system is underlain by a low-permeability liner constructed of soil-bentonite. Water that collects in the leak detection system is pumped to the wastewater treatment system. Further investigation is required to determine the source of the water.

PW99-5 was not operational between 9 April 2019 to 3 September 2019 and 3 December 2019 into 2020. Dry season sampling occurred in September during the period where PW99-5 was offline, and the wet season sampling occurred in November 2019 while the system was online. During both sampling events, concentrations were generally consistent with previous years, including the trends at 89-5 noted above. The effects of PW99-5 being offline from 3 December 2019 into 2020 will be monitored during the 2020 sampling events.

A laboratory error in preparing the algaecide for the dustfall sampling program resulted in the distorted dustfall results during the 2019 program and reduced confidence in the measurements. The 2019 dustfall monitoring data was disregarded.

The TEOMs at the Wildwood School air quality monitoring station were replaced in January 2019 with a API T640 that measures both $PM_{2.5}$ and PM_{10}. Concentrations measured at the Wildwood School remained well below all applicable air quality objectives over the 2019 monitoring period. The forest fire season was less active in 2019 and thus elevated concentrations that would typically be observed were not seen in summer and early fall of 2019. The average PM_{10} results from Hi-Vol sampling program at the Landfill remained well below the BCO at approximately 9% of the BCO. Amendments in the addendum letter to the Landfill permit allow for the cessation of PM_{10} monitoring at Wildwood School following favourable results in 2019. The contribution of the Landfill to ambient $PM_{2.5}$ and PM_{10} remains minimal.

It is recommended that the groundwater and air quality monitoring program for the Wildwood Landfill continue for 2020. Additional recommendations for the 2020 monitoring year are presented below:

- In addition to monitoring the leakage detection system beneath the Mini-Landfill for the presence of flow, efforts should be made to monitor for the presence of leachate beneath the Phase 2 liner on a semi-annual basis.

- To assess the source of water collected in the Mini-Landfill leak detection system, further investigation is required. To aid in determining the source of water in this system the following is recommended.
  - Addition of chloride and sodium to the analytes sampled from the Mini-landfill leakage detection system; and
  - Installation of a surveyed water level gauge in the truck wash pond, periodic monitoring of the pond water level, and comparison of this water level elevation data with the surveyed elevation of the geomembrane liner edge.
- Leachate samples from the Mini-Landfill and Phase 2 Landfill are currently analysed for pH, specific conductance, BOD_5 and dioxins and furans on an annual basis. Analysis of both the Mini-Landfill and Phase 2 Landfill leachate is recommended in 2020 for further leachate characterization. Recommended analyses consist of dissolved chloride, dissolved metals, alkalinity, sulphate, nitrate, total dissolved solids (TSS) and total organic carbon (TOC).
- It is recommended that a provision for discontinuing the sulphide analysis be incorporated into the Design and Operating Plan.
- The addition of chloride to the analytes sampled in the snapshot sampling program, and the addition of four monitoring wells (94-2, 94-3, 94-10 and AH4) to the snapshot sampling network is recommended for 2020. Once the results of the 2020 monitoring program have been reviewed, a determination can be made as to whether these additional sampling provisions are required moving forward or should be refined.
- The recovery well rehabilitation program conducted in March 2014 has resulted in increased efficiency of the pumping wells, particularly PW99-5. However, the increased efficiency at PW99-5 has only resulted in a limited response to pumping at nearby monitoring well AH6L. Additional efforts to recover impacted groundwater from this location may require the installation of an additional recovery well.
- Recommended changes to the air quality and dust monitoring programs include discontinuing the dustfall monitoring program and focusing on particulate monitoring at the James Thompson School. PM_{10} HiVol sampling on-site is also considered redundant and should be removed from the program.
- Despite the disregarded 2019 dustfall data, proactive investigation into other potential causes for elevated dustfall seen occasionally in previous monitoring years should be undertaken during future dustfall monitoring. This investigation could include:
  - Observation of dust generation on the Landfill access road, particularly on the Landfill itself, and consideration of a basic dust suppression program (potentially including water and/or gravel application).
  - Trimming and maintenance of trees in close proximity to the dustfall stations prior to the dustfall monitoring period to minimize plant/animal contamination and ensure Landfill activities are being accurately captured in the dustfall measurements.
- Section 6.1 of the closure plan contained in the 2012 D&O Plan should be updated and the updated plan should be presented in a standalone report in 2020.
- The operating plan should be updated to account for the large quantity of sand being placed in the Landfill. The updated plan should be presented in the 2020 standalone report.
- Further testing and assessment of the sand disposed at the Landfill is recommended in 2020 to evaluate the feasibility of diverting the sand from the Landfill through beneficial use.

8.0 CLOSURE

We trust that this annual report meets your current requirements. Should you have any questions or comments, please do not hesitate to call. Connie Romano, PGeo, was the professional responsible for the groundwater aspects of this report. Colin L.Y. Wong, PEng, was the professional responsible for the landfill operational planning, geotechnical stability and landfill settlement aspects of the report. Chris Madland, BSc, Senior Air Quality Scientist, was the professional responsible for the air quality aspects of this report.

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